Ecological Studies 212

Michael Bredemeier, Shabtai Cohen
Douglas L. Godbold, Elve Lode
Viliam Pichler, Patrick Schleppi
Editors

Forest Management and the Water Cycle

An Ecosystem-Based Approach

Springer

Ecological Studies, Vol. 212

Analysis and Synthesis

Ecological Studies

Further volumes can be found at springer.com

Volume 191
Wetlands: Functioning, Biodiversity Conservation, and Restoration (2006)
R. Bobbink, B. Beltman, J.T.A. Verhoeven, and D.F. Whigham (Eds.)

Volume 192
Geological Approaches to Coral Reef Ecology (2007)
R.B. Aronson (Ed.)

Volume 193
Biological Invasions (2007)
W. Nentwig (Ed.)

Volume 194
Clusia: A Woody Neotropical Genus of Remarkable Plasticity and Diversity (2007)
U. Lüttge (Ed.)

Volume 195
The Ecology of Browsing and Grazing (2008)
I.J. Gordon and H.H.T. Prins (Eds.)

Volume 196
Western North American Juniperus Communites: A Dynamic Vegetation Type (2008)
O. Van Auken (Ed.)

Volume 197
Ecology of Baltic Coastal Waters (2008)
U. Schiewer (Ed.)

Volume 198
Gradients in a Tropical Mountain Ecosystem of Ecuador (2008)
E. Beck, J. Bendix, I. Kottke, F. Makeschin, R. Mosandl (Eds.)

Volume 199
Hydrological and Biological Responses to Forest Practices: The Alsea Watershed Study (2008)
J.D. Stednick (Ed.)

Volume 200
Arid Dune Ecosystems: The Nizzana Sands in the Negev Desert (2008)
S.-W. Breckle, A. Yair, and M. Veste (Eds.)

Volume 201
The Everglades Experiments: Lessons for Ecosystem Restoration (2008)
C. Richardson (Ed.)

Volume 202
Ecosystem Organization of a Complex Landscape: Long-Term Research in the Bornhöved Lake District, Germany (2008)
O. Fränzle, L. Kappen, H.-P. Blume, and K. Dierssen (Eds.)

Volume 203
The Continental-Scale Greenhouse Gas Balance of Europe (2008)
H. Dolman,R.Valentini, and A. Freibauer (Eds.)

Volume 204
Biological Invasions in Marine Ecosystems: Ecological, Management, and Geographic Perspectives (2009)
G. Rilov and J.A. Crooks (Eds.)

Volume 205
Coral Bleaching: Patterns, Processes, Causes and Consequences (2009)
M.J.H van Oppen and J.M. Lough (Eds.)

Volume 206
Marine Hard Bottom Communities: Patterns, Dynamics, Diversity, and Change (2009)
M. Wahl (Ed.)

Volume 207
Old-Growth Forests: Function, Fate and Value (2009)
C. Wirth, G. Gleixner, and M. Heimann (Eds.)

Volume 208
Functioning and Management of European Beech Ecosystems (2009)
R. Brumme and P.K. Khanna (Eds.)

Volume 209
Permafrost Ecosystems: Siberian Larch Forests (2009)
A. Osawa, O.A. Zyryanova, Y. Matsuura, T. Kajimoto, R.W. Wein (Eds.)

Vol. 210
Amazonian Floodplain Forests (2011)
Junk, W.J.; Piedade, M.T.F.; Wittmann, F.; Schöngart, J.; Parolin, P. (Eds.)

Vol. 211
Mangrove Dynamics and Management in North Brazil (2010)
Saint-Paul, Ulrich; Schneider, Horacio (Eds.)

Michael Bredemeier · Shabtai Cohen
Douglas L. Godbold · Elve Lode
Viliam Pichler · Patrick Schleppi
Editors

Forest Management and the Water Cycle

An Ecosystem-Based Approach

Springer

Editors

Michael Bredemeier
Universität Göttingen
Forest Ecosystems Research Centre
University of Göttingen
Büsgenweg 2, D-37077 Göttingen
Germany
mbredem@gwdg.de

Douglas L. Godbold
Environment Centre Wales
Bangor University
Gwynedd, LL57 2UW
UK
d.l.godbold@bangor.ac.uk

Viliam Pichler
Department of Natural Environment
Faculty of Forestry
Technical University in Zvolen
T.G. Masaryka 24, 960 53 Zvolen
Slovakia
pichler@vsld.tuzvo.sk

Shabtai Cohen
Institute of Soil, Water and Environmental Sciences
ARO Volcani Center
POB 6, Bet Dagan 50250
Israel
vwshep@agri.gov.il

Elve Lode
Tallinn University
Institute of Ecology
Uus Sadama 5, 10120 Tallinn
Estonia
and
Swedish University of Agricultural Sciences, Department of Soil and Environment
P.O.Box 7001, SE-75007 Uppsala
Sweden
elve.lode@tlu.ee

Patrick Schleppi
Swiss Federal Institute for Forest Snow and Landscape Research
Zürcherstr. 111, CH-8903 Birmensdorf
Switzerland
schleppi@wsl.ch

ISSN 0070-8356
ISBN 978-90-481-9833-7 e-ISBN 978-90-481-9834-4
DOI 10.1007/978-90-481-9834-4
Springer Dordrecht Heidelberg London New York

Library of Congress Control Number: 2010938711

Cover illustration: A reductionist view of forest-water interactions. Two foci of forest management are provisions of pure drinking water and the danger of floods. Painting by Shalhavit S. Cohen.

Printed on acid-free paper

Springer is part of Springer Science+Business Media (www.springer.com)

Contents

Section II Forest Structures, Management and Water Fluxes

Section III Scales and Approaches for Forest – Water Interactions

Forest Management and the Water Cycle – Introduction to the Challenge

Michael Bredemeier

How and how much can forest management contribute to optimize the water cycle with respect to resource provision on the one hand, and mitigation of water related hazards on the other? This question is the golden thread along which the contributions in this volume are strung. The book synthesizes research on the interactions of forest management and the water regime of forests in Europe and beyond, covering geographic and climatic gradients from Iceland to Israel and from southern Spain to Estonia and Finland. The collection of people and their ideas on this topic was made possible through the COST Action “Forest Management and the Water Cycle (FORMAN)”, which was launched in 2007 and will run until 2011. The acronym COST stands for “cooperation in the field of scientific and technological research”. It is the longest established funding scheme of the EU for scientific cooperation and exchange in Europe and beyond, and has run for more than 35 years (http://www.cost.esf.org/).

The scientific consortium of FORMAN aims at the enhancement of knowledge on forest–water interactions in Europe, and the elaboration of science-based guidelines for the improvement of the management of forests in their delivery of the ecosystem service “purification and storage of water”.

Water is unquestionably a key resource for all life on earth, probably the most important one besides of solar radiation. These two together basically constitute climate at any given place on earth, which in turn determines natural potential productivity of ecosystems.

However, besides the resource aspect, a potential hazard to the human population is also related to water. Along river systems and in their larger floodplains, peak flow rates can entail devastating floods. It is speculated that the frequency and intensity of such floods are currently increasing because of global climatic change and a concomitant amplification of extreme weather situations (IPCC 2007).

Forests are an essential element of the European cultural landscapes. They cover about a third of Europe’s land area (27% in central, 32% in southern and 50% in

---

M. Bredemeier
Forest Ecosystems Research Centre, University of Göttingen,
Büsgenweg 2, D-37077, Göttingen, Germany

northern Europe according to the UN Food and Agricultural Office's 2000 Forest Resources Assessment Report). With respect to the water resource, the importance of forests has to be weighted even higher than their proportion of total area, since forested landscapes are the preferred sources for generation, storage and export of drinking water to supply the human population.

Almost all forest land in Europe is more or less intensively managed, "natural forest" in a strict sense can hardly be found, a fact which is not well known to the general public. Nevertheless, forests are highly valued for their relative closeness to natural ecosystems, and for the various services that they provide. Among those services, water regulation and supply from forest soils and aquifers is one of the most prominent. Forest ecosystems supply the high quality drinking- and process-water resource for the human population and also safeguard against both flooding and erosion by retaining water and thus delaying and mitigating peak flows.

The protective function of forests with respect to water quality and water-related hazards is potentially at risk under a changing climate and changing land-management practices. The water budgets of forest ecosystems are heavily dependent on climate and forest structure. The latter is determined by the management measures applied in the forestry sector, such as tree species selection, stand structure and density management, harvesting methods, etc. In the next decades, both climate and forest-management practices are expected to change. The intention in many European countries will be to establish forest stands that are closer to the potential natural forest vegetation. This means a general trend of increasing the area of broad-leaved and mixed stands while reducing the area of mono-specific conifer plantations that had been favoured in many European countries in the past. In other European countries, however, there are large campaigns for (re)afforestation, which partly comprise monocultures. All these different trends have to be considered when assessing the overall forest–water interaction in Europe. The chapters in this volume highlight them, both in the form of thematic reviews and of selected regional case studies.

Climatic change as the major driver will most probably result in a stronger precipitation gradient from (north)western to (south)eastern Europe – corresponding to the amplification of an already given gradient (Fig. 1).

The combination of changing forest-management practices with a changing physical and chemical climate is forming an unprecedented collection of management constraints for European forests. The contributions in this book cover environmental gradients from the humid Atlantic to the dry continental and Mediterranean climates and from lowlands to high mountains. Forest–water relations over these climatic gradients, scenarios of climatic changes, and the influence of forest-management options are evaluated, with scales of study ranging from local intensive plot measurements to regional and continental (European) integration.

The book is divided into **three sections**, reflecting three levels of forest and water interactions. The first section deals with the basic physical, chemical and biological processes. The topics cover the soil/water-biota-atmosphere continuum in the forest. The second section deals with individuals and communities, and with management practices that are usually applied at this scale. Thus, this section

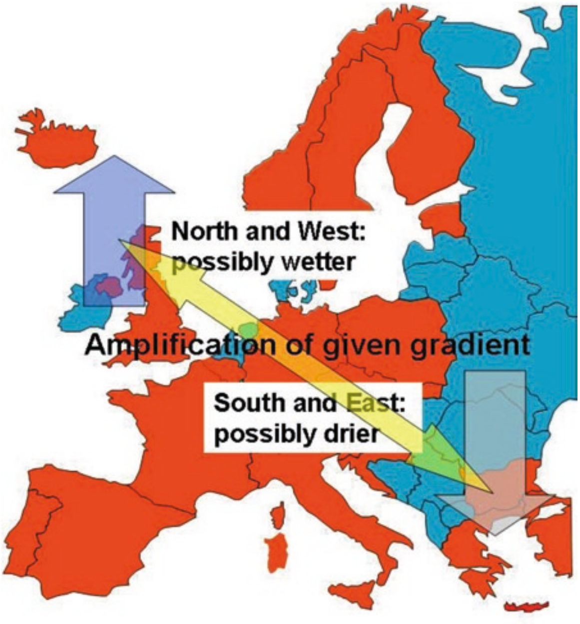


Fig. 1 Simplified general scenario of amplification of a climatic gradient across Europe; countries in red are COST Action "Forman" members, Israel is a member out of the map range

describes the species composition of the various plant communities and afforestation campaigns, management of these communities and soil management. It is the section closest to the application of scientific expertise in practical forest management. The third **section** targets the large scale view of forests, e.g. river sized watersheds. First, the different approaches for applying small scale understanding of the systems to larger scale problems are addressed. Then, topics like the hydrological consequences of forestry practices are considered.

The **first** section of this book comprises a systematic review of the affected processes in different ecosystem compartments, from the physiologically active interfaces below- and aboveground, to the forest catchment scale, and further to freshwater chemistry and biology in forest areas, which are likely to be affected by climatic- and system changes.

The second part of the section turns to case studies in this context, resulting from original research contributed by members of the COST consortium.

Rewald et al. put a focus on "Hydrological Effects on Below Ground Processes in Temperate and Mediterranean Forests". The belowground part of a forest ecosystem is intrinsically difficult to study but more and more good methods have become available in addition to longer established parameters. A challenge is to quantify the variability of physiological and anatomical root traits in heterogeneous soil environments as well as interactions with mycorrhizas.

Centritto et al. turn to the complementary part, “Above Ground Processes – Anticipating Climate Change Influences”. They look in detail at leaf to tree level processes and the effects that rising temperature, drought and extreme weather events exert upon them, but also widen the focus to address species differences in water relations and canopy structure across Europe.

Jan Čermák and Nadezhda Nadezhdina present a strong methodological scope in their chapter “Field Studies of Whole-Tree Leaf and Root Distribution”, starting from individual organs (roots or branches) but finally covering whole tree and stand level measurements up to the landscape scale.

Schleppi et al. in their contribution “forested water catchments” bring in the whole-catchment scale and demonstrate the usefulness of experiments at that level, as catchments are usually “integrators” of different ecological and biogeochemical effects in producing their runoff.

Christine Alewell and Peter Bebi write on “Forest Development in the European Alps and Potential Consequences on Hydrological Regime”. They cite the fine metaphor of mountains as our “water towers” and investigate how forest regrowth, often occurring after pasture land abandonment in the Alps will affect their function.

Ernst Leitgeb and Karl Gartner take us back down from the mountains, turning to “vulnerability of Norway spruce in Austria’s lowlands”. The extremely warm summer of 2003 highlighted the reduced vitality of Norway spruce in the lower parts of Austria during hot and dry years. It is very important to find alternative silvicultural options to Norway spruce stands in this and many other European regions.

Boris Vrbek and Ivan Pilaš work on water table management in lowland forests of south-eastern Europe. They show how indices for recognition of the onset of drought should be developed and implemented, and advocate conservation of water surplus from the winter season further into the growing season when unfavourable conditions of water scarcity occur.

Gabriel Schiller turns our attention to the dry environments in his chapter “The Case of Yatir Forest”. This case study is located in Israel and consists of *Pinus halepensis* (Mill.) forest, where he measured water use employing the heat pulse method.

Finally in the first section, Moreno et al. show us “How Mediterranean Deciduous Trees Cope with Long Summer Drought: the Case of *Quercus pyrenaica* Forests in Western Spain”. This species seems well adapted to surviving summer drought while maintaining photosynthetic activity and apparently having a non-conservative water consumption. The authors explain the strategy and mechanism of survival of those forests.

The **second** section examines more closely the forest management side of the overall issue with due consideration of its potentials as well as its limitations. As water availability might change dramatically in Europe and adjacent regions, it will affect existing tree species and their ability to cope with a lack or excess of water, soil salinity and fire frequency and intensity. Tree species and provenances which are the most suitable to deal with water stress (both under excess and scarcity) are identified and options for their future management are evaluated.

Considerable changes in European forests’ tree species composition and structures will be driven by silvicultural adaptation strategies to climatic change over the next decades. Also optimisation of afforestation and reforestation with respect to forest–water interactions is comprised, as well as mitigation of detrimental consequences for surface- and groundwater quality, water yield, groundwater recharge, runoff and erosion control in forested landscapes.

Pichler et al. start with an overview of "Changes in Forest Cover and its Diversity". This chapter provides an outline of the natural forest cover dynamics in Europe, as well as of its past, current and projected changes, which are driven by the two main factors climatic change and land use change.

Raftoyannis et al. contribute a chapter on "Afforestation Strategies with Respect to Forest–Water Interactions". Afforestation may offer attractive ecological perspectives, such as better peak flow control combined with biodiversity support, along with attractive economic returns, for instance from energy wood grown in short rotations. This is particularly the case in the moister environments of Europe with ample precipitation supply. However, where water supply gets more limited, a situation of distinct trade-off may emerge very quickly.

Lukac et al. give information on "Tree Species' Tolerance to Water Stress, Salinity and Fire". They describe tree species and provenances used in European forestry practice which are the most suitable to deal with those stressors and constraints.

Ben-Hur et al. write on "Overland Flow, Soil Erosion and Stream Water Quality in Forest Under Different Perturbations and Climate Conditions". They delineate the relationships between forest and climate conditions, and their effects on runoff, soil erosion and stream water quality, when subjected to various perturbations. The discussion addresses water-driven soil erosion.

Novák et al. make a contribution on the "Forest Management Effects on Below-Ground Hydrological Processes". They point out the value of comparative studies in managed forests and forests following natural dynamics. Important factors related to the forests' management are identified and their influence described both qualitatively and quantitatively.

Concluding the second section, Planinšek et al. expand on "Adjustment of Forest Management Strategies to Changing Climate". They put a focal point on the impacts of forest management and climate change on the interaction of forests and surface and ground water in boreal, temperate and Mediterranean regions of Europe and discuss measures to mitigate negative impacts. Approaches can be quite different as the forest management goals and strategies differ between regions.

The third section has questions of scaling at its core, starting with the important task of up-scaling from small resolution process information and ending at large spatial scale assessments. Scaling is an important tool for integration of research results with management options and society needs. The different scales from small headwater catchment to major river basin are all relevant in the assessment of forest–water interactions. The authors demonstrate that scales can serve as a structuring principle, both in the data-gathering process and in evaluation and assessment exercises. For smaller catchments and at plot-scale, there are frequently hydrological case studies available, whereas on the larger scale, remote sensing techniques and blending of available information layers from maps and GIS are often applied.

Starting from the fundamentals of scaling, authors in Section III review upscaling approaches for the water fluxes in forests, and in particular small scale and large scale approaches to forest and water interactions. General papers are followed by case studies related to scaling applications.

The section starts with an introductory overview on fundamental concepts and techniques of scaling including application of GIS in forest ecosystems and wetlands given by Elve Lode, Matthias Langensiepen, Jüri Roosaare, Gebhard Schueler and Harri Koivusalo.

Ivan Pilaš et al. refer to the EU Floods Directive in their chapter "Multidimensionality of Scales and Approaches for Forest–Water Interactions". The Directive (EC 2007) emphasises better utilization of forest hydrological functions. The mitigation potential of forests should be developed and be aimed at reducing runoff formation or mitigating adverse impacts of running waters. To alleviate the effect of drought in the future, forest management has to evaluate various adaptive strategies for drought impact management.

Matthias Langensiepen expands on the "Fundamentals of Model Scaling in Forest Ecology". He summarizes statistical techniques for scale identification and presents the establishment of scaling laws from a mathematical point of view. Structure and function are stressed as important concepts for understanding tree responses to changing environments. Methods of translating models across spatial scales are categorized in the final part of this chapter.

Chris Eastaugh et al. put the focus on "Up- and Downscaling Model Approaches". This is indeed a very central and crucial scale range as management measures are usually applied at the plot scale and ecological (as well as aesthetical) effects of forest management are prominently recognized at the landscape level. The authors put a special focus on how water-related processes can be integrated so that point or plot information may be generalised to larger scale applications.

Harri Koivusalo et al. write on "A Small Catchment Scale Approach … in Boreal Landscape". This chapter demonstrates how a forest growing on peatland or mineral soil in the boreal region can be parameterised as a hydrological response unit (HRU), and how environmental effects and effects of forestry on the water cycle can be discerned and described.

Gebhard Schüler et al. contribute a "Large Scale Approach to Forest and Water Interactions". Integrated land and water management at larger scale needs to find a way of combining technical (engineering) watershed management and a scientific based precautionary land-use management. A large scale European approach across political borders was laid down by the European Water Framework and Floods Directives.

Andreas Wahren et al. write on "Model-Based Assessment of Forest Land Management on Water Dynamics at Various Hydrological Scales – A Case Study". They point out that emphasis should be put on the detection of synergy effects between a good ecological status of the water bodies and flood protection. This approach combines socio-economic methods with state-of-the-art hydrological modelling and the available information about climatic change.

Elve Lode et al. contribute information on “Typological Up-Scaling of Wooded Peatlands”. This chapter introduces GIS based case study results of the small wooded Estonian Selisoo mire, located in the eastern part of Estonia and belonging to the Ramsar sites’ Muraka mire complex.

In the final chapter of Section III, Sultan Bekiroglu and Ömer Eker go deepest into the societal context with their contribution “Difficulties of Scaling in Forest and Water Management in Urban Areas: Social and Institutional Dimension”. In this study, interactive connections between water and forest management on the one hand and the complexities among socio-economic-environmental and institutional systems on the other are highlighted with some examples from European urban environments.

Acknowledgement All authors who have contributed to this volume greatly and gratefully acknowledge the support through COST Action FP 0601 “Forest Management and the Water Cycle (FORMAN)”, and through the numerous national and international funding programs and institutions by which their work was supported.

Section I

Forest and Water – Tree to Small Catchment Processes in a Changing Environment

Chapter 1
Introduction

Patrick Schleppi

Environmental changes induced by mankind are numerous in nature and in effects. Changes affecting the hydrosphere and the atmosphere are spread geographically by water and air circulation, with the potential of creating not only local but also regional and global problems. Ecosystems far from pollution sources are thus also affected, which is the case of many forests. The biggest concern at the beginning of this twenty-first century is certainly the global climatic change driven by human activities. This change includes not only physical components like temperature and precipitation, but also chemical components like air quality and deposition of pollutants.

Among terrestrial ecosystems, forests may be particularly affected by physical and chemical climate changes because of two essential characteristics of trees: their size and their longevity. The size of trees creates a bigger, more complex interface between the soil and the air, with larger microclimatic gradients between both. The resulting aerodynamic roughness enhances the capture of pollutants and thus their deposition into the ecosystem. The long generation time of trees, on the other hand, makes them slow to adapt to changes in a decadal time scale.

How forests will be affected by climate change is largely unknown (Bonan 2008). There are several reasons for this deficit. Firstly, the general magnitude as well as the time and space variability of the changes can be predicted only with a broad margin of error. Secondly, many mechanisms are known how physical and chemical environmental factors affect trees in the short-term, but their interactions in the long-term are difficult or impossible to test experimentally. In the present book section, we intend to review which processes are the most likely to be affected. We examine first below-ground, above-ground and whole-tree processes, then we look at the hydrologic scale of elementary forested catchments. After the four review chapters, we present several case-studies which enlighten some problems encountered in specific geographical regions within Europe, ranging from Alpine

---

P. Schleppi (✉)
Swiss Federal Institute for Forest, Snow and Landscape Research, Zürcherstr. 111,
CH-8903 Birmensdorf, Switzerland
e-mail: schleppi@wsl.ch

M. Bredemeier et al. (eds.), *Forest Management and the Water Cycle: An Ecosystem-Based Approach*, Ecological Studies 212, DOI 10.1007/978-90-481-9834-4_1,

to arid Mediterranean climates. Looking more closely at some sensitive systems is important because of the lessons that can be learnt from them and which may be important elsewhere in the future.

Given the complexity of the problems addressed, this first book section can certainly not achieve a complete coverage, and give all the recommendations that could be wished. Our goal is to give the reader the most important tools to understand present challenges and to anticipate, as well as possible, future changes likely to affect the water relations of forests. How we can adapt to theses changes will then be the topic of the next book section, where sylvicultural measures will be discussed. The scope of the present section is also limited in scale, leaving larger water catchments in complex landscapes to be treated by the third section of this book.

Reference

Bonan GB (2008) Forests and climate change: forcings, feedbacks, and the climate benefits of forests. Science 320:1444–1449

Chapter 2
Hydrological Effects on Below Ground Processes in Temperate and Mediterranean Forests

Boris Rewald, Panagiotis Michopoulos, Lise Dalsgaard,
David L. Jones, and Douglas L. Godbold

Introduction

Water is a critical factor limiting forest growth, either by absence (Stephenson 1990; Pigott and Pigott 1993) or excess (Kreuzwieser et al. 2004). In most parts of Central Europe, moderate water stress is typically to be expected during short rainless periods that occur at irregular intervals in most summers (Backes and Leuschner 2000; Czajkowski et al. 2005) while severe droughts are episodic events (Lloyd-Hughes and Saunders 2002). In the Mediterranean Basin water strongly limits plant growth and survival (Lloret et al. 2004). This is especially true for some Mediterranean tree genera, which are thought to have evolved before the onset of the present summer-dry climate (Petit et al. 2005). Floods, temporary cover of land by water, occur in most parts of Europe on a regular basis and originate among other things from rivers, mountain torrents and Mediterranean ephemeral water courses (EXCIMAP 2007).

B. Rewald (✉)
Plant Ecology, University of Göttingen, Untere Karspüle 2, 37073 Göttingen, Germany
and
Blaustein Institutes for Desert Research, Ben-Gurion University of the Negev, 84990
Midreshet Ben Gurion, Israel
e-mail: brewald@rootecology.de

P. Michopoulos
Forest Research Institute of Athens, Terma Alkmanos, Athens, 115 28, Greece
e-mail: mipa@fria.gr

L. Dalsgaard
Norwegian Forest and Landscape Institute, Høgskoleveien 8, box 1151431 Ås, Norway
e-mail: Lise.Dalsgaard@skogoglandskap.no

D.L. Jones and D.L. Godbold
Environment Centre Wales, Bangor University, Gwynedd, LL57 2UW, UK
e-mail: d.jones@bangor.ac.uk; d.l.godbold@bangor.ac.uk

M. Bredemeier et al. (eds.), *Forest Management and the Water Cycle: An Ecosystem-Based Approach*, Ecological Studies 212, DOI 10.1007/978-90-481-9834-4_2,

There is now convincing evidence that the global climate is changing at an unprecedented rate, putting forest health in jeopardy (Chapter 3, this volume). While total annual precipitation is predicted to be stable in most Northern and Central European regions, model calculations of the future climate predict an increasing frequency and severity of exceptional summer droughts and heatwaves, and floods during winter and spring, thus increasing the risks of both water-logging (Bárdossy and Caspary 1990; Nisbet 2002; Hundecha and Bárdossy 2005) and drought (Hulme et al. 2002; Kunstmann et al. 2004; Rowell 2005; Frei et al. 2006). In the Mediterranean Basin, a disproportionately strong increase of summer temperature and aridity is predicted under future climates (Iglesias et al. 2000; IPCC 2007). While existing meteorological models still embody considerable uncertainty in predicting future precipitation pattern in detail, especially for mountain regions, changes in the seasonal pattern of precipitation, severer summer droughts and winter flooding will lead to changes in a range of below ground processes. These changes include effects on gas exchange, nutrient mineralization, interactions between trees and mycorrhizal fungi, as well as effects on root growth and physiology. Such changes may potentially alter the distribution and composition of European forest by affecting the water- and nutrient-uptake directly, and indirectly by changing the competitive abilities of tree species to obtain these resources (Kozlowski 1997; Geßler et al. 2007; Rewald and Leuschner 2009b).

In this chapter we discuss the effects of soil water on below ground processes, first briefly describing the spatial heterogeneity of tree roots and soil water, and then discuss the effects of soil water deficit and soil water excess.

Biogenic Causes of Soil Moisture Heterogeneity

In predictions of the effects of changing water availability on below ground processes, soil water availability within a stand is often assumed to be relatively homogeneous. However, it has become increasingly apparent that soil moisture varies greatly within a stand (Göttlein and Manderscheid 1998). These authors could show that variability in the hydraulic properties of soil and tree root distribution causes substantial heterogeneity of soil water tension. Furthermore, the degree of heterogeneity increased at high water tension compared to low water tension. Much of the heterogeneity was due to preferential water flow. Preferential water flow has been shown to be due to factors such as macropore flow (Beven and Germann 1982), fingering (Hillel and Baker 1988) and funnelled flow (Kung 1990). In addition stand structural factors such as tree rooting density (Pärtel and Helm 2007; Lange et al. 2009) and tree canopy effects on throughfall and stemflow will influence the spatial and temporal variability of soil moisture (Staelens et al. 2006; Dalsgaard 2007). This may be due to individual crown interception efficiency in relation to crown structure (Staelens et al. 2006) and to the formation of canopy gaps (Dalsgaard 2007). An example of the influence of canopy gaps on soil moisture is shown in Fig. 2.1. While soil water content was relatively homogenous in January to April, it decreased

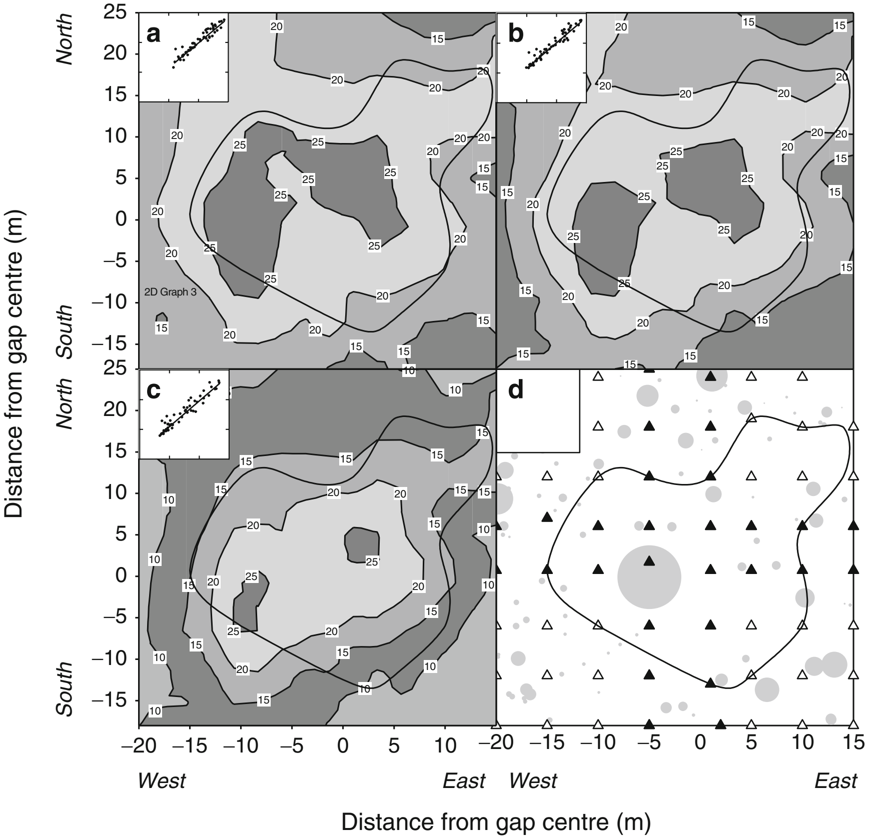


Fig. 2.1 Smoothed values of volumetric soil water content (Θ) at 0–0.5 m depth in and around a canopy gap during the summer of 2000 in a *Fagus sylvatica* forest at Suserup Skov, Denmark. The maps show values for July (**a**), August (**b**) and September (**c**). The Θ values are placed at the boundaries between the different zones of Θ (see Dalsgaard 2007 for details). Field capacity (FC) varied across positions, but there was no significant difference in FC between zones. Panel (**d**) shows the location of trees in and around the gap (*gray circles*). The size of the circles is scaled from tree diameter. The gap centre is at (0.0) m, a *black line* indicates the edge of the gap

steadily during spring and summer in two growing seasons to 64% and 68% of field capacity in the closed forest (June–September; n = 10, 0–0.9 m depth) but remained high, 90% and 93% of field capacity in gap positions (n = 10). Differences were significant (Tukey–Kramer adjusted t-test; $P < 0.05$; Dalsgaard 2007). Throughfall was significantly higher in gaps than in closed forest positions in spring (20%, April–May), summer (30%, June–September) and annually (17%).

Root architecture of structural roots and distribution of fine roots in the soil are of great importance as they determine plant access to water. Fine root density generally shows an exponential decrease with soil depth (Gale and Grigal 1987; Leuschner et al. 2004b), however many species of trees have a structural coarse root system which penetrates into deeper soil layers, thus enabling access to subsoil resources of water (Köstler et al. 1968; Kozlowski et al. 1991; Jackson et al. 1999; Chapters 7 and 10, this volume). For example, *Quercus* spp. have been found to successfully reduce water stress by deep rooting (Čermák et al. 1980; Badot et al. 1994; Bréda et al. 1995), whereas *Fagus sylvatica* roots are less frequent in greater depths (Leuschner et al. 2001) and *Fagus* might thus be more susceptible to drought.

The capability of root systems to explore the soil for heterogeneously distributed moisture is crucial for successful water uptake (Cole and Mahall 2006). Although it is known that water uptake of individual roots can differ within soil horizons (Coners and Leuschner 2005), possibly resulting from moisture differences, studies about species-specific differences in the detection of water, e.g. via hydrotropism, are lacking. Such foraging traits might be very important in densely rooted mixed forests (Rewald and Leuschner 2009a), where competitors could potentially pre-empt moist soil patches. In Loblolly pine (*Pinus taeda*), Parker and Van Lear (1996) investigated the effect of soil moisture and heterogeneity on root distribution. Fine root density was higher on xeric and sub-xeric soils than soil with intermediate water contents. However, on all soil types fine root density was ca. 17 times higher in old root channels and ca. four times higher on rock surfaces compared to the bulk soil matrix. Root channels provide not only easier root penetration paths through the soil but were also preferential flow paths for water, as were rock surfaces. Rooting in soil patches with different moisture contents as shown in a study by Parker and Van Lear (1996), can allow a transfer of water from wet soil to dry soil via ‘hydraulic lift’ (Caldwell et al. 1998) or ‘hydraulic redistribution’ (Burgess et al. 1998; Schulze et al. 1998). Hydraulic redistribution can re-wet dry topsoil layers, possibly facilitate nutrient uptake and buffer plants against water deficits (Richards and Caldwell 1987; Ryel 2004; Pereira et al. 2006; Chapter 4, this volume).

Effects of Soil Water Deficit

Soil

Water deficit causes major changes in the biological, chemical and physical nature of the soil. For example, extreme drought conditions typically induce a dramatic reduction in the amount, structure and activity of the soil microbial community. In general, however, soil microorganisms can survive and persist at much lower soil water potentials than plant roots, particularly as they have the ability to enter a dormant state and survive in very thin water films. Unlike roots, the soil microbial community can also rapidly recover from this inactive state within minutes of rewetting (Jones and Murphy 2007). The impact of drought on soil processes and ecosystem functioning and resilience remains somewhat controversial (Borken and Matzner 2009). Drought tends to reduce mineralization of soil organic matter while subsequent rewetting can significantly enhance the turnover of carbon (C) and nitrogen (N) in soils (Sardans et al. 2008; Chapter 3, this volume). However, the net balance of greenhouse gas exchange remains unclear as it depends greatly on forest type, soil type and the duration and severity of the drought (Almagro et al. 2009). It is well documented that drought induces lysis of root and microbial cells and that rewetting-induces a flush of nutrients and a stimulation of soil respiration (Inglima et al. 2009). Originally, it was thought that this enhanced CO_2 evolution and associated loss of soil organic matter would cause a decline in soil quality. Recent evidence, however, suggests that the loss of CO_2 from soil is much less than if the soil had been maintained at an optimal water content for microbial activity (i.e. −0.05 to −0.5 MPa; Borken et al. 2006). The benefit of drought in reducing greenhouse gas emissions has also been seen for N_2O where water deficit turned a Norway spruce forest from a N_2O source to a sink (Goldberg and Gebauer 2009). A similar response has also been observed for CH_4 (Muhr et al. 2008). In a comprehensive review, Borken and Matzner (2009) concluded that organic matter stocks are progressively preserved with increasing duration and intensity of drought periods; however, increased fire risk may ultimately enhance the risk of organic matter losses under dry conditions. Another major issue associated with drought periods is that water deficit induces organic surfaces in soil to become hydrophobic and water repellent (Cerdà et al. 1998). Consequently, upon rewetting, water tends to travel down macropores resulting in hydrological bypass of the soil's upper horizons, resulting e.g., in losses of nitrate contained in rain and snow-melt water (Schleppi et al. 2004). Alternatively, the hydrophobic organic horizons can prevent infiltration inducing surface runoff and erosion (Doerr et al. 2009). This hydrophobicity also tends to reduce soil organic matter cycling and can reduce potential leaching losses due to bypass flow (Hentschel et al. 2007).

Mycorrhizal Fungi

Mycorrhizas are among the most widespread associations between microorganisms and higher plants, and provide a range of benefits to forest trees of which the best demonstrated is an increased nutrient acquisition, but they also may be beneficial for water uptake and protection against pathogens. The extensive extramatrical or extraradical mycelium formed by ectomycorrhiza and arbuscular mycorrhiza, respectively, is thought to play a primary role in promoting nutrient acquisition (especially P; Smith and Read 2008). Both arbuscular- and ectomycorrhizas form associations with forest tree species, and both are more pronounced in infertile soils. Ectomycorrhizas are more common on moist soils with a high organic matter content, whereas trees infected with arbuscular mycorrhizas tend to be growing on more mineral soils (Smith and Read 2008). The degree of colonisation by mycorrhizas is often influenced by the availability of N and P in the soil. Paul and Clark (1996) argue that the total nutrient pool in soils is of no importance and it is the low concentration of a nutrient such as P in the aqueous phase that promotes the infection.

Mycorrhizas have the potential to affect tree water relations both through direct uptake of water and via nutritional factors. Considerable uncertainty exists about the influence of mycorrhizas on host hydraulics and water uptake (Hampp and Schaeffer 1999; Nardini et al. 2000), but also on the response of the fungal community to drought (Shi et al. 2002). Due to the large contact area between hyphae and soil particles, mycorrhiza formation has been suggested to improve water availability to the host plants (Duddridge et al. 1980; MacFall et al. 1991; Augé 2001), and to support acclimation to drought stress (Davies et al. 1996; George and Marschner 1996). However the benefits to the host tree may be species-specific. Steudle and Heydt (1997) found only a marginal influence of mycorrhization on water uptake of *Fagus sylvatica* saplings, however *Quercus* spp. showed a strong decline in transpired water after the extramatrical hyphae net of the mycorrhiza had been severed (Egerton-Warburton et al. 2003). Furthermore, the extramatrical hyphae were found to transfer water between roots of ‘donor’ *Quercus* individuals, performing hydraulic lift, and ‘receiver’ plants, possibly resulting in multiple benefits during drought. These benefits could include enhanced water and nutrient uptake and/or redistribution (Leake et al. 2004), a rapid recovery from desiccation, or refilling of embolized vessels. However, it remains an open question if similar mechanisms exist in other tree species, and if the quantities of redistributed water account for significant reduction of water stress.

Drought reduces both nutrient uptake by roots and transport from the roots to shoots, due to decreased transpiration rates and impaired active transport and membrane permeability (Alam 1999). The decline in soil moisture also results in a decrease in the diffusion rate of nutrients in the soil to the absorbing root surface (Raynaud and Leadley 2004). P is a nutrient the uptake of which is mostly affected by low soil moisture (Marschner 1997). Under conditions of reduced P mobility, the extensive hyphal network and increased P mobilisation through exudates from the hyphae and plant roots help to maintain P acquisition (Liebersbach et al. 2004). Another beneficial effect of the mycorrhizal association in dry conditions is the increased uptake of potassium (K) by trees especially in the seedling stage. K increases the drought resistance of plants through its function in stomatal regulation and osmoregulation (Marschner 1997). Maintenance of water balance at low soil water potentials is critical for maintenance of a positive carbon balance. In Mediterranean countries, where summers are dry, mycorrhizas help seedling survival among plants considered resistant to drought. Domínguez Núñez et al. (2006) used seedlings of *Quercus ilex* and *Quercus faginea* inoculated with the mycorrhizal fungus *Tuber melanosporum* to establish a stand in south-eastern Spain. These authors could show that mycorrhizal inoculation improved seedling growth and increased water and P uptake during summer drought.

Despite the beneficial effect of the mycorrhizal association on forest plants during drought periods, the development of the mycorrhizal association is affected by low soil moisture content. Valdéz et al. (2006) observed that drought years reduced biomass of both fine and ectomycorrhizal roots of *Pinus oaxacana* by almost 60%. In the field, Bell and Adams (2004) found that the number of root tips of *Pinus pinaster* and *Pinus radiata* associated with the fungus *Rhizopogon* was reduced in areas of low rainfall amounts (<380 mm). It seems that extreme moisture deficit affects mycorrhizas to such extent that below a certain moisture percentage mycorrhizal plants are equally affected as nonmycorrhizal ones. Similarly, in *Pinus pinaster*, Bakker et al. (2006) demonstrated an increase in long-distance exploration types of ectomycorrhizas (*Rhizopogon* and *Scleroderma* spp.) on drier sites and an increase in contact exploration types on wetter sites. In both the work of Bell and Adams (2004), and Bakker et al. (2006) soil organic matter was a co-variant, with the wetter sites having higher organic matter contents. In a growth chamber experiment, Kennedy and Peay (2007) found that *Pinus muricata* plants infected by the ectomycorrhizal fungus *Rhizopogon* had similar biomass, photosynthesis, conductance and total leaf biomass as non-mycorrhizal plants. However, all these parameters increased significantly in the mycorrhizal plants when soil moisture increased and reached a percentage of 13. This work emphasises that mycorrhizas may not be beneficial to plants at very low soil water contents.

Although mechanisms of improved drought tolerance by mycorrhization have been postulated, the effect of water stress on mycorrhizal community diversity is only poorly understood. Kernaghan (2005) reviewed the environmental factors influencing the mycorrhizal fungal communities. He argued that the most important factor is the structure of the associated plant communities through the variety of root exudates. It is probable that drought indirectly affects mycorrhizal diversity by selection of the most drought resistant plants.

Fine Roots

The root:shoot ratio is an important means of adjustment to altered soil moisture levels (Osunubi and Davies 1981; Chapter 4, this volume). Alteration of this ratio can be achieved by long term developmental changes in allocation, or through short term responses to changes in a specific environmental variable. It has been well documented that tree species adapted to more xeric conditions have higher root:shoot ratios (Joslin et al. 2000), thus increasing the ratio between water absorbing and transpiring surface. However, results for both seedlings and mature trees grown under different soil moisture regimes are contradictory (e.g. Thomas 2000; Pronk et al. 2002; Meier and Leuschner 2008a, b). Changes in root:shoot ratio are often achieved by alteration of fine root biomass. The direction of change, i.e. increase or decrease, and magnitude of root biomass response to drought largely depends on tree species or variety, but also on study duration and/or study design (e.g., light regime; Climent et al. 2006; Manes et al. 2006). However, two general trends can be noticed: (i) an increase of root biomass in response to drought have mostly been found in conifer species (e.g., Gower et al. 1992; Parker and Van Lear 1996), which are known to differ in their root growth strategies from deciduous broad-leaved trees (Bauhus and Messier 1999), and (ii) a decrease of root biomass in European deciduous tree species (e.g., Fort et al. 1998; Chiatante et al. 2006).

Table 2.1 Case study within two unmanaged forest patches stocked with 200 years-old *Quercus petraea* trees (see Rewald 2008 for details). Root density (RD$_{org.}$) and proline concentration of fine roots in the organic layer after 3-month (June–August 2004) of two different soil moisture treatments (Control and Reduced soil moisture). Specific conductivity (k$_s$) and degree of embolism of fine roots (diameter d = 0.7–2 mm) and coarse roots (d = 2–6 mm) after three subsequent summers differing in water availability (June–August, 2004–2006). Significant differences are indicated by different letters (RD$_{org.}$ and proline: Scheffé test, $P < 0.05$; k$_s$ and embolism: Kruskal–Wallis H test, $P < 0.05$; mean ± SE; n = sample size; n.d. = no data)

	n	RD$_{org.}$ [g d.wt l^{-1}]	n	Proline [mmol g d.wt^{-1}]	n	k$_s$ [10^{-3} m^2 MPa^{-1} s^{-1}]	Degree of embolism [%]
Control							
Fine roots	20	6.5 ± 0.1 a	10	10.4 ± 3.3 a	9	1.2 ± 0.3 a	20.8 ± 6.7 ab
Coarse roots	–	n.d.	–	n.d.	15	3.0 ± 1.7 ac	10.5 ± 5.3 b
Reduced soil moisture[a]							
Fine roots	20	2.3 ± 0.1 b	10	93.0 ± 29.4 b	11	2.5 ± 0.8 b	36.7 ± 9.3 ac
Coarse roots	–	n.d.	–	n.d.	10	10.0 ± 3.1 c	46.3 ± 10.9 c

[a] A sub-canopy roof (11 m × 11 m × 2 m) was used to reduce soil moisture during summer months (June–August), resulting in a soil moisture reduction of 5–15 vol% compared to ambient-watered control.

Change in root biomass is highly species-specific and connected to altered root turnover rates. Although data on changes in root turnover rates under soil drought are scarce, both unaltered (Joslin et al. 2000) and increased (Pietikäinen et al. 1999; Chiatante et al. 2006) turnover rates have been found in Mediterranean and temperate forests. For example, the fine root biomass in the organic layer (0.5–6 cm thick) of an unmanaged mature *Quercus petraea* stand (Unterlüß, Lower Saxony, Germany; 52°83′N, 10°26′E; stem density: 44 ha^{-1}, tree height: 28 m, stem basal area: 12.3 m^2 ha^{-1}) was significantly reduced after three months of experimentally-induced summer drought (Table 2.1). In contrast, more drought sensitive tree species like *Fagus sylvatica* and *Quercus robur* were found to retain their root biomass in the upper soil horizons even under severe drought, resulting in high turnover rates (Konôpka et al. 2005; Mainiero and Kazda 2006). It is possible that a less flexible carbon-investment strategy during drought and the subsequently increased fine root turnover are partly involved in a higher drought sensitivity of tree species.

Change in fine root morphology is another potential adaptation to altered soil moisture. However, the detection of adaptation mechanisms is hampered by the large variation of root morphology and architecture within species or individuals, possibly caused by soil heterogeneity (Fitter 1994; Meier and Leuschner 2008a). The finest (first order) roots are most important parts of the root system for water uptake (Rieger and Litvin 1999; Lindenmair et al. 2004; Rewald et al. 2010). Thus, specific root area (SRA) and length (SRL) could be expected to increase during drought. In contrast to this assumption, both the SRA and the SRL of *Betula pendula*, *Fagus sylvatica* and other tree species decreased under reduced water availability (Aspelmeier and Leuschner 2006; Ostonen et al. 2007; Meier and Leuschner 2008b).

The underlying mechanisms must remain speculative, but may include increased mortality/shedding of root tips and finest roots, stimulated ethylene production resulting in larger root diameter, (re-) growth of less ramified roots by increased turnover rates and higher soil mechanical impedance, and increased root tissue densities (Clark et al. 2003; Manes et al. 2006; Trubat et al. 2006).

Root elongation rate, which is positively correlated with root diameter, could be a more important attribute for maximizing uptake rates of low-diffusive nutrients such as P than root surface area (Silberbush and Barber 1983; Raven and Edwards 2001). In order to exploit more soil regions with plant-available water, higher elongation rates are suggested to be most favourable in soil with a heterogeneous distribution of moisture or in the case of inter-specific competition for water. Furthermore, roots of larger diameter, and consequently a lower SRA:SRL ratio have lower construction and maintenance costs per unit biomass than thinner roots (Eissenstat and Yanai 1997). Most likely, there must be a trade-off between the benefits of a large absorbing surface area per unit biomass and an increased contact with the soil, and the benefits of increased ‘long-distance’ foraging and reduced maintenance costs under water shortage. However, rates of water uptake per root surface area have been found to vary significantly between individual root branches and species (Korn 2004; Burk 2006). For example, *Fagus sylvatica* has been found to possess higher root-surface-area related sap-flow rates than *Quercus petraea* (Coners and Leuschner 2002). Furthermore, even under well-watered conditions, *Fagus sylvatica* roots showed higher root surface-specific flows on a site with a more continental climate than on a more oceanic site (Burk 2006), indicating a yet unknown, but highly plastic uptake pattern on drier sites. Unfortunately, no information is available about such plasticity in uptake kinetics for other tree species.

Physiological Adaptation of Roots

The capacity of roots for water uptake is determined not only by root surface area and foraging, but by the resistance of tissues to water transport as well. Drought is known to induce short- and long-term alterations of the radial pathway, usually resulting in an increase of radial resistance (Huang and Nobel 1993; Steudle 2000). Water channel proteins in the cell membranes (aquaporins) mediate the short-term adjustment of the symplastic pathway to drought stress (Yamada et al. 1997). Although data for woody plants is scarce, studies on herbaceous plants suggest that aquaporins are present in virtually all root types (Kirch et al. 2000; Kaldenhoff and Fischer 2006), and especially in cells that control water uptake and radial water flow (Schäffner 1998). The regulation of root aquaporins enables a very tight coupling between root water uptake and whole plant physiology, e.g. by facilitating water flow under moist conditions, or reducing water loss to the soil via unintended hydraulic redistribution by ‘more tight’ membranes. A higher expression of aquaporins, and, thus, higher root surface area-specific conductance, is suggested to compensate for a reduced root system size in water-stressed olive trees, explaining in part the above-average drought-tolerance of this species (Lovisolo et al. 2007; Secchi et al. 2007). However, knowledge of drought-induced adaptations in root physiology, resulting in altered water uptake rates, is still rare for most European tree species.

Osmotic adjustment is another mechanism, allowing plants to tolerate periods of drought (Chaves et al. 2003; Aranda et al. 2004; Chapter 3, this volume). Osmotic adjustment enables sustained root growth under moderate levels of drought stress by partial turgor recovery and maintenance of the ability to loosen cell walls (Hsiao and Xu 2000). Proline is an important component of this osmoregulation; its concentrations have been found to increase strongly in response to drought stress in roots of mature *Fagus sylvatica* and *Quercus petraea* trees (Table 2.1; Rewald 2008) and has been suggested to explain differences in drought-tolerance (Hare et al. 1998; Ennajeh et al. 2006; Garcia-Sánchez et al. 2007). Drought is thought to facilitate suberization of the root rhizodermis, subsequently limiting the apoplastic by-pass and reducing the radial conductivity of roots (Zimmermann and Steudle 1998). However, previous studies provide convincing evidence of water uptake even by strongly suberized regions of woody roots (Chung and Kramer 1975; MacFall et al. 1990, 1991). Increased root suberization under drought stress might therefore correlate better with general stress-tolerance (Schreiber et al. 2005), instead of explaining different water uptake rates (Leuschner et al. 2003).

Changes in Root Axial Conductivity and Hydraulic Safety Under Drought

Different species or even genotypes of woody plants may differ substantially with respect to root axial conductivity (e.g., Huber 1956; Larcher 2001). A majority of previous studies has reported reduced root conductivities in response to stress (e.g., Machado and Tyree 1994; Nardini and Pitt 1999; Trubat et al. 2006), thereby improving plant water status by reducing water loss to the atmosphere and the soil (Meinzer et al. 1996; Trillo and Fernández 2005). In contrast, the rarely observed increase of root axial conductivity (Table 2.1) is suggested to facilitate water uptake by reducing the flow resistance (Nardini and Pitt 1999). Although axial conductivity has previously been considered a minor limiting factor of whole-root conductivity as compared to radial conductivity (e.g., Steudle 1994), Hacke et al. (2000) demonstrated that whole-plant water use and axial conductivity of woody plants during water stress were in accordance. Especially the large root systems of mature trees with greater path lengths (West et al. 1999; Addington et al. 2006) and reduced potential gradients due to large root:leaf area ratios (Grier et al. 1981; Vanninen et al. 1996) illustrate the importance of sufficient root axial conductivities. Increased root conductivities could be a particularly effective adaptation in plants that respond to drought with a reduction of fine root biomass.

Another plastic adaptation of the tree root hydraulic system to soil water shortage might be the development of root branches with diverging hydraulic properties, thereby exploring the spatial heterogeneity of water reserves, as is typical for temporally drought-exposed soils (Göttlein and Manderscheid 1998). Several previous studies have documented that root axial conductivity and water uptake rate can differ substantially within root systems, e.g. between deep and shallow roots (Pate et al. 1995; Korn 2004; Leuschner et al. 2004a; Chapter 4, this volume). Since individual roots or root branches are thought to act as ‘physiologically autonomous units’ (Shani et al. 1993), the differentiation of the root population into high-conductivity and low-conductivity roots has been postulated as a favourable adaptation to heterogeneous environments.

While some embolism may occur even in roots of well-watered plants (Alder et al. 1996; Domec et al. 2004; Table 2.1), fine roots have been suggested to be weak, replaceable segments of the soil-plant-atmosphere continuum (SPAC; Sperry and Saliendra 1994; Domec et al. 2004; Chapter 3, this volume), expedient to uncouple parts of the SPAC in response to more severe water shortage. Acting as ‘hydraulic fuses’ (Zimmermann 1983), they are thought to prevent plant-wide cavitation and water loss to the soil, as resulting from unintended hydraulic redistribution (Sperry and Ikeda 1997; Hacke et al. 2000). ‘Hydraulic fuse’-mechanisms that are assumed to prevent or reduce such leakage are: (i) shedding of fine root branches (Head 1973; Pereira et al. 2004), (ii) suberization of the rhizodermis and/or aquaporin regulation (Vera-Estrella et al. 2004; Schreiber et al. 2005). Because there is now convincing evidence that embolism is reversible in many cases (Pickard 1989; Zwieniecki and Holbrook 1998; Lovisolo and Schubert 2006), cavitation is suggested to be another, possibly reversible, ‘hydraulic fuse’ mechanism in roots (Rewald 2008). Thus, the recently found increased vulnerability to cavitation in drought-stressed *Quercus petraea* roots (Rewald 2008) seems to be a straightforward adaption to drought rather than impairment by drought.

Effects of Excess Soil Water

Soil

In comparison to water deficit, the effects of excess soil water on below ground soil processes are relatively well understood (Richardson and Vepraskas 2000). Briefly, saturation of soil with water (waterlogging) for prolonged periods of time generates a negative impact on nearly all forests that are not adapted to hydromorphic or riparian soil conditions (White 2007). The incidence of periodic waterlogging is predicted to increase in European forests due to increases in the frequency and intensity of winter rainfall events (Fuhrer et al. 2006). The resultant changes in nutrient cycling induced by waterlogging may have profound consequences on forest development with some species more resilient to flooding (e.g., oak) than others (e.g., beech; Geßler et al. 2007). In Mediterranean woodlands, precipitation can be very abundant during the wet season, inducing temporary waterlogging. When the soil becomes saturated the rate of O_2 diffusion into the soil is greatly reduced leading to rapid depletion of soil O_2 and an inducement of hypoxic and anoxic conditions. These conditions reduce seed germination, root growth, soil microbial activity and may promote the attack of trees by pathogenic soil organisms (Burgess et al. 1999;

Gómez-Aparicio et al. 2008; Pérez-Ramos and Marañón 2009). Prolonged waterlogging tends to lead to a reduction in microbial activity, an increase in soil organic matter and greater emissions of N_2O and CH_4 (Pinay et al. 2000; Eglin et al. 2008). Flooding may also stimulate the input and redistribution of soil particles which can strongly influence C storage and greenhouse gas emissions (Pinay et al. 2000). To some extent this is driven by a reduced microbial biomass and changes in its community structure (i.e. reduction of aerobic bacteria, Gram-negative and Gram-positive bacteria, mycorrhizal fungi and mesofauna) and an inhibition of O_2-requiring enzymes which are central to the breakdown of forest litter (e.g., phenol oxidase; Langer and Rinklebe 2009; Unger et al. 2009). One study indicates that while CO_2 emissions from soil decline in flooded conditions, this is more than offset by the concomitant increase in N_2O and CH_4 making some wetland forests greater net emitters of greenhouse gases (GHG) than non-flooded ones (Yu et al. 2008). However, the overall response of soil flooding on GHG emission is likely to be highly dependent on a range of factors (e.g., geographical location, topography, climatic regime, catchment size, land use and management) making generalisations across landscapes difficult. In some regions of Europe deliberate flooding of organic soils is now occurring in an attempt to try and preserve soil organic matter stocks whose loss has been stimulated by historical drainage and oxygenation. Whilst this management approach appears to reduce CO_2 emissions and losses of dissolved organic carbon, it may stimulate emissions of N_2O and CH_4 and may reduce forest growth and therefore above and below ground C storage (Silvan et al. 2002).

Mycorrhizal Fungi

In *Pterocarpus officinalis,* Fougnies et al. (2007) found that arbuscular mycorrhizas contributed to flood tolerance and P acquisition. In contrast, Ray and Inouye (2006) found that the development of arbuscular fungus associated with *Typha latifolia* was adversely affected by flooding. Stenström (1991) could show that species of ectomycorrhizas vary greatly in tolerance to waterlogging. Based on the formation of ectomycorrhizas with *Pinus sylvestris*, the fungi *Thelephora terrestris, Laccaria laccata,* and *Hebeloma crustuliniforme* were not sensitive to flooding, whereas *Suillus flavidus* and *S. bovinus* were highly sensitive. It is highly probable that fungal species and periodicity of flooding play a crucial role in the final mycorrhizal distribution in forest soils.

Fine Roots

In soils with permanent or seasonal waterlogging, hypoxic conditions prevent deep rooting (Glenz et al. 2006) by inhibiting root exploration of those soils layers (Chapter 4, this volume). This may be due to both physical factors such as water preventing gas diffusion in soil pores and chemical factors such as high levels of reduced iron. In *Picea sitchensis*, the thickness of the root plate was lower in stago-orthic gleys than humic stagno-orthic gleys (Ray and Nicoll 1998), and decreased individual tree stability. Bakker et al. (2006) showed that in *Pinus pinaster* growing on a humid and dry site, root distribution was significantly shallower and root diameter increased more with depth at the humid site. This was suggested to be due to more adverse soil conditions such as the presence of a hardpan, higher amounts of aluminium oxides and/or anoxia at depth. Similarly, Xu et al. (1997) showed that seasonal waterlogging of subsoils restricted the growth of fine roots of *Picea abies* and *Abies grandis*, but this influence was greater in *Picea abies* than *Abies grandis*. During the water logging period, in *Abies grandis*, the dead fine root biomass was greater than the live fine root biomass, but the fine root biomass increased by 50% during the non-waterlogged period. Figure 2.2 shows the effect of a high soil water table on the fine root distribution of *Picea abies* growing at three sites in the Krkonoše mountains, Czech Republic. At the mesic Modrý Důl and Alžbétinka

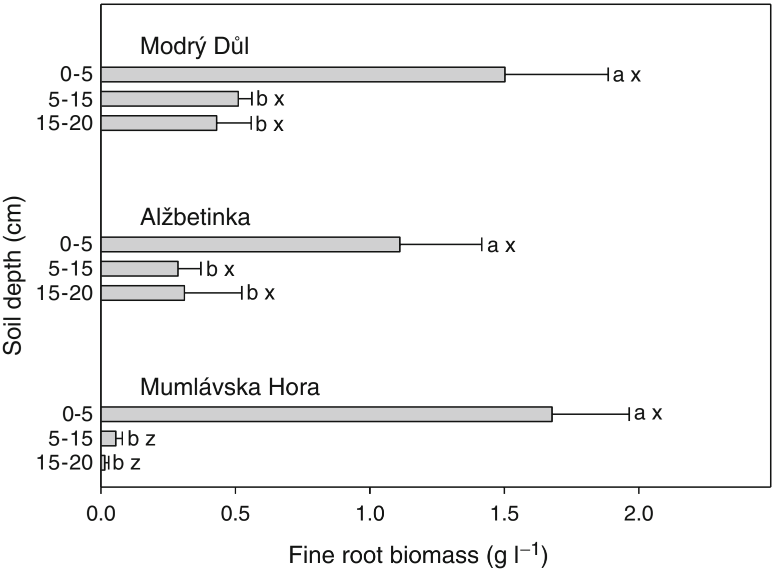


Fig. 2.2 Vertical distribution of fine roots of *Picea abies* at three sites with a different degree of forest decline in the Krkonoše mountains, Czech Republic. The damage to the forest cover in Modrý Důl, Alžbétinka and Mumlávska Hora were classified according to the Czech Forest Authority as second, third and fourth degree respectively. The three sites are relatively homogeneous in soil parent material (granite and gneiss), elevation (1,190–1,220 m a.s.l.), average annual precipitation (1,390–1,500 mm year^{-1}) and temperature (2.5–3.8°C). The original forest cover consist of plantations of Norway spruce approximately 130 years old in Modrý Důl, older (210 and 190 years) in Alžbétinka and Mumlávska Hora. The sites differ in the geomorphic position, Modrý Důl and Alžbétinka are located at mid slope, while Mumlávska Hora is on a summit. Within a site, bars not followed by the same indices (a, b) vary significantly between soil depths. Within a soil depth, bars not followed by the same indices (x, z) vary significantly between sites (Two-way ANOVA on log transformed data, Holms-Sidik corrected, $P < 0.05$; mean + SE)

sites fine roots are found to a depth of 20 cm. In contrast at the Mumlávska Hora site which is subject to regular waterlogging, the fine roots are restricted mostly to the upper 5 cm of soil.

In sensitive tree species (e.g., *Picea sitchensis*), waterlogging results in death and dieback of fine root tips (Nicoll and Coutts 1998). However, the timing of fine root growth and dormancy influences the degree to which the roots are affected. Provenances of *Picea sitchensis* with earlier root dormancy were less sensitive to autumn waterlogging than late-dormant provenances. This has also been shown in floodplain tree species where root growth of *Quercus pagodaeifolia*, which has early season root growth, was more sensitive to spring floods than *Quercus lyrata* which had delayed root growth (Burke and Chambers 2003).

On soils with a permanently high water table even deep rooting species develop shallower rooting systems (Burke and Chambers 2003). The effect on rooting varies between species depending upon tolerance to water-logging (Kozlowski 1984), but even tolerant trees tend to have shallower root systems (Lehnardt and Brechtel 1980) in permanently wet soils. In heterogeneously waterlogged soils, root biomass of more flooding tolerant species (e.g., *Fraxinus pennsylvanica*) decreased, but biomass of the less tolerant *Liquidambar styraciflua* was not affected. These species-specific differences illustrate that a response of root biomass is related to a complex interplay of root proliferation, altered uptake kinetic, and nutrient diffusion rate (Neatrour et al. 2007).

However, the rooting pattern can change rapidly due to death of roots after prolonged inundation (Polomski and Kuhn 1998). Further, these authors have suggested in *Salix* spp., *Populus canadensis* and *Alnus glutinosa* that the dieback of the root system due to flooding led to promotion of renewal of the fine root system. However, under non-waterlogged conditions these tree species also tend to have high rates of root turnover (Lukac et al. 2003; Ostonen 2003), which may be a reflection of high plasticity of the fine root system to deal with large fluctuations in environmental conditions. Again similar relationships have been shown in floodplain forests. Burke and Chambers (2003) could show that in *Quercus laurifolia* habitat with a shallow root zone and episodes of flooding and drought greater changes in root structure and physiology occurred than in *Nyssa sylvatica* habitat that had a deeper rooting zone and a more consistently moist to flooded hydroperiod.

The most frequently reported reactions of tree root systems to waterlogging, and hence hypoxia, are increased formation of lenticels (Coutts 1982; Angeles et al. 1986), followed by the differentiation of adventitious and flood-adapted roots (Topa and McLeod 1986; Colin-Belgrand et al. 1991). These adaptations have been described for a broad range of temperate tree species (compare Kozlowski 1997). Adventitious roots are produced on the original root system and on the submerged portions of stems and have been found to have larger SRA and more intercellular space (aerenchyma) than roots growing in well-aerated soils (*Nyssa sylvatica* var. *biflora*; Hook et al. 1971). They are thought to increasing water absorption (Hook and Scholtens 1978; Tsukahara and Kozlowski 1985; Herrera et al. 2008), oxidizing the rhizosphere and venting soil-bound ethylene and CO_2 (Hook and Brown 1973; Drew 1997; Colmer 2003). Under waterlogged conditions, Havens (1997) found a 30% higher soil redox-potential within the rhizosphere of *Fraxinus pennsylvanica* seedlings than in non-rhizosphere soil areas.

Physiological Adaptation of Roots

As stated above, drought is thought to facilitate suberization of the root rhizodermis. Under waterlogging, an increased suberization of the rhizodermis is thought to reduce the radial loss of oxygen in herbaceous species (Enstone et al. 2003). Unfortunately, no studies on suberization of the roots of European tree species under waterlogging are known to these authors. However, it was shown that water logging results in an increased osmotic permeability and a decreased solute reflection coefficient of *Larix laricina* and *Picea glauca* (Reece and Riha 1991). A decrease of membrane permeability and subsequently a decrease of root conductivity (i.e. sum of radial and axial hydraulic conductivity) was measured on waterlogged *Fagus sylvatica* and *Quercus* spp. seedlings (Schmull and Thomas 2000). Decreases in the hydraulic conductance of the root system due to hypoxic conditions, as one would expect to find during flooding, have previously been measured in different *Populus* species (Harrington 1987; Smit and Stachowiak 1988).

In *Fagus sylvatica* and *Quercus petraea* seedlings, nitrate reductase activity of the roots was found to be increased under hypoxic conditions (Schmull and Thomas 2000). Nitrate reduction can act as a sink for protons, thus helping to avoid damaging cytoplasmic acidosis (Fan et al. 1997). Armstrong et al. (1994) summarized that survival of flooding by woody plants depends on more than one of the following metabolic adaptations: (1) control of energy metabolism, (2) availability of energy resources, (3) provision of essential gene products and synthesis of macromolecules, and (4) protection against post-hypoxic injury. See Drew (1997) and Kreuzwieser et al. (2002, 2004) for extensive reviews about metabolic adaptations of roots and whole trees to flooding. Decreased xylem sap osmotic potentials due to waterlogging (e.g., in *Quercus petraea* seedlings, Folzer et al. 2006) are not considered to be adaptive mechanisms but have been attributed to enriched solutes provided by degenerating roots, a reduction in root water absorption (Jackson et al. 1996) or decreased membrane selectivity (Barrett-Lennard 2003; Kolb et al. 2004).

Studies about the influence of flooding on root hydraulics properties of European tree species are virtually lacking. However, in a study on *Picea sitchensis* and *Pinus contorta* seedlings, xylem growth in woody roots was found to be ceased in the centre of the water-logged area and highly reduced in non-flooded parts of root system (Coutts 1982). Thus, when the flood water drains away, the previously flooded plants may be more drought-sensitive as their smaller and less conductive root systems cannot adequately replenish transpirational losses (Kozlowski 1997; Chapter 8, this volume).

Conclusion and Outlook

The climate is changing, which may affect forest health and composition by increasing flood and drought events. Although, below ground traits are likely to play significant roles as adaptation mechanisms of current European forest communities, the current state of knowledge of the reaction of soils, tree roots and their symbiotic fungi to changing soil water status is poor. Where controlled drainage and flooding are undertaken there is also an urgent need to translate research findings into policy and practice to prevent negative impacts of forest management (see Part II). In addition to the traditional research on root biomass and root:shoot ratio, the variability and plasticity of physiological and anatomical root traits in heterogeneous soil environments as well as interactions with mycorrhizas should be considered in future studies. Furthermore, to predict the development of European forest under a future warmer climate there is an urgent need to evaluate the outcome of below and above ground competition and subsequently seedling establishment in forest communities under changed environmental conditions. Under increasing drought conditions heterogeneity of soil moisture is likely to increase, but will be strongly linked to changes in canopy structure (Chapter 3, this volume) and the wetting characteristics of the forest floor, thus introducing both positive and negative ecological feed-back mechanisms. In addition, increased spatial heterogeneity may change the outcome of competition between tree species, benefiting tree species with a greater plasticity and precision of root development rather than physiological adaptation per se. Much of the current knowledge is derived from single species growing in homogeneous environments. This knowledge base is insufficient to make meaningful predictions of the effects of changes of soil water status on forests and forest processes.

References

Addington RN, Donovan LA, Mitchell RJ, Vose JM, Pecot SD, Jack SB, Hacke UG, Sperry JS, Oren R (2006) Adjustments in hydraulic architecture of *Pinus palustris* maintain similar stomatal conductance in xeric and mesic habitats. Plant Cell Environ 29:535–545

Alam SM (1999) Nutrient uptake by plants under stress conditions. In: Pessaraki M (ed) Handbook of plant and crop stress. Marcel Dekker, New York, pp 285–314

Alder NN, Sperry JS, Pockman WT (1996) Root and stem xylem embolism, stomatal conductance, and leaf turgor in *Acer grandidentatum* populations along a soil moisture gradient. Oecologia 105:293–301

Almagro M, López J, Querejeta JI, Martínez-Mena M (2009) Temperature dependence of soil CO_2 efflux is strongly modulated by seasonal patterns of moisture availability in a Mediterranean ecosystem. Soil Biol Biochem 41:594–605

Angeles G, Evert RF, Kozlowski TT (1986) Development of lenticels and adventitious roots in flooded *Ulmus americana* seedlings. Can J For Res 16:585–590

Aranda I, Gil L, Pardos JA (2004) Osmotic adjustment in two temperate oak species [*Quercus pyrenaica* WILLD. and *Quercus petraea* (MATT.) LIEBL.] of the Iberian Peninsula in response to drought. Invest Agrarian: Sist Recur Forestales 13:339–245

Armstrong W, Braendle R, Jackson MB (1994) Mechanisms of flood tolerance in plants. Acta Bot Neerl 43:307–358

Aspelmeier S, Leuschner C (2006) Genotypic variation in drought response of Silver birch (*Betula pendula* ROTH.): leaf and root morphology and carbon partitioning. Trees 20:42–52

Augé RM (2001) Water relations, drought and vesicular-arbuscular mycorrhizal symbiosis. Mycorrhiza 11:3–42

Backes K, Leuschner C (2000) Leaf water relations of competitive *Fagus sylvatica* and *Quercus petraea* trees during 4 years differing in soil drought. Can J For Res 30:335–346

Badot PM, Lucot E, Bruckert S (1994) Soil-moisture in deep levels is the main source of mid-day water potential variations in oak (*Quercus sp*.). C R Acad Sci III-Vie 317:341–345

Bakker MR, Augusto L, Achat DL (2006) Fine root distribution of trees and understory in mature stands of maritime pine (*Pinus pinaster*) on dry and humid sites. Plant Soil 286:37–51

Bárdossy A, Caspary HJ (1990) Detection of climate change in Europe by analyzing European atmospheric circulation patterns from 1881 to 1989. Theor Appl Climatol 42:155–167

Barrett-Lennard EG (2003) The interaction between waterlogging and salinity in higher plants: causes, consequences and implications. Plant Soil 253:35–54

Bauhus J, Messier C (1999) Soil exploitation strategies of fine roots in different tree species of the southern boreal forest of eastern Canada. Can J For Res 29:260–273

Bell TL, Adams MA (2004) Ecophysiology of ectomycorrhizal fungi associated with *Pinus* spp. in low rainfall areas of Western Australia. Plant Ecol 171:35–52

Beven KJ, Germann P (1982) Macropores and water-flow in soils. Water Resour Res 18:1311–1325

Borken W, Savage K, Davidson EA, Trumbore SE (2006) Effects of experimental drought on soil respiration and radiocarbon efflux from a temperate forest soil. Global Change Biol 12:177–193

Borken W, Matzner E (2009) Reappraisal of drying and wetting effects on C and N mineralization and fluxes in soils. Global Change Biol 15:808–824

Bréda N, Granier A, Barataud F, Moyne C (1995) Soil-water dynamics in an oak stand. 1. Soil-moisture, water potentials and water-uptake by roots. Plant Soil 172:17–27

Burgess SSO, Adams MA, Turner NC, Ong CK (1998) The redistribution of soil water by tree root systems. Oecologia 115:306–311

Burgess T, McComb JA, Colquhoun I, Hardy GES (1999) Increased susceptibility of *Eucalyptus marginata* to stem infection by *Phytophthora cinnamomi* resulting from root hypoxia. Plant Pathol 48:797–806

Burk D (2006) Physiological, anatomical and chemical aspects of the regulation of water uptake by beech, pine and birch roots in two different water-supplying locations (In German). Ph.D. thesis, University of Göttingen, Germany. (http://webdoc.sub.gwdg.de/diss/2006/burk)

Burke MK, Chambers J (2003) Root dynamics in bottomland hardwood forests of the southeastern United States coastal plain. Plant Soil 250:141–153

Caldwell MM, Dawson TE, Richards JH (1998) Hydraulic lift: consequences of water efflux from the roots of plants. Oecologia 113:151–161

Cerdà A, Schnabel S, Ceballos A, Gomez-Amelia D (1998) Soil hydrological response under simulated rainfall in the Dehesa land system (Extremadura, SW Spain) under drought conditions. Earth Surf Proc Land 23:195–209

Čermák J, Huzulák J, Penka M (1980) Water potential and sap flow-rate in adult trees with moist and dry soil as used for the assessment of root-system depth. Biol Plantarum 22:34–41

Chaves MM, Maroco JP, Pereira JS (2003) Understanding plant responses to drought – from genes to the whole plant. Funct Plant Biol 30:239–264

Chiatante D, Di Iorio A, Sciandra S, Scippa GS, Mazzoleni S (2006) Effect of drought and fire on root development in *Quercus pubescens* WILLD. and *Fraxinus ornus* L. seedlings. Environ Exp Bot 56:190–197

Chung HH, Kramer PJ (1975) Absorption of water and ^{32}P through suberized and unsuberized roots of Loblolly pine. Can J For Res 5:229–235

Clark LJ, Whalley WR, Barraclough PB (2003) How do roots penetrate strong soil? Plant Soil 255:93–104

Climent JM, Aranda I, Alonso J, Pardos JA, Gil L (2006) Developmental constraints limit the response of Canary Island pine seedlings to combined shade and drought. For Ecol Manage 231:164–168

Cole ES, Mahall BE (2006) A test for hydrotropic behavior by roots of two coastal dune shrubs. New Phytol 172:358–368

Colmer TD (2003) Long-distance transport of gases in plants: a perspective on internal aeration and radial oxygen loss from roots. Plant Cell Environ 26:17–36

Colin-Belgrand M, Dreyer E, Biron P (1991) Sensitivity of seedlings from different oak species to waterlogging – effects on root-growth and mineral-nutrition. Ann For Sci 48:193–204

Coners H, Leuschner C (2002) *In situ* water absorption by tree fine roots measured in real time using miniature sap-flow gauges. Funct Ecol 16:696–703

Coners H, Leuschner C (2005) *In situ* measurement of fine root water absorption in three temperate tree species - Temporal variability and control by soil and atmospheric factors. Basic Appl Ecol 6:395–405

Coutts MP (1982) The tolerance of tree roots to waterlogging. V. Growth of woody roots of Sitka spruce and Lodgepole pine in waterlogged soil. New Phytol 90:467–476

Czajkowski T, Kuhling M, Bolte A (2005) Impact of the 2003 summer drought on growth of beech sapling natural regeneration (*Fagus sylvatica* L.) in north-eastern Central Europe. Ger J For Res 176:133–143

Dalsgaard L (2007) Above and below ground gaps – the effects of a small canopy opening on throughfall, soil moisture and tree transpiration in Suserup Skov, Denmark. Ecol Bull 52:81–102

Davies Jr. FT, Svenson SE, Cole JC, Phavaphutanon L, Duray SA, Olalde-Portugal V, Meier CE, Bo SH (1996) Non-nutritional stress acclimation of mycorrhizal woody plants exposed to drought. Tree Physiol 16:985–993

Doerr SH, Woods SW, Martin DA, Casimiro M (2009) Natural background' soil water repellency in conifer forests of the north-western USA: its prediction and relationship to wildfire occurrence. J Hydrol 371:12–21

Domec JC, Warren JM, Meinzer FC, Brooks JR, Coulombe R (2004) Native root xylem embolism and stomatal closure in stands of Douglas fir and Ponderosa pine: mitigation by hydraulic redistribution. Oecologia 141:7–16

Domínguez Núñez JA, Salva Serrano J, Rodríguez Barreal JA, Omeñaca González JAS (2006) The influence of mycorrhization with *Tuber melanosporum* in the afforestation of a Mediterranean site with *Quercus ilex* and *Quercus faginea*. For Ecol Manage 231:226–233

Drew MC (1997) Oxygen deficiency and root metabolism: Injury and acclimation under hypoxia and anoxia. Annu Rev Plant Physiol Plant Mol Biol 48:223–250

Duddridge JA, Malibari A, Read DJ (1980) Structure and function of mycorrhizal rhizomorphs with special reference to their role in water transport. Nature 287:834–836

Egerton-Warburton LM, Graham RC, Hubbert KR (2003) Spatial variability in mycorrhizal hyphae and nutrient and water availability in a soil-weathered bedrock profile. Plant Soil 249:331–342

Eglin T, Walter C, Nys C, Follain S, Forgeard F, Legout A, Squividant H (2008) Influence of waterlogging on carbon stock variability at hillslope scale in a beech forest (Fougères forest, West France). Ann For Sci 65:202

Eissenstat DM, Yanai RD (1997) The ecology of root lifespan. Adv Ecol Res 27:1–60

Enstone DE, Peterson A, Ma FS (2003) Root endodermis and exodermis: structure, function, and responses to the environment. J Plant Growth Regul 21:335–351

Ennajeh M, Vadel AM, Khemira H, Ben Mimoun M, Hellali R (2006) Defence mechanisms against water deficit in two olive (*Olea europaea* L.) cultivars 'Meski' and 'Chemlali'. J Horti Sci Biotech 81:99–104

EXCIMAP (2007) Annex 2: Atlas of flood maps - Examples from 19 European countries, USA and Japan. In: Martini F, Loat R (eds) Handbook on good practice for flood mapping in Europe. Netherlands Ministry of Transport, Public Works and Water Management, The Hague, The Netherlands, 197 pp

Fan TWM, Higashi RM, Frenkiel TA, Lane AN (1997) Anaerobic nitrate and ammonium metabolism in flood-tolerant rice coleoptiles. J Exp Bot 48:1655–1666

Fitter AH (1994) Architecture and biomass allocation as components of the plastic response of root systems to soil heterogeneity. In: Caldwell MM, Pearcy RW (eds) Exploitation of environmental heterogeneity by plants: ecophysiological processes above- and belowground. Academic, San Diego, CA, pp 305–323

Frei C, Scholl R, Fukutome S, Schmidli R, Vidale PL (2006) Future change of precipitation extremes in Europe: Intercomparison of scenarios from regional climate models. J Geophys Res-Atmo 111

Folzer H, Dat JF, Capelli N, Rieffel D, Badot PM (2006) Response of Sessile oak seedlings (*Quercus petraea*) to flooding: an integrated study. Tree Physiol 26:759–766

Fort C, Muller F, Label P, Granier A, Dreyer E (1998) Stomatal conductance, growth and root signaling in *Betula pendula* seedlings subjected to partial soil drying. Tree Physiol 18:769–776

Fougnies L, Renciot S, Muller F, Plenchette C, Prin Y, de Faria SM, Bouvet JM, Sylla SN, Dreyfus B, Bâ AM (2007) Arbuscular mycorrhizal colonization and nodulation improve flooding tolerance in *Pterocarpus officinalis* JACQ. seedlings. Mycorrhiza 17:159–166

Fuhrer J, Beniston M, Fischlin A, Frei C, Goyette S, Jasper K, Pfister C (2006) Climate risks and their impact on agriculture and forests in Switzerland. Clim Change 79:79–102

Gale MR, Grigal DF (1987) Vertical root distributions of northern tree species in relation to successional status. Can J For Res 17:829–834

Garcia-Sánchez F, Syvertsen JP, Gimeno V, Botia P, Perez-Perez JG (2007) Responses to flooding and drought stress by two citrus rootstock seedlings with different water-use efficiency. Physiol Plant 130:532–542

George E, Marschner H (1996) Nutrient and water uptake by roots of forest trees. Z Pflanzenernährung Bodenkunde 159:11–21

Geßler A, Keitel C, Kreuzwieser J, Matyssek R, Seiler W, Rennenberg H (2007) Potential risks for European beech (*Fagus sylvatica* L.) in a changing climate. Trees 21:1–11

Glenz C, Schlaepfer R, Iorgulescu I, Kienast F (2006) Flooding tolerance of Central European tree and shrub species. For Ecol Manage 235:1–13

Göttlein A, Manderscheid B (1998) Spatial heterogeneity and temporal dynamics of soil water tension in a mature Norway spruce stand. Hydrol Proc 12:417–428

Goldberg SD, Gebauer G (2009) Drought turns a Central European Norway spruce forest soil from an N_2O source to a transient N_2O sink. Global Change Biol 15:850–860

Gómez-Aparicio L, Pérez-Ramos IM, Mendoza I, Matías L, Quero JL, Castro J, Zamora R, Marañón T (2008) Oak seedling survival and growth along resource gradients in Mediterranean forests: implications for regeneration in current and future environmental scenarios. Oikos 117:1683–1699

Gower ST, Vogt KA, Grier CC (1992) Carbon dynamics of Rocky-Mountain Douglas fir – influence of water and nutrient availability. Ecol Monogr 62:43–65

Grier CC, Vogt KA, Keyes MR, Edmonds RL (1981) Biomass distribution and above-ground and below-ground production in young and mature *Abies amabilis* zone ecosystems of the Washington Cascades. Can J For Res 11:155–167

Hacke UG, Sperry J, Pittermann J (2000) Drought experience and cavitation resistance in six shrubs from the Great Basin, Utah. Basic Appl Ecol 1:31–41

Hampp R, Schaeffer C (1999) Mycorrhiza-carbohydrate and energy metabolism. In: Hock B, Varma A (eds) Mycorrhiza, 2nd edn. Springer, Berlin, Germany, pp 273–303

Hare PD, Cress WA, Van Staden J (1998) Dissecting the roles of osmolyte accumulation during stress. Plant Cell Environ 21:535–553

Harrington CA (1987) Responses of Red alder and Black cottonwood seedlings to flooding. Physiol Plant 69:35–48

Havens KJ (1997) The effect of vegetation on soil redox within a seasonally flooded forested system. Wetlands 17:237–242

Head GC (1973) Shedding of roots. In: Kozlowski TT (ed) Shedding of plant parts. Academic, New York, pp 237–293

Hentschel K, Borken W, Matzner E (2007) Leaching losses of inorganic N and DOC following repeated drying and wetting of a spruce forest soil. Plant Soil 300:21–34

Herrera A, Tezara W, Rengifo E, Flores S (2008) Changes with seasonal flooding in sap flow of the tropical flood-tolerant tree species, *Campsiandra laurifolia*. Trees 22:551–558

Hillel D, Baker RS (1988) A descriptive-theory of fingering during infiltration into layered soils. Soil Sci 146:51–56

Hook DD, Brown CL, Kormanik PP (1971) Inductive flood tolerance in swamp tupelo (*Nyssa sylvatica* var. *biflora* (Walt.) Sarg.). J Exp Bot 22:78–89

Hook DD, Brown CL (1973) Root adaptations and relative flood tolerance of five hardwood species. For Sci 19:225–229

Hook DD, Scholtens JR (1978) Adaptation and flood tolerance of tree species. In: Hook DD, Crawford RMM (eds) Plant life in anaerobic environments. Ann Arbor Science Publication, Ann Arbor, MI, pp 299–331

Hsiao TC, Xu LK (2000) Sensitivity of growth of roots versus leaves to water stress: biophysical analysis and relation to water transport. J Exp Bot 51:1595–1616

Huang BR, Nobel PS (1993) Hydraulic conductivity and anatomy along lateral roots of cacti: changes with soil water status. New Phytol 123:499–507

Huber B (1956) Die Gefäßleitung. In: Ruhland W (ed) Handbuch der Pflanzenphysiologie, vol 2. Springer, Berlin, Germany, pp 541–582

Hulme M, Jenkins GJ, Lu X, Turnpenny JR, Mitchell TD, Jones RG, Murphy JM, Hassell D, Boorman P, Mcdonald R, Hills S (2002) Climate change scenarios for the United Kingdom: The UKCIP02 Scientific Report. Tyndall Centre for Climate Change Research. School of Environmental Sciences, University of East Anglia, Norwich, UK, 112 pp

Hundecha Y, Bárdossy A (2005) Trends in daily precipitation and temperature extremes across western Germany in the second half of the 20th century. Int J Climatol 25:1189–1202

Iglesias A, Rosenzweig C, Pereira D (2000) Agricultural impacts of climate change in Spain: developing tools for a spatial analysis. Global Environ Change 10:69–80

Inglima I, Alberti G, Bertolini T, Vaccari FP, Gioli B, Miglietta F, Cotrufo MF, Peressotti A (2009) Precipitation pulses enhance respiration of Mediterranean ecosystems: the balance between organic and inorganic components of increased soil CO_2 efflux. Global Change Biol 15:1289–1301

IPCC (2007) Climate change 2007: the physical science basis – 4AR. The Intergovernmental Panel on Climate Change, Geneva, Switzerland

Jackson MB, Davies WJ, Else MA (1996) Pressure-flow relationships, xylem solutes and root hydraulic conductance in flooded tomato plants. Ann Bot 77:17–24

Jackson RB, Moore LA, Hoffmann WA, Pockman WT, Linder CR (1999) Ecosystem rooting depth determined with caves and DNA. PNAS 96:11387–11392

Jones DL, Murphy DV (2007) Microbial response time to sugar and amino acid additions to soil. Soil Biol Biochem 39:2178–2182

Joslin JD, Wolfe MH, Hanson PJ (2000) Effects of altered water regimes on forest root systems. New Phytol 147:117–129

Kaldenhoff R, Fischer M (2006) Aquaporins in plants. Acta Physiol 187:169–176

Kennedy PG, Peay KG (2007) Different soil moisture conditions change the outcome of the ectomycorrhizal symbiosis between *Rhizopogon* species and *Pinus muricata*. Plant Soil 291:155–165

Kernaghan G (2005) Mycorrhizal diversity: cause and effect? Pedobiologia 49:511–520

Kirch HH, Vera-Estrella R, Golldack D, Quigley F, Michalowski CB, Barkla BJ, Bohnert HJ (2000) Expression of water channel proteins in *Mesembryanthemum crystallinum*. Plant Physiol 123:111–124

Kolb RM, Dolder H, Cortelazzo AL (2004) Effects of anoxia on root ultrastructure of four neotropical trees. Protoplasma 224:99–105

Konôpka B, Yuste JC, Janssens IA, Ceulemans R (2005) Comparison of fine root dynamics in Scots pine and Pedunculate oak in sandy soil. Plant Soil 276:33–45

Korn S (2004) Experimental investigation of water uptake and hydraulic properties of the root system of six European tree species (In German). Ph.D. thesis, University of Göttingen, Germany (http://webdoc.sub.gwdg.de/diss/2004/korn)

Köstler JN, Brückner E, Bibelriether H (1968) Die Wurzeln der Waldbäume. Parey, Berlin, Germany, 284 pp

Kozlowski TT (1984) Response of woody plants to flooding. In: Kozlowski TT (ed) Flooding and plant growth. Academic, Orlando, FL, pp 129–193

Kozlowski TT, Kramer PJ, Pallardy SG (1991) The physiological ecology of woody plants. Academic, San Diego, CA, 657 pp

Kozlowski TT (1997) Responses of woody plants to flooding and salinity. Tree Physiol Monogr 1:1–29

Kreuzwieser J, Furniss S, Rennenberg H (2002) Impact of waterlogging on the N-metabolism of flood tolerant and non-tolerant tree species. Plant Cell Environ 25:1039–1049

Kreuzwieser J, Papadopoulou E, Rennenberg H (2004) Interaction of flooding with carbon metabolism of forest trees. Plant Biol 6:299–306

Kung KJS (1990) Preferential flow in a sandy vadose zone 2 – mechanisms and implications. Geoderma 46:59–71

Kunstmann H, Schneider K, Forkel R, Knoche R (2004) Impact analysis of climate change for an Alpine catchment using high resolution dynamic downscaling of ECHAM4 time slices. Hydrol Earth Syst Sci 8:1030–1044

Lange B, Luescher P, Germann PF (2009) Significance of tree roots for preferential infiltration in stagnic soils. Hydrol Earth Syst Sci 13:1809–1821

Langer U, Rinklebe J (2009) Lipid biomarkers for assessment of microbial communities in floodplain soils of the Elbe River (Germany). Wetlands 29:353–362

Larcher W (2001) Ökophysiologie der Pflanzen, 6th edn. Ulmer, Stuttgart, Germany, 408 pp

Leake JR, Johnson D, Donnelly DP, Muckle GE, Boddy L, Read DJ (2004) Networks of power and influence: the role of mycorrhizal mycelium in controlling plant communities and agroecosystem functioning. Can J Bot 82:1016–1045

Lehnardt F, Brechtel HM (1980) Durchwurzelungs- und Schöpftiefen von Waldbeständen verschiedener Baumarten und Altersklassen bei unterschiedlichen Bodenbedingungen. AFZ 151:120–127

Leuschner C, Backes K, Hertel D, Schipka F, Schmitt U, Terborg O, Runge M (2001) Drought responses at leaf, stem and fine root levels of competitive *Fagus sylvatica* L. and *Quercus petraea* (MATT.) LIEBL. trees in dry and wet years. For Ecol Manage 149:33–46

Leuschner C, Coners H, Icke R, Hartmann K, Effinger ND, Schreiber L (2003) Chemical composition of the periderm in relation to in situ water absorption rates of oak, beech and spruce fine roots. Ann For Sci 60:763–772

Leuschner C, Coners H, Icke R (2004a) In situ measurement of water absorption by fine roots of three temperate trees: species differences and differential activity of superficial and deep roots. Tree Physiol 24:1359–1367

Leuschner C, Hertel D, Schmid I, Koch O, Muhs A, Hölscher D (2004b) Stand fine root biomass and fine root morphology in old-growth beech forests as a function of precipitation and soil fertility. Plant Soil 258:43–56

Liebersbach H, Steingrobe B, Claasen N (2004) Roots regulates ion transport in the rhizosphere to counteract reduced mobility in dry soil. Plant Soil 260:79–88

Lindenmair J, Matzner E, Zimmermann R (2004) The role of woody roots in water uptake of mature spruce, beech and oak trees. In: Matzner E (ed) Biogeochemistry of forest catchments in a changing environment. Springer, Berlin, Germany, pp 279–289

Lloret F, Peñuelas J, Estiarte M (2004) Experimental evidence of reduced diversity of seedlings due to climate modification in a Meditarranean-type community. Global Change Biol 10:248–258

Lloyd-Hughes B, Saunders MA (2002) A drought climatology for Europe. Int J Climatol 22:1571–1592

Lovisolo C, Schubert A (2006) Mercury hinders recovery of shoot hydraulic conductivity during grapevine rehydration: evidence from a whole-plant approach. New Phytol 172:469–478

Lovisolo C, Secchi F, Nardini A, Salleo S, Buffa R, Schubert A (2007) Expression of PIP1 and PIP2 aquaporins is enhanced in olive dwarf genotypes and is related to root and leaf hydraulic conductance. Physiol Plant 130:543–551

Lukac M, Calfapietra C, Godbold DL (2003) Production, turnover and mycorrhizal colonisation of root systems of three *Populus* species grown under elevated CO_2 (POPFACE). Global Change Biol 9:838–848

MacFall JS, Johnson GA, Kramer PJ (1990) Observation of a water-depletion region surrounding Loblolly pine roots by magnetic-resonance-imaging. PNAS 87:1203–1207

MacFall JS, Johnson GA, Kramer PJ (1991) Comparative water-uptake by roots of different ages in seedlings of Loblolly pine (*Pinus taeda* L.). New Phytol 119:551–560

Machado JL, Tyree MT (1994) Patterns of hydraulic architecture and water relations of two tropical canopy trees with contrasting leaf phenologies: *Ochroma pyramidale* and *Pseudobombax septenatum*. Tree Physiol 14:219–240

Mainiero R, Kazda M (2006) Depth-related fine root dynamics of *Fagus sylvatica* during exceptional drought. For Ecol Manage 237:135–142

Marschner H (1997) Mineral nutrition of higher plants, 2nd edn. Academic, London, UK, 889 pp

Manes F, Vitale M, Donato E, Giannini M, Puppi G (2006) Different ability of three Mediterranean oak species to tolerate progressive water stress. Photosynthetica 44:387–393

Meier IC, Leuschner C (2008a) Genotypic variation and phenotypic plasticity in the drought response of fine roots of European beech. Tree Physiol 28:297–309

Meier IC, Leuschner C (2008b) The belowground drought response of European beech: fine root biomass and carbon partitioning in 14 mature stands across a precipitation gradient. Global Change Biol 14:2081–2095

Meinzer FC, Fownes JH, Harrington RA (1996) Growth indices and stomatal control of transpiration in *Acacia koa* stands planted at different densities. Tree Physiol 16:607–615

Muhr J, Goldberg SD, Borken W, Gebauer G (2008) Repeated drying-rewetting cycles and their effects on the emission of CO_2, N_2O, NO, and CH_4 in a forest soil. J Soil Sci Plant Nutr 171:719–728

Nardini A, Pitt F (1999) Drought resistance of *Quercus pubescens* as a function of root hydraulic conductance, xylem embolism and hydraulic architecture. New Phytol 143:485–493

Nardini A, Salleo S, Tyree MT, Vertovec M (2000) Influence of the ectomycorrhizas formed by *Tuber melanosporum* Vitt. on hydraulic conductance and water relations of *Quercus ilex* L. seedlings. Ann For Sci 57:305–312

Neatrour MA, Jones RH, Golladay SW (2007) Response of three floodplain tree species to spatial heterogeneity in soil oxygen and nutrients. J Ecol 95:1274–1283

Nicoll BC, Coutts MP (1998) Timing of root dormancy and tolerance to root waterlogging in clonal Sitka spruce. Trees 12:241–245

Nisbet NR (2002) Implications of climate change: soil and water. In: Broadmeadow M (ed) Climate change: impacts on UK forests. Forestry Commission, Edinburgh, UK, pp 53–67

Ostonen I (2003) Fine root structure, dynamics and proportion in net primary production of Norway spruce forest ecosystems in relation to site condition. Dissertationes Biologicae Universitatis Tartuensis 84, Tartu University Press

Ostonen I, Püttsepp Ü, Biel C, Alberton O, Bakker MR, Lõhmus K, Majdi H, Metcalfe D, Olsthoorn AFM, Pronk A, Vanguelova E, Weih M, Brunner I (2007) Specific root length as an indicator of environmental change. Plant Biosyst 141:426–442

Osunubi O, Davies WJ (1981) Root growth and water relations of oak and birch seedlings. Oecologia 51:343–350

Parker MM, Van Lear DH (1996) Soil heterogeneity and root distribution of mature Loblolly pine stands in Piedmont soils. SSSAJ 60:1920–1925

Pärtel M, Helm A (2007) Invasion of woody species into temperate grasslands: relationship with abiotic and biotic soil resource heterogeneity. J Veg Sci 18:63–70

Pate JS, Jeschke WD, Aylward MJ (1995) Hydraulic architecture and xylem structure of the dimorphic root systems of South-West Australian species of Proteaceae. J Exp Bot 46:907–915

Paul EA, Clark FE (1996) Soil microbiology and biochemistry. Academic, London, UK, 340 pp

Pereira JS, David JS, David TS, Caldeira MC, Chaves MM (2004) Carbon and water fluxes in Mediterranean-type ecosystems – constraints and adaptations. In: Esser K, Lüttge U, Beyschlag W, Murata J (eds) Progress in botany 65. Springer, Berlin, Germany, pp 468–495

Pereira JS, Chaves MM, Caldeira MC, Correia A (2006) Water availability and productivity. In: Morison JIL, Morecroft MD (eds) Plant growth and climate change. Blackwell, Oxford, UK, pp 118–145

Pérez-Ramos IM, Marañón T (2009) Effects of waterlogging on seed germination of three Mediterranean oak species: ecological implications. Int J Ecol 35:422–428

Petit RJ, Hampe A, Cheddadi R (2005) Climate changes and tree phylogeography in the Mediterranean. Taxon 54:877–885

Pickard WF (1989) How might a tracheary element which is embolized by day be healed by night. J Theor Biol 141:259–279

Pietikäinen J, Vaijärvi E, Ilvesniemi H, Fritze H, Westman CJ (1999) Carbon storage of microbes and roots and the flux of CO_2 across a moisture gradient. Can J For Res 29:1197–1203

Pigott CD, Pigott S (1993) Water as a determinant of the distribution of trees at the boundary of the Mediterranean zone. J Ecol 81:557–566

Pinay G, Black VJ, Planty-Tabacchi AM, Gumiero B, Décamps H (2000) Geomorphic control of denitrification in large river floodplain soils. Biogeochemistry 50:163–182

Polomski J, Kuhn N (1998) Wurzelsysteme. Haupt, Bern, Switzerland, 290 pp

Pronk AA, De Willigen P, Heuvelink E, Challa H (2002) Development of fine and coarse roots of *Thuja occidentalis* 'Brabant' in non-irrigated and drip irrigated field plots. Plant Soil 243:161–171

Raven JA, Edwards D (2001) Roots: evolutionary origins and biogeochemical significance. J Exp Bot 52:381–401

Ray AM, Inouye RS (2006) Effects of water-level fluctuations on the arbuscular mycorrhizal colonization of *Typha latifolia* L. Aquat Bot 84:210–216

Ray D, Nicoll BC (1998) The effect of soil water-table depth on root-plate development and stability of Sitka spruce. Forestry 71:169–182

Raynaud X, Leadley PW (2004) Soil characteristics play a key role in modelling nutrient competition in plant communities. Ecology 85:2200–2214

Reece CF, Riha SJ (1991) Role of root systems of Eastern larch and White spruce in response to flooding. Plant Cell Environ 14:229–234

Rewald B (2008) Impact of climate change-induced drought on tree root hydraulic properties and competition belowground. Ph.D. thesis, University of Göttingen, Germany (http://webdoc.sub.gwdg.de/diss/2008/rewald)

Rewald B, Leuschner C (2009a) Belowground competition in a broad-leaved temperate mixed forest – pattern analysis and experiments in a four species stand. Eur J Forest Res 128:387–398

Rewald B, Leuschner C (2009b) Does root competition asymmetry increase with water availability? Plant Ecol Div 2:255–264

Rewald B, Ephrath JE, Rachmilevitch S (2010) A root is a root is a root? - Water uptake rates of *Citrus* root orders. Plant Cell Environ DOI: 10.1111/j.1365-3040.2010.02223.x

Richards JH, Caldwell MM (1987) Hydraulic lift – substantial nocturnal water transport between soil layers by *Artemisia tridentata* roots. Oecologia 73:486–489

Richardson MJ, Vepraskas JL (2000) Wetlands soils: genesis, hydrology, landscapes and classification. CRC Press, Boca Raton, FL

Rieger M, Litvin P (1999) Root system hydraulic conductivity in species with contrasting root anatomy. J Exp Bot 50:201–209

Rowell DP (2005) A scenario of European climate change for the late twenty-first century: seasonal means and interannual variability. Clim Dyn 25:837–849

Ryel RJ (2004) Hydraulic redistribution. In: Esser K, Lüttge U, Beyschlag W, Murata J (eds) Progress in botany 65. Springer, Berlin, Germany, pp 413–435

Sardans J, Peñuelas J, Ogaya R (2008) Drought-induced changes in C and N stoichiometry in a *Quercus ilex* Mediterranean forest. For Sci 54:513–522

Schäffner AR (1998) Aquaporin function, structure, and expression: are there more surprises to surface in water relations? Planta 204:131–139

Schleppi P, Hagedorn F, Providoli I (2004) Nitrate leaching from a mountain forest ecosystem with Gleysols subjected to experimentally increased N deposition. Water Air Soil Pollut Focus 4:453–467

Schreiber L, Franke R, Hartmann KD, Ranathunge K, Steudle E (2005) The chemical composition of suberin in apoplastic barriers affects radial hydraulic conductivity differently in the roots of rice (*Oryza sativa* L. cv. IR64) and corn (*Zea mays* L. cv. Helix). J Exp Bot 56:1427–1436

Schmull M, Thomas FM (2000) Morphological and physiological reactions of young deciduous trees (*Quercus robur* L., *Q. petraea* [MATT.] LIEBL., *Fagus sylvatica* L.) to waterlogging. Plant Soil 225:227–242

Schulze ED, Caldwell MM, Canadell J, Mooney HA, Jackson RB, Parson D, Scholes R, Sala OE, Trimborn P (1998) Downward flux of water through roots (i.e. inverse hydraulic lift) in dry Kalahari sands. Oecologia 115:460–462

Secchi F, Lovisolo C, Uehlein N, Kaldenhoff R, Schubert A (2007) Isolation and functional characterization of three aquaporins from olive (*Olea europaea* L.). Planta 225:381–392

Shani U, Waisel Y, Eshel A, Xue S, Ziv G (1993) Responses to salinity of grapevine plants with split root systems. New Phytol 124:695–701

Shi LB, Guttenberger M, Kottke I, Hampp R (2002) The effect of drought on mycorrhizas of beech (*Fagus sylvatica* L.): changes in community structure, and the content of carbohydrates and nitrogen storage bodies of the fungi. Mycorrhiza 12:303–311

Silberbush M, Barber SA (1983) Sensitivity analysis of parameters used in simulating K uptake with a mechanistic mathematical model. Agron J 75:851–854

Silvan N, Regina K, Kitunen V, Vasander H, Laine J (2002) Gaseous nitrogen loss from a restored peatland buffer zone. Soil Biol Biochem 34:721–728

Smit BA, Stachowiak M (1988) Effects of hypoxia and elevated carbon-dioxide concentration on water flux through *Populus* roots. Tree Physiol 4:153–165

Smith SE, Read DJ (2008) Mycorrhizal symbiosis, 3rd edn. Academic, London, UK, 787 pp

Sperry JS, Saliendra NZ (1994) Intra- and inter-plant variation in xylem cavitation in *Betula occidentalis*. Plant Cell Environ 17:1233–1241

Sperry JS, Ikeda T (1997) Xylem cavitation in roots and stems of Douglas fir and White fir. Tree Physiol 17:275–280

Staelens JA, De Schrijver A, Verheyen K, Verhoest NEC (2006) Spatial variability and temporal stability of throughfall water under a dominant beech (*Fagus sylvatica* L.) tree in relationship to canopy cover. J Hydrol 330:651–662

Stenström E (1991) The effects of flooding on the formation of ectomycorrhizae in *Pinus sylvestris* seedlings. Plant Soil 131:247–250

Stephenson NL (1990) Climatic control of vegetation distribution: the role of the water balance. Am Nat 135:649–669

Steudle E (1994) Water transport across roots. Plant Soil 167:79–90

Steudle E, Heydt H (1997) Water transport across tree roots. In: Rennenberg H, Eschrich W, Ziegler H (eds) Trees – contributions to modern tree physiology. Backhuys, Leiden, The Netherlands, pp 239–255

Steudle E (2000) Water uptake by roots: effects of water deficit. J Exp Bot 51:1531–1542

Thomas FM (2000) Growth and water relations of four deciduous tree species (*Fagus sylvatica* L., *Quercus petraea* [MATT.] LIEBL., *Q. pubescens* WILLD., *Sorbus aria* [L.] CR.) occurring at Central-European tree-line sites on shallow calcareous soils: physiological reactions of seedlings to severe drought. Flora 195:104–115

Topa MA, McLeod KW (1986) Responses of *Pinus clausa*, *Pinus serotina* and *Pinus taeda* seedlings to anaerobic solution culture. I. Changes in growth and root morphology. Physiol Plant 68:523–531

Trillo N, Fernández R (2005) Wheat plant hydraulic properties under prolonged experimental drought: stronger decline in root-system conductance than in leaf area. Plant Soil 277:277–284

Trubat R, Cortina J, Vilagrosa A (2006) Plant morphology and root hydraulics are altered by nutrient deficiency in *Pistacia lentiscus* (L.). Trees 20:334–339

Tsukahara H, Kozlowski TT (1985) Importance of adventitious roots to growth of flooded *Platanus occidentalis* seedlings. Plant Soil 88:123–132

Unger IM, Kennedy AC, Muzika RM (2009) Flooding effects on soil microbial communities. Appl Soil Ecol 42:1–8

Valdéz M, Asbjornsen H, Cárdenas MG, Juârez M, Vogt KA (2006) Drought effects on fine-root and ectomycorrhizal-root biomass in managed *Pinus oaxacana* MIROV. stands in Oaxaca, Mexico. Mycorrhiza 16:117–124

Vanninen P, Ylitalo H, Sievanen R, Mäkelä A (1996) Effects of age and site quality on the distribution of biomass in Scots pine (*Pinus sylvestris* L.). Trees 10:231–238

Vera-Estrella R, Barkla BJ, Bohnert HJ, Pantoja O (2004) Novel regulation of aquaporins during osmotic stress. Plant Physiol 135:2318–2329

West GB, Brown JH, Enquist BJ (1999) A general model for the structure and allometry of plant vascular systems. Nature 400:664–667

White TCR (2007) Flooded forests: death by drowning, not herbivory. J Veg Sci 18:147–148

Xu YJ, Röhrig E, Fölster H (1997) Reaction of root systems of grand fir (*Abies grandis* LINDL) and Norway spruce (*Picea abies* KARST.) to seasonal waterlogging. For Ecol Manage 93:9–19

Yamada S, Komori T, Myers PN, Kuwata S, Kubo T, Imaseki H (1997) Expression of plasma membrane water channel genes under water stress in *Nicotiana excelsior*. Plant Cell Physiol 38:1226–1231

Yu KW, Faulkner SP, Baldwin MJ (2008) Effect of hydrological conditions on nitrous oxide, methane, and carbon dioxide dynamics in a bottomland hardwood forest and its implication for soil carbon sequestration. Global Change Biol 14:798–812

Zimmermann MH (1983) Xylem structure and the ascent of sap. Springer, Berlin, Germany, 143 pp

Zimmermann HM, Steudle E (1998) Apoplastic transport across young maize roots: effect of the exodermis. Planta 206:7–19

Zwieniecki MA, Holbrook NM (1998) Diurnal variation in xylem hydraulic conductivity in White ash (*Fraxinus americana* L.), Red maple (*Acer rubrum* L.) and Red spruce (*Picea rubens* SARG.). Plant Cell Environ 21:1173–1180

Chapter 3
Above Ground Processes: Anticipating Climate Change Influences

Mauro Centritto, Roberto Tognetti, Ernst Leitgeb, Katarina Střelcová, and Shabtai Cohen

Introduction

Over the past 250 years since the beginning of the industrial revolution the amount of CO_2 in the atmosphere has been gradually increasing from ca. 275 to 390 ppm, largely as a result of land-use change and anthropogenic emissions from the burning of fossil fuels. The atmospheric CO_2 concentration ($[CO_2]$), which is now higher than it was at any time in the past 20–25 million years (Pearson and Palmer, 2000), rose at the rate of 1.9 ppm year^{-1} for the 1995–2005 decade (i.e. the largest average increase observed for any decade in at least the last 200 years) (IPCC 2007) and is expected to double during this century. The most important consequence of this rise in $[CO_2]$ is a change in the long wave radiation balance and warming of the surface temperature of the Earth. The total temperature increase over the past 150 years was

---

M. Centritto (✉)
Institute of Agro-Environmental and Forest Biology, National Research Council, Via Salaria km 29,300, 00015, Monterotondo Scalo (RM), Italy
e-mail: mauro.centritto@cnr.it

R. Tognetti (✉)
EcoGeoFor Lab, Department of Science and Technology for the Environment and Territory (STAT), University of Molise, Contrada Fonte Lappone 86090, Pesche (IS), Italy
e-mail: tognetti@unimol.it

E. Leitgeb (✉)
Department of Forest Ecology and Soil, Federal Research and Training Centre for Forests, Natural Hazards and Landscape, Seckendorff Gudent Weg 8 1131, Vienna, Austria
e-mail: ernst.leitgeb@bfw.gv.at

K. Střelcová (✉)
Department of Natural Environment, Faculty of Forestry Technical University in Zvolen, T.G. Masaryka 24, 960 53, Zvolen, Slovakia
e-mail: strelcov@vsld.tuzvo.sk

S. Cohen (✉)
Institute of Soil Water and Environmental Sciences, ARO Volcani Center, POB 6, Bet Dagan 50250, Israel
e-mail: vwshep@agri.gov.il

M. Bredemeier et al. (eds.), *Forest Management and the Water Cycle: An Ecosystem-Based Approach*, Ecological Studies 212, DOI 10.1007/978-90-481-9834-4_3,

0.76°C. Moreover, the steady increase in the concentration of tropospheric O_3 and other air pollutants, including various kinds of aerosols have caused other climate changes. The IPCC Fourth Assessment Report (IPCC 2007) suggests that changes in atmospheric constituents and in radiative forcing of the climate system are likely to continue. The global average surface warming following a doubling of carbon dioxide concentrations is likely to be in the range of 2–4.5°C by the end of the century. Increasing temperature and atmospheric [CO_2] along with associated changes in the hydrological cycle will most likely alter the structure and function of forest ecosystems.

The three most important climate features that influence forests are precipitation, atmospheric and soil dryness, and temperature. Climate classifications are usually based on these three features, and especially the ratio of precipitation to atmospheric dryness. Thus, decreases in precipitation and/or increased atmospheric dryness (which can also result from increased temperatures unaccompanied by increases in absolute humidity) can change climate from mesic to semi-arid or arid. Climate change is expected to exacerbate and reiterate regional drought events, especially mid-latitude aridity (Jentsch et al. 2007). Dieback of forest trees in response to extreme climate events can have long-term impacts on community dynamics and species interactions (Condit et al. 1995; Breshears et al. 2005; Gitlin et al. 2006; Allen and Breshears 2007), and may feed back upon atmospheric CO_2 and climate.

There are several lines of evidence that temperate and boreal forests influence the physical and chemical properties of the atmosphere through evapotranspiration, albedo and carbon exchange, which may have positive and negative forcings on regional and continental climate (Bonan 2008; Rotenberg and Yakir 2010). The carbon cycle is the most important of the biogeochemical cycles implicated in the greenhouse effect, accounting for more than 63% of greenhouse forcing (IPCC 2007). The natural biogeochemical movement of carbon to and from the terrestrial vegetation is larger than that from anthropogenic activities (fossil-fuel use and deforestation). Of the 762 Pg of carbon in the atmosphere, about 122 Pg C are annually exchanged between the atmosphere and terrestrial vegetation, i.e. removed from the atmosphere through photosynthesis, and returned to the atmosphere by plant respiration and organic mass decomposition (Denman et al. 2007). More than 16% of the atmospheric CO_2 each year reacts with Rubisco (ribulose bisphosphate carboxylase-oxygenase, the primary photosynthetic enzyme which converts inorganic carbon, as CO_2, into organic compounds in terrestrial plants with the C3 photosynthetic pathway) in more than 95% of earth's plant species, including all temperate and boreal tree species.

Worldwide, forests play an important role in the global carbon cycle (Fig. 3.1) because they cover about 30% of the Earth's land surface (Bonan 2008). Forests are estimated to comprise about 95% of all aboveground and 40% of belowground terrestrial pools of organic carbon. They therefore contribute significantly to the terrestrial carbon sink (Körner 2006; Denman et al. 2007). Forests also play a major role in regulating the global hydrologic cycle (Fig. 3.2). Together with carbon sequestration, evapotranspiration, through feedbacks with clouds and precipitation, exerts a negative "physiological" forcing on regional and continental climate (Bonan 2008; Rotenberg

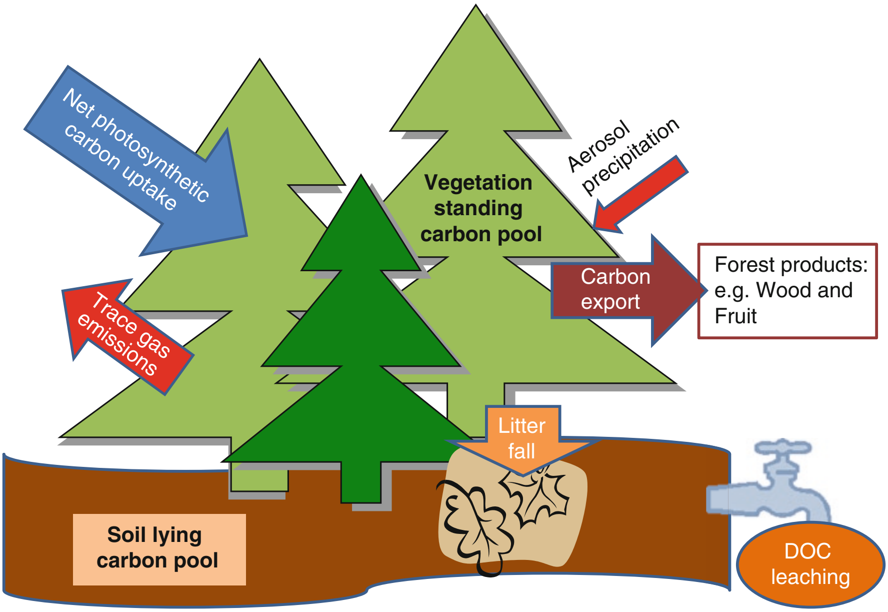


Fig. 3.1 Main components of the forest carbon cycle. The forest is a major factor in global carbon sequestration and represents a large standing pool of carbon. Changes in climate can lead to far reaching consequences for the forest carbon cycle and global CO_2 levels. Although carbon sequestration is a negative radiative forcing (blue), forest emission of isoprenoids and other greenhouse trace gases, and precipitation of aerosols in forest are positive forcings (red). Carbon leaves the forest in forest products or as dissolved organic carbon (DOC). Net photosynthetic uptake is described in the text

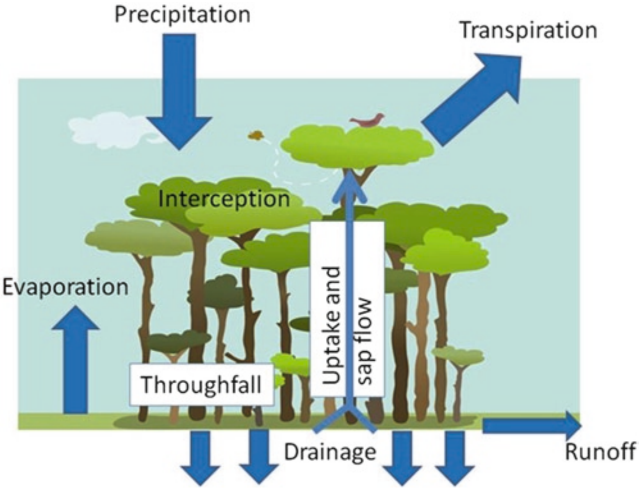


Fig. 3.2 Main components of the forest hydrological cycle. Small changes in any of the components of the forest hydrological cycle can have large influences outside the forest, e.g. changing flooding and water outflow, or groundwater and aquifer recharge. Quantifying these components is a major challenge

and Yakir 2010). Climate change may critically alter the biogeophysical and biogeochemical functioning of forests. Our current ability to predict when regional-scale plant stress will exceed a threshold that results in rapid and large-scale shifts in ecosystem structure and function is lacking. However, it is fundamentally needed to assess potential climate-change impacts (McDowell et al. 2008), including changes in vegetation and associated ecosystems and their feedbacks to the climate system (Keane et al. 2001; Scholze et al. 2006). Thus, understanding the effects of climate change on carbon assimilation and transpiration is critical to predict the future physiological feedbacks of forests on both the biosphere-atmosphere interactions (Bonan 2008; Rotenberg and Yakir 2010) and continental runoff (Betts et al. 2007).

This chapter discusses the interactive influences of climate change on forest processes at leaf (i.e. primary physiological and secondary metabolic responses), whole-plant (e.g. tertiary growth responses), and ecosystem levels (e.g. influences of forest on climate).

Elevated [CO_2] Influences on Leaf to Tree Level Processes

Along with land use transformation, changes in the chemical composition of the atmosphere, with increasing greenhouse gases, is the most important component of global change. Of the several anthropogenic greenhouse gases emitted globally, CO_2 is pre-eminent as an agent of potential future climate, as it accounts for about 63% of the gaseous radiative forcing responsible for anthropogenic climate change. Unlike temperature, precipitation and pollution concentrations, which have high spatial variations, rising [CO_2] is globally remarkably uniform and is likely to affect forest growth worldwide and, consequently, their “physiological” forcings on atmospheric temperature and hydrologic cycles. Therefore, studies on the effects of elevated [CO_2] on tree growth and resource use efficiency are crucial to understand the impact of rising [CO_2] on the biogeophysical and biogeochemical functioning of forests.

In the short-term, increasing levels of [CO_2] influence directly the physiology of terrestrial C3 plants via increased net photosynthesis (A) and decreased transpiration (E). Stomata modulate these primary physiological processes, because they act as control valves in the *pathways* of gaseous diffusion for the incoming CO_2 and the outgoing transpirational water vapour, enabling optimisation of CO_2 uptake per water loss. Notwithstanding this, A and E can themselves affect stomatal conductance (g_s) through several feedback loops (Wong et al. 1979). The implication of the complex direct or indirect feedback effects on g_s is that there are significant uncertainties about the physiological controls of stomatal behavior and it is not always apparent whether g_s controls gas exchange or vice versa. The first part of this section addresses the direct influence of rising [CO_2] on leaf gas exchange, by analysing the quantitative links between leaf biochemistry and gas exchange kinetics; the second part then reviews the main responses of forest trees to elevated [CO_2].

In C_3 species, short-term response of A to changes in intercellular CO_2 concentrations (C_i) are well known. In the model of Farquhar et al. (1980), A is given as:

$$A = v_c - 0.5v_o - R_d = v_c(1 - \Gamma^*/C_i) - R_d \tag{3.1}$$

where v_c and v_o are the carboxylation rate and the oxygenation rate of Rubisco, respectively, 0.5 is the stoichiometry between O_2 uptake by RubP (ribulose bisphosphate) oxygenase and photorespiratory efflux of CO_2 (Jordan and Ogren 1984), and Γ* is the photosynthetic compensation point, i.e. the [CO_2] at which the photorespiratory CO_2 evolution equals the rate of photosynthetic CO_2 uptake. Using Fick's first law of diffusion, it is possible to measure E as:

$$E = g_{tw}(\chi_i - \chi_a) \tag{3.2}$$

where χ_i and χ_a are the water vapor concentrations inside the leaf and in the ambient air, respectively, and g_{tw} is the total leaf conductance to water vapour, which is given by:

$$g_{tw} = g_s \cdot g_{bl} / (g_s + g_{bl}) \tag{3.3}$$

where g_{bl} is the boundary layer conductance to water vapour. Stomatal conductance to water vapour can then be obtained from g_{tw} by removing the g_{bl} contribution:

$$g_s = 1/(1/g_{tw} - 1/g_{bl}) \tag{3.4}$$

Because CO_2 diffuses along the same pathway as water, and considering that the ratio of the binary molecular diffusivities of CO_2 and water vapor in air is taken as 1.6 in the stomata and 1.37 in the boundary layer, it is possible to calculate the combined boundary layer-stomatal conductance to CO_2 (g_{sc}) as:

$$g_{sc} = 1/(1.6/g_s - 1.37/g_{bl}) \tag{3.5}$$

Then net steady state A can be also expressed as:

$$A = g_{sc}(C_a - C_i) = g_m(C_i - C_c) = g_t(C_a - C_c) \tag{3.6}$$

where:

$$g_t = g_{sc} \cdot g_m / (g_{sc} + g_m) \tag{3.7}$$

g_m and g_t are the mesophyll conductance and the total conductance to CO_2 diffusion, respectively, while C_c is [CO_2] at the Rubisco binding sites in the chloroplast. C_c is proportional to the gradient between [CO_2] in the air (C_a) and in the chloroplasts. C_c is, therefore, inversely related to the total resistance to CO_2 diffusion from air through leaf boundary layer and stomata into both the substomatal cavities and the intercellular air spaces present in the mesophyll, i.e. boundary layer-stomatal resistance to CO_2 diffusion in the gas phase, and from the cell walls to the sites of carboxylation, i.e. mesophyll resistances to CO_2 diffusion in the gas and liquid phase (Centritto et al. 2003; Niinemets et al. 2009).

According to Eq. 3.1, A is dependent on the carboxylation-photorespiration balance and on respiration, and is ultimately driven by C_c (Eq. 3.6); whereas, assuming that the sub-stomatal cavity is saturated with water vapour, and that g_{bl} is not affected by growth in elevated [CO_2], leaf transpiration is controlled by g_s at any given absolute humidity of the outside atmosphere (Eqs. 3.2 and 3.4). Because respiration is not inhibited by growth in elevated [CO_2], contrary to what was reported in earlier studies as a result of an artefact of the way respiration measurements were made (Davey et al. 2004), Ainsworth and Rogers (2007) have recently pointed out that rising [CO_2] affects plants and ecosystems via two fundamental processes: enhanced A and reduced g_s. Because the kinetics sensitivity of these two physiological processes to climate change factors affects both the carbon and the hydrological cycles, they are becoming embedded in models of the biogeochemical and of land surface feedbacks on climate (Bonan 2008).

CO_2 may be directly sensed by the surface of the guard cells in response to variations in C_i. Mott (1988) showed that stomatal aperture responds to C_i, such that the C_i/C_a ratio remains approximately constant. This conservative ratio indicates that changes in C_a, by causing proportional changes in C_i, make responses to C_i effective sensors of changes of C_a (Mott 1988). However, conservative C_i/C_a ratios imply that stomatal conductance (Eq. 3.6) and, in turn, leaf-level transpiration (Eq. 3.2), decrease as [CO_2] increases.

At ambient [CO_2], the operating C_c is generally at the transition between the limitations to photosynthesis caused by Rubisco activity and RubP regeneration capacity (Farquhar et al. 1980). However, because g_t is usually lower in forest trees than in herbaceous and shrub species (Niinemets et al. 2009), C_c of non-stressed trees is well below the transition from Rubisco carboxylation-limitation to RuBP-regeneration limitation (Fig. 3.3). This implies that Rubisco is not CO_2-saturated at current atmospheric [CO_2] and, consequently, A is limited by substrate supply. Moreover, CO_2 is in competition with O_2 for the active sites of Rubisco, which consequently can react with either CO_2 or oxygen, the latter leading to photorespiration, which generally accounts for about 30% of carbon loss in C3 leaves at 25°C (von Caemmerer and Quick 2000). Despite the decline in stomatal conductance and, consequently, in total diffusional limitations to photosynthesis (Eq. 3.6), increases in [CO_2] will result in higher C_c which will not only reduce photorespiratory loss by decreasing the oxygenation rate of Rubisco (Stitt and Krapp 1999), but will also concomitantly increase its carboxylation reaction rate. Thus, C3 photosynthesis is stimulated in elevated [CO_2], although its marginal increment declines as [CO_2] increases (Fig. 3.3) (Körner 2006; Loreto and Centritto 2008). Decreased g_s associated with high C_i is an adaptive response to C_a, by which diffusional limitations to A are adjusted in response to changes in mesophyll demand for CO_2 (i.e. the biochemical limitations to A) resulting in an increase in instantaneous transpiration efficiency (ITE) (Centritto et al. 2002; Wullschleger et al. 2002; Hu et al. 2010).

The effects of growth in elevated [CO_2] on g_m have surprisingly received little attention, and the few published studies have reported either unaffected g_m, in *Betula pendula* (Eichelmann et al. 2004), in shade leaves of *Liquidambar styraciflua* and in *Populus tremuloides* (Singsaas et al. 2003), or increased g_m, in sun leaves of *Liquidambar styraciflua* (Singsaas et al. 2003), in response to elevated [CO_2]. On the

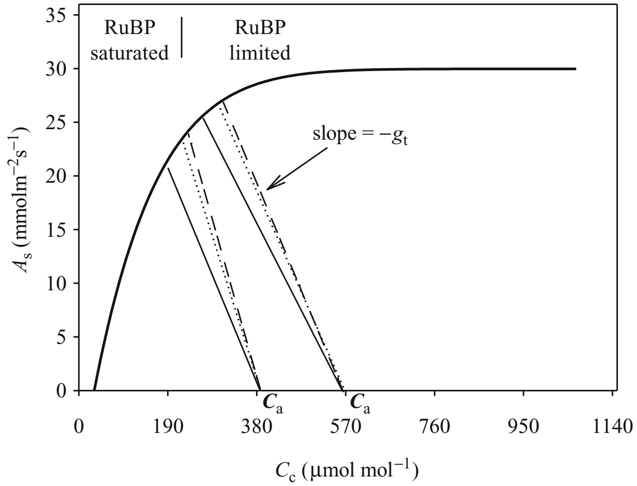


Fig. 3.3 Summary of A_s/C_c (ratio of assimilation to CO_2 concentration at the chloroplast) curves, plotted from average *J*, *V* and maximum *A* at saturating [CO_2] and PPFD, in plants of *Populus nigra* (Centritto et al. 2010 unpublished results). The initial slope of the curve is limited by Rubisco carboxylation efficiency, and is therefore ribulose-1,5-bisphosphate (RuBP) saturated. Beyond the inflection curve, A_s is assumed to be limited by the potential RuBP regeneration rate. The lines relating either ambient or elevated C_a (ambient CO_2) to A_s are the supply function for CO_2 in trees (—), crops and legumes (·······), and grasses (----), with the slopes of these lines equal to $-g_t$

contrary, the effects of elevated [CO_2] on *A*, g_s and ITE of forest species have been investigated in many studies (see for reviews: Curtis and Wang 1998; Ainsworth and Long 2005; Körner 2006). A meta-analysis of many FACE (Free Air Carbon dioxide Enrichment) experiments on forest trees (Ainsworth and Long 2005) showed that elevated [CO_2] resulted in a 47.4% increase in PPFD-saturated *A*, a 28.6% increase in the diurnal photosynthetic carbon assimilation, a 73.7% increase in ITE, and a 15.9% decrease in g_s (Fig. 3.4). It has been frequently reported that long-term growth in elevated [CO_2] may induce loss of photosynthetic capacity in C_3 species (Stitt and Krapp 1999). Downward acclimation of photosynthetic capacity may represent an optimisation of the distribution of the resources within the chloroplast to avoid the situation where either Rubisco or the apparatus for the regeneration of RuBP are in excess. Although some acclimation of photosynthesis capacity has been demonstrated in studies of trees grown in open-air field conditions, e.g. in fast growing poplar clones (Bernacchi et al. 2003) and in 1-year-old pine foliage (Crous et al. 2008), it is noteworthy that virtually no significant downward acclimation of photosynthesis of tree species, as expressed by maximum Rubisco carboxylation rate (*V*), maximum rate of electron transport (*J*) and *V*/*J* ratio (Fig. 3.4), was found by Ainsworth and Long (2005) in their literature review of FACE experiments. This is further evidence that when plants growing in elevated [CO_2] are rooted in the ground and adequate sinks are available so that N uptake keeps pace with carbon uptake and the source-sink functional balance is not altered,

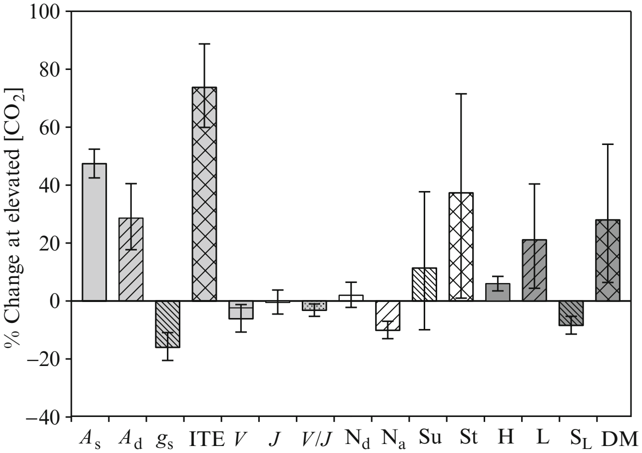


Fig. 3.4 Mean response (±95% confidence interval) of forest trees to elevated [CO_2] (data derived from the meta-analysis of many FACE experiments by Ainsworth and Long 2005): light saturated photosynthesis (A_s), diurnal carbon assimilation (A_d), stomatal conductance (g_s), instantaneous transpiration efficiency (ITE), maximum Rubisco carboxylation rate (V), maximum electron transport rate (J), V/J ratio, nitrogen content per unit of dry mass (N_d), nitrogen content on leaf area basis (N_a), sugar (Su) and starch (St) content on area basis, plant height (H), leaf-area index (L), specific leaf area (S_L), above-ground dry matter production (DM)

downward acclimation of photosynthetic capacity does not occur (Centritto et al. 2004; Springer et al. 2005).

The main tree responses to elevated [CO_2] are summarized in Fig. 3.4. In general, trees are more responsive to growth in elevated [CO_2] than plants belonging to other functional groups (Ainsworth and Long 2005; Körner 2006). In fact, despite the fact that height increased by only 6%, trees showed the largest response in terms of LAI (+21.1%) and aboveground dry mass (+28%) in response to growth under FACE (Fig. 3.4). Furthermore, Norby et al. (2005) found that the stimulation of forest net primary productivity in response to elevated [CO_2] is highly conserved across a broad range of productivity. The largest response of trees to elevated [CO_2] has been ascribed to a number of concomitant factors. First of all, g_{sc} and g_m are generally lower in forest trees than in other plants (Niinemets et al. 2009), resulting in lower g_t and, consequently, in lower C_c (Fig. 3.3). Because at current atmospheric [CO_2] the operating C_c of C3 plants lies in the curvilinear region of the photosynthetic-C_c response curve, the lower C_c of forest trees implies that as C_a increases the stimulation in A is higher than the increase for other functional groups (Fig. 3.3). In addition, for the most part, trees grown under FACE conditions were not only young and fast-growing, but were grown in either decoupled (in terms of nutrient supply, i.e. plants non depending on a natural nutrient cycle) or expanding (plants given ample space and time to access the available resources per unit land area) systems (see Körner 2006, for a thorough review). These trees could then accumulate "signals" (Körner 2006) propagating their effect over growing seasons

(Blackman 1919; Centritto et al. 1999b), resulting finally in their being more responsive to elevated [CO_2] than crops and grassland (Ainsworth and Long 2005). It is noteworthy that no lasting growth stimulation by CO_2 enrichment was found in 32-to 35-m-tall, mature trees in a near-natural deciduous forest (i.e. in a coupled systems': steady-state nutrient cycle and full canopy development - Körner 2006) in Switzerland after 4 years (Körner et al. 2005).

Drought, Rising Temperature, and Extreme Events Influences on Leaf to Tree Level Processes

Recent drought events, which are part of a widespread pattern of drying throughout the Northern Hemisphere, and which appear to be the worst since at least the middle of the fifteenth century, have the potential to become more frequent and exert an impact on larger mid-latitude areas, as projected by general circulation models. Warmer air temperature will exacerbate the impact of drought on plant water loss by elevating the vapour pressure deficit (VPD) in the atmosphere, thereby increasing the potential transpiration (see review by Mc Dowell et al. 2008). Mc Dowell et al. (2008) pointed out that elevated temperature may increase hydraulic failure and carbon starvation, and that changes in demographics of mortality agents (Thomas et al. 2002), such as insects and pathogens, may amplify carbon starvation, the main mechanism for drought driven mortality, when a species and site specific critical threshold of evapotranspiration is surpassed. The progressive massive dieback of woody plants, primarily Norway spruce (*Picea abies* L.) in the central Europe regions has recently become a well-known reality (Střelcová et al. 2009). Similar damaged stands are in boundary regions of Poland, in the Czech Republic, Germany and Italy. Furthermore warming is predicted to cause earlier snow melt and this is likely to increase the length and severity of summer droughts.

Drought influences plant growth in a number of different ways, i.e. through a decrease in the water potential of cambial cells, resulting in subsequent inhibition of cell growth, reduction of metabolic activity, inhibition of stomatal conductance and consequently of photosynthesis. Physiological processes are sensitive indicators of stress in plants, especially in extreme environmental conditions. However, it is difficult to identify all of the relevant factors influencing the water regime of forest stands, although Schwalm et al. (2010) have recently shown that assimilation is more sensitive to drought than respiration at the ecosystem level. To describe the water demands of tree species precisely it is important to know their response to various water stress levels and to characterise symptoms and consequences of the drought effect on forest trees and stands. At the tree level, sap flow measurements provide accurate estimates of the tree water supply and sap flow is also a good indicator for tree water stress (Nadezhdina 1999). Transpiration and consequently sap flow is controlled by atmospheric demand and soil water content. Air temperature, radiation, wind speed and air humidity are the main driving forces of the atmospheric demand, which is characterized by the potential evapotranspiration

(PET). PET can be calculated by using the FAO Penman-Monteith equation (Allen et al. 1998). A threshold for soil water deficit can be derived by using the definition of "relative extractable soil water" (R_{ew}) of Bréda et al. (1995):

$$R_{\mathrm{ew}} = \left(S_{\mathrm{act}} - S_{\min}\right) / W_{\mathrm{ext}} \tag{3.8}$$

where S_{act} is the actual soil water content; $S_{\min}$ is the minimum observed soil water content and W_{ext} is the maximum observed extractable water. A R_{ew} threshold between "control by demand" and "control by offer"of 0.4 indicates soil water deficit. Transpiration, photosynthesis and respiration are sharply decreased when the R_{ew} drops below this threshold (Granier et al. 2007). If sufficient soil water is available for the trees, transpiration is dominated by PET and the correlation between PET and sap flow is strong. As soil water reserves shrink due to drought, available soil water decreases and the correlation between PET and the transpiration weakens. Consequently, soil water content is the major determinant of transpiration. After soil water reserves are recharged the atmospheric demand returns to playing a major role in transpiration (Leitgeb et al. 2002).

Plant responses to [CO_2] can be either amplified or reduced by water and nutrient limitations (McCarthy et al. 2010) and by rising temperature, which is already increasing growing season length over Europe (Menzel and Fabian 1999). Curtis and Wang (1998) showed that growth of woody plants in elevated [CO_2] was halved under nutrient limitations. Ainsworth and Long's synthesis (2005) of the results from FACE experiments supports these conclusions. Similarly, McCarthy et al. (2010) found that the absolute enhancements of net primary productivity of trees growing in elevated [CO_2] became progressively smaller as nitrogen availability decreased and were not observable when nitrogen availability was very low. In contrast, low water availability is often shown to amplify tree growth responses to elevated [CO_2] (Wullschleger et al. 2002; Seiler et al. 2009). Amplifications of CO_2 responses in water stressed conditions is caused by reduced g_s and, in turn, by decreased leaf level transpiration under elevated [CO_2], which may lead to an increase in plant water potential and water use efficiency (Centritto et al. 2002), a delay in the onset of drought (Centritto et al. 1999c), and a conservation of soil water (Wullschleger et al. 2002).

Many studies have addressed the interactions that arise between elevated [CO_2] and drought, and most have focused on one or more components of plant water relations (Tschaplinski et al. 1993; Tognetti et al. 1998, 2000b; Centritto et al. 1999c; Ellsworth 1999), addressing the potential interaction between elevated [CO_2] and drought by direct multifactor manipulations (Johnsen 1993; Centritto et al. 1999a), comparing drought-induced changes in plant water relations at natural CO_2 springs (Tognetti et al. 1999, 2000a) or inferring CO_2-drought interactions by observing seasonal patterns of response (Ellsworth 1999; Tognetti et al. 2000b; Domec et al. 2009; McCarthy et al. 2010). These studies attempted to interpret results in the context of the potential ameliorating effects that elevated [CO_2] may have on the drought response of trees. The most direct impact is the reduction in transpiration caused by lower stomatal conductance commonly found under elevated [CO_2], which may ameliorate drought tolerance by increasing leaf or whole-plant water-use efficiency, thus enabling plants to better exploit water-limited environments. Increased allocation of carbon to root growth and osmotic adjustment in plants exposed to elevated [CO_2] may, for example, alleviate the negative impacts of water stress by improving the capacity to extract soil water. Elevated [CO_2] may also influence water relations and plant responses to drought by altering developmental processes, including root and shoot architecture (Miao et al. 1992) and leaf morphology (Thomas and Harvey 1983). However, because elevated [CO_2] often increases leaf area index and its negative effect on g_s tend to be reduced under water stress (Centritto et al. 1999c; Tognetti et al. 1999; Centritto et al. 2002; Gunderson et al. 2002; Domec et al. 2009), the benefits of CO_2-improved conservation of soil water in terms of maintaining growth or carbon gain during drought appear relatively minor (Wullschleger et al. 2002). McCarthy et al. (2010), by re-assessing 10 years of data from the Duke FACE experiments, found that the amelioration of drought effects by increased [CO_2] was observed only in the presence of very high nitrogen availability.

Rising temperature will have contrasting influences on A and g_s with respect to elevated [CO_2], and may also affect respiration. Rising temperature will increase the solubility of O_2 and especially the specificity of Rubisco for O_2, relative to CO_2, and this will decrease the RuBP-saturated and the RuBP-limited rates of carboxylation, favouring oxygenation and thus increasing the ratio of photorespiration to A (Jordan and Ogren 1984). However, because carboxylation by Rubisco will be favoured in elevated [CO_2], the depression of the rate of oxygenation relative to carboxylation by elevated [CO_2] will produce an upward shift in the temperature optimum of A (Long and Drake 1992). Moreover, Ehleringer and Björkman (1977) have shown that the maximum quantum yield (φ) of C_3 species decreases with increase in temperature, since increasing amounts of the NADPH and ATP produced by electron transport are diverted into photorespiration. By decreasing photorespiration elevated [CO_2] will reduce the decline in φ at all temperatures (Ehleringer and Björkman 1977). Consequently, the compensation photon flux density of A is also depressed at all temperatures by elevated [CO_2], and, as for A and φ, the effect will be largest at higher temperatures (Long and Drake 1992). Eventually, the impact of elevated temperature on A is dependent on whether temperature will increase beyond the thermal optimum of photosynthesis (Long and Drake 1992). Respiration, like photosynthesis, follows a general temperature-response curve: increases exponentially with temperature in its low range, reaches a maximum at an optimal temperature, and then declines. In short-term studies, respiration is usually stimulated by rising temperature. However, long-term studies show that respiration acclimates to growth temperature (Atkin et al. 2005), resulting generally in a respiration to photosynthesis ratio remarkably insensitive to rising temperature (Gifford 1995; Arnone and Körner 1997).

The effect of lower g_s on transpiration under elevated [CO_2] may be partially offset by a rise in canopy temperature, actual transpiration and hence canopy vapour pressure. This effect may be offset by an increase in leaf area and hence absorption of radiation. Another direct impact of climate change on water use is the increase in transpiration caused by larger canopy–atmosphere vapour pressure gradients that develop under rising air temperatures. The increase in these gradients is believed to be only partially offset by rising atmospheric humidity caused by more rapid evapotranspiration. Thus, regional soil drying is a projected consequence of some climate change scenarios unless accompanied by substantial increases in precipitation. There is much uncertainty about the net effects of elevated [CO_2] and temperature on stand water requirements. During the day the plant is under a heavy energy load consisting mainly of the incident solar radiation and ambient air temperature. While some of this energy is important for photosynthesis, most of it is not utilized and must be dissipated. It is partly dissipated by thermal radiation emission and sensible heat transfer to the air, but most of it is dissipated by transpiration ('latent heat'). Transpiration causes leaves to cool relative to ambient temperature when the environmental energy load on the plant is high. The rate of transpiration is also directly related to the air VPD (which is negatively related to relative humidity) and wind speed.

Experimental evidence shows contrasting results with respect to tree responses to combined increases in [CO_2] and temperature. Norby et al. (1995) found that above-ground biomass of sugar maples was decreased in response to warming, whereas Teskey (1997) and Wang et al. (2003) found inconsistent responses in enhancement of photosynthesis to elevated CO_2 and temperature in pine trees. Thus, our recognition of the myriad of interactions between plant and environment, with various feedbacks, indicates that only with advanced modelling of tree processes will we be able to fully assess the influences of climate change scenarios.

Extreme events, which are likely to increase in frequency and magnitude, are predicted to have a significant impact on forests. High temperatures usually occurring in concert with drought can cause large-scale declines in productivity. In the temperate zone of Europe, the 2003 summer heat wave, with its exceptionally hot and dry spell, caused a 195 g C m^{-2} $year^{-1}$ decline in ecosystem photosynthesis and a reduction in ecosystem respiration of 77 g C m^{-2} $year^{-1}$, resulting in a net annual loss of 0.5 Pg of carbon across the continent (Ciais et al. 2005; Bréda et al. 2006), roughly corresponding to 4 years of net ecosystem carbon storage, and increased forest mortality (Bigler et al. 2006). It has been suggested that such a crash of about 30% in gross primary productivity over Europe was unprecedented during the last century. Battisti and Naylor (2009) recently pointed out that in temperate regions extreme seasonal heat, such as that during the 2003 heat wave in central Europe, could become the median seasonal temperature in many locations by the end of the twenty-first century. Thus, climate-change driven frequency of droughts, or climatic variability (IPCC 2007), can lead to vegetation failure (Swetnam and Betancourt 1998; Martínez-Vilalta et al. 2002). Warmer air temperature will exacerbate the impact of drought on plant water loss by elevating the VPD of the atmosphere, thereby increasing the atmospheric demand for transpiration. Altered net radiation associated with climate change could also increase transpiration (see discussion below). Given the potential risks of climate-induced forest dieback, increased management attention to adaptation options for enhancing forest resistance and resilience to projected climate stress can be expected (Allen et al. 2010).

Influences on Hydraulic Structure of Trees

Three main components affect the forest water system: *stock*, *flow*, and *service*. Each of these is influenced by the intensity and duration of water stress. The water-availability (*stock*) mechanism indicates that drought drives changes in the amount of water held in the soil. This flow of water (and nutrients) is processed through the root system to become the flow of transpired water. Trees function within a physical system consisting of the soil-plant-atmosphere continuum. Tree water deficit develops as the demand exceeds the amount of water available in the soil to the depth of the root system. The demand for water is set by potential evapotranspiration, which influences both plant transpiration and soil evaporation. The energy for transpiration is provided mainly by solar radiation. Adjustments in water supply and demand are influenced over decades in response to climate, plant size, edaphic properties such as soil texture and depth, and stand density (Mencuccini 2003; McDowell et al. 2006). Transpiration from canopy surfaces, as the cohesion-tension theory states (Dixon and Joly 1894), pulls water from soil to leaves, and causes a variable gradient of water potential (Ψ) within the plant. Thus, according to the Fick's first law of diffusion, water movement is a passive process occurring along a complex network of fine capillaries (vessels and tracheids), forming the xylem conducting system. Water flow through stems specifically conforms to Darcy's law, where volume flow rate (Q) is a function of the hydraulic conductance (k) and the pressure difference between the ends of the flow path ($\Delta\Psi$):

$$Q = k \cdot \Delta\Psi \tag{3.9}$$

In analysing the components of the soil-leaf continuum conductance may be distinguished from conductivity (K); k can be measured directly, or derived from the integration of K with respect to the distance (x) along the flow path. The k is thus a function of flow path length, whereas K is independent of length:

$$K = -Q / (\mathrm{d}\Psi / \mathrm{d}x) \tag{3.10}$$

Although osmotic forces contribute to water flow into the root xylem (Passioura 1988) and may influence water flow from leaf xylem to mesophyll cells (Canny 1993), longitudinal transport in mature stem xylem introduces symplastic barriers and osmotic potential does not participate in driving flow (Pickard 1981). As water transpires from the leaf, leaf Ψ is reduced. If water is available in the soil (high Ψ), water will flow into the leaf to replenish the evaporative loss with a small reduction in leaf Ψ. As soil Ψ declines, leaf Ψ must decrease further in order to create the necessary gradient differential to drive the water up from the drying soil to the transpiring leaf (Fig. 3.5a). The negative pressures (tension) that continuous columns of water through the xylem can withstand before breaking (cavitation) is critical to the ability of a plant to tolerate dry periods (Tyree and Sperry 1989).

The hydraulic-failure (*flow*) mechanism predicts that reduced soil water supply coupled with critically high evaporative demand causes xylem conduits and the

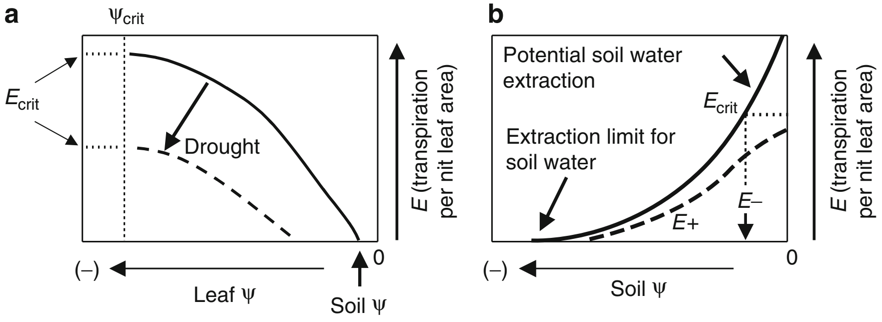


Fig. 3.5 (**a**) Draft of transpiration per unit leaf area as a function of leaf water potential (Ψ) for a plant with relatively abundant soil moisture (*solid line*) and the same plant with reduced soil moisture availability (*dashed line*). Evapotranspiration (E) exceeding critical rates (E_{crit}, maximum transpiration beyond which hydraulic failure occurs and soil-to-leaf hydraulic conductance falls to zero) results in xylem water potentials associated with hydraulic and symplastic failure (Ψ_{crit}). As bulk soil Ψ declines (drought), the relationship E/Ψ flattens and hydraulic limits become more severe. (**b**) Draft of transpiration per unit leaf area as a function of soil water potential. The *solid line* represents the transpiration threshold beyond which hydraulic failure occurs, and the *dashed line* represents realized transpiration, with the difference between the two lines representative of a hydraulic margin of safety. Regulation of E (E+) is necessary for the plant to stay within its hydraulic limits and fully exploit its potential for soil water uptake. Without regulation, (E-) once E surpasses E_{crit} there is no further water uptake, and the ability of the plant to extract water is compromised (extraction, – regulation). Redrawn from Sperry et al. (2002)

rhizosphere to cavitate (become air-filled), stopping the flow of water and desiccating plant tissues. The hydraulic-failure mechanism is based on the tenet that complete desiccation leads to cellular death. Hydraulic failure may be particularly likely if drought is sufficiently intense that plants run out of water before they run out of carbon. There is a complex relationship between stem-specific hydraulic conductivity and climate, related to life history (Maherali et al. 2004), which may bring to a virtual stasis in stem-specific hydraulic conductivity across sites with contrasting seasonality of rainfall. Bhaskar et al.'s (2007) global meta-analysis revealed that in deciduous angiosperms, evolution of increasing stem-specific hydraulic conductivity was correlated with decreasing precipitation, but water availability did not explain variation in stem-specific hydraulic conductivity in evergreen angiosperms or conifers. The evolution of stem-specific hydraulic conductivity within the evergreen angiosperms emerges as unrelated with climate parameters, including atmospheric demand and temperature. Under contrasting humidity conditions, intraspecific comparisons have found higher leaf-specific hydraulic conductivity in tree populations experiencing higher VPD (Maherali and De Lucia 2001). Higher leaf-specific hydraulic conductivity may be part of a hydraulic strategy to balance leaf water supply with the high evaporative demand during the dry season, when predawn Ψ is particularly negative (Fig. 3.5b). A highly conductive soil-leaf transport pathway can prevent excessive drops in late season leaf Ψ and allow continued carbon gain under high VPD (Addington et al. 2006). Leaf-specific hydraulic conductivity is determined by the interaction between stem-specific hydraulic conductivity and the leaf to sapwood area ratio; changes in leaf-specific hydraulic conductivity can be achieved through various combinations of change in the other two traits (Maherali and DeLucia 2000):

$$Q = (k_\mathrm{s} A_\mathrm{s} / h \eta A_\mathrm{l}) \Delta \Psi \tag{3.11}$$

where k_s is saturated permeability of the conducting path, A_s is sapwood area, A_l is leaf area, h is height, η is the viscosity of water, and $\Delta\Psi$ is $\Psi_\mathrm{s} - \Psi_\mathrm{l} - h\rho g$ (soil and leaf water potentials, respectively, and the gravitational pull on a water column of height h and density ρ). The complicated interplay among these traits may explain the varying direction of correlation: in some cases higher transport efficiency (stem-specific hydraulic conductivity) may allow maintenance of larger leaf deployment (higher leaf to sapwood area ratio), but the resulting leaf-specific hydraulic conductivity is dependent on their relative changes (Bhaskar et al. 2007).

The water-demand (*service*) mechanism predicts that stomata close to prevent hydraulic failure. Plants have to down-regulate their transpiration (through stomatal action) when soil Ψ decreases to stay within the hydraulic limits of the soil-to-leaf conducting system (Bond and Kavanagh 1999; Sperry et al. 2002; Buckley 2005), which decouples the canopy from the water-demanding atmosphere by increasing the resistance for water vapour leaving the crown (Meinzer et al. 1997):

$$\Psi_\mathrm{l} - \Psi_\mathrm{s} = -cDg_\mathrm{s} / k_\mathrm{l} \tag{3.12}$$

where D is defined as the imposed water VPD at the leaf surface, c is a constant (cDg_s equals the transpiration rate), and k_l is the leaf-specific hydraulic conductivity.

While reducing g_s reduces water loss, it also reduces CO_2 diffusion from the atmosphere to the site of carboxylation, thereby constraining photosynthetic CO_2 uptake (Eq. 3.6) (Wong et al. 1979). This may eventually lead to carbon starvation as a result of continued metabolic demand for carbohydrates. The lack of carbon may be particularly detrimental if drought is not intense enough to cause hydraulic failure, but lasts longer than the amount of time that the plant can survive with its existing carbon reserves. Down regulation of transpiration due to hydraulic constraints leads to 'isohydric' behaviour, i.e. regulation of transpiration in order to prevent leaf Ψ from decreasing to levels that endanger the integrity of the hydraulic system. This type of behaviour has been shown to result in a direct relationship between canopy resistance to water vapour and VPD (e.g. Bond and Kavanagh 1999; Sperry 2000; Cohen and Naor 2002). Since branches fix and store carbon, an established branch could be autonomous (i.e., not dependent on the tree) with respect to carbohydrates.

Branches usually cannot be autonomous with respect to water because water flows from the roots via the stem and is not recycled, unless branches are insulated from factors affecting other branches or if branches are insulated from each other (Sprugel et al. 1991). Schenk et al. (2008) showed that in contrast to shrubs in humid environments which tend to be hydraulically integrated, shrubs adapted to dry environments have modular hydraulic systems, achieved anatomically at the vessel-network scale or developmentally at the whole-plant scale. Hydraulic isolation of conduits allows independent stomatal regulation and, consequently, reduces the spread of runaway embolisms by allowing embolism repair while other parts of the xylem are under tension. Thus, branch autonomy with respect to water could benefit trees. Some isolation of individual branches might prevent a localized stress from decreasing plant Ψ as a whole (or decreasing it to some critical level), so that stomata of the affected branch would close while the rest of the tree would remain functional. In extreme cases, e.g. desert plants, branch dieback is a common behaviour for plant survival because the modular hydraulic systems allow isolated plant modules to be able to survive if these have access to small pockets of water in heterogeneous soil (Schenk et al. 2008). Thus, branch autonomy is a way to deal with the danger of runaway cavitation.

Influences on Tree Growth and Water Use Efficiency

There is evidence for a strong dependence of growth on ongoing tree water relations. In contrast to dry periods, wet periods may promote susceptibility to future drought via increased growth of leaf area and reduced growth of roots and sapwood, resulting in trees that have high ratios of hydraulic demand (leaf area) to supply (root area) (McDowell et al. 2006) (Fig. 3.6a). Stem tissues (bark, phloem, xylem) serve as a water reservoir for transpiration, and short-time oscillations in the stem dimensions reflect the water status of these tissues (Zweifel et al. 2000; Deslauriers et al. 2003), although Gall et al. (2002) found that in Norway spruce reversible variations of stem diameters did not reflect changes in internal water relations. Adaptations to water stress occur at several different temporal scales. In the short-term, water loss and leaf water status is controlled by g_s, keeping Ψ within the hydraulic limits necessary to maintain water transport through xylem (Tyree and Ewers 1991). Medium-term responses to water supply include osmotic adjustment to maintain gradients for water movement from the soil to leaves and changes to elasticity of leaf tissue to maintain turgor at low leaf water content (White 2000). Over longer time frames, plants may change growth rates in response to water supply, or there may be preferential allocation to roots so that capacity for water transport is maintained (Whitehead et al. 1984).

Diameter growth of woody species records plant-environment interactions throughout life span. Inter- and intra-annual variability of the diameter of tree trunks may be conveniently used to detect the seasonal growth patterns of trees as a result of changing environmental conditions, especially climate variation (Tatarinov and Cermák 1999; Deslauriers et al. 2003; Zweifel et al. 2006). At high elevation and latitude, the main factor controlling the seasonal pattern of diameter growth is temperature, although during dry years growth is limited even though high summer temperatures favour high growth potential. Strongly reduced radial growth during

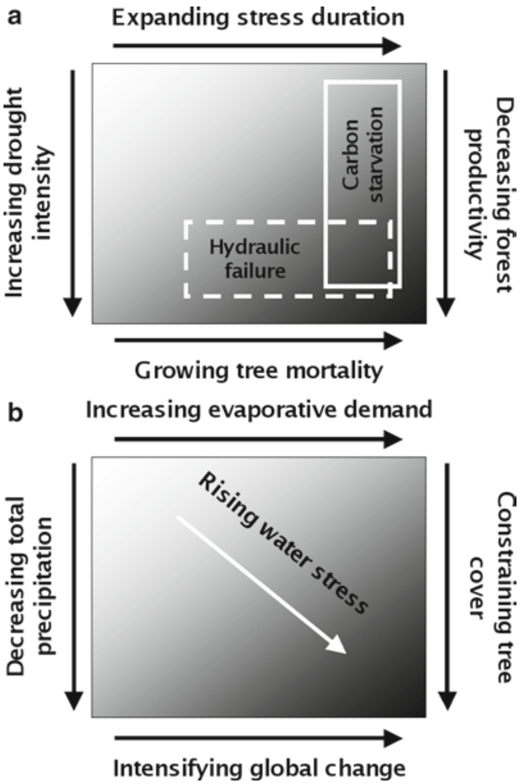


Fig. 3.6 (**a**) Theoretical relationship based on the hydraulic framework, between stress duration, drought intensity, forest productivity and tree mortality. Carbon starvation is hypothesized to occur when drought duration is long enough to curtail photosynthesis longer than the equivalent storage of carbon reserves for maintenance of metabolism. Hydraulic failure is hypothesized to occur if drought intensity is sufficient to push a plant past its threshold for irreversible desiccation before carbon starvation occurs. Inspired by McDowell et al. (2008). (**b**) Forest structure, water supply and water requirement network under varying climate change scenario

the extraordinary drought in 2003, in Europe, was registered in beech (Löw et al. 2006; Werf et al. 2007) and in oak, pine and spruce (Zweifel et al. 2006).

The hydraulic architecture of a tree shows three general qualitative properties: integration, compartmentalization and redundancy. The conducting system is built from a large number of integrated elements, tracheids and vessels, each being a unit of conduction, in communication with other elements by pits, which play a major role in protecting the conducting system from entrance of air. In all axes (trunk, branch, twig, petiole), at any given level, several xylem elements are present, like pipes in parallel, and a track of conducting elements is in close lateral contact with other tracks of vessels or tracheids. A correlation between wood density and cavitation resistance (Hacke et al. 2000) could mean that wood density is not only related to mechanical support requirements of the plant, but also to protecting the xylem pipeline from collapsing under large negative pressure. In a climate change scenario, the more drought-tolerant the plant, the more negative the xylem pressure can become without cavitation and the higher the wood density (Fig. 3.6b). Denser and stronger wood is necessary to balance the higher negative pressure within the xylem conduits (Hacke et al. 2001). Thus, although cavitation resistance is not always associated with reduced saturated xylem conductivity (Tyree et al. 1994), it would demand a price by reducing growth rate through higher xylem density (Enquist et al. 1999). The long-term structural adjustments that maintain homeostasis between water supply, water demand, and plant metabolism (Whitehead and Jarvis 1981; Katul et al. 2003; Bréda et al. 2006) may play a role in the survival or mortality of plants during drought.

Trees undergo seasonal and diurnal fluctuations in water content, as water goes into and out of storage. Water-storage capacity (or hydraulic capacitance of a plant tissue, C) is the mass of water that can be extracted, without irreversible wilting, per unit change in Ψ of the tissue:

$$C = \Delta R_{\mathrm{wc}} / \Delta \Psi \tag{3.13}$$

where R_{wc} is the relative water content of wood (stem, branches, etc.), calculated as $(W_{\mathrm{f}} - W_{\mathrm{d}})/(W_{\mathrm{s}} - W_{\mathrm{d}})$; W_{s} is the saturated weight determined after overnight hydration and blotting of excess water, W_{f} is the fresh weight, and W_{d} is the dry weight determined after oven-drying to constant weight.

The hydraulic conductance of a stem depends on stem length, transverse area of xylem, number and size distribution of xylem conduits, and extent of cavitation. In transpiring woody plants, most of the pressure drop in shoots occurs in the minor branches because leaf-specific conductance decreases as branch diameter decreases (Zimmermann 1978). When soil moisture declines, unrestrained and elevated midday transpiration rapidly leads to an exceedingly negative xylem Ψ inducing catastrophic embolism (Tyree and Sperry 1989) and reducing leaf-specific hydraulic conductivity to zero. The critical Ψ value causing full cavitation varies widely among species (Pockman et al. 1995; Pockman and Sperry 2000; Maherali et al. 2004) and is thought to be a function of interconduit pit structure (Pittermann et al. 2005), stems being more energetically costly and less vulnerable to cavitation than roots (Sperry et al. 2002). Hydraulic failure also occurs within soils and is functionally similar to xylem cavitation (McDowell et al. 2008). The hydraulic conductance of soils is a function of texture and structure, water content, hydraulic conductivity, and water table depth. Higher tension is required to pull water through fine-textured soils because of their small pore sizes, and thus fine-textured soils have lower conductance than sandy soils when water is abundant. However, fine-textured soils retain hydraulic conductance longer and at more negative Ψ than coarse-textured soils because the low conductance of fine soils results in slower water loss to transpiration and drainage (Sperry et al. 1998); besides, in coarse-textured soils hydraulic conductivity drops when water films become discontinuous (something like “soil embolism”).

Species Differences in Water Relations and Canopy Structure Across Europe

Transpiration equals approximately half of the total annual precipitation under temperate conditions in Europe (Denmead and Shaw 1962). The energetic equivalent of this amount of transpired water represents an important contribution to the energy balance of the Earth's surface. Soil drought may be a factor significantly affecting the transpiration rate via stomata and consequently the partitioning of energy in the energy budget of evaporating surfaces and, in turn, the energy exchange between vegetation and the atmosphere. Since this partitioning of energy determines the properties of the planetary boundary layer (Wilson and Baldocchi 2000), transpiration, reduced by water stress may have a significant influence on the climate (Shukla and Mintz 1982). For these reasons, research on transpiration has become important for understanding climate and climate change, especially in recent decades when the frequency of extreme weather phenomena has risen (Karl et al. 1995). VPD is an important environmental factor, which, together with low soil moisture, affects the gas exchange between vegetation and the atmosphere. A close statistical relationship exists between evaporative demand and canopy resistance for water vapour transfer to the atmosphere (Granier et al. 2000), which is related to the 'isohydric' behaviour discussed below. Consequently, evapotranspiration is not proportional to VPD (Bunce 1996).

Plants fall into two categories across the continuum of stomatal regulation of water status (Tardieu and Simonneau 1998). Isohydric plants reduce g_s as soil Ψ decreases and atmospheric conditions dry, maintaining relatively constant midday leaf Ψ regardless of drought conditions (Fig. 3.7). Isohydric behaviour has been observed in temperate hardwoods, Australasian and neotropical trees, and other species of gymnosperms (Loewenstein and Pallardy 1998a, b; Bonal and Guehl 2001; Fisher et al. 2006). Anisohydric species, by contrast, allow midday leaf Ψ to decline as soil Ψ declines with drought. Anisohydric species tend to occupy more drought-prone habitats compared with isohydric species and have xylem that is more resistant to negative Ψ (Franks et al. 2007).

Relating the hydraulic structure to the plant death is based on the premise that whole-plant hydraulic failure will be lethal. This premise may be false in cases of resprouting or xylem refilling. Resprouting has been observed following cavitation-induced shoot dieback in shrubs (Davis et al. 2002), mesic hardwoods (Tyree et al. 1993), and riparian trees (Horton et al. 2001). A benefit of reducing leaf area via shoot dieback is the resulting improvement in water status of the remaining foliage and subsequent survival of the individual (Tyree and Sperry 1989; Bréda et al. 2006). Refilling of cavitated elements may occur in some species when drought is relieved by precipitation, although the mechanisms and frequency of refilling remain a debated issue (e.g. Borghetti et al. 1991). Genetic differences could potentially play an important role in these mechanisms. The species-specific difference in regulation of the hydraulic safety margins occurs in part via differential relationships between leaf Ψ and g_s. Although isohydric species appear more vulnerable to

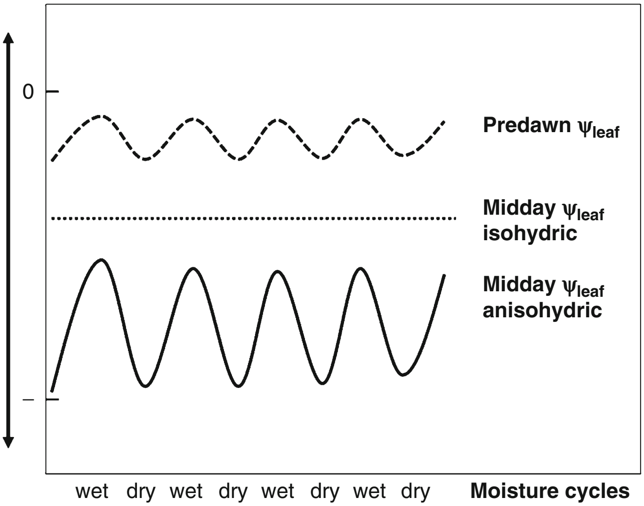


Fig. 3.7 Approximation of the two classical forms of water status control in vascular plants (isohydric and anisohydric). Midday leaf water potential (Ψ_{leaf}, *solid line*) relates to the same predawn water potential (*dashed line*). The vertical positioning of the midday water potential lines relative to each other is arbitrary. In isohydric plants, the midday Ψ_{leaf} is maintained relatively constant despite fluctuations in predawn leaf (and therefore soil) water potential. In anisohydric plants, the difference between predawn and midday Ψ_{leaf} is usually larger in drier periods because of a combination of moderate stomatal regulation of transpiration rate and the usually higher transpiration demand in drier periods. Redrawn from Franks et al. (2007)

embolism, they should actually be less likely to experience hydraulic failure because they close their stomata rather than risk cavitation. Anisohydric trees, instead, have higher rates of gas exchange during drought, but run a higher risk of cavitation as a consequence.

Leaf water availability limits plant productivity, and influences the adaptation of plants to environmental conditions. For a given size, angiosperms transport considerably larger quantities of water than conifers (Meinzer et al. 2005). The reduction in leaf-specific hydraulic conductance has been indicated amongst the mechanisms responsible for reduced growth in trees as they age and increase in height (Yoder et al. 1994), through reductions in g_s and therefore photosynthesis (Ryan and Yoder 1997). Hence, the ability to move water to the site of evaporation with a minimum investment is a major factor driving the architecture and physiology of trees, including the function of stomatal regulation. Stomatal regulation is a complex process as it depends on how microclimate, C_i, plant hormones, leaf Ψ and soil Ψ (Whitehead 1998) induce a variety of physiological responses that may regulate g_s (Dodd 2003). Besides other factors, leaf Ψ has been recognized as playing a key role in stomatal regulation (Bond and Kavanagh 1999). Studies of possible trade-offs between hydraulic conductivity and mechanical strength of wood (secondary xylem) indicate that as individual vessel water conduction is increased via larger vessel lumen area, mechanical strength of the wood may be reduced due to the reduced cross-sectional area of fibers (Wagner et al. 1998). However, other anatomical variables may confound the influence of the number and diameter of conduits. These variables include pith diameter, ray width, and fiber cell wall thickness.

The total volume (V, m^3) of a stem results from those of gaseous (V_g), liquid (V_l) and solid (V_s) spaces within the stem; $V_\mathrm{g} + V_\mathrm{l}$ represent the maximum available space for hydraulic networking (see Roderick and Berry 2001). The volumetric fraction of a stem, potentially available for the hydraulic network, can be derived from:

$$\left(V_\mathrm{g} + V_\mathrm{l}\right)/V = 1 - 0.97\left[D\right] \sim 1 - \left[D\right] \tag{3.14}$$

where [D, $\mathrm{g\ m^{-3}}$] is the basic density of woody stems, defined as the ratio of dry mass to fresh volume.

Species differences in patterns of water use and response to soil water stress are two areas of uncertainty in determining tree transpiration (Pataki and Oren 2003), because of the spatial variability in species composition. This fact complicates scaling water use in mixed stands from tree to stand level. Tree specific transpiration under water stress is strongly influenced by root formation and stomatal closure, whereas tree specific root penetration depends on soil properties (review by Rewald et al., Chapter 2, this volume). In a limited rooting zone, decreasing soil water reduces sap flow significantly, including its response to VPD. Therefore shallow rooted tree species may show large reductions of sap flow in response to VPD, while deeply rooted species show only gradual reductions in transpirations as the soil dries (Oren and Pataki 2001). In addition, topography can play a major role in drought driven mortality (Guarin and Taylor 2005).

Much literature deals with the effect of water stress on tree transpiration, but the number of studies on species differences is still too limited to draw general conclusions for relating water use to tree species attributes (Hölscher et al. 2005). Gartner et al. (2009) compared the hydrological regime of Norway spruce and birch growing on heavy soils during a pronounced drought stress period. In spite of having significantly higher transpiration rates, birch trees could more easily adapt their transpiration to soil water stress. An internal redistribution of sap flow in the xylem under drought stress may be taken as an indication of water stress (Cermák and Nadezdhina 1998). Gartner et al. (2009) also found evidence of such an internal redistribution in spruce trees under soil water stress. They conclude that the reason for the better performance of birch trees in drought stress periods is due to the exploitation of soil water reserves in deeper soil layers and in a more efficient adsorption of soil water. Burk (2006) found that broadleaved trees in general could overcome high negative water potentials much better than conifers. Coners (2001) observed root water potentials of only −0,6 MPa in spruce roots, but −1,8 MPa in beech roots and −1,2 MPa in oak roots during drought stress periods. Remarkably, in warm and dry regions at low elevation, where Norway spruce was artificially planted far beyond its natural range, the fine roots of spruce are not supported by mycorrhizal communities, which play an important role in water uptake.

Hölscher et al. (2005) studied sap flow of broad leaved tree species during a seasonal drought and found a reduction of average daily sap flux during a dry period of 44% in *Tilia cordata*, 39% in *Fagus silvatica*, 37% in *Acer pseudoplatanus*, 31% in *Carpinus betulus* compared to the sap flow in a wet period. A higher influence of soil moisture in dry periods was detected for *Fagus silvatica* than in the other species, which were more effectively controlled by the VPD. However, they argue that the relative reduction of the sap flux density does not sufficiently characterize the drought sensitivity of broad leaved trees.

For forests in Europe, Granier et al. (2007) concluded that net ecosystem exchange is reduced by soil water depletion, but to a lesser extent than gross ecosystem production, due to the compensatory effect of the decreased ecosystem respiration and the fact that coniferous species in general seem to be less affected than broadleaved species. Drought impacts not only the annual growth; it also influences the growth and vitality in the following years (Dobbertin et al. 2010). Van Mantgem et al. (2009) found increased mortality rates in old growth forests in the Western United States. They conclude that regional warming and increases in water deficits are likely contributors to that phenomenon.

Canopy Atmosphere Interactions: Forest Influences on Climate

Forest influences on evapotranspiration and other factors (e.g. albedo and carbon storage) can exacerbate or mitigate anthropogenic climate change. While tropical forests might have a role in mitigating global warming through evaporative cooling, increased atmospheric water vapour (that does not increase cloudiness) leads to a positive radiative forcing, thus, even for tropical forests the influence on climate is not straightforward. The effect of temperate forests is unclear and that of boreal forests would be weak (Bonan 2008). Tropical forests have lower albedo, higher net radiation and higher evapotranspiration, compared with pasture, producing a shallow boundary layer, thus sustaining forest transpiration in the dry season (Da Rocha et al. 2004). In boreal forests, conifers have low summer latent heat flux (evaporative fraction) compared with broadleaved deciduous trees, producing large sensible heat fluxes and a deep boundary layer (Baldocchi et al. 2000) (Fig. 3.8). Competing factors from low albedo during winter and evapotranspiration during summer influence annual mean temperature, making the net climate forcing of temperate forests highly uncertain (Bonan 2008). However, Rotenberg and Yakir (2010) and Schiller (Chapter 9, this volume) have recently demonstrated that the Yatir forest, a planted pine forest at the dry timberline (285 mm mean precipitation) at the edge of the Negev desert in southern Israel, adjusted its metabolism to reduce the impact of severe temperature and water stress. This homeostatic-like ecosystem-level behaviour resulted in a high net ecosystem CO_2 exchange to gross primary productivity ratio and in displacement of the timing of biological activity (i.e. peak of carbon uptake) to early spring, leading to net carbon uptake slightly lower than mean global pine forests and slightly larger than average European pine forests, i.e. 2.3, 2.5 and

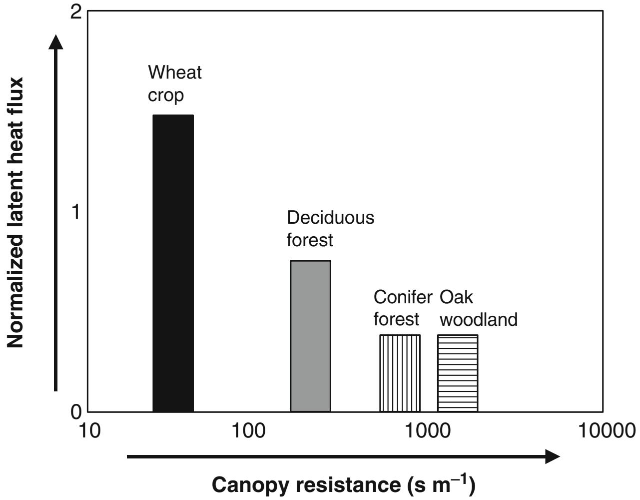


Fig. 3.8 Evapotranspiration normalized by its equilibrium rate in relation to canopy resistance for wheat crop, temperate deciduous forest, boreal conifer forest, and Mediterranean oak woodland. Latent heat exchange rates are normalized by the equilibrium *evaporation* rate ($\lambda E/\lambda E_{eq}$). Inspired by Bonan (2008)

2.0 metric tons per hectare, respectively. The substantial amount of carbon sequestered modified the surface energy balance: the plantation of the Yatir forest initially caused a regional warming because of the decreased albedo, but about 40 years after planting the balance between the albedo heating effect and the carbon sequestration driven cooling effect was reached. However, considering the positive radiative forcing caused by the observed suppression in longwave radiation, the time needed to reach a net cooling effect would be about 80 years after planting in the worst-case scenario (Rotenberg and Yakir 2010). These analyses focus on the influence of this type of forest on the energy budget of Earth's land surface, while it is clear that because of the decreased albedo this dry forest causes local warming.

The interactions between forest canopies and the atmosphere are in both directions since forests have a significant impact on the atmosphere. Recent reviews of forests, land use changes and climate targeted the influences on climate change (Betts 2007; Bonan 2008), which are important factors to consider as forest composition and extent change in the future. The basic interactions of forests with climate have been introduced into climate models (Sellers et al. 1997). Canopy atmosphere interactions can be viewed from several viewpoints, e.g. surface energy fluxes and radiative forcing (Fig. 3.9), the hydrological cycle (Fig. 3.2), and the carbon cycle (Fig. 3.1). Of course these are not independent of each other, but different viewpoints are necessary in order to focus on the relevant issues. Here we limit the discussion to aspects relevant to the water cycle and the European context, which ranges from Boreal Scandinavian forest through mid-latitude humid temperate forests and to semi-arid to arid Mediterranean, and excluding tropical forest.

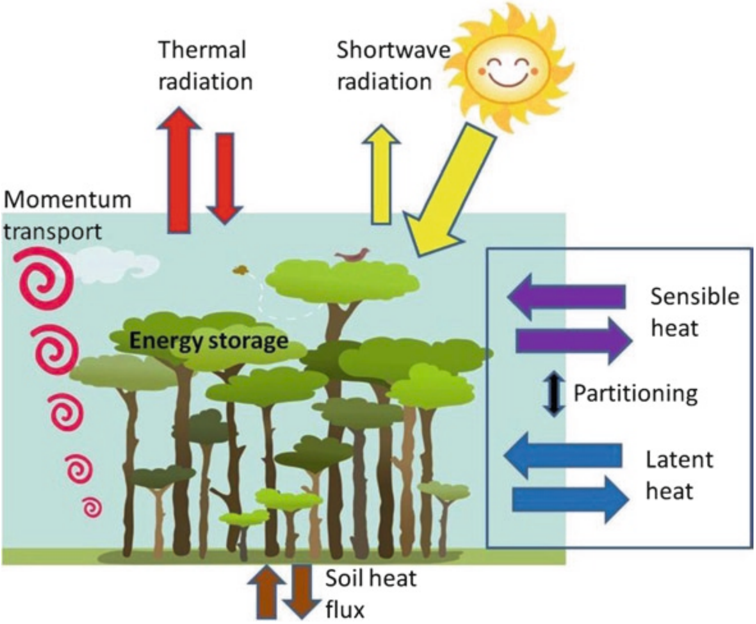


Fig. 3.9 Energy exchange of forest (inspired by Bonan 2008). Solar energy is the main input to the forest system and the fraction reflected is the albedo. A small amount of energy is stored in the forest, net thermal radiation and soil heat flux are responsible for some energy loss, while most energy is dissipated by sensible and latent (or evaporative) heat exchange with the atmosphere. Heat exchanges are facilitated by momentum transport in the surface boundary layer, which increases with wind speed and surface roughness

Climate always influences water issues through the hydrological cycle, i.e. precipitation patterns and evapotranspiration demand (Fig. 3.2). Soil drought accompanied by dry air can affect substantially the transpiration and consequently the soil water dynamics. High evaporative demands of the atmosphere can compensate partially for the reduction in transpiration rates caused by reduced soil water in the root zone (Střelcová et al. 2009). Large-scale impacts on the climate system include influences on the earth’s radiation balance through influences on albedo, aerosols and CO_2 exchange, where aerosols include dust and smoke. Another impact is that on wind speed, which changes turbulence and momentum transport.

Solar energy drives climate processes, as well as photosynthesis, powering the biosphere. Net solar radiation depends on the albedo, or the ratio of reflected radiation to that reaching the earth (i.e. the solar constant). Albedo depends mostly on cloudiness, but also on land use. Planetary albedo has been relatively constant in recent decades, but a change of 1% can have a large impact on climate (Raval and Ramanathan 1989; Wielicki et al. 2005; Ramanathan 2008). Forests in general have lower surface albedo, due to the dark colour of leaves and trapping of radiation in the canopy; the latter explaining their lower albedo even when snow covered. This leads to increased temperatures relative to non-forested landscape. Values of albedo for different forest types and other land uses are presented in Table 3.1.

Table 3.1 Albedo of various forests and other land use types

Land use type	Albedo[a]	Albedo (Eurasia) February, 1995[b]	Albedo (Eurasia) July, 1995[b]
Deciduous forest	0.10–0.20		
Coniferous forest	0.05–0.15		
Boreal forest with snow	0.12–0.30		
Rain forest	0.12		
Grass (July, August)	0.25		
Lawns	0.18–0.23		
Not cultivated fields	0.26		
Fresh snow	0.80–0.90		
Old snow	0.45–0.70		
Free water surface and solar elevation > 45°	0.05		
Wet dark soil	0.08		
Dry dark soil	0.13		
Dry sand	0.35		
Evergreen needleleaf forest		0.30	0.14
Evergreen broadleaf forest		0.14	0.15
Deciduous needleleaf forest		0.23	0.14
Deciduous broadleaf forest		0.18	0.18
Mixed forest		0.20	0.15
Closed shrublands		0.27	0.23
Open shrublands		0.27	0.27
Woody savannas		0.20	0.17
Croplands		0.18	0.16
Urban and built-up		0.23	0.18

[a] Textbook values; Donatelli et al. (2005).

[b] AVHRR satellite measurements; from Strugnell et al. (2001).

Forests are very efficient radiators of long wave radiation and their emissivity, 0.96–0.98, is slightly higher than that of other land uses, which also improves their dissipation of excess absorbed energy (Jin and Liang 2006). Forests enhance turbulence and reduce wind speed near the ground. This enhances the dissipation of energy through convection (sensible heat transfer to the atmosphere), mixing in the lower atmosphere, but reduces wind erosion and dust accumulation in the atmosphere. In addition, dust and other aerosols are deposited in the forest, resulting in "cleaner" air.

Natural and anthropogenic aerosols are today recognized as playing a major role in radiative forcing both directly through reducing atmospheric transmissivity and indirectly through their role as cloud condensation nuclei (e.g. Cohen 2009). Some aerosols change cloud frequency and properties. The forest's role in reducing atmospheric aerosol load is therefore important. However, forest fires add large amounts of smoke to the atmosphere having the opposite effect. Even so, smoke has a short-term impact on climate when compared to the long-term reductions of aerosols by forests.

Two more general features of forests are their large standing carbon pool, which if released would significantly increase atmospheric CO_2 concentrations, their large uptake of CO_2 (equivalent to about a third of anthropogenic carbon emissions) and that they are relatively efficient in removing soil moisture.

Conclusions

Increasing [CO_2] along with associated changes in temperature will most likely alter the structure and function of forest ecosystems and thus will affect their productivity and their role as stable sinks to CO_2 sequestration and as regulators of the global hydrologic cycle. However, models predict that Earth's surface temperatures will increase along with shifts in precipitation that result in greater drought severity and frequency (IPCC 2007; Seager et al. 2007). As an example, maximum summer temperatures are likely to increase more than the average in southern and central Europe, whereas increasing water stress will dramatically affect mainly south-eastern Europe. Thus, forest ecosystems will experience a combination of numerous environmental stresses, which may significantly alter their physiological feedback on regional and continental climate. However, there is a great deal of uncertainty with regard to tree responses to interactive effects of global change scenarios. Models focusing on the interactions between climate change factors might help the scientific community to fill in the gaps in knowledge of how forest trees will respond to interacting effects. However, model accuracy depends to a large extent on our understanding of forest responses to climate changes. We conclude that there is an urgent need for multifactor climate change experimental studies examining the kinetic sensitivity of photosynthesis, stomatal conductance-transpiration, and respiration to the interactive effects of rising temperature, elevated [CO_2], and environmental stress, in order to improve our ability to predict the physiological forcing of forest ecosystems on climate change.

References

Addington RN, Donovan LA, Mitchell RJ, Vose JM, Pecot SD, Jack SB, Hacke UG, Sperry JS, Oren R (2006) Adjustments in hydraulic architecture of *Pinus Palustris* maintain similar stomatal conductance in xeric and mesic habitats. Plant Cell Environ 29:535–545

Ainsworth EA, Long SP (2005) What have we learned from 15 years of free-air CO_2 enrichment (FACE)? A meta-analytic review of the responses of photosynthesis, canopy properties and plant production to rising CO_2. New Phytol 165:351–372

Ainsworth EA, Rogers A (2007) The response of photosynthesis and stomatal conductance to rising [CO_2]: mechanisms and environmental interactions. Plant Cell Environ 30:258–270

Allen CD, Breshears DD (2007) Climate-induced forest dieback as an emergent global phenomenon. Eos Trans Am Geophys Union 88:504–505

Allen RG, Pereira LS, Raes D, Smith M (1998) Crop evapotranspiration. FAO Irrig. And Drain, Paper, 56, Rome

Allen CD, Macalady AK, Chenchouni H, Bachelet D, McDowell N, Vennetier M, Kitzberger T, Rigling A, Breshears DD, Hogg EH (Ted), Gonzalez P, Fensham R, Zhang Z, Castro J, Demidova N, Lim J-H, Allard G, Running SW, Semerci A, Cobb N (2010) Drought-induced forest mortality: a global overview reveals emerging climate change risks. For Ecol Manag 259:660–684

Arnone JA III, Körner C (1997) Temperature adaptation and acclimation potential of leaf dark respiration in two species of *Ranunculus* from warm and cold habitats. Arct Alp Res 29:122–125

Atkin OK, Bruhn D, Hurry VM, Tjoelker MG (2005) The hot and the cold: unravelling the variable response of plant respiration to temperature. Funct Plant Biol 32:87–105

Baldocchi D, Finnigan JJ, Wilson K, Paw KT, Falge E (2000) On measuring net ecosystem carbon exchange over tall vegetation on complex terrain. Boundary-Layer Meteorol 96:257–291

Battisti DS, Naylor RL (2009) Historical warnings of future food insecurity with unprecedented seasonal heat. Science 323:240–244

Bernacchi CJ, Calfapietra C, Davey PA, Wittig VE, Scarascia-Mugnozza GE, Raines CA, Long SP (2003) Photosynthesis and stomatal conductance responses of poplars to free-air CO_2 enrichment (PopFACE) during the first growth cycle and immediately following coppice. New Phytol 159:609–621

Betts RA (2007) Implications of land ecosystem-atmosphere interactions for strategies for climate change adaptation and mitigation. Tellus 59B:602–615

Betts RA, Boucher O, Collins M, Cox PM, Falloon PD, Gedney N, Hemming DL, Huntingford C, Jones CD, Sexton DMH, Webb MJ (2007) Projected increase in continental runoff due to plant responses to increasing carbon dioxide. Nature 448:1037–1041

Bhaskar R, Valiente-Banuet A, Ackerly DD (2007) Evolution of hydraulic traits in closely related species pairs from mediterranean and nonmediterranean environments of North America. New Phytol 176:718–726

Bigler C, Braker OU, Bugmann H, Dobbertin M, Rigling A (2006) Drought as an inciting mortality factor in Scots pine stands of the Valais, Switzerland. Ecosystems 9:330–343

Blackman VH (1919) The compound interest law and plant growth. Ann Bot 33:353–360

Bonal D, Guehl J (2001) Contrasting patterns of leaf water potential and gas exchange responses to drought in seedlings of tropical rainforest species. Funct Ecol 15:490–496

Bonan GB (2008) Forests and climate change: Forcings, feedbacks, and the climate benefits of forests. Science 320:1444–1449

Bond BJ, Kavanagh KL (1999) Stomatal behaviour of four woody species in relation to leaf-specific hydraulic conductance and threshold water potential. Tree Physiol 19:503–510

Borghetti M, Edwards WRN, Grace J, Jarvis PG, Raschi A (1991) The refilling of embolized xylem in *Pinus sylvestris* L. Plant Cell Environ 14:357–369

Bréda N, Granier A, Aussenac A (1995) Effects of thinning on soil and tree water relations, transpiration and growth in an oak forest (*Quercus petraea* (Matt.) Liebl.). Tree Physiol 15:295–306

Bréda N, Huc R, Granier A, Dreyer E (2006) Temperate forest trees and stands under severe drought: a review of ecophysiological responses, adaptation processes and long-term consequences. Ann For Sci 63:625–644

Breshears DD, Cobb NS, Rich PM, Price KP, Allen CD, Balice RG, Romme WH, Kastens JH, Floyd ML, Belnap J, Anderson JJ, Myers OB, Meyer CW (2005) Regional vegetation die-off in response to global-change type drought. Proc Natl Acad Sci USA 102:15144–15148

Buckley TN (2005) The control of stomata by water balance. New Phytol 168:275–291

Bunce JA (1996) Does transpiration control stomatal responses to water vapour pressure deficit? Plant Cell Environ 19:131–135

Burk D (2006) Physiologische, anatomische und chemische Aspekte der Regulation der Wasseraufnahme bei Rotbuche, Kiefer und Birke auf unterschiedlich wasserversorgten Standorten. Dissertation Georg August Universität Göttingen

Canny MJ (1993) The transpiration stream in the leaf apoplast: water and solutes. Philos Trans R Soc London B 341:87–100

Centritto M, Lee HSJ, Jarvis PG (1999a) Interactive effects of elevated [CO_2] and drought on cherry (*Prunus avium*) seedlings. I. Growth, whole-plant water use efficiency and water loss. New Phytol 141:129–140

Centritto M, Lee HSJ, Jarvis PG (1999b) Increased growth in elevated CO_2: an early, short-term response? Global Change Biol 5:623–633

Centritto M, Magnani F, Lee HSJ, Jarvis PG (1999c) Interactive effects of elevated $[CO_2]$ and drought on cherry (*Prunus avium*) seedlings. II. Photosynthetic capacity and water relations. New Phytol 141:141–153

Centritto M, Lucas ME, Jarvis PG (2002) Gas exchange, biomass, whole-plant water-use efficiency and water uptake of peach (*Prunus persica*) seedlings in response to elevated carbon dioxide concentration and water availability. Tree Physiol 22:699–706

Centritto M, Loreto F, Chartzoulakis K (2003) The use of low $[CO_2]$ to estimate diffusional and non-diffusional limitations of photosynthetic capacity of salt-stressed olive saplings. Plant Cell Environ 26:585–594

Centritto M, Nascetti P, Petrilli L, Raschi A, Loreto F (2004) Profiles of isoprene emission and photosynthetic parameters in hybrid poplars exposed to free-air CO_2 enrichment. Plant Cell Environ 27:403–412

Cermák J, Nadezhdina N (1998) Sapwood as the scaling parameter – defining according to xylem water content or radial pattern of sap flow? Ann For Sci 55:509–521

Ciais P, Reichstein M, Viovy N, Granier A, Ogee J, Allard V, Aubinet M, Buchmann N, Bernhofer C, Carrara A, Chevallier F, De Noblet N, Friend AD, Friedlingstein P, Grunwald T, Heinesch B, Keronen P, Knohl A, Krinner G, Loustau D, Manca G, Matteucci G, Miglietta F, Ourcival JM, Papale D, Pilegaard K, Rambal S, Seufert G, Soussana JF, Sanz MJ, Schulze ED, Vesala T, Valentini R (2005) Europe-wide reduction in primary productivity caused by the heat and drought in 2003. Nature 437:529–533

Cohen S (2009) The role of widespread surface solar radiation trends in climate change: dimming and brightening (pp 21–41). In: Letcher T (ed) Climate change: observed impacts on planet earth. Elsevier, New York, 492 pages

Cohen S, Naor A (2002) The effect of three rootstocks on water use, canopy conductance and hydraulic parameters of apple trees and predicting canopy from hydraulic conductance. Plant Cell Environ 25:17–28

Condit R, Hubbell SP, Foster RB (1995) Mortality rates of 205 neotropical tree and shrub species and the impact of a severe drought. Ecol Monogr 65:419–439

Coners H (2001) Wasseraufnahme und artspezifische hydraulische Eigenschaften von Buche, Eiche und Fichte. In situ Messungen an Altbäumen. Dissertation Georg August Universität Göttingen

Crous KY, Walters MB, Ellsworth DS (2008) Elevated CO_2 concentration affects leaf photosynthesis-nitrogen relationships in *Pinus taeda* over nine years in FACE. Tree Physiol 28:607–614

Curtis PS, Wang XZ (1998) A meta-analysis of elevated CO_2 effects on woody plant mass, form, and physiology. Oecologia 113:299–313

Da Rocha HR, Goulden ML, Miller SD, Menton MC, Pinto LDVO, de Freitas HC, Figueira AMES (2004) Seasonality of water and heat fluxes over a tropical forest in eastern Amazonia. Ecol Appl 14:22–32

Davey PA, Hunt S, Hymus GJ, DeLucia EH, Drake BG, Karnosky DF, Long SP (2004) Respiratory oxygen uptake is not decreased by an instantaneous elevation of $[CO_2]$, but is increased with long-term growth in the field at elevated $[CO_2]$. Plant Physiol 134:520–527

Davis SD, Ewers FW, Sperry JS, Portwood KA, Crocker MC, Adams GC (2002) Shoot dieback during prolonged drought in *Ceanothus* (*Rhamnanceae*) Chaparral of California: a possible case of hydraulic failure. Am J Bot 89:820–828

Denman KL, Brasseur G, Chidthaisong A, Ciais P, Cox PM, Dickinson RE, Hauglustaine D, Heinze C, Holland E, Jacob D, Lohmann U, Ramachandran S, da Silva Dias PL, Wofsy SC, Zhang X (2007) Couplings between changes in the climate system and biogeochemistry. In: Solomon S, Qin D, Manning M, Chen Z, Marquis M, Averyt KB, Tignor M, Miller HL (eds) Climate change 2007: the physical science basis. Contribution of Working Group I to the Fourth Assessment Report of the Intergovernmental Panel on Climate Change. Cambridge University Press, Cambridge/New York, pp 499–587

Denmead OT, Shaw RH (1962) Availability of soil water to plants as affected by soil moisture content and meteorological conditions. Agron J 54:385–390

Deslauriers A, Morin H, Begin Y (2003) Cellular phenology of annual ring formation of *Abies balsamea* in the Quebec boreal forest (Canada). Can J For Res 33:190–200

Dixon HH, Joly J (1894) On the ascent of sap. Philos Trans R Soc London Biol Sci 186:563–576

Dobbertin M, Eilmann B, Bleuler P, Giuggiola A, Graf Pannatier E, Landolt W, Schleppi P, Rigling A (2010) Effect of irrigation on needle morphology, shoot and stem growth in a drought-exposed *Pinus sylvestris* forest. Tree Physiol 30:346–360

Dodd IC (2003) Hormonal interactions and stomatal responses. J Plant Growth Reg 22:32–46

Domec J-C, Palmroth S, Ward E, Maier CA, Thérézien M, Oren R (2009) Acclimation of leaf hydraulic conductance and stomatal conductance of *Pinus taeda* (loblolly pine) to long-term growth in elevated CO_2 (free-air CO_2 enrichment) and N-fertilization. Plant Cell Environ 32:1500–1512

Donatelli M, Carlini L, Bellocchi G (2005) GSRad, Global Solar Radiation estimates. Agricultural Research Council, ISCI, Italy. http://www.apesimulator.it/help/models/solarradiation/

Ehleringer J, Bjorkman O (1977) Quantum yields for CO_2 uptake in C_3 and C_4 plants. Plant Physiol 59:86–90

Eichelmann H, Oja V, Rasulov B, Padu E, Bichele I, Pettai H, Möls T, Kasparova I, Vapaavuori E, Laisk A (2004) Photosynthetic parameters of birch (*Betula pendula* Roth) leaves growing in normal and CO_2- and O_3-enriched atmospheres. Plant Cell Environ 27:479–495

Ellsworth DS (1999) CO_2 enrichment in a maturing pine forest: are CO_2 exchange and water status in the canopy affected? Plant Cell Environ 22:461–472

Enquist BJ, West GB, Charnov EL, Brown JH (1999) Allometric scaling of production and life-history variation in vascular plants. Nature 401:907–911

Farquhar GD, von Caemmerer S, Berry JA (1980) A biochemical model of photosynthetic CO_2 assimilation in leaves of C3 species. Planta 149:78–90

Fisher RA, Williams M, Do Vale RL, Da Costa AL, Meir P (2006) Evidence from Amazonian forests is consistent with isohydric control of leaf water potential. Plant Cell Environ 29:151–165

Franks PJ, Drake PL, Froend RH (2007) Anisohydric but isohydrodynamic: seasonally constant plant water potential gradient explained by a stomatal control mechanism incorporating variable plant hydraulic conductance. Plant Cell Environ 30:19–30

Gall R, Landolt W, Schleppi P, Michellod V, Bucher JB (2002) Water content and bark thickness of Norway spruce (*Picea abies*) stems: phloem water capacitance and xylem sap flow. Tree Physiol 22:613–623

Gartner K, Nadezdhina N, Englisch E, Cermák J, Leitgeb E (2009) Sap flow of birch and Norway spruce during the European heat and drought in summer 2003. For Ecol Manag 258:590–599

Gifford RM (1995) Whole plant respiration and photosynthesis of wheat under increased CO_2 concentration and temperature: long-term vs short-term distinctions for modeling. Global Change Biol 1:249–263

Gitlin AR, Sthultz CM, Bowker MA, Stumpf S, Paxton KL, Kennedy K, Munoz A, Bailey JA, Whitham TG (2006) Mortality gradients within and among dominant plant populations as barometers of ecosystem change during extreme drought. Conserv Biol 20:1477–1486

Granier A, Loustau D, Bréda N (2000) A generic model of forest canopy conductance dependent on climate, soil water availability and leaf area index. Ann For Sci 57:755–765

Granier A, Reichstein M, Bréda N, Janssens IA, Falge E, Ciais P, Grünwald T, Aubinet M, Berbigier P, Bernhofer C, Buchmann N, Facini O, Grassi G, Heinesch B, Ilvesniemi H, Keronen P, Knohl A, Köstner B, Lagergren F, Lindroth A, Longdoz B, Loustau D, Mateus J, Montagnani L, Nys C, Moors E, Papale D, Pfeiffer M, Pilegaard K, Pita G, Pumpanen J, Rambal S, Rebmann C, Rodrigues A, Seufert G, Tenhunen J, Vesala T, Wang Q (2007) Evidence for soil water control on carbon and water dynamics in European forests during the extremely dry year 2003. Agr For Meteorol 143:123–145

Guarin A, Taylor AH (2005) Drought triggered mortality in mixed conifer forests in Yosemite National Park, California, USA. For Ecol Manag 218:229–244

Gunderson CA, Sholtis JD, Wullschleger SD, Tissue DT, Hanson PJ, Norby RJ (2002) Environmental and stomatal control of photosynthetic enhancement in the canopy of a sweetgum (*Liquidambar styraciflua* L.) plantation during 3 years of CO_2 enrichment. Plant Cell Environ 25:379–393

Hacke UG, Sperry JS, Ewers BE, Ellsworth DS, Schafer KVR, Oren R (2000) Influence of soil porosity on water use in *Pinus taeda*. Oecologia 124:495–505

Hacke UG, Sperry JS, Pockman WT, Davis SD, McCulloch KA (2001) Trends in wood density and structure are linked to prevention of xylem implosion by negative pressure. Oecologia 126:457–461

Hölscher D, Koch O, Korn S, Leuschner Ch (2005) Sap flux of five co-occurring tree species in a temperate broad-leaved forest during seasonal drought. Trees 19:628–637

Horton JL, Kolb TE, Hart SC (2001) Physiological response to groundwater depth varies among species and with river flow regulation. Ecol Appl 11:1046–1059

Hu H, Boisson-Dernier A, Israelsson-Nordström M, Böhmer M, Xue S, Ries A, Godoski J, Kuhn JM, Schroeder JI (2010) Carbonic anhydrases are upstream regulators of CO_2-controlled stomatal movements in guard cells. Nat Cell Biol 12:87–93

IPCC (2007) Climate change 2007: the physical science basis. In: Solomon S, Qin D, Manning M, Chen Z, Marquis M, Averyt KB, Tignor M, Miller HL (eds) Contribution of Working Group I to the Fourth Assessment Report of the Intergovernmental Panel on Climate Change. Cambridge University Press, Cambridge, UK/New York, USA

Jentsch A, Kreyling J, Beierkuhnlein C (2007) A new generation of climate-change experiments: events, not trends. Front Ecol Environ 5:365–374

Jin M, Liang S (2006) An improved land surface emissivity parameter for land surface models using global remote sensing observations. J Climatol 19:2867–2881

Johnsen KH (1993) Growth and ecophysiological responses of black spruce seedlings to elevated CO_2 under varied water and nutrient additions. Can J For Res 23:1133–1142

Jordan DB, Ogren WL (1984) The CO2/O2 specificity of ribulose 1, 5-bisphosphate carboxylase/oxygenase. Planta 161:308–313

Karl TR, Knight RW, Plummer N (1995) Trends in high-frequency climate variability in the twentieth century. Nature 377:217–220

Katul G, Leuning R, Oren R (2003) Relationship between plant hydraulic and biochemical properties derived from a steady-state coupled water and carbon transport model. Plant Cell Environ 26:339–350

Keane RE, Austin M, Field C, Huth A, Lexer MJ, Peters D, Solomon A, Wyckoff P (2001) Tree mortality in gap models: application to climate change. Clim Change 51:509–540

Körner Ch (2006) Plant CO_2 responses: an issue of definition, time and resource supply. New Phytol 172:393–411

Körner Ch, Asshoff R, Bignucolo O, Hättenschwiler S, Keel SG, Pelaez-Riedl S, Pepin S, Siegwolf RTW, Zotz G (2005) Carbon flux and growth in mature deciduous forest trees exposed to elevated CO_2. Science 309:1360–1362

Leitgeb E, Gartner K, Nadezdina N, Englisch M, Cermák J (2002) Ecological effects of pioneer species on soil moisture regime in an early successional stage following wind-throw in a spruce stand. Proceedings of the IUFRO Conference on Restoration of Boreal and Temperate Forests, Gardiner ES, Breland LJ [Comp.] Reports/Skov & Landskab (11), pp 193–194

Loewenstein NJ, Pallardy SG (1998a) Drought tolerance, xylem sap abscisic acid and stomatal conductance during soil drying: a comparison of young plants of four temperate deciduous angiosperms. Tree Physiol 18:421–430

Loewenstein NJ, Pallardy SG (1998b) Drought tolerance, xylem sap abscisic acid and stomatal conductance during soil drying: a comparison of canopy trees of three temperate deciduous angiosperms. Tree Physiol 18:431–440

Long SP, Drake BG (1992) Photosynthetic CO_2 assimilation and rising atmospheric CO_2 concentrations. In: Baker NR, Thomas H (eds) Crop photosynthesis: spatial and temporal determinants. Elsevier, Amsterdam, pp 69–103

Loreto F, Centritto M (2008) Leaf carbon assimilation in a water-limited world. Plant Biosyst 142:154–161

Löw M, Herbinger K, Nunn AJ, Häberle K-H, Leuchner M, Heerdt C, Werner H, Wipfler P, Pretzsch H, Tausz M, Matyssek R (2006) Extraordinary drought of 2003 overrules ozone impact on adult beech trees (*Fagus sylvatica*). Trees 20:539–548

Maherali H, DeLucia EH (2000) Xylem conductivity and vulnerability to cavitation of ponderosa pine growing in contrasting climates. Tree Physiol 20:859–867

Maherali H, DeLucia EH (2001) Influence of climate-driven shifts in biomass allocation on water transport and storage in ponderosa pine. Oecologia 129:481–491

Maherali H, Pockman WT, Jackson RB (2004) Adaptive variation in the vulnerability of woody plants to xylem cavitation. Ecology 85:2184–2199

Martínez-Vilalta J, Piñol J, Beven K (2002) A hydraulic model to predict drought-induced mortality in woody plants: an application to climate change in the Mediterranean. Ecol Model 155:127–147

McCarthy HR, Oren R, Johnsen KH, Gallet-Budynek A, Pritchard SG, Cook CW, LaDeau SL, Jackson RB, Finzi AC (2010) Re-assessment of plant carbon dynamics at the Duke free-air CO_2 enrichment site: interactions of atmospheric [CO_2] with nitrogen and water availability over stand development. New Phytol 185:514–528

McDowell NG, Adams HA, Bailey JD, Hess M, Kolb TE (2006) Homeostatic maintenance of ponderosa pine gas exchange in response to stand density changes. Ecol Appl 16:1164–1182

McDowell N, Pockman WT, Allen CD, Breshears DD, Cobb N, Kolb T, Plaut J, Sperry J, West A, Williams DG, Yepez EA (2008) Mechanisms of plant survival and mortality during drought: why do some plants survive while others succumb to drought? New Phytol 178:719–739

Meinzer FC, Andrade JL, Goldstein G, Holbrook NM, Cavelier J, Jackson P (1997) Control of transpiration from the upper canopy of a tropical forest: the role of stomatal, boundary layer and hydraulic architecture components. Plant Cell Environ 20:1242–1252

Meinzer FC, Bond BJ, Warren JM, Woodruff DR (2005) Does water transport scale universally with tree size? Funct Ecol 19:558–565

Mencuccini M (2003) The ecological significance of long-distance water transport: short-term regulation, long-term acclimation and the hydraulic costs of stature across plant life forms. Plant Cell Environ 26:163–182

Menzel A, Fabian P (1999) Growing season extended in Europe. Nature 397:659

Miao SL, Wayne PM, Bazzaz FA (1992) Elevated CO_2 differentially alters the responses of co-occurring birch and maple seedlings to a moisture gradient. Oecologia 90:300–304

Mott KA (1988) Do stomata respond to CO_2 concentrations other than intercellular? Plant Physiol 86:200–203

Nadezhdina N (1999) Sapflow as an indicator of plant water stress. Tree Physiol 19:885–891

Niinemets U, Díaz-Espejo A, Flexas J, Galmés J, Warren CR (2009) Role of mesophyll diffusion conductance in constraining potential photosynthetic productivity in the field. J Exp Bot 60:2249–2270

Norby RJ, Gunderson CA, Edwards NT, Wullschleger SD, O'Neill EG (1995) TACIT: temperature and CO_2 interactions in trees. Photosynthesis and growth. Ecol Soc Am Bull 76(Suppl):197

Norby RJ, DeLucia EH, Gielen B, Calfapietra C, Giardina CP, King JS, Ledford J, McCarthy HR, Moore DJP, Ceulemans R, De Angelis P, Finzi AC, Karnosky DF, Kubiske ME, Lukac M, Pregitzer KS, Scarascia-Mugnozza GE, Schlesinger WH, Oren R (2005) Forest response to elevated CO_2 is conserved across a broad range of productivity. Proc Natl Acad Sci USA 102:18052–18056

Oren R, Pataki D (2001) Transpiration in response to variation in microclimate and soil moisture in southeastern deciduous forests. Oecologia 127:549–559

Passioura JB (1988) Water transport in and to roots. Annu Rev Plant Physiol Plant Mol Biol 39:245–265

Pataki DE, Oren R (2003) Species differences in stomatal control of water loss at the canopy scale in a mature bottomland deciduous forest. Adv Water Resour 26:1267–1278

Pearson PN, Palmer MR (2000) Atmospheric carbon dioxide concentrations over the past 60 million years. Nature 406:695–699

Pickard WF (1981) The ascent of sap in plants. Prog Biophys Mol Biol 37:181–229

Pittermann J, Sperry JS, Hacke UG, Wheeler JK, Sikkema EH (2005) Torus-margo pits help conifers compete with angiosperms. Science 310:1924

Pockman WT, Sperry JS (2000) Vulnerability to cavitation and the distribution of Sonoran desert vegetation. Am J Bot 87:1287–1299

Pockman WT, Sperry JS, O'Leary JW (1995) Sustained and significant negative water pressure in xylem. Nature 378:715–716

Ramanathan V (2008) Why is the earth's albedo 29% and was it always 29%? iLEAPS Newsl 5:18–20

Raval A, Ramanathan V (1989) Observational determination of the greenhouse effect. Nature 342:758–761

Roderick ML, Berry SL (2001) Linking wood density with tree growth and environment: a theoretical analysis based on the motion of water. New Phytol 149:473–485

Rotenberg E, Yakir D (2010) Contribution of semi-arid forests to the climate system. Science 327:451–454

Ryan MG, Yoder BJ (1997) Hydraulic limits to tree height and tree growth. Bioscience 47:235–242

Schenk HJ, Espino S, Goedhart CM, Nordenstahl M, Cabrera HIM, Jones CS (2008) Hydraulic integration and shrub growth form linked across continental aridity gradients. Proc Natl Acad Sci USA 105:11248–11253

Scholze M, Knorr W, Arnell NW, Prentice IC (2006) A climate-change risk analysis for worlds ecosystems. Proc Nat Acad Sci USA 103:13116–13120

Schwalm CR, Williams CA, Schaefer K, Arneth A, Bonal D, Buchmann N, Chen J, Law BE, Lindroth A, Luyssaert S, Reichstein M, Richardson AD (2010) Assimilation exceeds respiration sensitivity to drought: A FLUXNET synthesis. Global Change Biol 16:657–670

Seager R, Ting M, Held I, Kushnir Y, Lu J, Vecchi G, Huang H-P, Harnik N, Leetmaa A, Lau N-C, Li C, Velez J, Naik N (2007) Model projections on an imminent transition to a more arid climate in southwestern North America. Science 316:1181–1184

Seiler TJ, Rasse DP, Li J, Dijkstra P, Anderson HP, Johnson DP, Powell TL, Hungate BA, Hinkle CR, Drake BG (2009) Disturbance, rainfall and contrasting species responses mediated aboveground biomass response to 11 years of CO_2 enrichment in a Florida scrub-oak ecosystem. Global Change Biol 15:356–367

Sellers PJ, Dickinson RE, Randall DA, Betts AK, Hall FG, Berry JA, Collatz GJ, Denning AS, Mooney HA, Nobre CA, Sato N, Field CB, Henderson-Sellers A (1997) Modeling the exchanges of energy, water, and carbon between continents and the atmosphere. Science 275:502–509

Shukla J, Mintz Y (1982) Influence of land-surface evapotranspiration on the earth's climate. Science 215:1498–1501

Singsaas EL, Ort D, DeLucia E (2003) Elevated CO_2 effects on mesophyll conductance and its consequences for interpreting photosynthetic physiology. Plant Cell Environ 27:41–50

Sperry JS (2000) Hydraulic constraints on plant gas exchange. Agric For Meteorol 104:13–23

Sperry JS, Adler FR, Campbell GS, Comstock JP (1998) Limitation of plant water use by rhizosphere and xylem conductance: results from a model. Plant Cell Environ 21:347–359

Sperry JS, Hacke UG, Oren R, Comstock JP (2002) Water deficits and hydraulic limits to leaf water supply. Plant Cell Environ 25:251–263

Springer CJ, DeLucia EH, Thomas RB (2005) Relationships between net photosynthesis and foliar nitrogen concentrations in a loblolly pine forest ecosystem grown in elevated atmospheric carbon dioxide. Tree Physiol 25:385–394

Sprugel DG, Hinckley TM, Schaap W (1991) The theory and practice of branch autonomy. Ann Rev Ecol Syst 22:309–334

Stitt M, Krapp A (1999) The interaction between elevated carbon dioxide and nitrogen nutrition: the physiological and molecular background. Plant Cell Environ 22:583–621

Střelcová K, Mátyás Cs, Kleidon A, Lapin M, Matejka F, Blaženec M, Škvarenina J, Holécy J (2009) Bioclimatology and natural hazards. Springer, Berlin

Strugnell NC, Lucht W, Schaaf C (2001) A global albedo data set derived from AVHRR data for use in climate simulations. Geophys Res Lett 28:191–194

Swetnam TW, Betancourt JL (1998) Mesoscale disturbance and ecological response to decadal climatic variability in the American southwest. J Climate 11:3128–3147

Tardieu F, Simonneau T (1998) Variability among species of stomatal control under fluctuating soil water status and evaporative demand: modelling isohydric and anisohydric behaviours. J Exp Bot 49:419–432

Tatarinov F, Cermák J (1999) Daily and seasonal variation of stem radius in oak. Ann For Sci 56:579–590

Teskey RO (1997) Combined effects of elevated CO_2 and air temperature on carbon assimilation of Pinus taeda trees. Plant Cell Environ 3:373–380

Thomas FM, Blank R, Hartmann G (2002) Abiotic and biotic factors and their interactions as causes of oak decline in Central Europe. Forest Pathol 32:277–307

Thomas JF, Harvey CN (1983) Leaf anatomy of four species grown under continuous CO_2 enrichment. Bot Gazette 144:303–309

Tognetti R, Longobucco A, Miglietta F, Raschi A (1998) Transpiration and stomatal behaviour of *Quercus ilex* plants during the summer in a Mediterranean carbon dioxide sprint. Plant Cell Environ 21:613–622

Tognetti R, Longobucco A, Miglietta F, Raschi A (1999) Water relations, stomatal response and transpiration of *Quercus pubescens* trees during summer in a Mediterranean carbon dioxide spring. Tree Physiol 19:261–270

Tognetti R, Minnocci A, Penuelas J, Raschi A, Jones MB (2000a) Comparative field water relations of three Mediterranean shrub species co-occurring at a natural CO_2 vent. J Exp Bot 51:1135–1146

Tognetti R, Raschi A, Jones MB (2000b) Seasonal patterns of tissue water relations in three Mediterranean shrubs co-occurring at a natural CO_2 spring. Plant Cell Environ 23:1341–1351

Tschaplinski TJ, Norby RJ, Wullschleger SD (1993) Responses of loblolly pine seedlings to elevated CO_2 and fluctuating water supply. Tree Physiol 13:283–296

Tyree MT, Ewers FW (1991) The hydraulic architecture of trees and other woody plants. New Phytol 119:345–360

Tyree MT, Sperry JS (1989) Vulnerability of xylem to cavitation and embolism. Ann Rev Plant Physiol Plant Mol Biol 40:19–38

Tyree MT, Cochard H, Cruiziat P, Sinclair B, Ameglio T (1993) Drought-induced leaf shedding in walnut: evidence for vulnerability segmentation. Plant Cell Environ 16:879–882

Tyree MT, Davis SD, Cochard H (1994) Biophysical perspectives of xylem evolution – is there a tradeoff of hydraulic efficiency for vulnerability to dysfunction? IAWA J 15:335–360

van der Werf GW, Sass-Klaassen U, Mohren GMJ (2007) The impact of the 2003 summer drought on the intra-annual growth pattern of beech (*Fagus sylvatica* L.) and oak (*Quercus robur* L.) on a dry site in the Netherlands. Dendrochronologia 25:103–112

Van Mantgem P, Stephenson NL, Byrne JC, Daniels LD, Franklin JF, Fule PZ, Harmon ME, Larson AJ, Smith JM, Taylor AH, Veblen TT (2009) Widespread increase of tree mortality rates in the Western United States. Science 323:521–524

von Caemmerer S, Quick PW (2000) Rubisco, physiology in vivo. In: Leegood RC, Sharkey TD, von Caemmerer S (eds) Photosynthesis: physiology and metabolism. Kluwer, Dordrecht, pp 85–113

Wagner KR, Ewers FW, Davis SD (1998) Tradeoffs between hydraulic efficiency and mechanical strength in the stems of co-occurring species of chaparral shrubs. Oecologia 117:53–62

Wang K-Y, Kellomäki S, Li C, Zha T (2003) Light and water-use efficiencies of pine shoots to elevated carbon dioxide and temperature. Ann Bot 92:1–12

White PJ (2000) Calcium channels in higher plants. Biochim Biophys Acta 1465:171–189

Whitehead D (1998) Regulation of stomatal conductance and transpiration in forest canopies. Tree Physiol 18:633–644

Whitehead D, Jarvis PG (1981) Coniferous forest and plantations. In: Kozlowski TT (ed) Water deficits and growth, vol 6. Academic Press, New York, pp 49–152

Whitehead D, Jarvis PG, Waring RH (1984) Stomatal conductance, transpiration, and resistance to water uptake in a *Pinus sylvestris* spacing experiment. Can J For Res 14:692–700

Wielicki BA, Wong T, Loeb N, Minnis P, Priestley K, Kandel R (2005) Changes in earth's albedo measured by satellite. Science 308:825

Wilson JB, Baldocchi DD (2000) Seasonal and interannual variability of energy fluxes over a broadleaved temperate deciduous forest in North America. Agric For Meteorol 100:1–18

Wong SC, Cowan IR, Farquhar GD (1979) Stomatal conductance correlates with photosynthetic capacity. Nature 282:424–426

Wullschleger SD, Tschaplinski TJ, Norby RJ (2002) Plant water relations at elevated CO_2 – implications for water-limited environments. Plant Cell Environ 25:319–331

Yoder B, Ryan MG, Waring RH, Schoettle AW, Kaufmann MR (1994) Evidence of reduced photosynthetic rates in old trees. For Sci 40:513–527

Zimmermann MH (1978) Hydraulic architecture of some diffuse porous trees. Can J Bot 56:2286–2295

Zweifel R, Item H, Häsler R (2000) Stem radius changes and their relation to stored water in stems of young Norway spruce trees. Trees 15:50–57

Zweifel R, Zeugin F, Zimmermann L, Newbery DM (2006) Intra-annual radial growth and water relations of trees – implications towards a growth mechanism. J Exp Bot 57:1445–1459

Chapter 4
Field Studies of Whole-Tree Leaf and Root Distribution and Water Relations in Several European Forests

Jan Cermak and Nadezhda Nadezhdina

The Importance of the Integrated Whole Tree Level Approach

Plants can die because of its structural or functional disbalance, even if all their individual organs or parts are healthy (Chapter 2 and Chapter 3, this volume). The whole tree approach is especially needed when working with large trees, which can be studied only in situ. Another reason for studying trees this way is, that landscape-level information (watersheds, forest enterprises, etc.) are frequently requested by regional managements, etc. It is certainly possible to measure dispersed organs, but this usually takes much longer time, than if working with integrating plant parts. Far not all processes or tree structures are measurable by the instrumental technologies at this level yet. Among the few exceptions are e.g. dendrometers (recording stem diameters), sap flow and electric conductivity measurement or several other technologies based on thermodynamics or electronics, increasingly applied for tree and stand-level studies recently. Application of the above methods includes up-scaling procedures e.g. for entire forest districts, watersheds, etc. (Chapter 5 and Section 3, this volume).

Whole Tree and Stand Level Measurements

Everyday forestry practice can operate with only rather rough data, such as ground-based forest inventory, or remote sensing. Such data are more difficult to introduce into functions-describing models, or semi-quantitative data such as geobiocenosis characteristics and soil composition when working on a landscape level (Zlatník 1976; Randuška et al. 1986; Waring and Running 1998; Buček and Vlčková 2009). There are currently several laboratories using special particular technologies in

---

J. Cermak (✉) and N. Nadezhdina
Institute of Forest Botany, Dendrology and Geobiocenology, Mendel University of Agriculture and Forestry, Zemědělská 3, CZ 61300, Brno, Czech Republic
e-mail: cermak@mendelu.cz

M. Bredemeier et al. (eds.), *Forest Management and the Water Cycle: An Ecosystem-Based Approach*, Ecological Studies 212, DOI 10.1007/978-90-481-9834-4_4,

their field studies, which are applicable for research at higher spatial scales. Mendel University in Brno performed such studies in over 50 species growing in over 60 experimental sites in Europe and U.S. between 1972 and 2009. Comparable studies evaluating simultaneously leaves and roots in relation to water regime at the whole tree level are unfortunately rare. Therefore we reviewed mostly our own literature relevant on this topic here. We tried to elaborate technology focusing especially on water relations, because water is the most frequent limiting factor of tree growth (if not considering high mountains and boreal regions, where temperature could prevail). Major exchanges of energy in ecosystems through water relations are achieved, when transpiration effectively cools plant surfaces, which will be otherwise overheated by solar radiation. Only easily movable instrumentation not requiring any supporting constructions (such as towers or greenhouses), which principally can work anywhere on the whole tree and stand levels including associated software and interpretation technology has been applied in broad scales.

When considering tree structure, studies of leaf distribution have been still based on destructive measurements. Basal area, sapwood basal area or other simply measurable parameters and corresponding allometric relationships have been used for leaf area estimates even earlier (Snell and Brown 1978; Whitehead 1978; Rogers and Hinckley 1979; Running 1980), remote sensing data were later applied for such purposes (Running et al. 1994). Sapwood area related to leaf area/stem growth relations were sometimes applied to evaluate tree vigor (Waring et al. 1980). Vertical and radial distribution of foliage (important for light penetration studies and evaluation of remote sensing data, respectively) can be estimated using the “cloud” technology based on 3D crown mapping (e.g., Morales et al. 1996a, b, c; Cermak 1989, 1998; Cermak et al. 1998a, 2008b; Urban et al. 2009). No suitable instrumentation applicable for leaf distribution measurements in sparse as well as dense forest stands was available. The “cloud” technology is based on spatial description of branch form included into a square network, which is then transferred into the grid (spreadsheet). Coordinates represented by the height of the whorl, azimuth of branch, length and point of beginning the leafed part of the branch have been applied (Fig. 4.1).

Total leaf dry weight and area (using sub-samples) is estimated and its part corresponding to one cell is written in the spreadsheet. Vertical and radial profile is then easily calculated. Ground projected crown pattern is estimated similarly. Certain inaccuracies occurring at the branch level are negligible at the whole tree level. Effective form of tree crowns can be also estimated. This is based on the comparing of diurnal patterns of sap flow and solar radiation absorbed at different – e.g., flat and spherical surfaces (Cermak and Kucera 1990). Recently available modern measurements based on canopy light transmission (LiDAR technology) including spatial scanning (hemispherical photography) the whole aboveground tree parts (Van der Zande et al. 2009) is promising and seems to have a good perspective in future as soon as problems associated with denser stands will be solved. Instrumental measurement of roots is paradoxically more developed than that of leaves at present. Coarse roots have been visualized usually by excavation in the past (Jenik 1957, Vyskot 1976, Kutschera and Lichtenegger 2002, Tatarinov et al. 2008). Similar studies have been done nowadays e.g., by the ground-penetrating radar (georadar - Hruska et al. 1999,

Fig. 4.1 Example of the technology applied for leaf distribution studies. Branch form representing certain volume occupied by foliage ("cloud" of foliage) is included into a square network (here 0.2 × 0.2 m), which is then transferred into the grid (spreadsheet). Coordinates are the height of the whorl, azimuth of branch axis, length and point of beginning the leafed part of the branch. Total leaf dry weight and area (using sub-samples) of each branch are estimated and their fractions corresponding to one cell at the scheme (here: total/17) are written in the spreadsheet. Vertical and radial profile of leaf distribution is then easily calculated. Ground projected crown patterns are estimated similarly. Certain inaccuracies visible on the scheme occurring at the branch level are negligible at the whole tree level (Cermak et al. 1998)

Cermak et al. 2000, Stokes et al. 2002) or by the acoustic technology (Divos and Szalai 2003). The supersonic air-stream (the air-spade) technology was found suitable for almost harmless root excavation allowing also visualization of most fine roots under suitable soil moisture conditions (Rizzo and Gross 2000, Nadezhdina and Cermak, 2003; Cermak et al. 2008, Nadezhdina et al. 2008).

The modified earth impedance (MEI) technology provided information on electrically conducting (ion absorbing) root surface areas (Aubrecht et al. 2006). This technology was first tested through allometric relations in a series of broadleaf and coniferous species, where consistent results were found (Cermak et al. 2006). The range of measured tree size (DBH), have been 5 up to over 1,000 mm (see details in Fig. 4.2). Results of the MEI measurements were tested on the basis of comparing with results obtained with the classical soil coring/root scanning and microscopic methods; conclusions are promising but testing still continue. Analysis of sap flow radial (or cross-sectional) patterns in stems provides information about approximate vertical and circumferential distribution of actually functioning roots, distinguishing superficial and sinker ones (Fig. 4.3; Nadezhdina et al. 2007a, b; Cermak et al. 2008b). Further methodical development is still going on.

When the physiological processes should be studied simultaneously in a large number of experimental sites, only automatically measuring systems not requiring human assistance for rather long periods of time are acceptable. Sap flow technology is one of the most important between them. For sap flow based studies

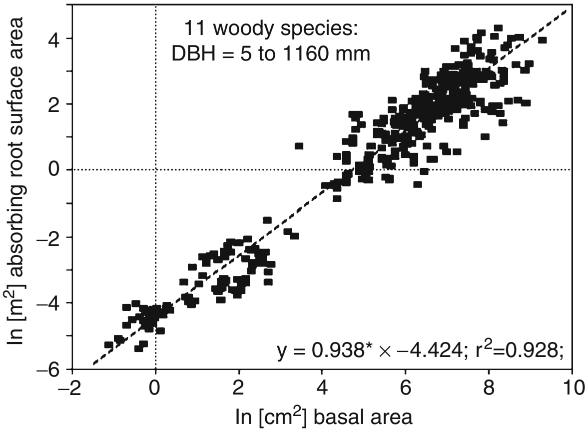


Fig. 4.2 Relationship of basal area and absorbing root surface area measured by the modified earth impedance method in ln-ln scale (the only one allowing to show all trees = points) in almost 500 sample trees of 11 woody species of different age (range about 2–150 years), diameter at breast height (DBH = 5–1,160 mm; proportional height was considered in seedlings), growing in contrasting conditions (such as dry or wet soil, different soil composition, its electric conductivity, etc.). Presented are rough data without corrections to soil conditions. Part of these data was published earlier (Cermak et al. 2006)

we preferably applied sensors installed on stems, because they reflect in an integrated way the behavior of huge number of individual aboveground as well as belowground tree organs (Cermak and Prax 2001; Nadezhdina and Cermak 2003; Nadezhdina et al. 2007a; Cermak et al. 2008a). However if necessary, sensors could be also easily installed in individual branches or roots (Cermak et al. 1984; Lindroth et al. 1995; Nadezhdina et al. 2006b). Two main technologies were applied for sap flow measurements: e.g., trunk heat balance (THB – Cermak et al. 1973, 1982, 2004; Kucera et al. 1977; Tatarinov et al. 2005b) and heat field deformation method (HFD – Nadezhdina et al. 1998, 2002, 2006a; Cermak et al. 2004). Single-point and multi-point HFD sensors (Cermak and Nadezhdina 1998) are suitable for short- as well as long-term field studies under any field conditions.

Studies of Water Storage and the Effectivity of Water Conducting Systems

As an example of further application of flow measurements we can take into account the whole tree (and stand) internal water storage, often estimated from the difference between diurnal courses of transpiration and sap flow (Waring and Running 1978; Hinckley et al. 1978; Cermak et al. 1982, 2007; Phillips et al. 2003; Meinzer et al. 2006). This storage represents a buffering capacity, allowing trees to overcome short periods of soil drought (up to several summer days with fine weather). Such situations are typical during certain periods without rain in summer

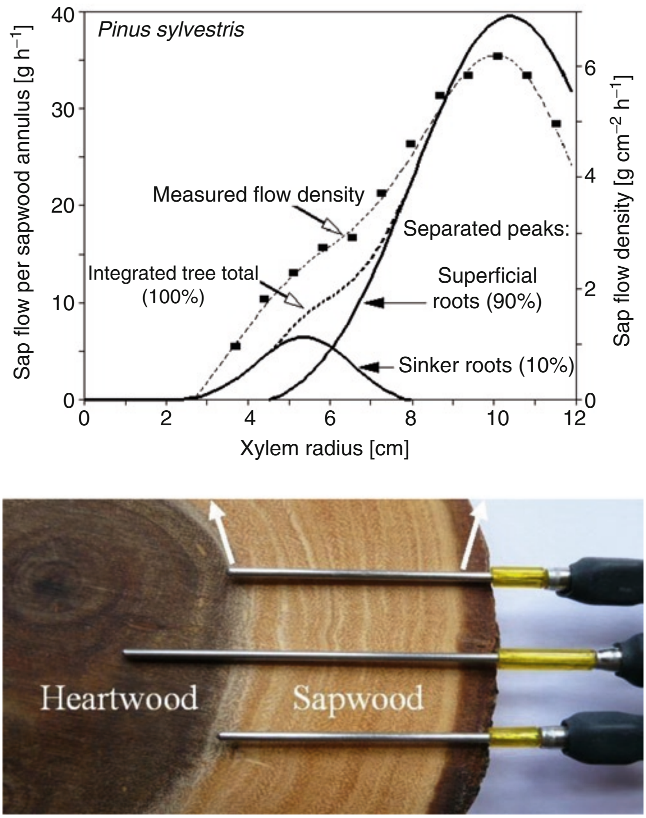


Fig. 4.3 Scheme of the measurement of sap flow density across stems using the multi-point sensors based on the heat field deformation method, HFD (*lower panel*). Sensors consisting of the heater (*middle needle*) and series of thermocouples (two needles symmetrically up and down the heater and one side needle, or two such needles – see Cermak et al. 2004) are installed in the sapwood along stem radius from different stem sides (e.g., cardinal points, up and down the slope, etc.) in order to cover natural variation. Typical results characterizes slightly asymmetric radial pattern of flow density related to root distribution (*upper panel*). Earlier studies confirm, that the flow in different sapwood layers near the stem base corresponds to water supply from roots growing in different depths – superficial and sinker. (Similarly it can be estimated flow into different crown parts, when sensors are situated at the crown base). Mathematical operations usually applied to separate poorly separated peaks allow separation of both fractions of flow (Nadezhdina et al. 2007; Cermak et al. 2008) roots are characterized according to their functions here. Superficial and sinker roots could have very different seasonal dynamics e.g., during drought

or also during sunny days in winter (when water is frozen in shallow soil layers, but still liquid in plant tissues).

The storage is used in a similar way also during different time of the day. Water is easier accessible for transpiration during morning hours from plant tissues (especially sapwood), when it must not overcome the soil and root resistances (Chapter 2, this volume). Therefore the storage of free water in the xylem and soft plant tissues is partially depleted via sap flow (following the gradient of water potentials) and the tissues become slightly dehydrated. Transpiration demands usually decrease in the afternoon, but the sap flow follows still in a rather high rate, refilling the partially depleted storage (Fig. 4.4). Even small storage may be crucial for tree survival under extreme environmental conditions.

Interesting information was also obtained during analysis of conducting efficiency of stems and petioles in trees. This was based on the detail analysis of vessel distribution on the whole tree and stand levels and comparing theoretical flow

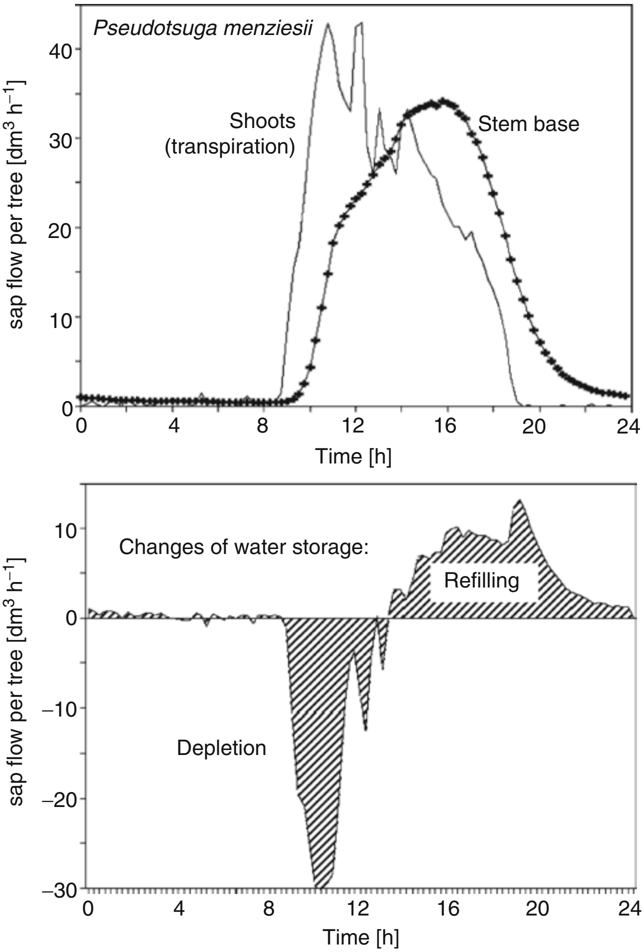


Fig. 4.4 Diurnal courses of sap flow in small shoots (practically equal to transpiration) and at the stem base of a large Douglas fir tree (*Pseudotsuga menziessii* (Mirb.) Franco) – *upper panel*. There is a delay of sap flow at the stem base behind transpiration, because tree utilizes stored water for transpiration from its own tissues first and only after its partial depletion starts extracting water from soil. Differences between both above mentioned flows (which can be very variable, from minutes to hours, such as on the figure) led to depletion of water storage in morning hours and its refilling in afternoon hours down to next morning (*lower panel*)

(calculated from the Hagen–Poiseuille equation including data from anatomical studies), with that measured via sap flow in the field (Krejzar and Kravka 1998; Morales et al. 2002; Cermak et al. 2002). Estimates of trees and stand biometry and leaf distribution (Cermak 1989, 1998; Morales et al. 1996b, c) as well as mean crown water potential (Morales et al. 1996a) were preconditions for such studies. Millions of vessels were found in stems and billions in petioles, when up-scaled for the stand level. Main vessel parameters and results of calculated and measured sap flow indicated in two studied contrasting species (ring-porous oak and diffuse-porous laurel) that theoretical sap flow in stems was about 40 times larger than maximal measured during the growing season, but theoretical flow in petioles was almost equal to the measured one at the tree and stand levels at very distant sites (Central Europe and Canary Islands – Table 4.1). This means, that petioles work almost at maximum of their possibilities (about 95%), while only very small part of stems (about 2%) is sufficient to supply water for transpiration under non-limiting soil water supply. However, leaves work for a short time only (growing seasons), while stems for years, therefore they must have conducting reserves enough to overcome unfavorable conditions (associated with embolism, etc. – Chapter 3, this volume), when the above given data could change.

Detail Studies of Water Redistribution Within Trees

Activity of different tree parts and functional tree architecture can be easily studied with the same instrumentation as mentioned above also on individual roots or branches. Such studies are important e.g., for explanation of tree behavior after some mechanical damage eventually jeopardizing tree survival. Experimental treatment based on several approaches was applied. This includes e.g., branch or root severing (Rychnovska et al. 1980; Cermak et al. 1984; Nadezhdina and Cermak 2000a), localized watering (Matyssek et al. 1991; Cermak et al. 1993; Nadezhdina and Cermak 2000b, 2006b), mechanical loading (Stokes et al. 2000) and others.

Responses of sap flow rate to drastic experimental treatment were observed in the experiment, when one large superficial root growing from the east stem side of a lime tree (*Tilia cordata* L.) was severed. Sap flow abruptly decreased in this root, but also abruptly increased in the sinker root growing near the cut root and remains higher in the sinker also during the next day there (compare 18 and 20 August in Fig. 4.5). This experiment showed the ability of the tree to partially compensate losses of one of its water sources by supplying water for transpiration from another sources. On the other hand and in contrast, at the same time sap flow abruptly decreased in the southern branch at height of 24 m, which indicates interconnection of this branch with the cut root. No changes were recorded in the other measured root and branch. Influence of drought in the topsoil was drastically reflected by changes of sap flow rate in the shallow root, which serves as a conduit for hydraulic redistribution from deep wet into dry soil for the larger part of the day (Fig. 4.6).

Table 4.1 Vessel parameters and sap flow in tree stems and petioles in two species: *Quercus robur* L. from the floodplain forests in southern Moravia (Krejzar and Kravka 1998) and *Laurus azorica* (Seub.) Franco, from the foggy mountain forests in Canary Islands (Morales et al. 2002; Cermak et al. 2002). Theoretical flow was calculated using the Hagen–Poiseuille equation using anatomical data and considering water filled vessels. Sap flow (maximum found during the growing season) was measured by the trunk heat balance method. Numbers of vessels are given in millions (10^6) or billions (10^9). Values of maximum measured flows in brackets represent percentage of the theoretical flows

Level	Species	Leaf area	Number of petioles (= leaves)	Number of vessels		Theoretical flow		Max. measured flow
				in stems	in petioles	in stems	in petioles	
per whole (mean) tree		[m^2]	[pieces]	[pieces]	[pieces]	[kg h^{-1}]	[kg h^{-1}]	[kg h^{-1}]
per stand area unit		LAI [m^2m^{-2}]	[pc ha^{-1}]	[pc ha^{-1}]	[pc ha^{-1}]	[mm h^{-1}]	[mm h^{-1}]	[mm h^{-1}]
Whole tree	Quercus	380	136,000	–	–	980	24.0	24.0
	Laurus	46	36,000	0.4×10^6	4.8×10^6	178	4.3	4.0
Stand 1 ha	Quercus	5.0	20×10^6	–	–	10.1	0.25	0.24 (96%)
	Laurus	7.8	40×10^6	686×10^6	8.2×10^9	30.1	0.72	0.68 (94%)

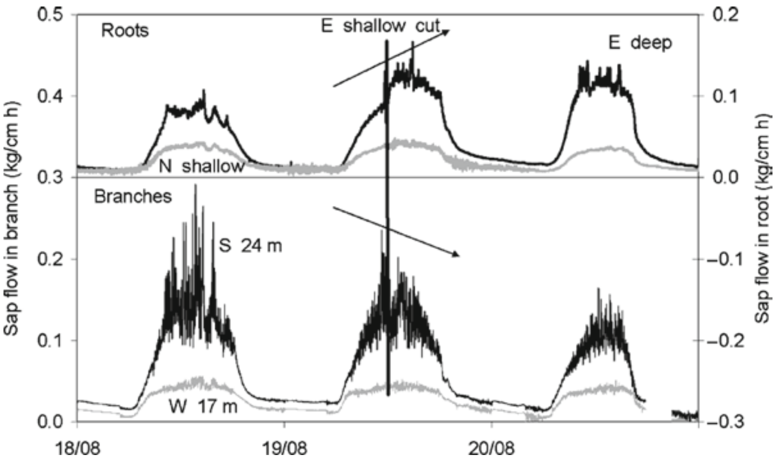


Fig. 4.5 Daily sap flow dynamics in two roots (superficial in shallow soil oriented to North and sinker coming to deep soil oriented to East) and two branches of a lime tree (Mendel University Forest Training Enterprise 1999) recorded with high time resolution (1 min). Three days are presented with similarly high evaporative demands. In the middle of August 19 a large superficial lateral root was cut from the Eastern tree side. Abbreviation E-Shallow and E-deep means East oriented roots of different depths. Flow in the Eastern deep root immediately increased and it was larger also during the next day while there were no changes in the northern root. Increase of flow in the nearest deep root indicates compensation mechanism on the root level. At the same time flow in the southern branch substantially decreased after large root severing. This indicates connection of the cut root with the southern branch

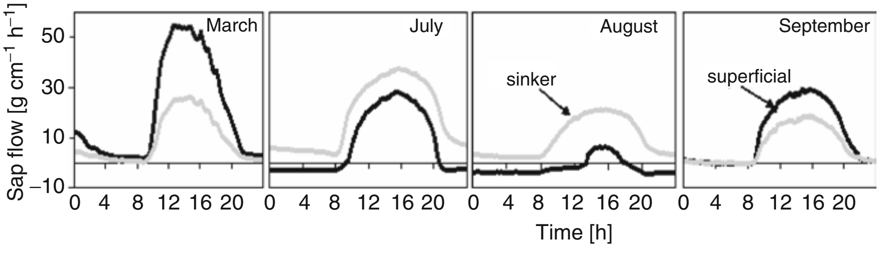


Fig. 4.6 Examples of daily sap flow dynamics in two small (3 cm in diameter) roots (superficial in shallow soil layers and sinker in deep soil layers) of *Quercus suber* tree measured by the HFD single-point sensors. Data are shown for typical days of the growing season 2003 with similar high evaporative demands, but different soil moisture: wet in March and September after rain events and gradually drying from June till end of August without precipitation. Sap flow in the superficial root drastically decreased with topsoil drying and during 2 months reverse flow was recorded in this root, indicating hydraulic redistribution from deep wet soil. Flow in this root increased again after rain when topsoil became wet

Spatial Application: Tree to Stand Level Studies

Leaf area index (LAI) or its most efficient part, the sunlit leaf area index (SLAI) are usual parameters applied in stand description. The above mentioned earth impedance technology allows also estimation of effectively absorbing root surface area index (RAI) and therefore calculation of the RAI/LAI ratio. This is an important parameter widely applied in studies of herbaceous species, where usually ratio of total weight of aboveground and belowground is applied (Hinckley et al. 1978; Pallardy and Kozlowski 1979). An example of this kind of stand description is demonstrated by Norway spruce (*Picea abies* (L.)Karst.) stands of different age, canopy opening and health state. There were rather small differences between healthy young (max 40 years) as well as mature (over 80 years) stands (young stands tend to have slightly higher values of both quantities). Opening of forest stand edge by removing about 50% of trees during thinning and or partial clear cutting lead to enhancing of root growth (having more space than). In contrast, mature stand with root systems (or better root/shoot area ratio) developed below a certain survival limit (originally adjusted to non-limiting soil water supply on peat, where the level of underground water suddenly decreased) appeared very drought-sensitive (Fig. 4.7).

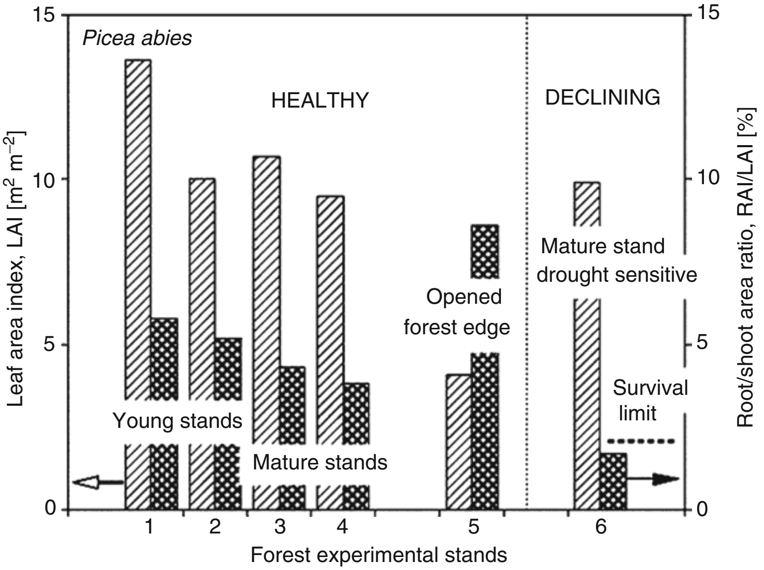


Fig. 4.7 Leaf area index (LAI) absorbing root surface area index (RAI) and their ratio (RAI/LAI) in several Norway spruce stands of different age, canopy opening and health state. Leaf area was derived from allometric relationships; absorbing root surface was measured by the modified earth impedance technology. There are rather small differences between healthy young (max 40 years) as well as mature (over 80 years) stands (young stands tend to have slightly higher values of both quantities). Opening of forest stand edge by removing about 50% of trees during reforestation lead to enhancing of root growth. In contrast, mature stand with root systems developed below a certain survival limit (originally adjusted to non-limiting soil water supply on peat, where the level of underground water suddenly decreased) appeared very drought-sensitive

Practical application of whole tree approach usually includes studies of tree functions and related structures. Differences between species and stands in canopy structure, root distribution, water uptake and seasonal transpiration can be demonstrated in two contrasting species/sites: mature Scots pine (*Pinus sylvestris* L.) plantation growing on sandy soil (Brasschaat, Belgium – Fig. 4.8) and a mixture of species in the frequently flooded floodplain forest (Lednice, Moravia) growing on heavy soil, mostly consisting of pedunculate oak (*Quercus robur* L.), ash (*Fraxinus excelsior* L.) and lime (*Tilia cordata* L.) – (Fig. 4.9). Vertical leaf distribution in canopy layers at different height above ground was estimated on the basis of destructive analysis of series of trees of different size and social positions, (for oak Čermák 1989, 1998, 2008b; for pine Čermák et al. 1998a). Results calculated for individual trees were up-scaled for the stand level on the basis of forestry inventory data (Čermák et al. 2004). Relative vertical distribution of absorbing roots was derived approximately from radial sap flow patterns in stems (flow rate in different sapwood layers – Nadezhdina and Čermák 2003; Čermák et al. 2008a).

We compared seasonal course of transpiration in broadleaf and coniferous trees estimated on the basis of long-term measurements of sap flow rate using the trunk heat balance (THB) method (Čermák et al. 1973, 2004; Kučera et al. 1977, Tatarinov et al. 2005b) or the heat field deformation (HFD) method (Nadezhdina et al. 1998, 2006). Series of sample trees were studied at both compared sites, using as much as 700 sap flow measuring points per stand. Typical years were selected out of 3–10 years of continual measurements there (pine: Meiresonne et al. 2003; Verbeeck et al. 2007; Čermák et al. 2008a; oak: Čermák et al. 1982, 1991, 2001a).

Fig. 4.8 Experimental site of Scots pine forest (composed of *Pinus sylvestris* L., with some under-story broadleaf species), northern Belgium, Brasschaat

Fig. 4.9 Experimental site of mixed floodplain forest (composed mostly from oak *Quercus robur* L. ash *Fraxinus excelsior* L. and lime *Tilia cordata* Mill.; southern Moravia, Lednice). *White belt* marks one of the sample trees

Results were up-scaled using easily measurable tree and stand biometric parameters, such as diameter at breast height, DBH or basal area (Čermák and Kucera 1990; Nadezhdina et al. 2002; Čermák et al. 2004). Seasonal total of transpiration in Scots pine reaching about 140 mm is relatively low, but even much lower values were found under extreme situations in other sites (e.g. pine stand transpired only 50 mm annually on very dry sand-rocks – Čermák et al. 1986). Seasonal total of transpiration was more than twice as large in the floodplain forest when comparing to Scots pine, it was more dispersed in different tree classes, having much larger range there (Fig. 4.10 – lower panels).

The pine forest was rather homogenous plantation with a single narrow canopy layer (main part only about 5 m deep) and leaf area index (LAI) of 3.0, which is very low for coniferous species. In contrast, the floodplain forest represented a multi-layer almost naturally growing canopy (including also frequent shrubs and herbaceous plants), reaching from the ground to over 30 m high tree tops and rather high LAI = 5.0, together with shrub and herbaceous layers LAI = 7, which was sometimes characterized as the "European jungle". Vertical leaf distribution (leaf area density) was very different when comparing both experimental sites, concentrated into a shallow spatial layer in pine and dispersed over the whole aboveground tree part in the floodplain forest, whose LAI was almost twice as large as in pine (Fig. 4.10 – upper panels).

Estimated fractions of absorbing roots seems very similar in the main contrasting species, pine and oak, reaching for superficial roots 82–84 % and for sinker roots 12–16%. These sap flow based and therefore functional characteristics of root systems have been supported by biometric studies in pine (fine roots: Janssens et al. 1999; Xiao et al. 2003). Sinker roots can be active only in periods of time, when

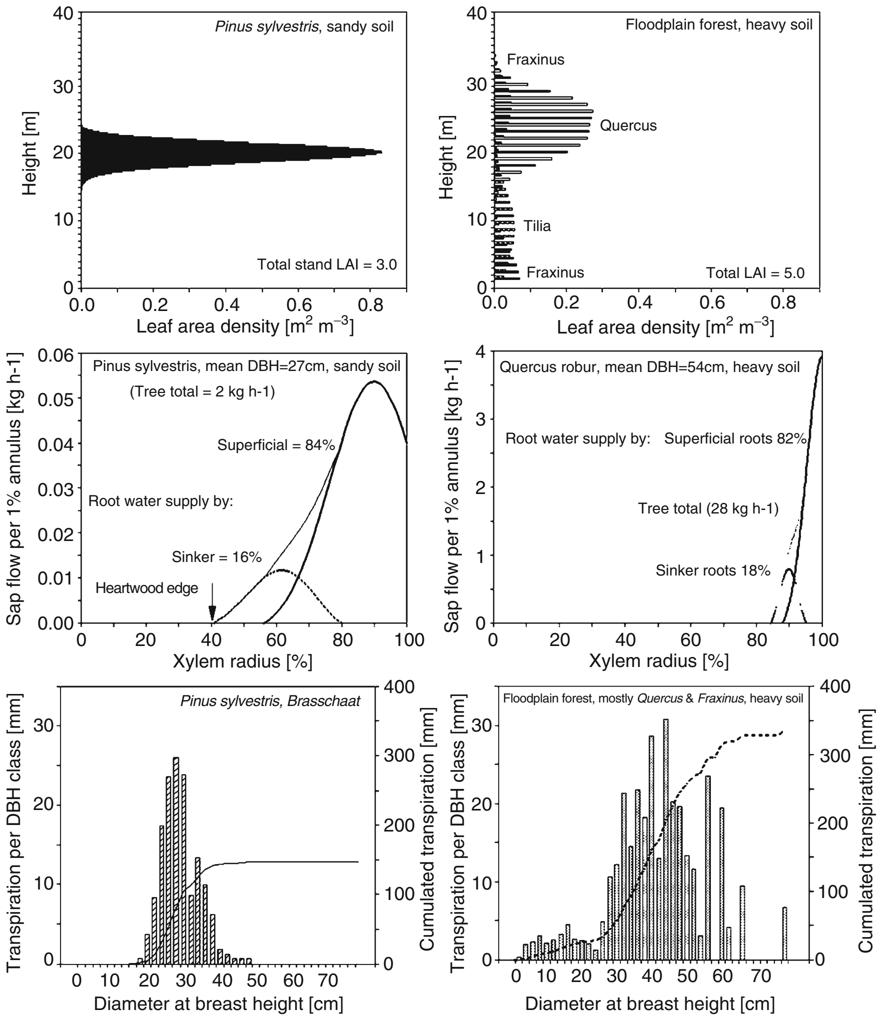


Fig. 4.10 Operative biometric parameters and seasonal transpiration in two contrasting species/sites. Mature Scots pine (*Pinus sylvestris* L.) plantation growing on sandy soil (*left panels*) and a mixture of species in the floodplain forest growing on heavy soil, mostly consisting of pedunculate oak (*Quercus robur* L.), partially of ash (*Fraxinus excelsior* L.) and lime (*Tilia cordata* L.) (*right panels*) were compared. Pine forest was a rather homogenous plantation with a single canopy layer, while floodplain forest represented a multi-layer canopy. Upper panels characterize vertical leaf distribution on the stand level (layer depths in pine 0.2 m, in oak 1.0 m). Medium panels show relative vertical distribution of superficial and sinker absorbing roots (such supplying water through specified layers of sapwood). Their y-axes are different, because the flow values are so contrasting, that anyone cannot be distinguishable when using a single scale only. *Lower panels* characterize transpiration of individual DBH classes over the growing season and their cumulated values giving stand total (Modified according to Cermak 1998, Cermak et al. 2001)

they can reach increasing underground water table, i.e., usually only in spring. Nevertheless soil water supply can quickly decrease during even slight drought in sandy soil. Relative vertical root distribution, shown at the level of typical trees, was rather similar in the same sites; although the patterns of sap flow radial profiles differed substantially (Fig. 4.10 – middle panels). Biometric studies of oak coarse roots in floodplain forests indicated their prevailing spatial distribution in medium to deep soil layers and also indicated, that in general its root system is relatively small, adjusted to supra-optimal water supply (see Vyskot 1976; Tatarinov et al. 2008). This size was sufficient under normal conditions of water supply, because roots have been mostly in permanent contact with underground water table, allowing high stand transpiration this way. Presented value of transpiration (measured via sap flow technology – see Čermák et al. 1982, 2001b; Čermák and Prax 2001) reaching 340 mm per the studied growing season is close to the average, but maximum values of 450 mm occurred under high evaporation demands, when water from underground sources represented up to 70% of the total. Sap flows significantly more intensively in the narrow outer sapwood than in deeper sapwood layers in ring-porous species. (This finding serves well to improved methodology of interpretation of measured data via procedures of root separation). Heavy soil in the floodplain holds huge amounts of water, which also represents a large water holding capacity. However, irrespectively of deep rooting of oak, soil properties can be critical for its transpiration because of small size of its root system. Such situation happened in the seventies, when the water table dropped by 1–2 m to the sandy gravel subsoil for almost 20 years, after regional water management measures. When soil water content decreased by a few percent, soil water potential dropped significantly more (by about 0.8 MPa), but especially soil hydraulic conductivity dropped many (up to 100) times. Therefore soil water supply became critical particularly in small trees with the lowest root/leaf area ratio, which suffered the highest mortality rate there. Some large trees with deep root system suffered too. Similar problems with increased mortality of trees with small root systems growing on heavy soils we also met in the mountain forest of beech (*Fagus sylvatica* L.) in Switzerland (Matyssek et al. 1991; Čermák et al.1993).

Spatial Application: Landscape – Level Studies

Sap flow measurement performed at the stand level can be applied also for landscape – level studies, particularly for calibration of models applied for assessment of regional transpiration (Chapter 8, this volume), especially if series of experimental sites representing the study areas are applied. Such results can represent rather large regions, hundreds or thousands of square kilometers. Representative experimental sites were selected on the basis of forest classification (so called forest types – e.g., Zlatnik 1976; Randuska et al. 1986; Waring and Running 1998) and other site characteristics available for the whole study area. This can also include data obtained using a remote sensing approach (Spanner et al. 1990; Henderson-Sellers et al. 1996;

Thornton and Running 1999). This approach is illustrated on the examples of two contrasting regions, which we studied, particularly two oak species in southern Europe (Italy – Tuscany: Čermák et al. 1998b; Chiesi et al. 2001; Fig. 4.11) and northern Europe (Northern Russia – Oltchev et al. 2002) in forests composed from different broadleaf and conifer species (Fig. 4.12).

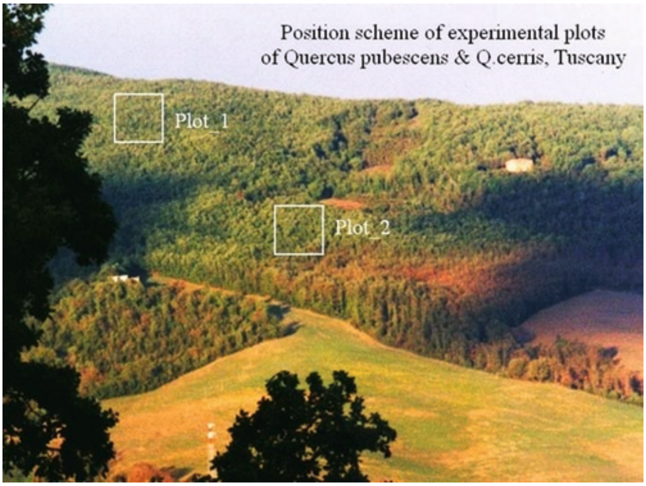


Fig. 4.11 Idealized positioning scheme of experimental plots of *Quercus pubescens* and *Q.cerris* forests near the small town of Radicondoli in Tuscany approximately drawn on an air-born image (orig. 1999)

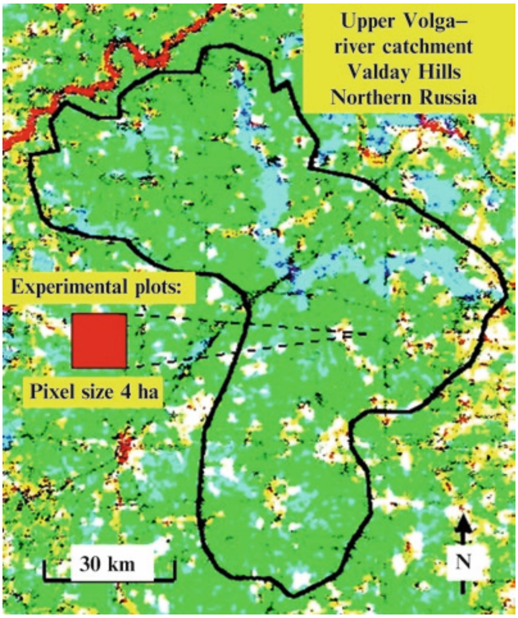


Fig. 4.12 Example of the Upper Volga river catchment in the Valday Hills area (Volga river source) in Northern Russia (Geodetic and Cartographical Survey of the USSR, 1991), where the general hydrological model and local sap flow measurements in selected experimental plots were performed. Marked area (3,500 km^2) (Modified according to Zappa et al. 2001)

We compared transpiration (1) in southern (Mediterranean – Tuscany, central Italy) and (2) northern (boreal – Russia) European watersheds. (1) Mixed spruce-aspen-birch forest in the Valday Hills of boreal region (composed mostly of *Picea abies* (L.) Karst., *Populus tremula* L. and *Betula alba* L.) was determined using sap flow measurements and applied for comparing with two different SVAT (Soil-Vegetation-Atmosphere-Transfer) models with pixel size of 4 ha (Oltchev et al. 2002), based on detail regional forest sites characteristics (Zappa et al. 2001) – (Fig. 4.13 – upper panel). The more sophisticated Mixed Forest multi-layer SVAT model (MF-SVAT) considered water uptake and transpiration of each tree species individually, and the simpler Multi-Layer (ML-SVAT) model described the forest stand using averaged effective parameters of canopy structure and tree physiology. Evapotranspiration and transpiration was represented in many SVAT models (e.g., Geyer and Jarvis 1991; Henderson-Sellers et al. 1996; Sellers et al. 1997) using both "big-leaf" and "distributed multi-layer" approaches (Raupach and Finnigan 1986). Comparisons of modeled and measured transpiration rates under sufficient soil moisture conditions did not show any significant differences between both models (Fig. 4.13 – middle and lower panels). However under limited soil moisture conditions the model MF-SVAT described forest transpiration still realistically, whereas ML-SVAT overestimated it by up to 50%. Drought in the upper soil layers reduced transpiration of spruce more than of deciduous tree species due to differences in their physiological properties and vertical root distribution. Individual transpiration control in different tree species is typical for mixed forests and cannot be accurately described with averaged parameterization as used in simple Multi-Layer model.

(2) The simulation model of forest ecosystem processes applied by Chiesi et al. (2001) in a Mediterranean environment at two distant deciduous forest stands. It was originally created for temperate areas (called FOREST-BCG, Running and Coughlan 1988) and was calibrated to local conditions. The experimental stands were composed of dominating oak species (*Quercus pubescens* Willd. and *Q.cerris* L.) with little admixture of other woody species. Conventional data including sap flow (Čermák et al. 1998b, 2001a – see Fig. 4.13 – upper panels) and remote sensing data were used as inputs. The model was modified using the "maximum canopy average leaf conductance", which is in strict connection with transpiration and therefore can control their xeric nature. First, information on the two stands needed to initialize the model was derived from different sources (such as Forest Management Plan – DREAM 1994, and ecophysiological species parameters derived from existing literature, e.g. Running and Coughlan 1988 and others), while meteorological data were extrapolated from a nearby station (Larderello, Italian National Hydrological Institute) by an existing procedure (MT-Clim). Temporal profiles of leaf area index (LAI, measured by the LAI-2000 plant canopy analyzer Li-Cor, NE, USA, corrected on the basis of individual leaf area measurements: correcting factor of 1.25) were than derived both from direct ground measurements and from processing of satellite NOAA images.

Seasonal maxims of LAI in both oak species estimated this way approached values obtained by the detail ground-based measurements (see Cermak et al. 2008b). Remote sensing images were transformed to the more suitable form (pixel size was 0.01°) by a multi-step procedure. The model was calibrated using stand

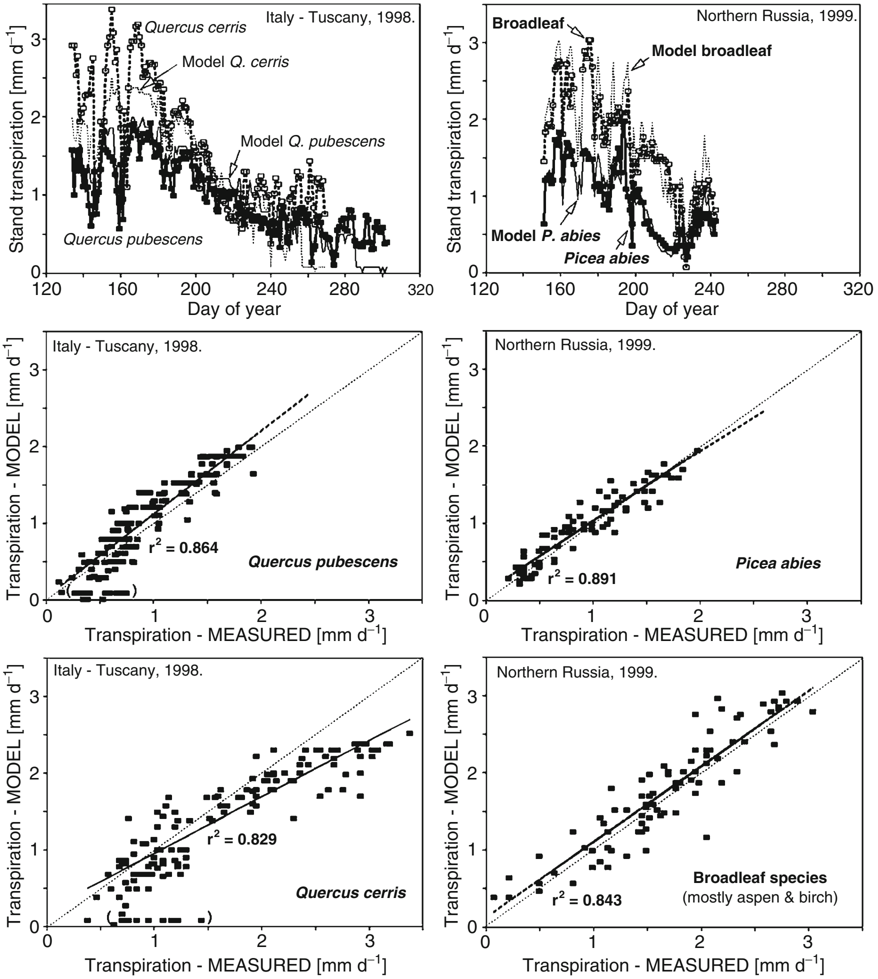


Fig. 4.13 Sap flow measurements as applied for checking different models applied in landscape –level studies for assessment of regional transpiration: Italy – Tuscany 1998 (*left*) and Northern Russia 1999 (*right*). *Upper panels*: seasonal courses of transpiration in Tuscany (where the calibrated model FOREST-BCG was applied) and in Northern Russia (models MF-SVAT and ML-SVAT). *Medium and lower panels*: relationship of data measured in different species (x) and data derived from different models (y). (Data from ends of growing seasons shown in brackets were not included in calculations of regressions) (Modified according to Cermak et al. 1998a, 2001; Chiesi et al. 2001; Oltchev et al. 2002)

transpiration values obtained for 1997 by a sap flow method (using both trunk heat balance and heat field deformation technology as mentioned above). Next, the model performances were tested against the same transpiration values measured in 1998 (see Fig. 4.13 – middle and lower panels). Nevertheless, combination of landscape-level modeling and local sap flow measurements was found very useful in both above-mentioned cases.

Temporal Application: Combining Present Data with History

Sessile and pedunculate oak (*Quercus petraea* Liebl. and *Q. robur* L.) trees represent about 25% of the forested area between Vienna, Bratislava and Brno (Schume 1993). Many of these stands are located along the floodplains of lowland rivers, usually well supplied by water. Changes in soil hydrology can be investigated with theoretical models, integrating the aboveground and belowground tree structure and main soil hydrological processes combined with mechanistic description of water flows in the system (Mualem 1986). Alternatively, biogeochemical ecosystem (BGC) models may be used. Such model (BIOME-BCG, Thornton 1998) is also based on long-term (within almost 30 years) records of sap flow rate (Cermak et al. 1982, 1991, 2001b). Long-term studies of stem growth were also found important for evaluating stand behavior (Tatarinov and Čermák 1999; Tatarinov et al. 2005a). Also indirect historical records were applied in the study of floodplain forest in the above-mentioned region (Pietsch et al. 2003): after the natural virgin phase of forest development the coppice management system with a 20-year rotation period was applied from the fourteenth century, when volume growth decreased by more tan 40%, whereas soil and litter carbon together with nitrogen fell by 50% during the following 500 years. Fortunately the historical records indicated, that forest management changed to high forest in 1880, which led to significant improvement of stand growth, when LAI increased from about 4.5 to 6.

However this positive trend as mentioned above was jeopardized by cessation of floods (usually occurring 4 up to 11 times annually) after 1972 for next almost 20 years due to water management measures in the region associated with canalisation of rivers (Schume 1992; Cermak and Prax 2001; Fig. 4.14). Stand water consumption, which was based up to 70% on underground water under normal conditions decreased by about 50%, trees with unfavourable root/shoot ratio (enveloping root area/sunlit leaf area) started to decline. Artificial watering after 1992 improved partially the situation, but hypoxia problems occurred in stands with permanently increased water table (Cermak and Prax 2001; Cermak et al. 2001b). Modelling over the long-term periods shows that proper management of balanced soil water and air supply keeping high, as much as possible close to natural forests seems the only way to assure well-being of these forests in the future.

Conclusions

Whole tree water relations and structure remain important topic for forest stand as well as landscape level studies. Several instrumental technologies for leaf and root distribution measurement are disposal at present, which allow their implementation in a much easier way for more complex field studies. This considers stand water consumption, water storage, response of sap flow to external disturbances and drought as well as topics related to functional tree architecture, water redistribution

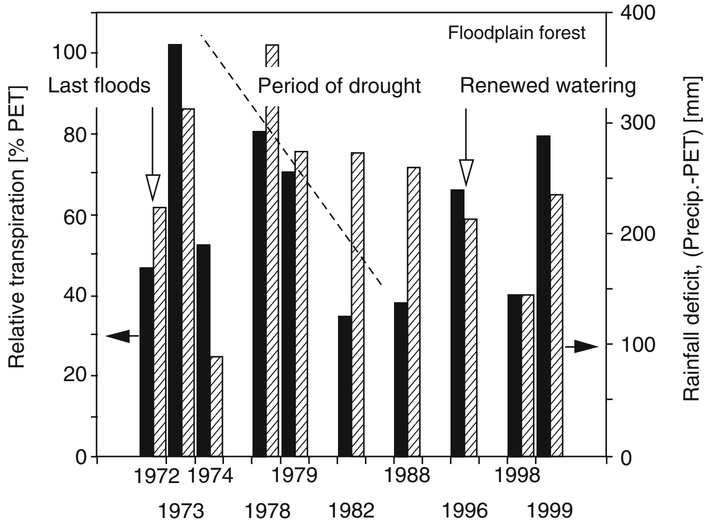


Fig. 4.14 Long-term course (10 years of measurements within the period of almost 30 years) of seasonal totals of transpiration (expressed in relative values to potential evapotranspiration – *left scale*) in floodplain forests in southern Moravia compared to rainfall deficit (precipitation minus potential evapotranspiration – *right scale*). Dramatic soil water changes caused by hydrological measures occurred in the region. Years with sufficient water supply during regular flooding (1972) and after renewed artificial watering (1992–1999) contrasted to the years with substantially decreased underground water table (by 1–2 m between 1973 and 1991). Therefore impact of relative drought on decreasing transpiration is clearly visible. Wet years with higher number of rainy days and foggy weather (1974) differed from dry years (1982 and 1988) (Modified from Cermak et al. 2001)

in trees and also studies including the macro- and micro-levels. Water relations and corresponding structures, starting from individual organs (roots or branches), through whole trees and stands and when using appropriate models up to landscapes are described. Measurements of leaf and root area distributions help to evaluate other eco-physiological processes and also to up scale the data (e.g., allowing connections with remote sensing, serve for calibration etc.). Sap flow measurement provides a tool suitable for short- as well as long-term studies. Whole tree water storage helps trees to overcome critical periods of time in summer and specify periods when actually trees grow. Combination of anatomical analysis with sap flow measurement serves for evaluation of the efficiency of the water conducting system. Analysis of hydraulic tree architecture including water redistribution can explain tree survival under drought. Application of biometric parameters helps to up-scale the sap flow data from trees to entire stands. Combination of maps and remote sensing data enable to work up to the landscape level. The applied technology naturally does not allow solve all problems occurring in the field, but the presented results illustrate that the complex approach provides much better background for deeper analysis of studied forest stands, easier modeling and therefore their better understanding and eventual reasonable local modifications of forest management measures.

Acknowledgements The work on this chapter was done within the framework of the Research Invention of Czech MSM No.6215648902 and project NAZV QC 60063.

References

Aubrecht L, Stanek Z, Koller J (2006) Electric measurement of the absorbing surfaces in whole tree roots by the earth impedance method – I. Theory. Tree Physiol 26:1105–1112

Buček A, Vlčková V (2009) Scenario of vegetation zone changes in the Czech Republic: ten years after. Nature Prot (Ochrana přírody) 64:8–11

Čermák J (1989) Solar equivalent leaf area as the efficient biometric parameter of individual leaves, trees and stands. Tree Physiol 5:269–289

Čermák J (1998) Leaf distribution in large trees and stands of the floodplain forests in southern Moravia. Tree Physiol 18:727–737

Čermák J, Kučera J (1990) Scaling up transpiration data between trees, stands and watersheds. Silva Carelica 15:101–120

Čermák J, Nadezhdina N (1998) Sapwood as the scaling parameter – defining according to xylem water content or radial pattern of sap flow? Ann Sci For 55:509–521

Čermák J, Prax A (2001) Water balance of the floodplain forests in southern Moravia considering rooted and root-free compartments under contrasting water supply and its ecological consequences. Ann Sci For 58:1–12

Čermák J, Deml M, Penka M (1973) A new method of sap flow rate determination in trees. Biol Plant 15:171–178

Čermák J, Úlehla J, Kučera J, Penka M (1982) Sap flow rate and transpiration dynamics in the full-grown oak (*Quercus robur* L.) in floodplain forest exposed to seasonal floods as related to potential evapotranspiration and tree dimensions. Biol Plant 24:446–460

Čermák J, Jeník J, Kučera J, Židek V (1984) Xylem water flow in a crack willow tree (*Salix fragilis* L.) in relation to diurnal changes of environment. Oecologia 64:145–151

Čermák J, Kučera J, Prax A, Balek J (1986) Transpiration and water regime of the pine stand in the sand-rock region of poor pine forests (in Czech). In: Proc. Symp. VSZ v Brne "Funkce lesù v životním prostredi" (pp 67–73), Brno

Čermák J, Kučera J, Štepánková M (1991) Water consumption of full-grown oak (*Quercus robur* L.) in a floodplain forest after the cessation of flooding. In: Penka M, Vyskot M, Klimo E, Vašícek F (eds) Floodplain forest ecosystem II, vol 15B, Development in Agriculture and Management Forest Ecology. Elsevier, Amsterdam, Oxford, New York, Tokyo, pp 397–417

Čermák J, Matyssek R, Kučera J (1993) Rapid response of large, drought stressed beech trees to irrigation. Tree Physiol 12:281–290

Čermák J, Riguzzi F, Ceulemns R (1998a) Scaling up from the individual trees to the stand level in Scots pine: 1. Needle distribution, overall crown and root geometry. Ann Sci For 55:63–88

Čermák J, Nadezhdina N, Raschi A, Tognetti R (1998b) Sap flow in *Quercus pubescens* and *Q. cerris* stands in Italy. In: Proceedings of the 4th international workshop on measuring sap flow in intact plants. Židlochovice, Czech Republic, IUFRO Publications, Publishing house of Mendel University, Brno, 3–5 Oct 1998, pp 134–141

Čermák J, Hruška J, Martinková M, Prax A (2000) Urban tree root systems and their survival near houses analyzed using ground penetrating radar and sap flow techniques. Plant Soil 219:103–115

Čermák J, Nadezhdina N, Jimenez M.S, Morales D, Raschi A, Tognetti R (2001a) Long-term sap flow and biometric studies in laurel and oak forests – Canary Islands and Italy. In: Radoglou K (ed) Proceedings of the international conference on "forest research: a challenge for an integrated European approach", Thesalonniki, Greece, 27 Aug–1 Sept 2001, Vol II., pp 489–494

Čermák J, Kučera J, Prax A, Bednářová E, Tatarinov F, Nadyezhdin V (2001b) Long-term course of transpiration in a floodplain forest in southern Moravia associated with changes of underground water table. Ekologia (Bratisl) 20(suppl 1):92–115

Čermák J, Jimenez MS, Gonzales-Rodriguez AM, Morales D (2002) Laurel forests in Tenerife, Canary Islands: efficiency of water conducting system in *Laurus azorica* trees. Trees 16:538–546

Čermák J, Kučera J, Nadezhdina N (2004) Sap flow measurements with two thermodynamic methods, flow integration within trees and scaling up from sample trees to entire forest stands. Trees 18:529–546

Čermák J, Ulrich R, Staněk Z, Koller J, Aubrecht L (2006) Electric measurement of the absorbing surfaces in whole tree roots by the earth impedance method – II. Verification based on allometric relationships and root severing experiments. Tree Physiol 26:1113–1121

Čermák J, Kučera N, Bauerle WL, Phillips J, Hinckley TM (2007) Tree water storage and its diurnal dynamics related to sap flow and changes of trunk volume in old-growth Douglas-fir trees. Tree Physiol 27:181–198

Cermák J., Ulrich R., Culek I., Cermák M. 2008. Visualization of root systems by the supersonic air stream. In: Neruda J. (ed.): Determination of damage to soil and root system of forest trees by the operation of logging machines. Mendel University of Agriculture and Forestry Publishing House, Brno 2008, pp 89–95

Čermák J, Nadezhdina N, Meiresonne L, Ceulemans R (2008a) Scots pine root distribution derived from radial sap flow patterns in stems of large leaning trees. Plant Soil 305:61–75

Čermák J, Tognetti R, Nadezhdina N, Raschi A (2008b) Stand structure and foliage distribution in *Quercus pubescens* and *Quercus cerris* forests in Tuscany (central Italy). For Ecol Managem 255:1810–1819

Chiesi M, Maselli F, Bindi M, Fibbi L, Bonora L, Raschi A, Čermák J, Nadezhdina N (2001) Calibration and application of forest-BCG in a Mediterraen area by the use of conventional and remote sensing data. Ecol Model 154:251–262

Divos F, Szalai L (2003) Tree evaluation by acoustic tomography. In: Beall FC (ed) Proceedings of the 13th international symposium on nondestructive testing of wood, University of California, Berkeley, CA, 19–21 Aug 2002, pp 251–256

Geyer B, Jarvis P (1991) A review of models of soil-vegetation-atmosphere transfer schemes (SVATS). A report to the TIGER III Committee, March 1991, Edinburgh, 69 pp

Henderson-Sellers A, Mc Guffie K, Pitman AJ (1996) The project of intercomparison of land-surface parametrization schemes (PILPS): 1992–1995. Clim Dyn 12:849–859

Hinckley TM, Lassoie JP, Running SW (1978) Temporal and spatial variations in the water status of forest trees. For Sci Monogr 20:72

Hruška J, Čermák J, Šustek S (1999) Mapping of tree root systems by means of the ground penetrating radar. Tree Physiol 19:125–130

Janssens IA, Sampson DA, Čermák J, Meiresonne L, Riguzzi F, Overloop S, Ceulemans R (1999) Above- and belowground phytomass and carbon storage in a Belgian Scots pine stand. Ann For Sci 56:81–90

Jeník J (1957) Root system of pedunculate and sessile oaks. (in Czech). Rozpr Cesk Akad 67:1–88

Krejzar T, Kravka M (1998) Sap flow and vessel distribution in annual rings and petioles of large oaks. Lesnictvi-Forestry 44:193–201

Kučera J, Čermák J, Penka M (1977) Improved thermal method of continual recording the transpiration flow rate dynamics. Biol Plant 19:413–420

Kutschera L, Lichtenegger E (2002) Wurzelatlas mitteleuropaischer Waldbäume und Sträucher. Leopold Stocker Verlag, Graz, p 604

Lindroth A, Čermák J, Kučera J, Cienciala E, Eckersten H (1995) Sap flow by heat balance method applied to small size *Salix*-trees in a short-rotation forest. Biomass Bioenergy 8:7–15

Matyssek R, Čermák J, Kučera J (1991) Ursacheneingrenzung eines lokalen Buchensterbens mit einer Messmethode der Kronentranspiration. Schweiz Z Forstwes 142:809–828

Meinzer FC, Brooks JR, Domec JC, Gartner BL, Warren JM, Woodruff DR, Bible KD, Shaw DC (2006) Dynamics of water transport and storage in conifers studied with deuterium and heat tracing techniques. Plant Cell Environ 29:105–114

Meiresonne L, Sampson DA, Kowalski AS, Janssens IA, Nadezhdina N, Čermák J, Van Slycken J, Ceulemans R (2003) Water flux estimates from a Belgian Scots pine stand: a comparison of different approaches. J Hydrol 270:230–252

Morales D, Gonzalez-Rodriguez AM, Čermák J, Jimenez MS (1996a) Laurel forests in Tenerife, Canary Islands: the vertical profiles of leaf characteristics. Phyton 36:1–13

Morales D, Jimenez MS, Gonzalez-Rodriguez AM, Čermák J (1996b) Laurel forests in Tenerife, Canary Islands: I. The site stand structure and leaf distribution. Trees 11:34–40

Morales D, Jimenez MS, Gonzalez-Rodriguez AM, Čermák J (1996c) Laurel forests in Tenerife, Canary Islands: II. Leaf distribution patterns in individual trees. Trees 11:41–46

Morales D, Jimenez MS, Gonzalez-Rodriguez AM, Čermák J (2002) Laurel forests in Tenerife, Canary Islands: Vessel distribution in stems and in petioles of *Laurus azorica* trees. Trees 16:529–537

Mualem Y (1986) Hydraulic conductivity of unsaturated soils: prediction and formulas. In: Kluthe A (ed) Methods of soil analyses. Part 1. Physical and mineralogical methods. Agronomy Monographs 9, 2nd edn. American Society of Agronomy, Madison, WI, pp 799–823

Nadezhdina,N, Čermák J (2000a) Responses of sap flow in spruce roots to mechanical injury. In: Klimo E, Hager H, Kulhavy J (eds) Spruce monocultures in Central Europe: problems and prospects. EFI Proceedings 33:167–175

Nadezhdina N, Čermák J (2000b) Responses of sap flow rate along tree stem and coarse root radii to changes of water supply. Plant Soil 12:1–12

Nadezhdina N, Čermák J (2003) Instrumental methods for studies of structure and function of root systems in large trees. J Exp Bot 54:1511–1521

Nadezhdina N, Čermák J, Nadezhdin V (1998) Heat field deformation method for sap flow measurements. In: Proceedings of the 4th international workshop on measuring sap flow in intact plants. Židlochovice, Czech Republic, IUFRO Publications, Publishing house of Mendel University, Brno, 3–5 Oct 1998, pp 72–92

Nadezhdina N, Čermák J, Ceulemans R (2002) Radial pattern of sap flow in woody stems related to positioning of sensors and scaling errors in dominant and understorey species. Tree Physiol 22:907–918

Nadezhdina N, Čermák J, Neruda J, Prax A, Ulrich R, Nadezhdin V, Gašpárek J, Pokorný E (2006a) Roots under the load of heavy machinery in spruce trees. Eur J For Res 125:111–128

Nadezhdina N, Čermák J, Gašpárek J, Nadezhdin V, Prax A (2006b) Vertical and horizontal water redistribution within Norway spruce (*Picea abies*) roots in the Moravian Upland. Tree Physiol 26:1277–1288

Nadezhdina N, Nadezhdin V, Ferreira MI, Pitacco A (2007a) Variability with xylem depth in sap flow in trunks and branches of mature olive trees. Tree Physiol 27:105–113

Nadezhdina N, Čermák J, Meiresonne L, Ceulemans R (2007b) Transpiration of Scots pine in Flanders growing on soil with irregular substratum. For Ecol Manage 243:1–9

Nadezhdina N., Čermák J., Nadezhdin V., Gašpárek J. 2008. Responses of sap flow in roots and tree stems. In: Neruda J. (ed.): Determination of damage to soil and root system of forest trees by the operation of logging machines. Mendel University of Agriculture and Forestry Publishing House, Brno 2008, pp 106–116

Oltchev A, Čermák J, Nadezhdina N, Tatarinov F, Tischenko A, Ibrom A, Gravenhorst G (2002) Transpiration of a mixed forest stand: field measurements and simulation using SVAT models. Boreal Environ Res 7:389–397

Pallardy SG, Kozlowski TT (1979) Early root and shoot growth of *Populus* clones. Silvae Genetica 28:153–156

Pietsch S, Hasenauer H, Kučera J, Čermák J (2003) Modeling the effects of hydrological changes on the carbon and nitrogen balance of oak in floodplains. Tree Physiol 23:735–746

Phillips NG, Ryan MG, Bond BJ, McDowell NG, Hinckley TM, Čermák J (2003) Reliance on stored water with tree size in three species in the Pacific Northwest. Tree Physiol 23:237–245

Randuška D, Vorel J, Plíva K (1986) Phytocenology and forest typology. State Agricultural Publishers (SZN), Prague, 340 pp

Raupach MR, Finnigan JJ (1986) Single-layer models of evapotranspiration from plant canopy are incorrect but useful, whereas multi-layer models are correct but useless. Discuss Aust J Plant Physiol 15:705–716

Rizzo DM, Gross R (2000) Distribution of *Armillaria melea* on pear root systems and comparison of excavation techniques. In: Stokes A (ed) The supporting roots of trees and woody plants: form, function and physiology, vol 87, Developments in plant and soil sciences. Kluwer, Dordrecht, The Netherlands, pp 305–311

Rogers R, Hinckley TM (1979) Foliar weight and area related to current sapwood area in oak. For Sci 25:298–306

Running SW (1980) Relating plant capacitance to the water relations of *Pinus contorta*. For Ecol Manage 2:237–252

Running SW, Coughlan JC (1988) A general model of forest ecosystem processes for regional applications. I. Hydrological balance, canopy gas exchange and primary production processes. Ecol Model 42:1425–1454

Running SW, Justice CO, Salomonson V, Hall D, Barker J, Kaufmann YJ, Strahler AH, Huete AR, Muller JP, Vanderbilt V, Wan ZM, Teillet P, Carneggie D (1994) Terrestrial remote sensing science and algorithms planned for EOS/MODISE. Int J Remote Sens 15:3587–3620

Rychnovská M, Čermák J, Šmíd P (1980) Water output in a stand of *Phragmites communis* Trin. A comparison of three methods. Acta Sci Nat (Brno) 14:1–27

Schume H (1992) Vegetations- und Standortskundliche Untersuchungen in Eichenwäldern des nordostlichen Niederösterreichs unter Zuhilfenahme multivariater Methoden. FIW-Forschungsbericht. University für Bodenkultur, Vienna, p 138

Schume H (1993) Standortliche Zonierung von Eichenwaldökosystemen in Österreich. In: Neuhuber F (ed) FIW-Forschungberichte 1993/95. University für Bodenkultur, Vienna, pp 118–156

Sellers PJ, Dickinson RE, Randall DA, Betts AK, Hall FG, Berry JA, Collatz GJ, Denning AS, Mooney HA, Nobre CA, Sato N, Field CB, Henderson-Sellers A (1997) Modeling the exchanges of energy, water and carbon between continents and the atmosphere. Science 275:502–509

Snell JAK, Brown JK (1978) Comparison of tree biomass estimators – dbh and sapwood area. For Sci 24:455–465

Spanner MA, Pierce LL, Pettersons DL, Running SW (1990) Remote sensing of temperate coniferous forest leaf area index. The influence of canopy closure, understorey vegetation and background reflectance. Int J Remote Sens 11:95–111

Stokes A, Berthier S, Nadezhdina N, Čermák J, Loustau D (2000) Sap flow in trees in influenced by stem movement. In: Spatz HC, Speck T (eds) Proceedings of the 3rd plant biomechanics conference, Freiburg-Badenweiler, 27 Aug–2 Sept 2000. Georg Thieme Verlag Stuttgart, New York, pp 272–277

Stokes A, Fourcaud T, Hruška J, Čermák J, Nadyezhdina N, Nadyezhdin V, Praus L (2002) An evaluation of different methods to investigate root system architecture of urban trees *in situ.* I. Ground penetrating radar. J Arboric 28:1–9

Tatarinov F, Čermák J (1999) Daily and seasonal variation of stem radius in oak. Ann Sci For 56:579–590

Tatarinov F, Bochkarev Y, Oltchev A, Nadezhdina N, Čermák J (2005a) Effect of contrasting water supply on the diameter growth of Norway spruce and aspen in mixed stands: a case study from the southern Russian taiga. Ann For Sci 62:1–10

Tatarinov FA, Kučera J, Cienciala E (2005b) The analysis of physical background of tree sap flow measurements based on thermal methods. Meas Sci Technol 16:1157–1169

Tatarinov F, Urban J, Čermák J (2008) The application of "clump technique" for root system studies of *Quercus robur* and *Fraxinus excelsior*. For Ecol Manage 255:495–505

Thornton PE (1998) Description of a numerical simulation model for predicting the dynamics of energy, water, carbon and nitrogen in a terrestrial ecosystems. Ph.D. thesis, University of Montana, Missoula, MT, 280 pp

Thornton PE, Running SW (1999) An improved algorithm for estimating incident daily solar radiation from measurements of temperature, humidity and precipitation. Agric For Meteoro 93:211–228

Urban J, Tatarinov F, Nadezhdina N, Čermák J, Ceulemans R (2009) Crown structure and leaf area of the understorey species *Prunus serotina*. Trees 23:391–399

Van der Zande D, Mereu S, Nadezhdina N, Cermak J, Muys B, Coppin P, Manes F (2009) 3D upscaling of transpiration from leaf to tree using ground based LIDAR: application on a Mediterraen Holm oak (*Quercus ilex* L.) tree. Agric For Meteorol 149:1573–1583

Verbeeck H, Steppe K, Nadezhdina N, Op de Beeck M, Deckmyn G, Meiresonne L, Lemeur R, Cermak J, Ceulemans R, Janssens IA (2007) Storage water use and transpiration in Scots pine: a modeling analysis using ANAFORE. Tree Physiol 27:1671–1685

Vyskot M (1976) Tree story biomass in lowland forests in South Moravia. Rozpravy CSAV (Praha) 86:86

Waring RH, Running SW (1978) Sapwood water storage: Its contribution to transpiration and effect upon water conductance through the stems of old growth Douglas-fir. Plant Cell Environ 1:131–140

Waring RH, Thies WG, Muscato D (1980) Stem growth per unit of leaf area: a measure of tree vigor. For Sci 26:112–115

Waring RH, Running SW (1998) Forest ecosystems, analysis at multiple scales. Academic, San Diego, CA, 370 pp

Whitehead D (1978) The estimation of foliage area from sapwood basal area in Scots Pine. Forestry 51:137–146

Xiao CW, Curiel Yuste J, Janssens IA, Roskams P, Nachtergale L, Carrara A, Sanchez BY, Ceulemans R (2003) Above- and belowground biomass and net primary production in a 73-year-old Scots pine forest. Tree Physiol 23:505–516

Zappa M, Gurtz J, Jasper K, Vitvar T (2001) The response of the water flows of the boreal forest region at the Volga's source area to climatic and land-use changes. Volga Forest Report of Swiss Federal Institute of Technology, 21 pp

Zlatník A (1976) Forest phytocenology. State Agricultural Publishers (SZN), Prague, 500 pp

Chapter 5
Forested Water Catchments in a Changing Environment

Patrick Schleppi

Introduction: Catchments as Integrators

Headwater catchments are interesting research objects not only as they provide essential measurements about the water cycle, but also because they integrate information on biogeochemical processes over a land area and bring it to a single point, the water outlet. This feature allows monitoring of short- and long-term ecological and geochemical changes in whole ecosystems (Moldan and Černý 1994; Jenkins et al. 1994; Likens and Bormann 1995; Church 1997). One of the main applications of studies on headwater catchments has ever been to elucidate the hydrological role of forests.

Small or very small catchments of less than 1 km^2 or even less than 1 ha are especially adapted to experimental research because their size allows for manipulations and comparisons. Homogeneity of soil and vegetation, along with the possibility of experimental replications, are further potential advantages. In large catchments, variations in discharge and water quality are smoothed and appear mainly as seasonality (e.g. Davis 1986). In small catchments, however, water discharge and water quality fluctuate more (Schleppi et al. 2006a). Short-term variations are not smoothed because of the short residence times and because of limited effects of pooling between subcatchments. The effect of an experimental treatment may thus be measurable in a small, but not in a larger catchment (Burch et al. 1996; Mundy et al. 2001). In the present chapter, we essentially limit our considerations to the scale of elementary catchments. Larger stream and river systems in composite landscapes will be treated in the third part of this book (Chapter 18, this volume).

A prerequisite of catchment studies is that they are well delimited and do not lose water by deep infiltration or gain water from outside by upwelling. This is, however, not always the case. From a statistical comparison between US experimental catchments, Verry (2003) concluded that more than one third of them was leaking significant

---

P. Schleppi (✉)
Snow and Landscape Research, Swiss Federal Institute for Forest, Zürcherstr. 111,
CH-8903, Birmensdorf, Switzerland
e-mail: schleppi@wsl.ch

M. Bredemeier et al. (eds.), *Forest Management and the Water Cycle: An Ecosystem-Based Approach*, Ecological Studies 212, DOI 10.1007/978-90-481-9834-4_5,

amounts of water by deep seepage. Keller (1985) reported of a paired-catchment study in Switzerland which failed because water was leaking from one of the catchments upstream of the weir. Also in Switzerland, the weir on the river Alp had to be rebuilt downstream because, at its first location, it was over a groundwater body with significant flow. Since unsuccessful experiments are seldom published, there are probably more unreported examples of this kind. Some problems certainly also remained unnoticed and caution is needed when interpreting single studies.

An Historical Controversy

In his review, Andréassian (2004) found records about the hydrological roles of forests back to the Antiquity, as Pliny the Elder wrote:

> It frequently happens that in spots where forests have been felled, springs of water make their appearance, the supply of which was previously expended in the nutrition of the trees. (…) Very often too, after removing the wood which has covered an elevated spot and so served to attract and consume the rains, devastating torrents are formed by the concentration of the waters. (Plinus Secundus, first century ad)

In France, an administrative link between water and forest ("Eaux et Forêts") was established during the Middle Age and maintained until recently. This allowed also a coordinated regulation of hunting and fisheries. However, the water cycle itself was not really understood until the late seventeenth century, as the importance of evaporation and infiltration were finally recognised (Mariotté 1674, cited by Verry 2003).

The hydrological effects of forests received much attention in Europe during the nineteenth century, as deforestation increased to meet the demand of a growing population and developing industries. Rougier de la Bergerie (1800, cited by Andréassian 2004) presented an opposite view compared to Pliny the Ancient: he associated droughts and drying out of springs with forest clear-cuttings taking place after the French revolution. Rauch (1821) defended the same opinion and believed that forests were attracting clouds, thus increasing precipitation, but had no effect on the amount of water "pulled by the sun" i.e., in modern terms, on evapotranspiration. Boussingault (1837) also attributed an important role to the forest, stating that it "regulates" water flow. This view became dominant during the nineteenth century and emerged as the "forest hydrological hypothesis" or "sponge theory", according to which forest soils can store much water during rain events and release it slowly during dry periods. This theory seemed to be confirmed by devastating floods in many deforested regions of Europe during the 1830s and 1860s. The view of forests acting both against drought and floods was so appealing to foresters that it found its way into many national forest policies, being for example at the very basis of the Swiss forest law. See also Section 2 of this book for a discussion of this in relation with forest management practices.

The regulating effect of forests, however, was always again questioned (German 1994). In France, it was described as a debate between "foresters" and "engineers"

(Andréassian 2004). The earliest systematic measurements trying to resolve the debate were done by Belgrand (1853) in Burgundy, France. He compared three rivers draining catchments with different forest coverage and found that their water stages reacted similarly to rainfall events. Jeandel et al. (1862) found contrasting results in Vosges valleys, France, showing moderating effects of the forest cover on discharge peaks (compared to other land-uses). At the beginning of the twentieth century, runoff measurements started from two streams in Emmental, Switzerland (Engler 1919; Burger 1943, 1954). Comparing the almost entirely forested Sperbelgraben with the Rappengraben forested to only 35%, these authors have found a strong reduction of peak discharges during short but intensive rain events (thunderstorms), but not during long-lasting rain events.

> After a period of dry weather, for example, alpine pastures produced surface runoff immediately after the rain began, while infiltration was first the dominating process in forests. Only after seven days did the runoff from forests exceed that from pastures. (Burger 1943)

As recognised by these authors themselves, the main drawback of their investigations was to compare catchments that did not differ only in their forest cover. Even by choosing them as similar as possible, it is never possible to avoid small differences in size, morphology, exposition, soil or geology. This limitation can be largely overcome if numerous different catchments are compared. An example is a study by Molchanov (1963), who compared more than 100 streams and rivers in former USSR and found that a forest cover around 50% was optimum to mitigate peak discharges. The challenge of such studies is to ensure that the examined factor (here the proportion of forest cover) is not confounded with any other factor like topography, soil or vegetation type. For example, forests are more frequently in mountainous areas and on slopes than agricultural land. Between forest types, the same applies when comparing conifers to broadleaves. There is thus a risk to ascribe to the vegetation a role actually played by the topography, or arising as interaction only when both factors are combined.

Paired-Catchment Approach

The interpretation of differences between catchments is much easier if a comparison is available both before and after an experimental intervention. This approach was first used at Wagon Wheel Gap, Colorado Mountains, USA (Bates and Henry 1928). In this kind of experiment, catchments are first selected to be as similar to each other as possible and measured in parallel for at least 1 year (to cover the seasonal variations) or for several years (to cover inter-annual variability). After this period of calibration, a single factor is changed experimentally in one of the catchments (treatment), while the other is left unchanged (control). For the Wagon Wheel Gap experiment, measurements were collected during 7 years before the aspen trees were cut in the treated catchment. After the clear-cut, both average and peak discharges were found to increase.

Although Bates and Henry (1928) did not have the formal statistical tools we have nowadays, their design was sound. The effect of the treatment can be tested statistically be subjecting the generated time series to an intervention analysis. The principle is to look at differences between catchments over time and to see if their distribution changes after the treatment compared to the calibration period. This way, both the spatial variability between catchments and the temporal variability among years are taken into account. If there is an interaction between years and catchments, however, it will produce an error on the estimated treatment effect. See Beschta et al. (2000) for a more detailed discussion on the accuracy of flow measurements in a paired-catchment design.

Many paired-catchment experiments were set up around the world during the last century. These studies were reviewed several times (Colman 1953; Hibbert 1967; Bosch and Hewlett 1982; Keller 1988; Hornbeck et al. 1993; Whitehead and Robinson 1993; Cosandey 1995; Stednick 1996; Sahin and Hall 1996; Chang 2002; Andréassian 2004; Brown et al. 2005). The reported results are summarised below, and then discussed in the perspective of environmental changes.

Water Yield and Low Flows

Water discharges are obviously variable in time. To describe the effect of any factor on the discharge of a stream, a "flow-duration curve" is a simple tool synthesising much information. Such a curve (Fig. 5.1 as an example) shows the cumulated frequency distribution of the discharge, i.e. how often a certain discharge is reached or exceeded. Often, however, more attention is given to the extremes of the statistical distribution, i.e. to floods and to low flows, of which severity and frequency are then interpreted. This is also the case when several paired-catchment experiments are compared, because flow duration curves are not always available, or because they do not refer to comparable periods of time relatively to the treatments.

As a rule, water flow increases in deforestation experiments and decreases after reforestation (Hibbert 1967). The size of this effect depends on the following parameters.

The Affected Area and the Intensity of the Treatment

The larger the area affected by tree felling or by replanting, the higher the effect on the annual water flow. Up to 700 mm differences in water yield were measured after a clearcut or after replanting an entire catchment (Bosch and Hewlett 1982; Andréassian 2004). Similarly, the intensity of the cutting is also important,

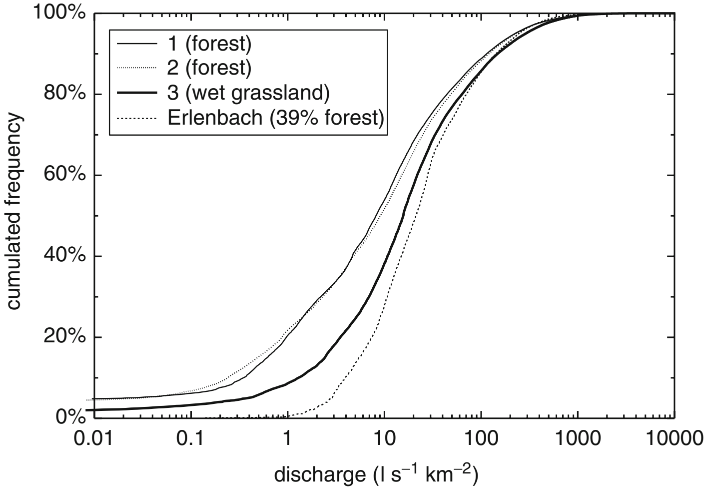


Fig. 5.1 Flow-duration curves of three small experimental catchments (~1,500 m^2) and of the surrounding Erlenbach catchment (70 ha) at Alptal, Switzerland. Averages of 6 years (1995–2000). Note that, in a logarithmic scale, the catchment size as well as the vegetation type affect the frequency of low flows more than that of peak flows. See Schleppi et al. (1998) for details about the experiment

clearcutting of a whole catchment increasing the water flow more than a partial harvest or a thinning (Chang 2002).

The Vegetation Type

Deciduous forests generally yield more water than evergreen forests because they transpire only during the growing season. During the dormant season, there is also less interception loss. A difference in albedo often adds to this effect, evergreens tending to have darker foliage, thus more energy available to evaporate water. The maximum stomatal conductance is generally lower for evergreen conifers compared to deciduous broadleaves (Bond et al. 2007), but the actual transpiration depends on stomatal closure and on the leaf area index (Chapter 3, this volume). Douglass and Swank (1974) found a 200-mm or 20% reduction of streamflow after replacing a mature hardwood forest by pines at Coweeta (Appalachians, USA). The evapotranspiration of grasslands is lower compared to forests, while shrubs take an intermediate position (Chang 2002), along with eucalypt forests (Sahin and Hall 1996) or open forests. Accordingly, the amount of water gained by removing the vegetation depends on how much its evapotranspiration was. Plantation density and thinning are thus considered very important means to control the water budget of managed forests (Chapters 7–9, 22, this volume).

The Soil and the Rooting Depth

Based on US deforestation experiments, Chang (2002) noted that the soil must have a depth of a least 1 m to see an effect on the annual water flow. In central Switzerland, comparing the discharges of catchments differing in their proportions of forest cover (Schleppi et al. 2003), we have found very few differences over the year, except during snowmelt (see below). This has been ascribed to the shallow gleyic soils of the catchments, where trees cannot develop deep roots and thus cannot access deeper water reserves than an herbaceous vegetation does. Cosandey (1995) also recalled the importance of the rooting depth as affecting the water cycle of forests compared to other types of vegetation. Lange et al. (2009) have demonstrated that tree roots are important for preferential flow paths and thus for water infiltration into stagnic soils. See Rewald et al. (Chapter 2, this volume) for further discussion on tree root morphology and physiology.

The Amount of Precipitation

In dry regions (Arizona, USA), Hibbert (1971) found that the absolute effect of removing deep-rooted shrubs depended on the annual precipitation, the relative change being approximately constant. Sahin and Hall (1996) as well as Brown et al. (2005) found increased gains with increasing precipitation also after felling eucalyptus forests. Like for the effect of the vegetation type, the potential for water saving appears to depend on the amount of water used before the vegetation change. According to Chang (2002), there is no measurable effect below 400 mm annual precipitation. In wetter climates, when the demand of the vegetation is met over the year, its water use will not change significantly with precipitation. The effects of precipitation and of a change in vegetation on the discharge can in this case be expected to be simply additive. The frequency of precipitations events obviously also plays a role because it affects the interception loss.

Snow and Other Seasonal Effects

Understanding and quantifying the interception of precipitation is generally a “difficult problem” (Cosandey 1995), but in the case of snow our knowledge is even scarcer. Modelling of snow water equivalent is generally more difficult in forests than in open sites (Rutter et al. 2009). On paired plots in Utah, USA, LaMalfa and Ryle (2008) observed that more snow accumulates on meadows than in coniferous forests, with deciduous forests taking an intermediate position. The total discharge during snowmelt was affected accordingly. We have found similar results when comparing the discharge from experimental catchments covered either by grassland or by conifers (Schleppi et al. 2003), but the duration rather than the intensity of

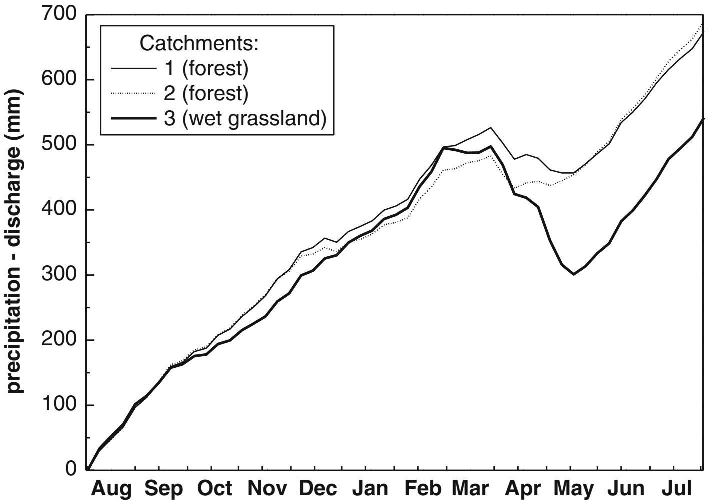


Fig. 5.2 Apparent water balance (precipitation minus discharge) of three small experimental catchments (~1,500 m^2) at Alptal, Switzerland. Averages of 6 years (1995–2000). Note the larger loss of water due to snowmelt from the grassland catchments. See Schleppi et al. (1998) for details about the experiment

snowmelt was reduced in the forest (Fig. 5.2). Snow measurements over 20 years in this area (López-Moreno and Stähli 2008) as well as aerial photographs from the area taken during snowmelt confirmed that forests are earlier free of snow than meadows and that the difference increases up to an altitude close to the timberline. In the deforested catchments of Wagon Wheel Gap (Bates and Henry 1928) and Fool Creek (Troendle and King 1985), both in Colorado, USA, the snowmelt period started earlier and was thus also longer. LaMalfa and Ryle (2008) have found no significant differences in sublimation rates from the snow pack. However, we can explain the reduced snow accumulation in the forest only by a combination of interception and sublimation from the tree crowns over the winter. These processes interact with the radiation regime within the forest (Musselman et al. 2008; López-Moreno and Stähli 2008; Stähli et al. 2009). They are thus different at high latitudes compared to high altitudes closer to the equator, especially for the relative importance of radiation versus convection energy fluxes. It can thus been found that forests become snow free earlier or later than adjacent open fields depending on the location (Rutter et al. 2009; see also Chapter 6, this volume, for further discussion on the role of mountain forests in relation to snow cover).

Differences between seasons are not only related to snow and to the weather itself but also to the seasonality of foliage (deciduous species). Flow-duration curves have been calculated on a seasonal basis by Hornbeck et al. (1997) at Hubbard Brook (New Hampshire, USA). There, the impact of deforestation was much less pronounced during the dormant season than during the vegetation period. At Coweeta (Appalachians, USA), the increased water consumption of pines compared to hardwoods was especially marked during the dormant season and at the beginning of the growing season (Douglass and Swank 1974).

The Time Factor

In the paired-catchment approach, an experimental treatment is done at a certain point in time or for a certain duration. Depending on the type and duration of the treatment, the effect on the water cycle may last for a few or for many years. In forest cutting experiments, this duration depends on the speed at which the new vegetation is growing. It takes typically around 10–15 years to reach again a water discharge close to the forested control (Hornbeck et al. 1993). Extreme values have been reported as short as 4 years (in Pennsylvania, USA; Lynch and Corbett 1990) or as long as 80 years (Colorado, USA; Troendle and King 1985). Compared to deforestation experiments, reports on afforestation experiments are very few. This may be because their effect on the water cycle is not immediate but develops over time, while scientists are nowadays often under pressure to get fast results. Over longer periods of time, it is necessary to take possible changes in the control catchment into account, especially those due to tree aging. An effect of age was for example reported by Hudson et al. (1997) for spruce trees in the Severn catchment (Wales, UK) and by Andréassian et al. (2003) for the maquis of the Valescure catchment (South France). In a long-term study in Australia, Langford (1976) noted a lower water yield in a young eucalypt forest growing after a fire compared to the old-growth forest. A reduction of the stand transpiration with tree age was also found in Douglas fir in northwest USA (Yoder et al. 1994). The leaf area index of aging forests often tends to slightly decrease after having reached a maximum (Ryan et al. 1997) and the hydraulic conductance of old trees is lower (review by Bond et al. 2007). These factors probably explain increased water yield in aging forests.

Hibbert (1967) concluded his review of paired-catchment experiments with three statements: (1) that deforestation increases water yield, (2) that afforestation decreases it, but (3) that the response is variable and largely unpredictable. The first and second conclusions are still generally accepted by most scientists. As shown above, several important factors could meanwhile be identified and more or less quantified, thus reducing the unpredictability deplored by Hibbert in his third statement.

When considering specifically the low flows, they appear to react in the same direction then annual flows but in a relatively more pronounced manner. This was for example reported from flow-duration curves of several paired-catchment experiments in Australia and New Zealand (Brown et al. 2005). Johnson (1998) also indicates that low flows are reduced as trees grow and increased if they are felled. In Russia, Finland and Great Britain, however, low flows were shown to increase for several years after afforestation if this was done by draining peatlands or by ploughing the soil (review by Johnson 1998). Altogether, however, the “sponge theory” appears to be falsified in most cases and forests generally deliver less water than other types of vegetation during periods of low flow.

Floods

Like other catastrophic events, floods attract strong public attention. Much has been published about floods, and the occurrence of floods is often related to the degradation of forests within river basins, in line with the "sponge theory" described above. In the past as well as recently, this theory was often questioned, sometimes even referred as a myth (Hamilton 1985). If forests can reduce average or low-water flows, can they also reduce floods?

To understand floods, the question of the spatial scale always has to be considered (e.g. FAO 2005; Sanford et al. 2007). Scale effects are the topic of Section 3 of this book, thus we will examine here only what happens in elementary catchments. We have to keep in mind, however, that knowledge from small catchments does not directly translate itself into larger, composite basins. Floods in large basins are indeed linked to large-scale, long-lasting precipitation events, while floods from small catchments are often due to local thunderstorms.

A clear increase in peak discharges and flood volumes was found in several experimentally deforested catchments like Wagon Wheel Gap, (Colorado, USA; Bates and Henry 1928), Fool Creek (Colorado, USA; Troendle and King 1985), Coweeta (North Carolina, USA; Swank et al. 1988) or Chiemseemoor (South Germany; Robinson et al. 1991). Increased peak discharges have also been reported after forest fires at Réal Collobrier (South France; Lavabre et al. 1993) or insect outbreaks (Schwarze and Beudert 2009). In the last case, the authors found an increased contribution of fast flow components of the discharge, especially when decaying roots left opened drainage channels in the soil.

Several other studies, however, have shown only small or no measurable effects of removing the trees, especially when focusing on the extreme precipitation events. This was for example the case at Coshocton (Ohio, USA; McGuinness and Harrold 1971) or at the H.J. Andrews Forest (Cascades, Oregon, USA; Beschta et al. 2000). Beside the Swiss studies mentioned above (Engler 1919; Burger 1943, 1954), several authors noted that the reduction of peak flows is proportionally less than for lower flows (e.g. Robinson et al. 1991; Beschta et al. 2000), which points to a saturation of the protective effect ascribed to forests. A similar conclusion has been drawn from a harvesting experiment in the Balsjö catchment (Sweden) (Sørensen et al. 2009). The main mechanism whereby forests can reduce floods is increasing the volume of empty soil pores, by enhancing either the evapotranspiration, the porosity of the soil or the rooting depth (or a combination of them). Calculations by Hegg (2006) have shown that the water retention capacity of a forest soil is rapidly filled up during an extreme event, especially if pre-event humidity was already high. The "sponge effect" is simply, physically limited. Simulations carried out by Leuppi and Forster (1990) have shown that pre-event soil moisture was more important than the vegetation type (forest or meadow) in an alpine catchment. This has been further demonstrated by the fact that the runoff coefficients of rainstorms increase with the pre-event groundwater levels (Burch et al. 1996).

Even if they are often considered in terms of a “forest effect”, tree-cutting experiments have a clear limitation: they do not compare forestry with other land uses, at least not as long as the new ecosystem (replacing the forest) had time to equilibrate its own function. The same applies for natural or experimental disturbances, which should always be considered as such and not as comparisons “forest versus non-forest”. Afforestation experiments allow for a more unbiased interpretation because they produce less short-term effects. Due to the slowness of pedogenic processes, however, landuse-change experiments can barely be considered as finished within a human lifetime. As for the low flows, if afforestation is accompanied by draining or ploughing, this can notably modify the effect on peak flows. In a comparison between three small catchments in central Switzerland, Burch et al. (1996) have found no direct relation between the proportion of forest cover on one hand and either peak flows or runoff coefficients during storms on the other hand. In this case, the catchment with the intermediate forest cover had the highest runoff peaks. In this catchment, drainage trenches had been dug with the purpose of enhancing forest development into the wet meadows. Even if there is no direct experimental evidence for this effect, it appears that this way of afforestation is actually increasing flood risks.

On the other hand, forests are less prone to erosion than most other forms of land use, and this may help to reduce floods (Piégay et al. 2004). Studies by Alewell and Bebi (Chapter 6, this volume) are good examples. Here again, one should not draw conclusions about “forest versus non-forest” by actually comparing “forest versus clearcut”. Grassland, for example, has also a permanent soil cover and its soil protection effect is closer to that of a forest than of a tilled agricultural field. As for the water yield, a land-use change can have transient effects that differ from the longer-term, steady-state effects. At Sleepers River (Vermont, USA), for example, McBride et al. (2008) have observed a deepening and broadening of stream channels along which spontaneous reforestation had occurred. In a paired-catchment study in Missouri (USA), Udawatta et al. (2002) found increased erosion under agroforestry compared to a crop rotation. Disastrous soil erosion is often observed in the tropics following deforestation, but removing the trees is only part of the problem, the new land use practices being decisive in terms of erosion (FAO 2005).

Water Quality

Older studies on small catchments did not include measurements of water quality. The clear-cutting experiments at Hubbard Brook (New Hampshire, USA) then showed clearly that a paired catchment design is also appropriate to study effects on water quality (Likens et al. 1970). This insight promoted experiments in headwater catchments especially when questions about acid deposition arose, later about eutrophication with nitrogen (van Breemen and Wright 2004). A good example was given by Neal (1997) as he reviewed all publications from research on the Plynlimon catchments (Wales): while quantitative hydrological studies dominated in the 1970, the focus shifted towards more qualitative studies in the 1980.

Water quality is described through physical, chemical and biological aspects. Among the physical parameters, the concentration of particles and the turbidity are the most important. They are mainly related to erosion and thus to floods (see above). Eroded soil material is often rich in nutrients and losses through erosion can be substantial or even dramatic. Biological aspects are mostly described in terms of biodiversity and integrity of food chains. Physical and biological components will not be discussed in detail, the focus being here on the chemical parameters of water quality.

Substances dissolved in water can be classified according to their chemistry into organic or inorganic compounds. Specific classes of pollutants can also be recognised, for example pesticides, hormone-like substances, mercury, radionuclides etc. They can also be classified according to their input/output ratio in the ecosystems, input being mainly deposition (wet and dry) and output being leaching from the soil (Likens and Bormann 1995). Elements enriched in the water leaving the ecosystem are typically calcium, magnesium and aluminium, originating from soil weathering (geogenic elements, input/output ratio below 1). Their concentrations are mainly determined by the content in the soil and by acidifying agents, from soil processes or from atmospheric deposition. In well-buffered catchments, these elements are typically less concentrated when the water discharge is high, due to a simple dilution effect. This was for example shown at Pluhuv Bor (Czech Republic) and Sleepers River (Vermont, USA) (Shanley et al. 2004) and at Alptal (Switzerland) (Schleppi et al. 2006b). Over more acidic parent material, biological cycling becomes relatively more important beside weathering, and the output patterns are more complex, as for example at Hubbard Brook (New Hampshire, USA) (Thorne et al. 1988). In contrast, ammonium is present in higher concentrations in precipitation than in drainage water or runoff because it is effectively retained in the soil and used by biota. Other elements have an input/output ratio sometimes above, sometimes below 1, depending on time and location (crossover patterns according to Likens and Bormann 1995). Nitrate, sulphate and potassium are examples. As nutrients, they can be taken up by plants and typically show a clear seasonality (see also Neal et al. 1997). Their concentrations generally have also a negative correlation with the water discharge, but not as closely as for geogenic elements (Schleppi et al. 2006b). Sulphate and especially chloride and sodium are present in sea spray and their fluxes can thus be highly affected by deposition of sea salt, even hundreds of kilometres downwind from the sea. The pH of stream water is obviously linked to the pH of the soil, which is mostly acidic in boreal forest and in temperate coniferous forests. For example, in a spruce afforestation at Mont Lozère (France) an increase in the loss of base cations was observed as compared to a beech forest (Lelong et al. 1990). However, van Breemen et al. (1984) were able to show that the observed acidification of soils and streams in Europe was dominated by acid deposition rather than by acids produced during litter decomposition. At Birkenes (southern Norway), Christophersen et al. (1990) observed decreasing concentrations of base cations over a 25-year period; they attributed this trend to a continuing soil acidification in the catchment in spite of reduced sulphate deposition. Improvements of stream water quality were obtained in Black Forest catchments (Germany) by liming forests growing on soils with a weak buffering capacity (Sucker et al. 2009). Stream acidification may be delayed compared to soil acidification if neutralisation occurs in the sub-soil, as observed in a catchment with spruce decline in Alsace (France) (Probst et al. 1990), but generally surface waters are a more sensitive indicator of acidification processes than soils. Starting in the 1980s, a roof experiment was set up at Risdalsheia (Norway), where a small catchment was protected from acid deposition (Wright et al. 1993). The resulting improvements in water quality played a major role in pollution control policies, especially within the Convention on Long-range Transboundary Air Pollution (CLRTAP) of the United Nations Economic Commission for Europe (UNECE) or in amendments of the Clean Air Act in the USA.

As a result of abatement policies in Europe and North America, emissions of sulphur were strongly reduced and those of reactive nitrogen also slightly. The reduced deposition have been shown to reduce the sulphate concentrations in streams on both continents (Stoddard et al. 1999; Prechtel et al. 2001), while the effect on nitrate, if any, was weak. In Scandinavia (Folster et al. 2003), Scotland (Soulsby et al. 1997), England (Tipping et al. 2000) and in the Czech Republic and Slovakia (Veselý et al. 2002), this led to a reversal of acidification, but in Germany the process was delayed by sulphur accumulated in the soil (Alewell et al. 2001). A decade later, however, the analysis of numerous German catchments showed a reversal of acidification in most of them, resulting also in less leaching of the weathering products calcium, magnesium, aluminium and manganese (Sucker et al. 2010).

In a study of stream water quality across the USA, Omernik (1977) found lowest concentrations of nitrate and phosphate in catchments with more than three quarters of forests. The effect of forests on water quality is often seen as consisting of a passive and an active component (Wenger 2002). The passive effect corresponds to the exclusion of practices that could impair water quality. This may be due to practical reasons (for example because the forest is in a remote area, difficult to access, or otherwise unsuitable for other landuse forms) or because of legal restrictions (for example on the use of fertilisers and pesticides in forests of many regions or countries). This passive effect is generally recognised as very important (Küchli and Meylan 2002; Chang 2002).

A passive effect was also deduced from our nitrate analyses in streams from three small catchments at Alptal (Switzerland), each with a different land cover (Fig. 5.3; Schleppi et al. 2003). Low nitrate concentrations were measured in catchments dominated either by spruce forests or by pastures, and even lower concentrations in the catchment dominated by wet grasslands (formerly used as extensive meadows and then abandoned). These positive results were interpreted as arising from a combination of (1) the perennial vegetation, (2) no use of commercial

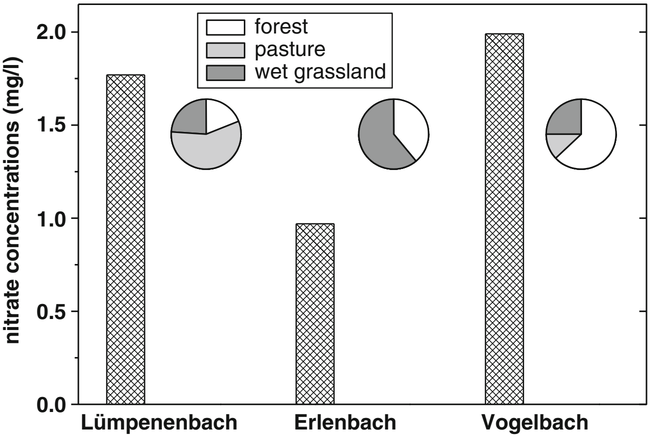


Fig. 5.3 Land use and nitrate leaching from three headwater catchment at Alptal, Switzerland (Lümpenenbach: 93 ha, Erlenbach: 70 ha, Vogelbach: 155 ha). Averages of 6 years (1995–2000)

fertilisers, (3) high precipitation amounts (dilution effect) and (4) wet soil favouring nitrate losses via denitrification.

Active effects are defined as arising directly or indirectly from the presence of trees. Because of the size and aerodynamic roughness of forest canopies, they retain more atmospheric pollutants, which corresponds to an increased dry deposition that can be washed from the foliage as throughfall (Lovett and Reiners 1986; Draaijers et al. 1992). The evaporation of water intercepted by the canopy also causes an increase in the concentration of solutes (Swanson et al. 2000). These effects are generally more pronounced with evergreen than with deciduous species (e.g. Rothe et al. 2002). In areas with high deposition rates, it may happen that a forest clearcut improves the water quality by reducing pollutant concentrations in the water reaching the soil (Frijns and Tietema 2002). Below ground, trees differ mainly from other plants by their ability to develop deeper roots and by their more frequent association with mycorrhizae, which enables them to better access water and nutrients from a larger soil volume. The high transpiration and lower water yield (see above) of forests are potentially negative for water quality by concentrating the solutes. Deep roots, however, (re)capture nutrients and especially base cations from deeper soil layers (Jobbágy and Jackson 2004). This can facilitate the closure of the element cycles and thus slow down soil acidification and improve water quality. These direct effects of trees on water quality were mainly studied at the plot level with soil solution sampled with lysimeters or suction cups. Results from studies on tree roots, mycorrhizae and their water relations are summarized by Rewald et al. (Chapter 2, this volume).

At Hubbard Brook (New Hampshire, USA), all major ions except sulphate, carbonate and ammonium showed increased concentrations in the stream after clearcutting headwater catchments. This was especially the case for the major plant nutrients nitrate and potassium (Likens et al. 1970). In the Plynlimon catchments (Wales), deforestation was shown to increase nutrient losses to the stream for 5 years, which was due to both a reduced plant uptake and a faster decomposition (Neal et al. 1992). At Coweeta (North Carolina, USA), element losses after a clearcut were limited by fast-growing early-succession species (Boring et al. 1981), but defoliation by insects increased nitrate losses (Swank et al. 1981). However, the effect of such disturbances was also limited in time, while long-term increases in nitrate leaching were attributed to N saturation from atmospheric N deposition (Swank and Vose 1997). Even a moderate forest thinning can lead to noticeably increased fluxes of sediments, base cations and chloride, as shown in paired catchments near Istanbul (Turkey) (Serengil et al. 2007). In small catchments in Québec (Canada), forest harvest and forest fires were shown to increase N, P and K nutrient exports (Lamontagne et al. 2000). For N and P, this was relevant for water quality, for K it was important for the nutrition of the forest ecosystem itself. More generally, increased nutrient concentrations in streams are observed after fire (reviews by Landsberg and Tiedemann 2000; Ice et al. 2004) as well as after forest cutting and harvesting. Dissolved P exports to streams are not much affected by forest management, but P is present in suspended matter and exports are thus increased when erosion occur (Stednick 2000).

Apart from fertilisation, the amount of nitrate lost after forest harvest has been related to N deposition and N saturation (Swank and Johnson 1994; Fenn et al. 1998). From a study of water bodies in the USA, Stoddard (1994) distinguished between different stages of N saturation, according to the seasonality of nitrate leaching, with advanced N saturation being characterised by high nitrate losses throughout the year. Nitrate leaching, however, cannot be predicted from N deposition only. Among forest sites across Europe, Dise and Wright (1995) distinguished between three levels of N deposition. At low rates (<10 kg ha^{-1} a^{-1}), there is practically no nitrate leaching and at high rates (>25 kg ha^{-1} a^{-1}) significant leaching is generalised. At medium rates, some forests lose nitrate but others do not. Their behaviour is then related to their internal N status, mainly to their soil C/N ratio, as shown by coniferous forests belonging to the European NITREX project (Gundersen et al. 1998). On three of these sites, experimentally increased N deposition was tested in small catchments. At Sogndal (Norway), the catchment treated for 9 years with nitric acid (7 kg ha^{-1} a^{-1} N) lost 10% of the added N as nitrate. This proportion was similar as in the control catchment and did not show increases with time, which was a sign that this shrub ecosystem did not reach N saturation (Wright and Tietema 1995). At Gårdsjön (Sweden), under Norway spruce, episodic nitrate leaching initially occurred mainly during the dormant season. It increased with time in the NH_4NO_3-treated catchment (40 kg ha^{-1} a^{-1} N), affecting also the growing season and reaching 10% of the added N after 13 years (Moldan et al. 2006). The authors considered an accumulation of ammonium as responsible for the incomplete retention of N on their site. At the Alptal (Switzerland), also a Norway spruce forest, a similar treatment (25 kg ha^{-1} a^{-1} N) resulted in higher loss rates, increasing from 10% to 30% of the added amounts within 7 years (Schleppi et al. 2004). The original rate has been explained by a direct loss of precipitation nitrate due to fast preferential flow through the soil, while the increase with time was accompanied by a decreasing C/N ratio. This was interpreted as a causal relationship, the lower C/N ratio hampering the immobilisation of further N deposition. At Bear Brook (Maine, USA), a site with mixed hard- and softwood forests, N was added at the same rate (25 kg ha^{-1} a^{-1} N) for 12 years, to but as $(NH_4)_2SO_4$. A strong increase in nitrate losses was also the consequence, in this case accompanied by clear signs of acidification in the short term (Norton et al. 2004) but without further deterioration in the longer term (Laudon and Norton 2010). At Fernow (West Virginia, USA), acidification has also been shown in the stream draining the treated hardwood forest receiving additional $(NH_4)_2SO_4$ (25 kg ha^{-1} a^{-1} N). Nitrate-N leaching was in the same proportion as at Alptal, initially 10% of the addition, later increasing to around 30% (Adams et al. 2007). Considering together these paired-catchment studies, it appears that, regardless of its chemical form, a long-term addition of N to forests leads to an accumulation of N in the soil and to a progressive increase in nitrate leaching. This general pattern can be described as N saturation, but the involved mechanisms are diverse across sites (with or without acidification, with or without decreasing C/N, with an initial retention rate higher or lower than the control). The mechanisms of N retention are also changing over time (Morier et al. 2010) and do not necessarily fit to the classical models of N saturation by Aber et al. (1989) or by Stoddard (1994).

The Importance of Long-Term Catchment Studies in a Changing World

Experiments dealing with climate changes are obviously difficult to conduct at the scale of headwater catchments. At Risdalsheia (Norway), the roof first used to exclude acid rain (see above) was converted to a greenhouse making it possible to study effects of an increased temperature and CO_2 concentration (van Breemen et al. 1998). In the treated small catchments, N mineralization increased and markedly more nitrate was lost to stream water, in spite of faster tree growth. At Storgama (also in Norway), Stuanes et al. (2008) conducted different manipulations on very small catchments, including removing snow or, on the contrary, artificially insulating the soil during winter. Water quality (especially nitrate concentration) was shown to change both with soil temperature and atmospheric deposition. In a warmer climate, soil temperatures may either increase or decrease during winter depending on snowfall and snow interception by trees (Stadler et al. 1996). As a consequence of such non-linear effects, trends in water quality may change locally as average temperature increases.

Whole-catchment experiments were very useful for the study of water and biogeochemical cycles. Compared to laboratory experiments and plot studies, they have the incomparable advantage to encompass a wider range of natural variability, and more interactions and feedbacks between components of the ecosystems. Avoiding many artifacts of smaller-scale experiments, they deliver results which can more easily be extrapolated to the real world. Plot studies within catchments, with the possibility to compare different scales can often combine this advantage with more insights into the underlying processes (e.g. Hagedorn et al. 2001; Knoepp et al. 2008). Depending on the factors studied, however, catchment research can be very expensive. This was already the case for the land-use and deposition experiments described above, but even more if realistic climate-change scenarios are to be simulated. The only perfectly realistic climate-change experiment is probably the one currently conducted by mankind on the whole planet Earth. An experiment without a control, however.

Due to the difficulties in establishing realistic climate-change experiments, some gap remains between small-scale experiments and long-term monitoring, the latter being often difficult to interpret because several factors change in parallel, confounding between causes underlying the observations. We can trust that scientists will always again find new approaches to tackle new questions and hypotheses. There were many successful studies in the past which actually based on unwanted disturbances like fire, insect outbreaks (see above) or climatic events like frost (e.g. Mitchell et al 1996) or windstorm (e.g. Badoux et al. 2006). Already shortly after the question of global warming arose, Driscoll et al. (1989) interpreted trends observed after 25 years of research at Hubbard Brook as a possible effect of climatic changes. For long-term studies, however, decisions about methods are actually partly taken before new questions arise. A great attention must thus be paid to the quality control of field measurements, sampling and laboratory analyses. In their classical work, Likens and Bormann (1995) note that a sampling strategy based on fixed times is sufficient, at least for their catchments where discharge and concentrations do not vary much over time. Hornung et al. (1990) also recommended taking weekly water samples, along with some storm sampling. Many other catchment studies followed or still follow such sampling schemes. Introducing flow-proportional sampling schemes, however, was shown to reduce both systematic and residual errors in flux calculations (Schleppi et al. 2006b). Here, we see a simple but real potential to improve the statistical power of the monitoring, one of the three key issues recently pointed out for the success of long-term monitoring (Lindenmayer and Likens 2009).

Several networks of catchment studies are active in different regions of the world or globally, for example the International Hydrological Programme (IHP, UNESCO), the Hydrology and Water Resources Programme (HWRP, WMO), the Global Terrestrial Network - Hydrology (GTN-H) or the Global Runoff Data Centre (GRDC). For the long term, catchments studies will probably need to be even better integrated in large network to be able to resolve at least the main interactions existing between temperature, precipitation, land-use, pollution etc. Because of the duration of forest life cycles and forest dynamics, and because progressive effects of environmental changes can take long to evolve, long data records are important to yield statistically significant trends. There is always a risk that this kind of research loses novelty and attractiveness with time, which does obviously not make it easier to finance. Forest research institutes outside the university structure are often the only ones to ensure the necessary longevity of research projects. Within the concept of adaptive monitoring by Lindenmayer and Likens (2009), however, scientists should be able to establish a productive cycle of new questions, adapted methods and new answers. Maintaining, improving and integrating high-quality networks thus remains of prime importance. Significant advances in our ability to understand and forecast the effects of environmental changes on forests and their water cycle will always require the combination of experiments and long-term monitoring at the catchment scale.

References

Aber JD, Nadelhoffer KJ, Steudler P, Melillo J (1989) Nitrogen saturation in northern forest ecosystems. BioScience 39:378–386

Adams MB, Kochenderfer JN, Edwards PJ (2007) The Fernow watershed acidification study: Ecosystem acidification, nitrogen saturation and base cation leaching. Water Air Soil Pollut Focus 7:267–273

Alewell C, Armbruster M, Bittersohl J, Evans CD, Meesenburg H, Moritz K, Prechtel A (2001) Are there signs of acidification reversal in freshwaters of the low mountain ranges in Germany? Hydrol Earth Syst Sci 5:367–378

Andréassian V, Parent E, Michel C (2003) A distribution-free test to detect gradual changes in watershed behavior. Water Resour Res 39:1252

Andréassian V (2004) Waters and forests: from historical controversy to scientific debate. J Hydrol 291:1–27

Badoux A, Jeisy M, Kienholz H, Luscher P, Weingartner R, Witzig J, Hegg C (2006) Influence of storm damage on the runoff generation in two sub-catchments of the Sperbelgraben. Swiss Emmental. Eur J For Res 125:27–41

Bates CG, Henry AJ (1928) Forest and streamflow experiment at Wagon Wheel Gap, Colorado. Mon Weather Rev Suppl 30:1–79

Belgrand E (1853) De l'influence des forêts sur l'écoulement des eaux pluviales. Annu Soc Météorol France 1:176–193

Beschta RL, Pyles MR, Skaugset AE, Surfleet CG (2000) Peakflow responses to forest practices in the western cascades of Oregon, USA. J Hydrol 233:102–120

Bond BJ, Meinzer FC, Brooks JR (2007) How trees influence the hydrological cycle in forest ecosystems. In: Wood PJ, Hannah DM, Sadler JP (eds) Hydroecology and ecohydrology: past, present and future. Wiley, Chichester, pp 7–35

Boring LR, Monk CD, Swank WT (1981) Early regeneration of a clear-cut southern Appalachian forest. Ecology 62:1244–1253

Bosch JM, Hewlett JD (1982) A review of catchment experiments to determine the effect of vegetation changes on water yield and evapotranspiration. J Hydrol 55:3–23

Boussingault J-B (1837) Mémoire sur l'influence des défrichements dans la diminution des cours d'eau. Ann Chim 64:113–141

Brown AE, Zhang L, McMahon TA, Western AW, Vertessy RA (2005) A review of paired catchment studies for determining changes in water yield resulting from alterations in vegetation. J Hydrol 310:28–61

Burch H, Forster F, Schleppi P (1996) Zum Einfluss des Waldes auf die Hydrologie der Flysch-Einzugsgebiete des Alptals. Schweiz Z Forstwes 147:925–938

Burger H (1943) Einfluss des Waldes aud den Stand der Gewässer, 3. Mitteilung: Der Wasserhaushalt im Sperbel- und Rappengraben von 1927/28 bis 1941/42. Mitt Schweiz Anst Forstl Versuchswes 23:167–222

Burger H (1954) Einfluss des Waldes aud den Stand der Gewässer, 5. Mitteilung: Der Wasserhaushalt im Sperbel- und Rappengraben von 1942/43 bis 1951/52. Mitt Schweiz Anst Forstl Versuchswes 35:179–224

Chang M (2002) Forest hydrology. An introduction to water and forests. CRC, Boca Raton, FL, p 474

Christophersen N, Robson A, Neal C, Whitehead PC, Vigerust B, Henriksen A (1990) Evidence for long-term deterioration of streamwater chemistry and soil acidification at the Birkenes catchment, southern Norway. J Hydrol 116:63–76

Church MR (1997) Hydrochemistry of forested catchments. Annu Rev Earth Planet Sci 25:23–59

Colman EA (1953) Vegetation and watershed management: an apraisal of vegetation management in relation to water supply, flood control and soil erosion. Ronald Press, New York, p 412

Cosandey C (1995) La forêt réduit-elle l'écoulement annuel? Ann Géogr 104:7–25

Davis JS (1986) Improving information utilization of data from rivers and streams – the role of seasonal factors and annual periodicity in the variance of biogeochemical parameters. Trends Anal Chem 5:247–251

Dise NB, Wright RF (1995) Nitrogen leaching from European forests in relation to nitrogen deposition. For Ecol Manag 71:153–161

Douglass WT, Swank JE (1974) Streamflow greatly reduced by converting deciduous hardwood stands to pine. Science 185:857–859

Draaijers GPJ, van Ek R, Meijers R (1992) Research on the impact of forest stand structure on atmospheric deposition. Environ Pollut 75:243–249

Driscoll CT, Likens GE, Hedin LO, Bormann FH (1989) Changes in the chemistry of surface waters. Environ Sci Technol 23:1028–1028

Engler A (1919) Untersuchungen über den Einfluss des Waldes aud den Stand der Gewässer. Mitt Schweiz Zentralanst Forstl Versuchswes 12:1–626

FAO (2005) Forests and floods - Drowning in fiction or thriving on facts? RAP Publication 2005/03, FAO, Bangkok. Thailand & CIFOR, Bogor Barat, Indonesia

Fenn ME, Poth MA, Aber JD, Baron JS, Bormann BT, Johnson DW, Lemly AD, McNulty SG, Ryan DE, Stottlemyer R (1998) Nitrogen excess in North American ecosystems: predisposing factors, ecosystem responses, and management strategies. Ecol Appl 8:706–733

Folster J, Bishop K, Kram P, Kvarnas H, Wilander A (2003) series of long-term annual fluxes in the streamwater of nine forest catchments from the Swedish environmental monitoring program (PMK 5). Sci Total Environ 310:113–120

Frijns E, Tietema A (2002) Nitrogen cycling following clear-cut harvest of a nitrogen limited Scots pine and a nitrogen saturated Douglas fir stand in the Netherlands. BIOGEOMON – 4th international symposium on ecosystem behaviour, University of Reading, UK, p 69 (abstract)

German P (1994) Do forests control run-off? Beitr Hydrol Schweiz 35:105–110

Gundersen P, Emmett BA, Kjønaas OJ, Koopmans CJ, Tietema A (1998) Impact of nitrogen deposition on nitrogen cycling in forests: a synthesis of NITREX data. For Ecol Manag 101:37–55

Hagedorn F, Schleppi P, Bucher JB, Flühler H (2001) Retention and leaching of elevated N deposition in a forested ecosystem with Gleysols. Water Air Soil Pollut 129:119–142

Hegg C (2006) Waldwirkung auf Hochwasser. Ber Bayer Landesanst Wald Forstwirtsch 55:29–33

Hibbert AR (1967) Forest treatment effects on water yield. In: Sopper WE, Lull HW (eds) Forest hydrology. Pergamon, Oxford, pp 527–543

Hibbert AR (1971) Increases in streamflow after converting chaparral to grass. Water Resour Res 7:71–80

Hornbeck JW, Adams MB, Corbett ES, Verry ES, Lynch JA (1993) Long-term impacts of forest treatments on water yield: a summary for northeastern USA. J Hydrol 150:323–344

Hornbeck JW, Martin CW, Eagar C (1997) Summary of water yield experiments at Hubbard Brook Experimental Forest, New Hampshire. Can J For Res 27:2043–2052

Hornung M, Roda F, Langan SJ (eds) (1990) A review of small catchment studies in Western Europe producing hydrochemical budgets. Air Poll Res Rep 28, Commission of the European Communities, Bruxelles, 186 pp

Hudson JA, Crane SB, Robinson M (1997) The impact of the growth of new plantation forestry on evaporation and streamflow in the Llanbrynmair catchments. Hydrol Earth Sys Sci 1:463–475

Ice GG, Neary DG, Adams PW (2004) Effects of wildfire on soils and watershed processes. J For 102(6):16–20

Jeandel F, Cantégril JB, Bellaud L (1862) Etudes expérimentales sur les inondations. Bureau des Annales Forestières, Paris, p 144

Jenkins A, Peters NE, Rodhe A (1994) Hydrology. In: Moldan B, Černý J (eds) Biogeochemistry of small catchments: a tool for environmental research. Wiley, New York, pp 31–54

Jobbágy EG, Jackson RB (2004) The uplift of soil nutrients by plants: biogeochemical consequences across scales. Ecology 85:2380–2389

Johnson R (1998) The forest cycle and low river flows: a review of UK and international studies. For Ecol Manag 109:1–7

Keller HM (1985) Die hydrologische Forschung an der EAFV seit 1889. Mitt Eidgenöss Forsch Anst Wald Schnee Landsch 61:746–756

Keller HM (1988) European experiences in long-term forest hydrology research. In: Swank WT, Crossley DA Jr (eds) Forest hydrology and ecology at Coweeta. Springer, New York, pp 407–414

Knoepp JD, Vose JM, Swank WT (2008) Nitrogen deposition and cycling across an elevation and vegetation gradient in southern Appalachian forests. Int J Environ Stud 65:389–408

Küchli C, Meylan B (2002) Wälder liefern das beste Trinkwasser. Wald Holz 10:52–54

Hamilton LS (1985) Overcoming myths about soil and water impacts of tropical forest land uses. In: El-Swaify SA, Moldenhauer WC, Lo A (eds) Soil erosion and conservation. Soil Conservation Society of America, Ankeny, IA, pp 680–690

LaMalfa EM, Ryle R (2008) Differential snowpack accumulation and water dynamics in aspen and conifer communities: implications for water yield and ecosystem function. Ecosystems 11:569–581

Lange B, Lüscher P, Germann PF (2009) Significance of tree roots for preferential infiltration in stagnic soils. Hydrol Earth Syst Sci 13:1809–1821

Lamontagne S, Carignan R, D'Arcy P, Prairie YT, Paré D (2000) Element export in runoff from eastern Canadian Boreal Shield drainage basins following forest harvesting and wildfires. Can J Fish Aquat Sci 57(suppl 2):118–128

Landsberg JD, Tiedemann AJ (2000) Fire management. In: Dissmeyer GE (ed) Drinking water from forests and grasslands – a synthesis of the scientific literature. Gen Tech Rep 39. USDA Southern Research Station, Asheville, NC, pp 124–138

Langford KJ (1976) Change in yield of water following a bushfire in a forest of *Eucalyptus regnans*. J Hydrol 29:87–114

Laudon H, Norton SA (2010) Drivers and evolution of episodic acidification at the Bear Brook Watershed in Maine, USA. Environ Monit Assess, DOI 10.1007/s10661-010-1526-0

Lavabre J, Sempere Torres D, Cernesson F (1993) Changes in the hydrological response of a small Mediterranean basin a year after a wildfire. J Hydrol 142:273–299

Lelong F, Dupraz C, Durand P, Didon-Lescot JF (1990) Effects of vegetation type on the biogeochemistry of small catchments (Mont Lozere, France). J Hydrol 116:125–145

Leuppi E, Forster F (1990) Zur Frage der Wirksamkeit des Waldes für den Hochwasserschutz – ein Beispiel aus dem oberen Reusstal. Schweiz Z Forstwes 141:943–954

Likens GE, Bormann FH (1995) Biogeochemistry of a forested ecosystem, 2nd edn. Springer, New York, p 159

Lindenmayer DB, Likens GE (2009) Adaptive monitoring: a new paradigm for long-term research and monitoring. Trends Ecol Evol 24:482–486

Likens GE, Bormann FH, Johnson NM, Fisher DW, Pierce RS (1970) Effects of forest cutting and herbicide treatment on nutrient budgets in the Hubbard Brook watershed-ecosystem. Ecol Monogr 40:23–47

López-Moreno J, Stähli M (2008) Statistical analysis of the snow cover variability in a subalpine watershed: assessing the role of topography and forest interactions. J Hydrol 348:379–394

Lovett GM, Reiners WA (1986) Canopy structure and cloud water deposition in subalpine coniferous forests. Tellus 38B:319–327

Lynch JA, Corbett ES (1990) Evaluation of best management practices for controlling nonpoint pollution from silvicultural operations. J Am Water Resour Assoc 26:41–52

Mariotté E (1674) L'origine des fontaines. Paris

McBride M, Hession WC, Rizzo DM (2008) Riparian reforestation and channel change: a case study of two small tributaries to Sleepers River, northeastern Vermont, USA. Geomorphology 102:445–459

McGuinness JL, Harrold LL (1971) Reforestation influences on small watershed streamflow. Water Resour Res 7:845–852

Mitchell MJ, Driscoll CT, Likens KJS, GE MPS, Pardo LH (1996) Climatic control of nitrate loss from forested watersheds in the northeast United States. Environ Sci Technol 30:2609–2612

Moldan B, Černý J (1994) Small catchment research. In: Moldan B, Černý J (eds) Biogeochemistry of small catchments: a tool for environmental research. Wiley, New York, pp 1–29

Moldan F, Kjønaas OJ, Stuanes AO, Wright RF (2006) Increased nitrogen in runoff and soil following 13 years of experimentally increased nitrogen deposition to a coniferous-forested catchment at Gårdsjön, Sweden. Environ Pollut 144:610–620

Molchanov AA (1963) Gidrologicheskaya Rol'lesa. Academy of Science USSR, Moscow, p 488, Russian

Morier I, Schleppi P, Saurer M, Providoli I, Guenat C (2010) Retention and hydrolysable fraction of atmospherically deposited nitrogen in two contrasting forest soils in Switzerland. Eur J Soil Sci 61:197–206

Mundy SC, Nathan RJ, Daamen CC, Cornish PM (2001) Development and application of a model to assess the impact of practical plantation forestry on water yields, MODSIM, vol 1: Natural Systems (Part One). Australian National University, Canberra, pp 449–454

Musselman KN, Molotch NP, Brooks PD (2008) Effects of vegetation on snow accumulation and ablation in a mid-latitude sub-alpine forest. Hydrol Proc 22:2767–2776

Neal C, Fisher R, Smith CJ, Hill S, Neal M, Conway T, Ryland GP, Jeffrey HA (1992) The effects of tree harvesting on stream-water quality at an acidic and acid-sensitive spruce forested area: Plynlimon, mid-Wales. J Hydrol 135:305–319

Neal C (1997) A view of water quality from the Plynlimon watershed. Hydrol Earth Sys Sci 1:743–753

Neal C, Wilkinson J, Neal M, Harrow M, Wickham H, Hill L, Morfitt C (1997) The hydrochemistry of the headwaters of the River Severn, Plynlimon. Hydrol Earth Syst Sci 1:583–617

Norton SA, Fernandez IJ, Kahl JS, Reinhardt RL (2004) Acidification trends and the evolution of neutralization mechanisms through time at the Bear Brook Watershed in Maine (BBWM), U.S.A. Water Air Soil Pollut Focus 4:289–310

Omernik JM (1977) Nonpoint source – stream nutrient level relationships: a nationwide study. EPA Ecol Res Ser EPA-600/3-77-105, U.S. Environ Prot Agency, Corvallis, 151 pp

Piégay H, Walling DE, Landon N, He Q, Liébault F, Petiot R (2004) Contemporary changes in sediment yield in an alpine mountain basin due to afforestation (the upper Drôme in France). Catena 55:183–212

Plinius Secundus G (first century AD) Naturalis historia, liber XXXI, caput XXX

Prechtel A, Alewell C, Armbruster M, Bittersohl J, Cullen JM, Evans CD, Helliwell R, Kopacek J, Marchetto A, Matzner E, Meesenburg H, Moldan F, Moritz K, Veselý J, Wright RF (2001) Response of sulphur dynamics in European catchments to decreasing sulphate deposition. Hydrol Earth Syst Sci 5:311–325

Probst A, Dambrine E, Viville D, Fritz B (1990) Influence of acid atmospheric inputs on surface-water chemistry and mineral fluxes in a declining spruce stand within a small granitic catchment (Vosges massif, France). J Hydrol 116:101–124

Rauch FA (ed) (1821) Annales Européennes de Physique Végétale et d'Économie Publique, Vol. 2. Eberhart, Paris, 491 pp

Robinson M, Gannon B, Schuch M (1991) A comparison of the hydrology of moorland under natural conditions, agricultural use and forestry. Hydrol Sci J 36:565–577

Rothe A, Huber C, Kreutzer K, Weis W (2002) Deposition and soil leaching in stands of Norway spruce and European Beech: results from the Höglwald research in comparison with other European case studies. Plant Soil 240:33–45

Rougier de la Bergerie F (1800) Mémoire et observations sur les abus de défrichements et la destruction des bois et forêts. Fournier, Auxerre, 76 pp

Rutter N, Essery R, Pomeroy J, Altimir N, Andreadis K, Baker I, Barr A, Bartlett P, Boone A,Deng H, Douville H, Dutra E, Elder K, Ellis C, Feng X, Gelfan A, Goodbody A, Gusev Y,Gustafsson D, Hellström R, Hirabayashi Y, Hirota T, Jonas T, Koren V, Kuragina A,Lettenmaier D, Li W-P, Luce C, Martin E, Nasonova O, Pumpanen J, Pyles RD, Samuelsson P, Sandells M, Schädler G, Shmakin A, Smirnova TG, Stähli M, Stöckli R, Strasser U, Su H Suzuki K, Takata K, Tanaka K, Thompson E, Vesala T, Viterbo P, Wiltshire A, Xia K, Xue Y, Yamazaki T (2009) Evaluation of forest snow processes models (SnowMIP2). J Geophys Res 114:D06111

Ryan MG, Binkley D, Fownes JG (1997) Age-related decline in forest productivity: pattern and process. Adv Ecol Res 27:213–262

Sahin V, Hall MJ (1996) The effects of afforestation and deforestation on water yields. J Hydrol 178:293–309

Sanford SE, Creed IF, Tague CL, Beall FD, Buttle JM (2007) Scale-dependence of natural variability of flow regimes in a forested landscape. Water Resour Res 43:W08411

Schleppi P, Hagedorn F, Providoli I (2004) Nitrate leaching from a mountain forest ecosystem with Gleysols subjected to experimentally increased N deposition. Water Air Soil Pollut Focus 4:453–467

Schleppi P, Muller N, Feyen H, Papritz A, Bucher JB, Flühler H (1998) Nitrogen budgets of two small experimental forested catchments at Alptal, Switzerland. For Ecol Manag 101:177–185

Schleppi P, Waldner PA, Fritschi B (2006a) Accuracy and precision of different sampling strategies and flux integration methods for runoff water: comparisons based on measurements of the electrical conductivity. Hydrol Proc 20:395–410

Schleppi P, Waldner P, Hegg C (2003) Einfluss des Waldes auf Nitrat-Gehalte im Wasser. Bündner Wald 56(4):27–30

Schleppi P, Waldner PA, Stähli M (2006b) Errors of flux integration methods for solutes in grab samples of runoff water, as compared to flow-proportional sampling. J Hydrol 319:266–281

Schwarze R, Beudert B (2009) Analyse der Hochwassergenese und des Wasserhaushalts eines bewaldeten Einzugsgebiets unter dem Einfluss eines massiven Borkenkäferbefalls. Hydrol Wasserbewirtsch 53:236–249

Serengil Y, Gökbulak F, Özhan S, Hizal A, Cĕngönül K (2007) Alteration of stream nutrient discharge with increased sedimentation due to thinning of a deciduous forest in Istanbul. For Ecol Manag 246:264–272

Shanley JB, Krám P, Hruška J, Bullen TD (2004) A biogeochemical comparison of two well-buffered catchments with contrasting histories of acid deposition. Water Air Soil Pollut Focus 4:325–342

Sørensen R, Ring E, Meili M, Högbom L, Seibert J, Grabs T, Laudon H, Bishop K (2009) Forest harvest increases runoff most during low flows in two boreal streams. Ambio 38:357–363

Soulsby C, Turnbull D, Hirst D, Langan SJ, Owen R (1997) Reversibility of stream acidification in the Cairngorm region of Scotland. J Hydrol 195:291–311

Stadler D, Wunderli H, Auckenthaler A, Flühler H (1996) Measurement of frost-induced snowmelt runoff in a forest soil. Hydrol Proc 10:1293–1304

Stähli M, Jonas T, Gustafsson D (2009) The role of snow interception in winter-time radiation processes of a coniferous sub-alpine forest. Hydrol Proc 23:2498–2512

Stednick JD (1996) Monitoring the effects of timber harvest on annual water yield. J Hydrol 176:79–95

Stednick JD (2000) Timber Management. In: Dissmeyer GE (ed) Drinking water from forests and grasslands - A synthesis of the scientific literature. USDA Forest Service, General Technical Report SRS-39, pp 103–119

Stoddard JL (1994) Long-term changes in watershed retention of nitrogen: its causes and aquatic consequences. In: Baker LA (ed) Environmental chemistry of lakes and reservoirs, vol 237, Advances in Chemical Series. American Chemical Society, Washington, pp 223–284

Stoddard JL, Jeffries DS, Lükewille A, Clair TA, Dillon PJ, Driscoll CT, Forsius M, Johannessen M, Kahl JS, Kellogg JH, Kemp A, Mannio J, Monteith DT, Murdoch PS, Patrick S, Rebsdorf A, Skjelkvåle BL, Stainton MP, Traaen T, Van Dam H, Webster KE, Wieting J, Wilander A (1999) Regional trends in aquatic recovery from acidification in North America and Europe. Nature 401:575–578

Stuanes AO, de Wit HA, Hole LR, Kaste O, Mulder J, Riise G, Wright RF (2008) Effect of climate change on flux of N and C: air-land-freshwater-marine links: synthesis. Ambio 37:2–8

Sucker C, Puhlmann H, Zirlewagen B, von Wilpert K, Feger K-H (2009) Hydrol Wasserbewirtsch 53:250–262

Sucker C, von Wilpert K, Puhlmann H (2010) Water quality in forested catchments: trends in stream acidification. Environ Monit Assess (accepted for publication)

Swank WT, Johnson CE (1994) Small catchment research in the evaluation and development of forest management practices. In: Moldan B, Černý J (eds) Biogeochemistry of small catchments: a tool for environmental research. Wiley, New York, pp 383–408

Swank WT, Swift LW, Douglass JE (1988) Streamflow changes associated with forest cutting, species conversions, and natural disturbances. In: Swank WT, Crossley DA (eds) Forest Hydrology and Ecology at Coweeta. Ecol Stud 66:297–312

Swank WT, Vose JM (1997) Long-term nitrogen dynamics of Coweeta forested watersheds in the southeastern United States of America. Global Biogeochem Cycles 11:657–671

Swank WT, Waide JB, Crossley DA, Todd RL (1981) Insect defoliation enhances nitrate export from forest ecosystems. Oecologia 51:297–299

Swanson FJ, Scatena FN, Dissmeyer GE, Fenn ME, Verry ES, Lynch JA (2000) Watershed processes – fluxes of water, dissolved constituents and sediments. In: Dissmeyer GE (ed) Drinking water from forests and grasslands – a synthesis of the scientific literature. Gen Tech Rep 39. USDA Southern Research Station, Asheville, CA, pp 26–41

Thorne JF, Anderson JE, Horiuchi KM (1988) Cation cycling in a base-poor and base-rich Northern hardwood forest ecosystem. J Environ Qual 17:95–101

Tipping E, Bettney R, Hurley MA, Isgren F, James JB, Lawlor AJ, Lofts S, Rigg E, Simon BM, Smith EJ, Woof C (2000) Reversal of acidification in tributaries of the River Duddon (English Lake District) between 1970 and 1998. Environ Pollut 109:183–191

Troendle CA, King RM (1985) The effect of timber harvest on the Fool Creek watershed, 30 years later. Water Resour Res 21:1915–1922

Udawatta RP, Krstansky JJ, Henderson GS, Garrett HE (2002) Agroforestry practices, runoff, and nutrient loss: a paired watershed comparison. J Environ Qual 31:1214–1225

Van Breemen N, Driscoll CT, Mulder J (1984) Acidic deposition and internal proton sources in acidification of soils and waters. Nature 307:599–604

Van Breemen N, Jenkins A, Wright RF, Beerling DJ, Arp WJ, Berendse F, Beier C, Collins R, van Dam D, Rasmussen L, Verburg PSJ, Wills MA (1998) Impacts of elevated carbon dioxide and temperature on a boreal forest ecosystem (CLIMEX project). Ecosystems 1:345–351

Van Breemen N, Wright RF (2004) History and prospect of catchment biogeochemistry: a European perspective based on acid rain. Ecology 85:2263–2268

Verry ES (2003) Ground water and small research basins: an historical perspective. Ground Water 41:1005–1007

Veselý J, Majer V, Norton SA (2002) Heterogeneous response of central European streams to decreased acidic atmospheric deposition. Environ Pollut 120:275–281

Wenger W (2002) Bedeutung des Waldes für die Trinkwassergewinnung. LWF Aktuell 34(11):21–24

Whitehead PG, Robinson M (1993) Experimental basin studies – an international and historical perspective of forest impacts. J Hydrol 145:217–230

Wright RF, Lotse E, Semb A (1993) RAIN project: results after 8 years of experimentally reduced acid deposition. Can J Fish Aquat Sci 50:258–268

Wright RF, Tietema A (1995) Ecosystem response to 9 years of nitrogen addition at Sogndal, Norway. For Ecol Manag 71:133–142

Yoder BJ, Ryan MG, Waring RH, Schoettle AW, Kaufmann MR (1994) Evidence of reduced photosynthetic rates in old trees. For Sci 40:513–527

Chapter 6
Forest Development in the European Alps and Potential Consequences on Hydrological Regime

Christine Alewell and Peter Bebi

Introduction

Mountains cover roughly one quarter of the planet's surface and are known as the Earth's water towers (Weingartner et al. 2003). They do not only produce a surplus of water that is transported to neighboring lowlands, bringing vital benefit for life, but also hold dangers as a destructive element in the form of floods (Weingartner et al. 2003). In view of their hydrological significance, mountain regions present a paradox: although they provide the bulk of the world's water resources, knowledge of these resources is generally much less extensive, reliable and precise than for other physiographic regions (Rodda 1994). Hydrological processes in mountains are strongly influenced by snow and snow melting processes, relatively frequent precipitation events with greater total volume compared to lowlands, decreasing soil depth with increasing altitude resulting in quick hydrological responses, relatively short vegetation growth with comparably low evapotranspiration rates, and steep slopes. All of the latter favor the susceptibility for floods and peak runoff events. At mid-latitudes, the elevational zone between 1,000 and 2,000 m a.s.l. has been identified as being particularly sensitive because effective snow storage reduces the flood hazard at altitudes above 2,000 m (Weingartner et al. 2003). For Alpine countries flood protection and water management play a key role in the overall hazard management and risk prevention.

Changes in vegetation and land use management may drastically affect stability of mountain ecosystems and their hydrology. On a global scale, changes in total forest cover have resulted in a net forest loss of 7.3 million hectares per year between 1990 and 2005 (FAO 2007). While deforestation (please see Box 6.1. for terminology) has occurred mainly in the tropics and outside of mountain ranges, a significant forest

---

C. Alewell (✉)
Institute of Environmental Geosciences, University of Basel, Switzerland
e-mail: christine.alewell@unibas.ch

P. Bebi
Swiss Fedral Institute for Forest, Snow and Landscape Research, WSL Davos, Switzerland
e-mail: bebi@slf.ch

M. Bredemeier et al. (eds.), *Forest Management and the Water Cycle: An Ecosystem-Based Approach*, Ecological Studies 212, DOI 10.1007/978-90-481-9834-4_6,

Box 6.1 Terminology

Forest growth: forest (or shrub) growth without intervention by humans in areas where naturally no trees were growing (it might or might not be due to climate change)
Forest regrowth: forest (or shrub) regrowth without intervention by humans in areas where trees have already been growing in former times (it might or might not be due to climate change)
Afforestation: planting of trees by humans in areas where naturally no trees were growing
Reforestation: planting of trees by humans in areas where trees have already been growing in former times

expansion has been observed during the twentieth century in several mountain ranges of developed countries including the Alps (Gellrich et al. 2007; Tasser et al. 2007), the Pyrenees (Vicente-Serrano et al. 2004), the Appennines (Piégay et al. 2004), the Carpathians (Kozak 2003), the British Uplands (Bunce 1991), and the Appalachian Highlands (Wear and Bolstad 1998). Responsible for the forest cover increases is mainly the marginalisation of agricultural areas (Baldock et al. 1996). The natural regrowth of forests in industrialized countries during the last decades is the expression of a profound economic and social change where resource scarcities (food, soil, labour, energy) have fundamentally changed. The return of forests accompanies the decrease of poverty and the respective growth of welfare (Baur et al. 2006).

Forest cover changes have drastic effects on a large number of ecosystem goods and services in the Alps (Schröter et al. 2005). Beside changes in their role as habitats for various plant and animal species, for wood production and recreation, forest development in mountainous ecosystems is also expected to influence hydrological processes, soil erosion and the occurrence of natural hazards such as rockfall, snow avalanches and shallow landslides (García-Ruiza et al. 1995; McDonald et al. 2000; Bebi et al. 2009).

There is overall agreement that climate as well as land use change in the Alps will be rapid and strong compared to other European regions, and that mountain areas will be the most vulnerable regions of Europe (Begert et al. 2005; Schröter et al. 2005). However, there is considerable uncertainty about the effect of these changes on alpine vegetation cover and especially on forest development, shrub encroachment, stability of soils and slopes and the concomitant effects on hydrological regime. From an ecological point of view, land-use abandonment and subsequent forest regrowth have a lot of positive impacts such as lowered inputs of pesticides and fertilizers, provision of clean water and an increase in soil stability. Other effects, like the loss of biodiversity, loss of cultural heritage elements and the change in landscape aesthetics have been controversially discussed (McDonald et al. 2000; Conti and Fagarazzi 2005). An important, but yet not well investigated or discussed change will be the change in hydrological regime and dynamics. In this contribution, we review the forest development in the European Alps with special emphasis on the potential consequences for the hydrological regime of alpine catchments.

Development of Forested Areas and Shrub Encroachment in the Alps

Naturally Alpine treelines seem to have been relatively stable throughout the Holocene with not more than ±100 m fluctuations (Heiri et al. 2006). Nevertheless, human land use already influenced the tree line and forested areas in the Alps since the early Holocene and lowered the treeline considerably since 4500 BP (Veit 2002; Heiri et al. 2006). Forested areas decreased in many alpine countries dramatically between the tenth and nineteenth centuries for firewood collection, construction, industrial needs and agricultural use and has led to severe soil erosion (Bundesamt für Statistik 2001; Descroix and Gautier 2002; Liébault et al. 2005; Fritsche et al. 2006). A representative land cover analysis of the Eastern Alps for the late 1970s yielded in a total of 26% area transformed from forested areas into anthropogenic grasslands (Körner 1989). Since the late nineteenth century the forested area is increasing again because reforestation was promoted for flood protection and erosion control or due to land abandonment. In the European Alps, 40% of all farm holdings were abandoned within 20 years (1980–2000). In addition, almost 70% of the farms that are still in operation today are run only as a secondary source of income (Tappeiner et al. 2003). In terms of land use, this means that an average of 20%, in some areas even as much as 70% of the agricultural land of the Alps has been abandoned (Tappeiner et al. 2003). The areas which have been abandoned were mostly former traditional land use types such as larch meadows, less intensively used meadows of the subalpine and alpine belt as well as the inaccessible and steep areas (Bundesamt für Statistik 2001; Tasser and Tappeiner 2002; Verbunt et al. 2005). After abandonment, vegetation succession occurs at varying rates that are dependent on site conditions (climatic factors, topography, soil degradational status, increasing with increasing soil depth), tree species, proximity of mature seeding trees, pressure by snow gliding and avalanches (Tasser et al. 2003). In Switzerland, forested area increased from 18% to 31% between 1855 and 1997 (Ritzmann-Blickenstorfer 1996; Bundesamt für Statistik 2001; Descroix and Gautier 2002; Piégay et al. 2004) with most of this change occurring in the Alps, and with a high regional heterogeneity in forest expansion within the Alps (Fig. 6.1). Even though changes in tree line dynamics in the Alps are a result of both, land-use and climate change, there is overall agreement that the abandonment of alpine meadows is the main driving factor for the increase in wooded area (Tasser and Tappeiner 2002; Krausmann et al. 2003; Heiri et al. 2006; Bigler et al. 2006; Bolli et al. 2007; Gehrig-Fasel et al. 2007; Tenhunen et al. 2009). However, when discussing the relative importance of climate versus land use change in affecting vegetation cover and forest regrowth, not only the expected upward shift of treeline with increase in temperature has to be considered.

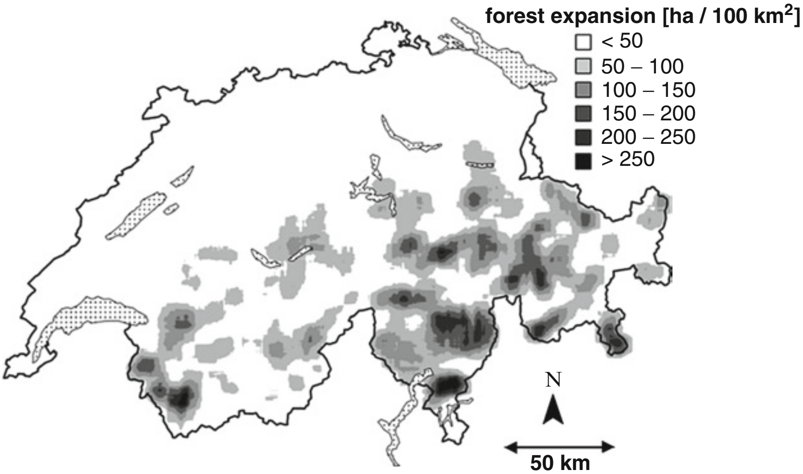


Fig. 6.1 Spatial distribution of forest expansion in the Swiss Alps between 1985 and 1997. Calculations are based on 1-ha raster inventory data of the Swiss Areal Statistics

Any change in snow dynamics (amount, duration and timing of the snow-pack) can have long lasting environmental and economic consequences (Beniston et al. 2003). Especially changes in the length of the snow season can have a strong impact on growth and survival rates of numerous plant species and are thus likely to be one of the most significant factors for alpine vegetation (Jones et al. 2001; Körner 2005; Cannone et al. 2007).

A quantitative analysis of forest expansion in the Swiss Alps based on inventory data has confirmed that natural regrowth of forests occurred mainly where the costs of cultivation might not be covered by yields. Natural regrowth occurred thus mainly in high elevated, steep, poorly accessible sites. This is consistent with the observation that about two thirds of the abandoned areas were alpine pastures, where only 4% of direct agricultural subsidies are invested (Baur et al. 2006).

The speed of forest succession may vary considerably according to regional climate, former land-use and different autogenic or allogenic factors. Long-term average return interval between successful tree establishment in the European Alps after land abandonment ranges from 15 to 100 years (Bugmann 2001; Bolli et al. 2007). Rapid succession processes after land abandonment are mainly an effect of favorable climatic conditions and are for example observed in the southern (insubric) valleys of the Swiss Alps (Walther and Julen 1986). Slow pathways are most often observed where exposure to grazing or to climatic factors are extreme (Schütz et al 2000) or where formation of dwarf shrubs build relatively stabile successional stages (Zoller et al 1984).

The future regrowth of forested areas might in some areas be impeded by the combined effects of increased insect disturbance and increased risk of droughts and wildfires under future climate change (Alewell and Bredemeier 2006;

Bigler et al. 2006; Schumacher and Bugmann 2006). In contrast, shrub encroachment may increase more rapidly as has been shown for the case study in the Urseren valley (30% increase of area covered by Alnus viridis between 1959 and 2004; Meusburger and Alewell 2008, 2009). Furthermore, spatially variable socio-economic and natural factors of forest development add to a pattern of forest expansion and forest development, which may considerably vary for different regions of the Alps (Gellrich et al. 2007).

Forests and forest regrowth have strong local and regional effects on climate (discussion see below). Vice versa, climate change affects forests and forest regrowth. Globally, forests are under tremendous pressure from global change (Bonan 2008), but mountain forests might be the winner of the combined effects of land use change (discussion see above) and climate change. Climate change is discussed to cause an upwards shift in alpine treelines. However, the observed treeline dynamics in the Alps are the results of both, land use and climate change with land use change having most likely the greater effect: Tree establishment is triggered by the abandonment of the agricultural use of the meadow, but then strongly favored by particular good growing seasons under warm conditions (Bolli et al. 2007).

Development of Soil Stability in Alpine Regions

Predicted and observed climate and land use change can be expected to have significant detrimental effect on soil stability in alpine environments. Soil stability and forest expansion are closely linked since soil degradation may hamper forest expansion but vice versa forest expansion might increase soil stability. Deforestation generally leads to a dramatic increase in natural rates of erosion and sediment production (Liébault et al. 2005). Furthermore, soil stability is related to hydrological regime, since degraded soils have increased runoff and lower evapotranspiration.

Rising air temperature, and especially the increasing number of days with air temperatures above zero, influence the occurrence of snowfall and time of snowmelt (Birsan et al. 2005), resulting in a rising snow line. At the same time snow melting is reported to occur earlier in spring due to rising temperatures but with no obvious shift in autumn (Laternser and Schneebeli 2003; Jasper et al. 2004; Beniston 2006; Horton et al. 2006). This indicates a higher amount of precipitation in the form of rainfall and runoff in winter and spring with potentially increasing soil erosion during times of sparse or no vegetation cover (Fuhrer et al. 2006). A permanent snow cover in winter protects the soil from freezing which is important for snowmelt infiltration and runoff, respectively (Stähli et al. 2001; Bayard et al. 2005). Thus, with early snow melt, changes in freezing/thawing cycles can be expected to increase soil erosion and mass movement because of low vegetation cover in early spring, but also due to changes in permafrost occurrence. Changes in snowmelt are considered to be mainly critical for altitudes between 500–800 (in winter) and 1,000–1,500 m a.s.l. (in spring) (Wielke et al. 2004). These have also been shown to be the most critical areas for increase in erosion damage due to land use intensification of lower, more accessible slopes (see below, Alewell et al. 2008; Meusburger and Alewell 2008, 2009).

Droughts are expected to be more frequent under future climate change (Brunetti et al. 2006; Fuhrer et al. 2006). Droughts are not only affecting vegetation stability but can furthermore cause the soil surfaces to become hydrophobic. Thus, the infiltration rates decrease and surface runoff increases. In combination with increased heavy rain events following such droughts, this might cause increases in soil erosion rates. Precipitation changes in winter are predominantly due to an increase in number, frequency and extension of heavy precipitation events (Schmidli and Frei 2005; Fuhrer et al. 2006). Torrential rain is widely recognized as considerably contributing to erosion (Summer 1989; Acornley and Sear 1999; Fraser et al. 1999; Descroix and Gautier 2002) especially in winter or spring when soils have low or no vegetation cover.

Even though there is no doubt that climate change, and especially the change in snow dynamics and the increase in intensity and duration of torrential rain events contributes to soil degradation, land use intensification and change in management practice of the lower, more easily accessible slopes was found to be the main driving force of increase in soil erosion (Alewell et al. 2008; Meusburger and Alewell 2008, 2009). A case study in the Urseren Valley (Canton Uri, Switzerland) with an analysis of aerial photographs between 1959 and 2004 has revealed a 92% increase of the area affected by landslides (Fig. 6.2; Meusburger and Alewell 2008, 2009).

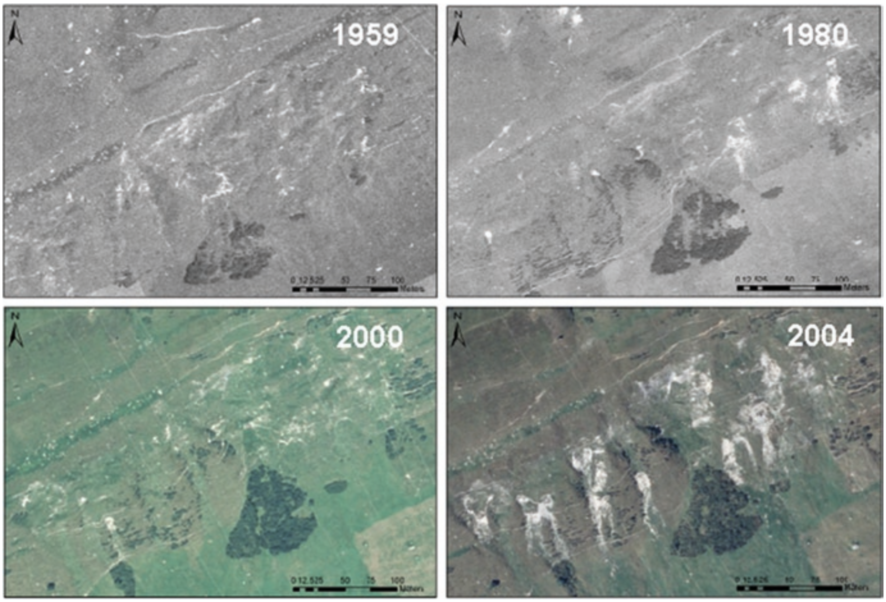


Fig. 6.2 Increase in land slide affected area in the Urseren Valley, Kanton Uri, Switzerland (Data from Meusburger and Alewell 2008)

With respect to land use change, the erosion risk in lowlands and in pre-alpine areas is assumed to have mainly increased during the last century (Scheurer et al. 2009). In alpine areas, it is not yet clear whether we can expect a reduced erosion risk due to reforestation or if the abandonment of sites will result in destabilization and an increased erosion risk (Tasser et al. 2003; Scheurer et al. 2009). While Tasser et al. (2003) found an increase in landslide susceptibility in abandoned areas which are invaded by shrubs or bushes (*Alnetum viridis*, *Calluna vulgaris*, *Junipero-Arctostaphyletum*), Meusburger and Alewell (2008) found lower landslide densities in areas invaded by *Alnus virids* in the past 45 years. It is likely that some alpine grasslands show a destabilized phase during the succession from grassland soils to scrub or forest.

There is no doubt that stable vegetation cover is one of the best protection measures to shield soils and slopes from degradation through erosion (e.g, Isselin-Nondedeu and Bédécarrats 2007). Thus, the development of vegetation and especially the development of forested areas are a crucial parameter when predicting soil erosion and slope instability.

Effects of Change in Forested Area on Hydrology

When discussing effects of forest area on hydrological regime, the effect of forests on climate in general is important. The worlds' forests influence climate warming positively through decreased albedo (= increased solar heating of land compared to crop, grassland, bare soil and snow) and negatively through carbon sequestration and evaporative cooling including feedbacks such as increased cloud formation (Bonan 2008). The net effect of forests on climate is unclear and will differ between climate zones. Boreal forests are considered a warming of climate mainly due to low surface albedo during the snow season, which offsets the negative climate forcing due to carbon sequestration (Bonan 2008). The latter might be extrapolated to winter time conditions of alpine forests even though scientific evidence is missing so far. Temperate forests have been shown to have lower surface radiative temperature due to greater aerodynamic conductance and evaporative cooling compared to grasslands (Bonan 2008). We suggest that the latter might also apply for summer time alpine conditions. The recent discussion about a regrowth of forests has often focused on a mere consideration of carbon sequestration and an assumed climate benefit. However, the neglect of other crucial impacts such us alteration of hydrological dynamic and changes in albedo might miss important environmental consequences.

While there is general agreement that forests reduce flood flow, the discussion is controversial about the effect of forests on overall runoff amount and base flow conditions (Chapter 5, this volume). The "forest hydrological hypothesis" states that forests increase base flow (German 1994; Weingartner et al. 2003). However, two studies in Alpine catchments concluded that surface runoff can be considered to be generally lower in forests due to high infiltration rates of humus layers and higher tree evapotranspiration compared to grasslands (Ammer et al. 1995; Leuppi and

Forster 1990; Leuppi et al. 1990). Tasser et al. (2005) concludes that a reforestation of formerly cultivated areas would result in an increase in evapotranspiration and a reduction of runoff by 7–52%. A comparison of alpine grasslands with alpine spruce forest in the Urseren Valley (Canton Uri, Switzerland) resulted in significantly higher simulated evapotranspiration for the forest (34% or 538 mm compared to 18% or 310 mm for forests and grasslands, respectively, Leuppi et al. 1990) and concomitant differences in runoff regime. Furthermore, simulated monthly runoff was not only generally lower throughout most of the year but maximum runoff rates due to snow melt are later in the year (June compared to early May) because snow melt is generally delayed and dampened by higher shielding of radiation in the forests. Thus, forests provide protection against peak runoff events especially during the critical time period of snow melt. Greatest relative differences between evapotranspiration rates of mountain forests in the Alps and grasslands were simulated in early spring when spruce forests are already transpiring while grasslands are still under closed snow cover and in winter when precipitation interception is effective in forests and/or forests are transpiring during warmer winter days while for the grassland only evaporation of the snow layer is active (Leuppi et al. 1990).

The highest probability for floods in the Swiss Alps is in late summer/ early fall (August–September; Weingartner et al. 2003). A runoff simulation of the detrimental rain storm event of August 1987 resulted in 22% decrease in runoff (104 mm compared to 134 mm) with a simulated reforested south slope of the Urseren Valley (Leuppi et al. 1990). However, Leuppi et al. (1990) pointed out that the net effect of a simulated reforestation for the whole catchment at three investigated alpine sites on the reduction of peak runoff was relatively small because only 15–20% of the catchments could be reforested under today's climatic conditions. No clear correlation between percentage of forest cover and specific mean annual flood discharge was found for mesoscale catchments (10–500 km^2, n = 37; Aschwanden and Spreafico 1995) because other factors like slope, soil characteristics, altitude, precipitation and snow dynamics play into this relationship. Nevertheless, the above discussion implies that a significant influence can be expected within a given catchment once percent of forest cover changes. At catchments above the tree line, up to 90% of the precipitation leaves the ecosystem by runoff (Schädler and Weingartner 2002). With increasing areas of forest, a decrease in erosion and sediment production was reported in different catchments in the Southern French Alps (Piégay et al. 2004; Liébault et al. 2005). Erosion rates in undisturbed forests are generally low as soil is stabilized by roots and protected against raindrops by leaves and humus layer (Wischmeier and Smith 1978; Bundesamt für Statistik 2001). In contrast, deforested areas yield higher runoff, and instable soils with low or non vegetation cover will increase percent runoff on total water balance. From forest clear cutting, planned hill slope afforestation as well as hydrological modeling studies it became obvious that the change in vegetation cover may have a tremendous effect on hydrological processes (water balance, floods), sediment production and mudflows (WSL 2005; Piégay et al. 2004).

In the eleventh century almost all forests in the Urseren Valley (Canton Uri, Switzerland) had already been converted into grasslands. The subsequent, detrimental consequences have been increases in avalanches, floods and severe landslides and led to the first written directive across the Alps, the so-called Waldbannbrief of 1397. The letter aimed at preserving and protecting the remaining forest islands in the Urseren Valley. Modeling of re-afforestation scenarios in the Urseren Valley indicated a 10% decrease in runoff for a water saturated soil situation and a 57% decrease for a dry soil situation (Leuppi and Forster 1990). Leuppi and Forster (1990) conclude that a complete regrowth of forests in the Urseren Valley within forests climatic boundaries would only insignificantly influence hydrograph dynamics because of the relatively small area of regrowth compared to forest free area above the tree line. Long term measurements of the Reuss catchment in the Urseren Valley (runoff data since 1904) indicate a reduction by about 30% during the last 10 years, particularly during the summer months, compared with long term means over the last 60 years (BAFU 2007). However, it is unclear to what extend these runoff changes are caused by shrub encroachment (30% increase of shrub area between1959 and 2004), soil degradation (Meusburger and Alewell 2008), by climate change or by changes in the water ways and management.

Afforestation in the southern French Alps induced a decrease in hydrograph peak events and duration increases during the twentieth century because of increase in water retention capacity of forested areas (Piégay et al. 2004). Even though the beneficial effects of reforestation or afforestation on decreased sediment production, land slide and flood risk has been shown in a variety of studies (for overview see Piégay et al. 2004 and Liébault et al. 2005), lower runoff coefficients and sediment supply comes along with negative side effects like channel narrowing and deepening and shortened period of soil moisture surplus (Pearce et al. 1987). The latter might cause water supply deficiencies in times of increased drought stress under future climate change. Large-scale reforestation often induces a proportionally greater decline in sediment supply than in runoff (Liébault et al. 2005). The reduction in sediment supply hampers replenishment of gravel bars and induces formation of forested terraces along gravel-bed rivers narrowing channel beds which may lead to further incision (Liébault et al. 2005). Channel narrowing and deepening has been observed as a typical morphology of reforested headwater catchments (Liébault et al. 2005; Piégay et al. 2004). Even though channel narrowing and incision might generally be connected to increased flood risk, this might not be true for headwater catchments at least not to a dramatic extent. However, if this phenomenon extends to rural and urban areas it might result in undercutting and destabilizing of dikes causing increased flood risks (Landon et al. 1995, 1998 quoted from Piégay et al. 2004). Headwater channel incision might also temporarily increase the supply of sediment to main rivers, thereby delaying the impact of reforestation on the fluvial system further down stream (Liébault et al. 2005). Apart from hydrological concerns, the disappearance of the braided pattern of headwater streams, which is predicted to be followed with the same trend by the main rivers in the coming decades, might raise concerns about biodiversity and nature conservation. However, the latter issues are beyond the scope of this paper.

Influences of Different Site Conditions and Forest Types

It is useful for further discussion and management options to delineate simple, straight forward cause–effect relationships between forest dynamics and hydrological regimes (Fig. 6.3). However, we have to keep in mind that the influence of forest cover on runoff is highly variable according to site conditions and forest type. Bosch and Hewlett (1982), who reviewed 94 catchment studies from different locations worldwide (but with a focus on catchments in the USA), found a large variability in the effect of the forest on runoff. With one exception, an increase in forest cover had a decreasing influence and a decrease in forest cover had an increasing influence on annual runoff. However, this influence was higher for coniferous forests (on average 40 mm change in water yield per 10% change in forest cover) compared with deciduous hardwoods (ca. 25 mm) and shrubs (ca. 10 mm).

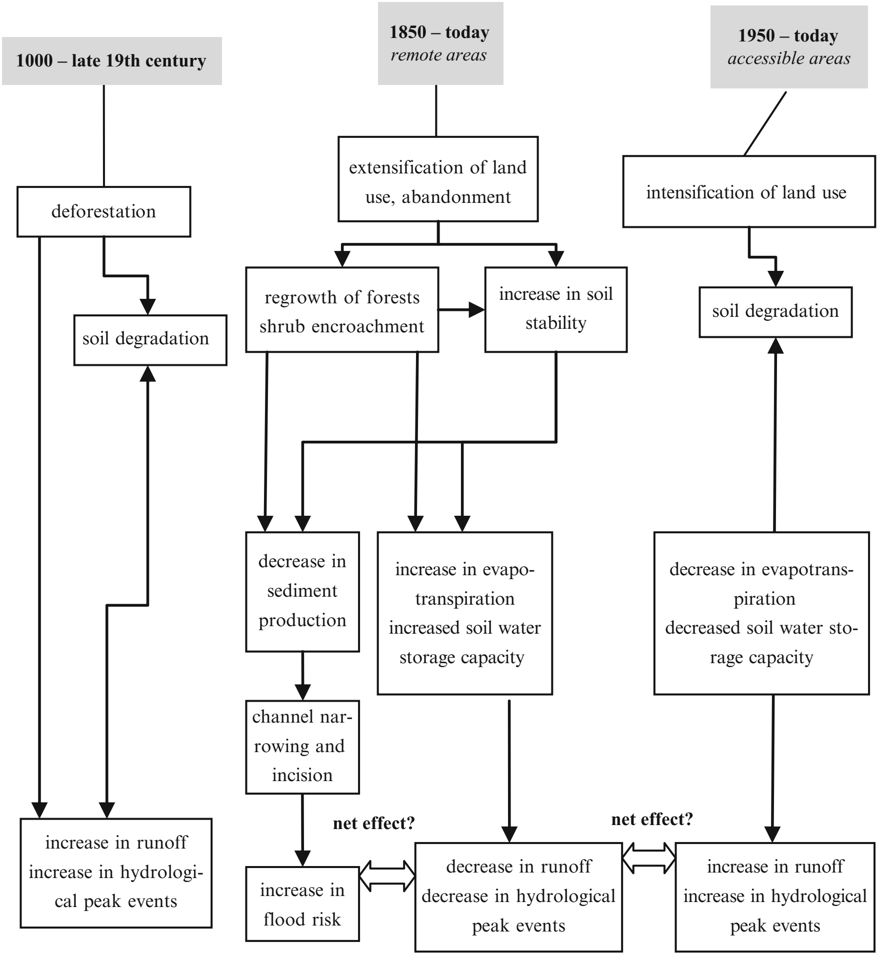


Fig. 6.3 Synthesis hypotheses on forest water interactions in the European Alps

Various authors associate the influence of forests on the hydrological regime and on flood hazard with soil properties (Chang 2003; Weinmeister 2003; Badoux et al. 2006). Forest soils have generally a larger water storage capacity than agricultural land due to a higher content of organic material, less compaction and usually more porous soil structure up to larger rooted depths (Badoux et al. 2006). However, these differences may be negligible under certain soil conditions, such as shallow and wet soils lying on impermeable bedrock (Burch et al. 1996). Accordingly, the influence of different forest types and tree species and thus the influence of forest management on hydrological regimes is also highly dependent on soil type and soil permeability (Frehner et al. 2005).

Duration and depth of snowpack are influenced by both, climatic changes and changes in forest cover. Expansion of forest cover and forest cover density have generally an increasing effect on snow cover duration in cold areas, where the forest canopy acts as a filter to incoming shortwave radiation (Hardy et al. 1997; Link and Marks 1999). However, under warmer conditions, lower altitudes and with a high exposure to radiation, the heat storage of the canopy is increased, which enhances the release of longwave radiation and subsequently accelerates the melting process below forest canopies (Davis et al. 1997; Lopez-Moreno and Latron 2008). For topographically complex mountain ranges with large elevation gradients like the Alps, this means that the combined effect of climate warming and forest cover expansion will reduce snow cover accumulation and duration in some lower areas and increase it in other areas at higher altitudes. Thus, the occurrence of areas with deeper snowpack during warmer conditions may increase the occurrence of damaging floods during intensive melting events, while a reduction in the snowpack across other areas is likely to impact upon the availability of water resources (Lopez-Moreno and Stähli 2008). A similar effect of a different distribution of future snowpack in combination with forest cover change is expected for the occurrence of snow avalanches and forest-avalanche interactions. It is probable that climatic changes in some mountain ranges may increase the ratio of wet snow avalanches in forested areas, while snow avalanches may become less frequent in some lower-elevation forests, where temperature increases will be combined with a decrease of heavy snowfall events and/or with an increase of forest cover and forest cover density (Bebi et al. 2009). Changes of snow cover as a consequence and in combination with changes in forest cover are thus likely to lead to spatial shifts of areas prone to different natural hazards and other consequences for the hydrological regime.

General Conclusions and Management Implications for the Alps

As discussed above, natural forest growth and forest regrowth in the Alps is highly heterogeneous on different spatial scales and is likely to continue in the future with areas tending towards almost complete forest regrowth and with other areas where forest expansion is limited due to natural and socio-economic causes. It is important to note that forest dynamics and forest regrowth patterns are regionally very diverse.

While in some mountain regions mainly low and medium slopes were and will be reclaimed by forests (e.g. the Pyrenees, Lasanta et al. 2006), the typical pattern for the Swiss Alps is abandonment and forest regrowth in remote areas (Baur et al. 2006). In many European mountain ecosystems completely new ecosystem dynamics will form and rural landscapes will be lost (Höchtl et al. 2005). The human assessment of this development is divided, with visitors/tourists judging mostly positive and inhabitants/farmers of Alpine valleys being critical to negative (Höchtl et al. 2005). The effects of forest regrowth on the provision of different ecosystem goods and services are even more variable, as they depend on a variety of additional socio-economic and biophysical factors. This is particularly true for hydrological effects, in a complex mountainous environment. However, we believe it valid to state, that forest regrowth will reduce runoff, magnitude of hydrological peak events, sediment production and land slide risk. As such, areas at risk of one of these hazards might consider promoting forest regrowth (Fig. 6.3). However, forest management and landscape planning should respect as much as possible the high spatial and temporal variability of forest cover changes and subsequent effects on hydrological regime. While site factors and soil properties relevant for hydrological processes in forests are increasingly respected in the management of existing protection forests (Frehner et al. 2005; Gauquelin and Courbaud 2006), land use planning and policy measures on the interface between agriculture land and forests are hardly adapted to these spatial variations. For example, in Switzerland centrally planned policy measures generally aim at avoiding land abandonment and forest/shrub regrowth, independent on the regional and local effects on hydrological regimes and on the provision of other ecosystem services. The large variability of these effects would suggest a change in the governance of forest- and scrub regrowth, towards more target orientation in terms of the effects on soil erosion and hydrology and a shift of competence from the federal level to a regional or even local level. The need to differentiate situations at a low geographical scale and with respect to potential local and regional amenities and the need for local and regional adaptation of measures to manage abandoned land has also been identified for other Alpine countries (McDonald et al. 2000; Dax 2001).

Regionally adapted measures to avoid land abandonment or to manage abandoned land would be in line with major features of contemporary forest governance world-wide, which heads towards decentralization of forest management to meet the growing demands on forest ecosystems services like food, biofuel, and timber production in addition to the above discussed environmental services (Agrawal et al. 2008). To ensure a healthy balance between provision of environmental services and production, and to guaranty effective protection, forest governance should, however, not be decentralized into private ownership.

References

Acornley RM, Sear DA (1999) Sediment transport and siltation of brown trout (*Salmo trutta L.*) spawning gravels in chalk streams. Hydrol Process 13:447–458

Agrawal A, Chhatre A, Hardin R (2008) Changing governance of the world's forests. Science 320:1460–1461

Alewell C, Bredemeier M (2006) Forest change under a changing climate: introducing the problem. Contrib For Sci 28:20–28

Alewell C, Meusburger K, Brodbeck M, Bänninger D (2008) Methods to describe and predict soil erosion in mountain regions. Landsc Urban Plan 88:46–53

Ammer U, Breitsameter J, Zander J (1995) Der Beitrag des Bergwaldes zum Schutz gegen Oberflächenabfluß und Bodenabtrag. Forstwiss Centbl 114:232–249

Aschwanden H, Spreafico M (1995) Flood discharge–analysis of long standing measurements series. Plate 5.6. Hydrological Atlas of Switzerland. Bundesamt für Umwelt BAFU, Bern

Badoux A, Jeisy M, Kienholz H et al (2006) Influence of storm damage on the runoff generation in two sub-catchments of the Sperbelgraben, Swiss Emmental. Eur J For Res 125:27–41

BAFU (2007) Hydrologisches Jahrbuch der Schweiz 2005. Umwelt-Wissen Nr. 0713. Bundesamt für Umwelt, Bern, 502 pp

Baur P, Bebi P, Gellrich M, Rutherford G (2006) WaSAlp – Waldausdehnung im Schweizer Alpenraum. Eine quantitative Analyse naturräumlicher und sozio-ökonomischer Ursachen unter besonderer Berücksichtigung des Agrarstrukturwandels. Schlussbericht. http://www.wsl.ch/projects/WaSAlp. Accessed 25 Nov 2008

Baldock D, Beaufoy G, Brouwer F et al (1996) Farming at the margins: abandonment or redeployment of agricultural land in Europe. Case studies. Institute for European Environmental Policy (IEEP), The Hague

Bayard D, Stähli M, Parriaux A et al (2005) The influence of seasonally frozen soil on the snowmelt runoff at two Alpine sites in southern Switzerland. J Hydrol 309:66–84

Bebi P, Kulakowski D, Rixen C (2009) Snow avalanche disturbances in forest ecosystems – state of research and implications for management. For Ecol Manage 257:1883–1892

Begert M, Schlegel T, Kirchhofer W (2005) Homogeneous temperature and precipitation series of Switzerland from 1864 to 2000. Int J Climatol 25:65–80

Beniston M (2006) Mountain weather and climate: a general overview and a focus on climatic change in the Alps. Hydrobiologia 5262:3–16

Beniston M, Keller F, Goyette S (2003) Snow pack in the Swiss Alps under changing climatic conditions: an empirical approach for climate impacts studies. Theor Appl Climatol 71:19–31

Bigler C, Bräker OU, Bugmann H et al (2006) Drought as an inciting mortality factor in Scots Pine stands of the Valais, Switzerland. Ecosystems 9:330–343

Birsan MV, Molnar P, Burlando P, Pfaundler M (2005) Streamflow trends in Switzerland. J Hydrol 314:312–329

Bolli JC, Rigling A, Bugmann H (2007) The Influence of changes in climate and land-use on regeneration dynamics of Norway Spruce at the treeline in the Swiss Alps. Silva Fenn 41:55–70

Bonan GB (2008) Forests and climate change: Forcings, feedbacks, and the climate benefits of forests. Science 320:1444–1449

Bosch JM, Hewlett JD (1982) A review of catchment experiments to determine the effect of vegetation changes on water yield and evapotranspiration. J Hydrol 55:3–23

Brunetti M, Maugeri M, Nanni T et al (2006) Precipitation variability and changes in the greater alpine region over the 1800–2003 period. J Geophys Res 111:1–29

Bugmann H (2001) A comparative analysis of forest dynamics in the Swiss Alps and the Colorado Front Range. For Ecol Manage 145:43–55

Bunce RGH (1991) Ecological implications of land abandonment in Britain: some comparison with Europe. Options Méditerr 15:53–59

Bundesamt für Statistik (2001) The changing face of landuse. Landuse statistics of Switzerland. BFS, Bern

Burch H, Forster F, Schleppi P (1996) Zum Einfluss des Waldes auf die Hydrologie der Flysch-Einzugsgebiete des Alptals. Schweiz Z Forstwes 147:925–937

Cannone N, Sgorbati S, Guglielmin M (2007) Unexpected impacts of climate change on alpine vegetation. Front Ecol Environ 5:360–364

Chang M (2003) Forest hydrology: an introduction to water and forests. CNC, Boca Raton, FL, p 474

Conti G, Fagarazzi L (2005) Forest expansion in mountain ecosystems: “environmentalist’s dream” or societal nightmare? Driving forces, aspects and impacts of one of the main 20th century’s environmental, territorial and landscape transformations in Italy. Planum 6:1–20

Davis RE, Hardy JP, Ni W et al (1997) Variation of snow cover ablation in the boreal forest: a sensitivity study on the effects of conifer canopy. J Geophys Res 102:29389–29396

Dax T (2001) Endogenous development in Austria's Mountain regions. From a source of irritation to a mainstream movement. Mt Res Dev 21:231–235

Descroix L, Gautier E (2002) Water erosion in the southern French Alps: climatic and human mechanisms. Catena 50:53–85

FAO (2007) State of the world's forests 2007. Food and Agriculture Organization of the United Nations, Rome, p 220

Fraser AI, Harrod TR, Haygarth PM (1999) The effect of rainfall intensity on soil erosion and particulate phosphorus transfer from arable soils. Water Sci Technol 39:41–45

Frehner M, Wasser B, Schwitter R (2005) Nachhaltigkeit und Erfolgskontrolle im Schutzwald. Wegleitung für Pflegemassnahmen in Wäldern mit Schutzfunktion Bundesamt für Unwelt. Wald und Landschaft, Bern, 564 pp

Fritsche B, Bollmann K, Graf RF et al (2006) Die Entwicklung der Wälder auf der Schwägalp im 20. Jahrhundert und ihre Bedeutung für den Lebensraum des Auerhuhns (*Tetrao urogallus L.*). Schweiz Z Forstwes 157:135–146

Fuhrer J, Beniston M, Fischlin et al (2006) Climate risks and their impact on agriculture and forests in Switzerland. Clim Change 79:79–102

Gauquelin X, Courbaud B (2006) Guide des sylvicultures de montagne. ONF/CRPF/Cemagref, France, p 289

García-Ruiza JM, Lasantaa T, Martía C et al (1995) Flañoc changes in runoff and erosion as a consequence of land-use changes in the Central Spanish Pyrenees. Phys Chem Earth 20:301–307

Gehrig-Fasel J, Guisan A, Zimmermann NE (2007) Tree line shifts in the Swiss Alps: climate change or land abandonment? J Veg Sci 18:571–582

Gellrich M, Baur P, Koch B et al (2007) Agricultural land abandonment and natural forest regrowth in the Swiss mountains: a spatially explicit economic analysis. Agric Ecosyst Environ 118:93–108

German P (1994) Do forest control run-off? In: Hydrologie kleiner Einzugsgebiete. Beiträge zur Hydrologie der Schweiz, Nr. 35, Bern

Hardy JP, Davis RE, Jordan R et al (1997) Snow ablation modeling at the stand scale in a boreal jack pine forest. J Geophys Res 102:397–406

Heiri C, Bugmann H, Tinner W et al (2006) A model-based reconstruction of Holocene treeline dynamics in the Central Swiss Alps. J Ecol 94:206–216

Höchtl F, Lehringer S, Konold W (2005) Wilderness: what it means when it becomes a reality – a case study from the southwestern Alps. Landsc Urban Plan 70:85–95

Horton P, Schaefl B, Mezghani A et al (2006) Assessment of climate-change impacts on alpine discharge regimes with climate model uncertainty. Hydrol Process 20:2091–2109

Isselin-Nondedeu F, Bédécarrats A (2007) Influence of alpine plants growing on steep slopes on sediment trapping and transport by runoff. Catena 71:330–339

Jasper K, Calanca P, Gyalistras D et al (2004) Differential impacts of climate change on the hydrology of two alpine river basins. Climate Res 26:113–129

Jones HG, Pomeroy JW, Walker DA et al (2001) Snow ecology: an interdisciplinary examination of snow-covered ecosystems. Cambridge University Press, Cambridge, 378 pp

Körner C (1989) Der Flächenanteil unterschiedlicher Vegetationseinheiten in den Hohen Tauern: eine quantitative Analyse grossmasstäblicher Vegetationskartierungen in den Ostalpen. Veröff. des Österr, vol 13, MaB-Hochgebirgsprogrammes. Wagner, Innsbruck, pp 33–47

Körner C (2005) The green cover of mountains in a changing environment. In: Huber UM (ed) Global change and mountain regions. Springer, Dordrecht, The Netherlands, pp 650–375

Kozak J (2003) Forest cover change in the Western Carpathians in the past 180 years. Mt Res Dev 23:369–375

Krausmann F, Haberl H, Schulz NB et al (2003) Land-use change and socio-economic metabolism in Austria – Part I: driving forces of land-use change: 1950–1995. Land Use Policy 20:1–20

Landon, N, Piégay H, Bravard JP (1995) Compte-rendu de la mission d'expertise realisée sur la Drôme au cours du printemps 1995 pour le compte du syndicat mixte de la rivière Drôme et de la Commission Locale de l’Eau du SAGE, propositions pour une gestion physique équilibrée du lit de la Drôme. Laboratoire de Géographie Rhodanienne, URA 260, CNRS. Unpublished report, Lyon

Landon N, Piégay H, Bravard JP (1998) The Drôme River incision (France): from assessment to management. Landsc Urban Plan 43:119–131

Lasanta T, Gonzalez-Hidalgo JC, Vicente-Serrano SM, Sferi E (2006) Using landscape ecology to evaluate an alternative management scenario in abandoned Mediterranean mountain areas. Landsc Urban Plan 78:101–114

Laternser M, Schneebeli M (2003) Long-term snow climate trends of the Swiss Alps (1931–99). Int J Climatol 23:733–750

Liébault F, Gomez B, Page M et al (2005) Land-use change, sediments production and channel response in upland regions. River Res Appl 21:739–756

Link T, Marks D (1999) Distributed simulation of snow cover mass and energy-balance in the boreal forest. Hydrol Proc 13:2439–2452

Leuppi E, Forster F (1990) Zur Frage der Wirksamkeit des Waldes für den Hochwasserschutz – ein Beispiel aus dem oberen Reusstal. Schweiz Z Forstwes 141:943–954

Leuppi E, Forster F, Tarian D (1990) Waldwirkungen: Abflussbildung – the forest effects: the water discharge. Mitt – Landeshydrol Geol 14:41–48

Lopez-Moreno J, Stähli M (2008) Statistical analysis of the snow cover variability in a subalpine watershed: assessing the role of topography and forest interactions. J Hydrol 348:379–394

Lopez-Moreno JI, Latron J (2008) Influence of canopy density on snow distribution in a temperate mountain range. Hydrol Proc 22:117–126

McDonald D, Crabtree JR, Wiesinger G, Dax T, Stamou N, Fleury P, Gutierrez Lapita J, Gibon A (2000) Agricultural abandonment in mountain areas of Europe: environmental consequences and policy responses. J Environ Manage 59:47–69

Meusburger K, Alewell C (2008) Impacts of anthropogenic and environmental factors on the occurrence of shallow landslides in an alpine catchment (Urseren Valley, Switzerland). Nat Hazards Earth Syst Sci 8:509–520

Meusburger K, Alewell C (2009) On the influence of temporal change on the validity of landslide susceptibility maps. Nat Hazards Earth Syst Sci 9:1495–1507

Pearce AJ, ÓLoughlin CL, Jackson RJ et al (1987) Reforestation: on site effects on hydrology and erosion, eastern Raukumara Range, New Zealand. Int Assoc Hydrol Sci 167:489–497

Piégay H, Walling DE, Landon N et al (2004) Contemporary changes in sediment yield in an alpine mountain basin due to afforestation (the upper Drome in France). Catena 55:183–212

Ritzmann-Blickenstorfer H (1996) Historische Statistik der Schweiz. Chronos Verlag, Zürich

Rodda JC (1994) Mountains – a hydrolgical paradox or paradise? Beitr Hydrol Schweiz 35:41–51

Schädler B, Weingartner R (2002) Ein detaillierter hydrologischer Blick auf die Wasserressourcen der Schweiz – Niederschlagskartierung im Gebirge als Herausforderung. Wasser Energ Luft 94:189–197

Scheurer K, Alewell C, Bänninger D et al (2009) Climate and land-use changes affecting river sediment and brown trout in alpine countries – a review. Environ Sci Pollut Res 16:232–242

Schmidli J, Frei C (2005) Trends of heavy precipitation and wet and dry spells in Switzerland during the 20th century. Int J Climatol 25:753–771

Schröter D, Cramer W, Leemans R et al (2005) Ecosystem service supply and vulnerability to global change in Europe. Science 310:1333–1337

Schumacher S, Bugmann H (2006) The relative importance of climatic effects, wildfires and management for future forest landscape dynamics in the Swiss Alps. Glob Chang Biol 12:1435–1450

Schütz M, Krüsi BO, Edwards P (2000) Succession research in the Swiss National Park. Natl park-Forsch Schweiz 89:1–259

Stähli M, Nyberg L, Mellander PE et al (2001) Soil frost effects on soil water and runoff dynamics along a boreal transect: 2. Simulations. Hydrol Process 15:927–941

Summer W (1989) Umfassende Betrachtung der Erosions- und Sedimentationsproblematik. Wiener Mitteilungen Nr. 86, Wien

Tappeiner U, Tappeiner G, Hilbert A et al (2003) The EU agricultural policy and the environment. Evaluation of the Alpine region. Blackwell, Berlin, Germany

Tasser E, Tappeiner U (2002) Impact of land use changes on mountain vegetation. Appl Veg Sci 5:173–184

Tasser E, Mader M, Tappeiner U (2003) Effects of land use in alpine grasslands on the probability of landslides. Basic Appl Ecol 4:271–280

Tasser E, Tappeiner U, Cernusca A (2005) Ecological effects of land use changes in the European Alps. In: Huber UM, Bugmann H, Reasoner MA (eds) Global change and mountain regions – a state of knowledge overview. Springer, Dordrecht, The Netherlands, pp 413–425

Tasser E, Walde J, Teutsch A et al (2007) Land-use changes and natural reforestation on the Eastern Central Alps. Agric Ecosyst Environ 118:115–129

Tenhunen JR, Geyer S, Tappeiner U et al (2009) Influences of landuse change on ecosystem and landscape level carbon and water balances in mountainous terrain of the stubai valley, Austria. Glob Planet Change 67:29–43

Veit H (2002) Die Alpen- Geoökologie und Landschaftentwicklung. Verlag Eugen Ulmer, Stuttgart, pp 1–352

Verbunt AM, Zwaaftink G, Gurtz J (2005) The hydrologic impact of land cover changes and hydropower stations in the Alpine Rhine basin. Ecol Model 187:71–84

Vicente-Serrano SM, Lasanta T, Romo A (2004) Analysis of spatial and temporal evolution of vegetation cover in the Spanish central Pyrenees: role of human management. Environ Manage 34:802–818

Walther P, Julen S (1986) Unkontrollierte Waldflächenvermehrung im Schweizer Alpenraum. Ber Eidgenöss Forsch Anst Wald Schnee Landsch 282:1–83

Wear DN, Bolstad P (1998) Land-use changes in southern Appalachian landscapes: spatial analysis and forecast evaluation. Ecosystems 1:575–594

Weingartner R, Barben M, Spreafico M (2003) Floods in mountain areas – an overview based on examples from Switzerland. J Hydrol 282:10–24

Weinmeister W (2003) Fähigkeiten des Waldes zur Verminderung von Hochwasser und Erosionsschäden. In: Hochwasserschutz im Wald, Ber Bayer Landesanst Wald Forstwirtsch (LWF) 40:15–29

Wielke LM, Haimberger L, Hantel M (2004) Snow cover duration in Switzerland compared to Austria. Meteorol Z 13:13–17

Wischmeier WH, Smith DD (1978) Predicting rainfall erosion losses – a guide to conservation planning. United States Department of Agriculture, Washington, p 58

WSL (2005) Waldbericht 2005. Zahlen und Fakten zum Zustand des Schweizer Waldes. Bundesamt für Umwelt, Wald und Landschaft, Bern und Eidg. Forschungsanstalt für Wald, Schnee und Landschaft WSL, Birmensdorf

Zoller H, Bischof N, Erhard A et al (1984) Remarks on succession, significance and treatment of abandoned, unfertilized meadows in Switzerland. In: Brugger (ed) The transformation of Swiss Mountain regions. Haupt, Bern

Chapter 7
Effects of Climate Change on the Vulnerability of Norway Spruce Stands – Soil Hydrological Constraints for Forest Management in Austria's Lowlands

Karl Gartner, Michael Englisch, and Ernst Leitgeb

Introduction

In the last century the global air temperature increased by 0.74°C (IPCC 2007). In Austria an increase of 1.8°C was observed in this period (Auer et al. 2001). A climate change scenario used for Austria (ECHAM4/OPYC3) indicates an increase of the air temperature from 1.1 to 2.9°C for the next 55 years, compared to the 1961–1995 climatology (Matulla et al. 2002). Due to the country specific relief characteristics of Austria the precipitation scenarios contain much more uncertainty and the regionalized precipitation changes are ranging between −29 and +26% (Matulla et al. 2002). Results from a recent analysis by Loibl et al. (2009) based on the ECHAM5 model comparing the decade 2041–2050 with the decade 1981–1990 show an increase of 2–2.5°C with higher temperatures during the summer and autumn months. A regional differentiated increase of precipitation occurs during winter and spring and a decrease during summer and autumn. In mountainous areas with high precipitation the effects of climate change for tree growth are expected to be minor. In high altitudes a high air temperature combined with an elevated CO_2 level may even result in better growing conditions for the trees, as the growing season is extended (Menzel and Fabian 1999). A precondition for that enhanced growth is, however, sufficient precipitation recharging the soil water reserves and consequently covering the transpiration demand of the trees. Otherwise changes of tree species and a shift of forest communities due to altered competition may occur (Lexer et al. 2007, 2002). However, the adaptive capacity and regional vulnerability to climate change in European forests is not well understood yet (Lindner et al 2010).

K. Gartner (✉), M. Englisch, and E. Leitgeb
Department of Forest Ecology and Soil, Federal Research and Training Centre for Forests, Natural Hazards and Landscape, Seckendorff - Gudentweg 8, A-1131 Vienna, Austria
e-mail: karl.gartner@bfw.gv.at

M. Bredemeier et al. (eds.), *Forest Management and the Water Cycle: An Ecosystem-Based Approach*, Ecological Studies 212, DOI 10.1007/978-90-481-9834-4_7,

The impact of such changing environmental conditions on Austrian forests is highly dependent on site characteristics. In the warm and dry regions of east and south east of Austria, where lowlands and hilly landscape are dominating, the annual precipitation is already very low (500–800 mm). Especially in these regions Loibl et al. (2009) expect a severe decrease of precipitation during the growing season posing a serious challenge for sustainable forest management.

Distribution and Vulnerability of Norway Spruce (*Picea abies* (L.)) in Lowlands

The management of spruce forests in low elevation sites is a challenge for silviculture in Central Europe, especially under the aspect of climate change. Spiecker (2000) give the following reasons for the artificial, man-made expansion of Norway spruce:

- A high value production
- Low cost of planting, including low need for repair planting and relatively low damage due to deer browsing
- High level of knowledge about Norway spruce

In the eighteenth century plantations of Norway spruce monocultures were initiated in the Czech Republic. As a consequence the composition of the tree species was changed fundamentally (Klimo 2007). At present, Norway spruce is the most important tree species in the Czech Republic, accounting for 52.8% of the total cadastral forest area. In striking contrast the natural amount of Norway spruce is estimated to be only 11% (Klimo and Kulhavy 2006). Climatic scenarios up to 2050 derived from global circulation models predict similar trends for temperature and precipitation compared to Austria (Dubrovsky et al. 2005). The species composition of the Slovak forest differs from that of the Czech Republic. The main species is beech (31.4%) and not Norway spruce (25.7%). Nevertheless, only an estimated quarter of the Norway spruce stands are considered to be natural. Hlasny and Balaz (2008) predict a negative climatic water balance for 2045 in large parts of the country. Therefore it can be assumed that in both countries the effects of climatic change will deteriorate the growth conditions for Norway spruce. In Germany also, Norway spruce is heavily affected by climate change. In many regions of Germany, especially in southern Germany the growth conditions for Norway spruce are already critical. By using “climate envelopes”, where the temperature and precipitation regimes of the natural distribution areas of the particular tree species are combined (Kölling 2007; Kölling and Zimmermann 2007; Bolte et al. 2008; Kölling et al. 2009) many spruce stands in Germany were identified as “problematic” and it is most likely that climate change will exacerbate this situation.

Austria is covered by 3.6 million hectares of forests that is 47.2% of the Austrian state area. Norway spruce is by far the dominating tree species, comprising 61.5%

of the total growing stock. In the lowlands and in the hilly regions of Austria many Norway spruce plantations were established in the last centuries, far outside of their natural range. Prskawetz and Schadauer (2000) estimate, that "secondary coniferous forests" comprise an area of approximately 355,000 ha. Secondary coniferous forests are defined as forests with an amount of more than 80% of conifers growing on sites suitable only for deciduous trees. On these sites already the current climatic conditions are often very unfavorable for the growth of Norway spruce.

The vulnerability of Norway spruce forests in Austria's lowlands was analyzed by Niedermair et al. (2007). Based on the fact that in dry areas spruce dominated stands are highly vulnerable, the expansion of such vulnerable areas under an assumed increase of the air temperature of 2.5–5°C until 2100 was investigated. In the eastern and south eastern part of the country as well as in the north the climatologic stress for Norway spruce will increase significantly. High air temperature combined with low precipitation triggers very often drought stress. Due to the reduced tree vitality bark beetle (*Ips typographus* (L.)) attacks become more frequent and consequently increase tree mortality (Niedermair et al. 2007). The development of bark beetles is highly correlated with high temperatures during the growing season (Wermelinger and Seifert 1998). The impact of bark beetle disturbances on forest systems under climate change was investigated by Seidl et al. (2008, 2009). Results of scenario analysis indicated a strong increase in bark beetle damage under climate change scenarios (up to +219% in terms of timber volume losses) compared to the baseline climate scenario. If these scenarios will come true, only higher elevations in mountainous regions will provide suitable growing conditions for Norway spruce.

Kazda and Englisch (2005) analysed the vulnerability of Norway spruce by using climatic parameters like temperature sum and precipitation and defined 3 risk classes (high- medium – low). A sum of temperatures exceeding 3,050 deg d (2 p.m. temperatures summed according to Harlfinger 1999) indicates that Norway spruce is not part of the potential natural woodland community (Tüxen 1956). A long term annual precipitation lower than 600 mm or lower than 300–350 mm within the growing season (Mayer 1980) makes a sustainable growth of Norway spruce more than questionable. Besides of the already mentioned increased susceptibility to bark beetle attack, the competitive power of Norway spruce in relation to other (broadleaved) trees is limited. In the "medium risk" class Norway spruce is a subordinate/co-dominant element of the potential natural woodland community. This zone is characterized by a temperature sum of more than 2,600°C and/or an annual precipitation of less than 800 mm. In the "low risk" class Norway spruce is a co-dominating/dominating element of the natural woodland community. The risk classes for Norway spruce in Austria are shown in Fig. 7.1. However, it is noteworthy that these risk classes represent only a very general approach. Local site parameters, like soil properties, elevation, etc. can modify the risk within each class to a certain extent. If the water storage capacity of a shallow soil, for instance, is low, the risk is increasing. On the other hand soil water movement in a slope or a high soil water storage capacity can compensate low precipitation.

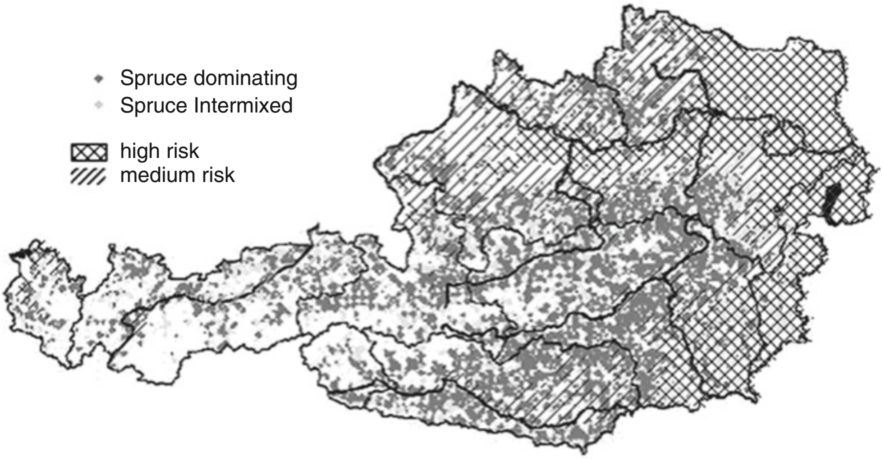


Fig. 7.1 "High" and "medium" risk zones of drought for Norway spruce in Austria based on temperature sums and long-term annual precipitation

Soil Water Regime in Norway Spruce Dominated Forests

Local soil properties play a crucial role for the survival of spruces in warm and dry areas. Shallow and sandy soils have only little soil water reserves. On the other hand also heavy, clayey soils are difficult to penetrate for spruce roots. Uniform spruce stands influence the stand hydrology in an unfavorable way. A considerable proportion of precipitation is additionally lost by crown interception. Moreover, spruce tends to build up rather hydrophobic humus layers (mor, raw humus), which impedes the infiltration of water. The following two case studies illustrate the water use of Norway spruce under drought stress conditions (case study Fürstenfeld) and the water demand of Norway spruce compared with European beech (case study Kreisbach).

The Fürstenfeld Study

In this study the transpiration of Norway spruce (*Picea abies*), birch (*Betula pendula*) and trembling aspen (*Populus tremula*) was compared and the interactions with the soil water regime were analysed. The study was performed from 2003 to 2004 in a uniform 20 year old stand, consisting mainly of these tree species. In 2003 the sap flow of spruce and birch was measured, in 2004 the measurements included also trembling aspen. It is noteworthy that the year 2003 was dominated by heat and drought in large parts of Europe (Granier et al. 2007; Rebetez et al. 2006; Eybl et al. 2005). This drought led to a decline of tree growth and vitality of many trees in different regions of Europe (e.g. Ciais et al. 2005; Dobbertin 2005). The stand originated from a large wind throw of a pure stand of Norway spruce. The study plot in “Fürstenfeld” is located in the southeast of Austria at an elevation of 320 m a.s.l. The climate has a continental aspect with relatively warm summer and cool winter months. Mean annual temperature is 8.8°C, the maximum precipitation occurs in the early summer and the long-term average annual rainfall is 765 mm. The trees grow on a heavy, clay rich soil, classified as a Stagnic Luvisol according to the World Reference Base classification system (FAO 2006). Sap flow measurements are a generally approved method to evaluate transpiration at tree level. For this study the heat field deformation (HFD) method (Nadezhdina et al. 1998; Čermák et al. 2004; Chapter 4, this volume) was used to measure the sap flow. HFD is based on the measurement of the deformation of a heat field around a needle-like linear heater due to the moving sap flow. Additional to these sap flow measurements soil moisture was registered by using the time domain reflectometry (TDR) principle. As tree transpiration is also driven by the atmospheric demand, the measurement of meteorological parameters, like global radiation, wind speed, air temperature and relative air humidity completed the data set. More details about the study design and the applied methods are given by Gartner et al. (2009).

Weather Conditions in the Year 2003 and 2004

The weather conditions in the summer months of 2003 and 2004 differed considerably. In 2003 the weather was extremely hot and dry. These extreme weather conditions at the study site, especially in August 2003, are shown in Fig. 7.2a and b. The mean air temperature of this month was more than 6°C above the long-term value. Although 72 mm of precipitation could be observed in August 2003, more than 90% of this amount fell only during the last 2 days of the month. In contrast, the summer of 2004 showed average weather conditions with a very humid June (166 mm). The precipitation in all 3 summer months was higher than in 2003 (see Fig. 7.2b). Although the average monthly air temperature in summer 2004 exceeded the long-term values in some cases, it was considerably lower than in 2003 (see Fig. 7.2a).

Soil Water Content

The differences in the weather conditions between 2003 and 2004 are mirrored by the soil water content (Table 7.1). Although the yearly mean values are similar, the yearly variation of both years differed considerably. The soil water content in 2003 started from field capacity in early spring (see Fig. 7.3). High temperature and a low precipitation in June 2003 were responsible for a sharp decline in soil water content. Due to a more humid period in the second half of July the soil was able to fill up the water reserves again. During the very hot and dry August of 2003 the lowest water content within the whole measurement period could be observed. Heavy rainfall at the end of the month finished this drought period. In 2004 there was a lot of rain and the soil water content remained at field capacity level till the mid of July (see Fig. 7.3). Then a characteristic seasonal pattern of

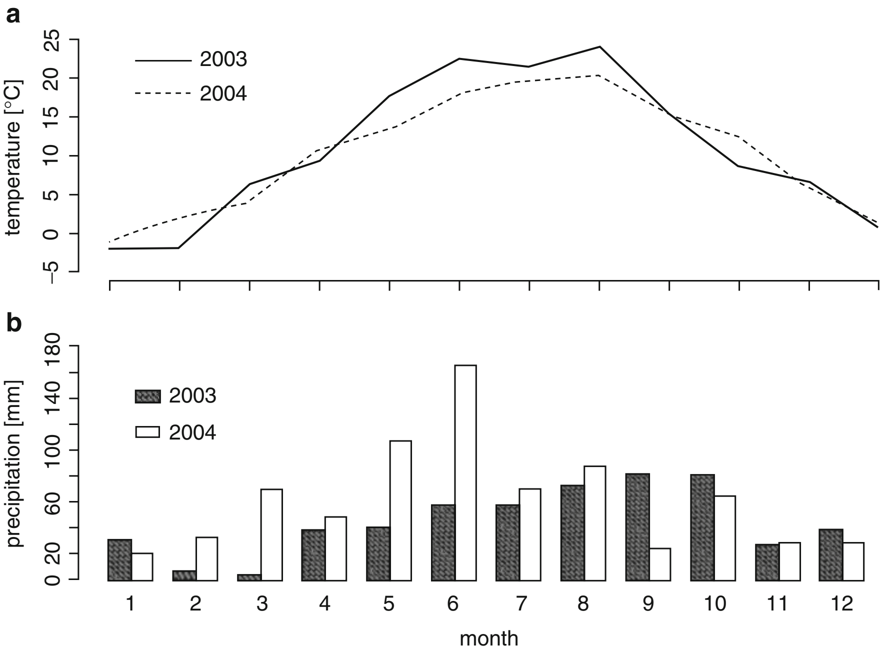


Fig. 7.2 (**a**) Mean monthly air temperature in the years 2003 and 2004, (**b**) monthly precipitation sums in the years 2003 and 2004 in Fürstenfeld (Source: ZAMG = Zentralanstalt für Meteorologie und Geodynamik)

the soil water content with distinct dry cycles with minimum soil water content in October could be observed. However, this minimum soil water content was much higher than the minimum values of 2003. Both tree species formed rather homogenous groups that allowed a comparison of the soil water content. In 15 cm soil depth, within the main rooting zone of both tree species, significant differences could be recognized. At the beginning of the summer drought, where high soil water contents close to field capacity still prevailed, the soil under spruce dried out more rapidly than under birch trees. After some days, however, the soil water content under birch fell below the corresponding water content under spruce. During the extreme drought conditions in August 2003 the water content under spruce declined more rapidly than that under birch resulting in nearly identical values at the end of the drought period. In both years 2003 and 2004 differences between the soil water content under spruce and birch could still be found in a soil depth of 30 cm, however, in an attenuated manner.

Sap Flow

The sap flow of the different tree species was controlled by the weather conditions in both years. In June 2003, the first month of the heat wave of this year, enough

Table 7.1 Soil water content [m^3/m^3] in the 20 year old mixed stand near Fürstenfeld (Austria) in the years 2003 and 2004 (absolute maximum, absolute minimum and average values of the year are *bold*)

			Month												Year		
	Depth [cm]		1	2	3	4	5	6	7	8	9	10	11	12	Max	Min	Avg
2003	15	Max	0.58	0.57	0.57	0.54	0.51	0.44	0.32	0.24	0.28	0.37	0.35	0.44	**0.58**	0.24	0.43
		Min	0.44	0.42	0.42	0.40	0.30	0.13	0.10	0.09	0.11	0.14	0.23	0.25	0.44	**0.09**	0.25
		Avg	0.49	0.47	0.48	0.46	0.41	0.28	0.17	0.12	0.18	0.26	0.31	0.32	0.49	0.12	**0.33**
	30	Max	0.45	0.45	0.46	0.46	0.45	0.39	0.35	0.29	0.34	0.40	0.40	0.43	**0.46**	0.29	0.40
		Min	0.41	0.42	0.41	0.41	0.33	0.20	0.15	0.13	0.19	0.19	0.32	0.34	0.42	**0.13**	0.29
		Avg	0.43	0.43	0.43	0.43	0.39	0.31	0.23	0.18	0.25	0.32	0.36	0.36	0.43	0.18	**0.34**
	50	Max	0.48	0.50	0.50	0.49	0.48	0.47	0.38	0.38	0.39	0.45	0.45	0.45	**0.50**	0.38	0.45
		Min	0.42	0.42	0.42	0.41	0.39	0.31	0.26	0.23	0.24	0.28	0.38	0.38	0.42	**0.23**	0.35
		Avg	0.44	0.44	0.44	0.44	0.43	0.38	0.32	0.28	0.32	0.39	0.42	0.41	0.44	0.28	**0.39**
2004	15	Max	0.44	0.41	0.60	0.65	0.65	0.71	0.72	0.34	0.22	0.24	0.31	0.35	**0.72**	0.22	0.47
		Min	0.29	0.27	0.27	0.40	0.38	0.42	0.29	0.15	0.13	0.12	0.19	0.21	0.42	**0.12**	0.26
		Avg	0.38	0.38	0.45	0.50	0.47	0.51	0.43	0.23	0.15	0.18	0.25	0.28	0.51	0.15	**0.35**
	30	Max	0.43	0.44	0.46	0.45	0.45	0.45	0.46	0.35	0.29	0.32	0.35	0.39	**0.46**	0.29	0.40
		Min	0.39	0.38	0.40	0.40	0.37	0.40	0.30	0.21	0.18	0.17	0.25	0.29	0.40	**0.17**	0.31
		Avg	0.41	0.41	0.42	0.43	0.42	0.42	0.39	0.28	0.22	0.25	0.31	0.33	0.43	0.22	**0.36**
	50	Max	0.46	0.48	0.49	0.49	0.50	0.51	0.50	0.43	0.36	0.40	0.41	0.43	**0.51**	0.36	0.45
		Min	0.42	0.41	0.41	0.41	0.41	0.42	0.38	0.31	0.28	0.28	0.33	0.35	0.42	**0.28**	0.37
		Avg	0.43	0.44	0.44	0.44	0.45	0.45	0.43	0.36	0.31	0.32	0.36	0.38	0.45	0.31	**0.40**

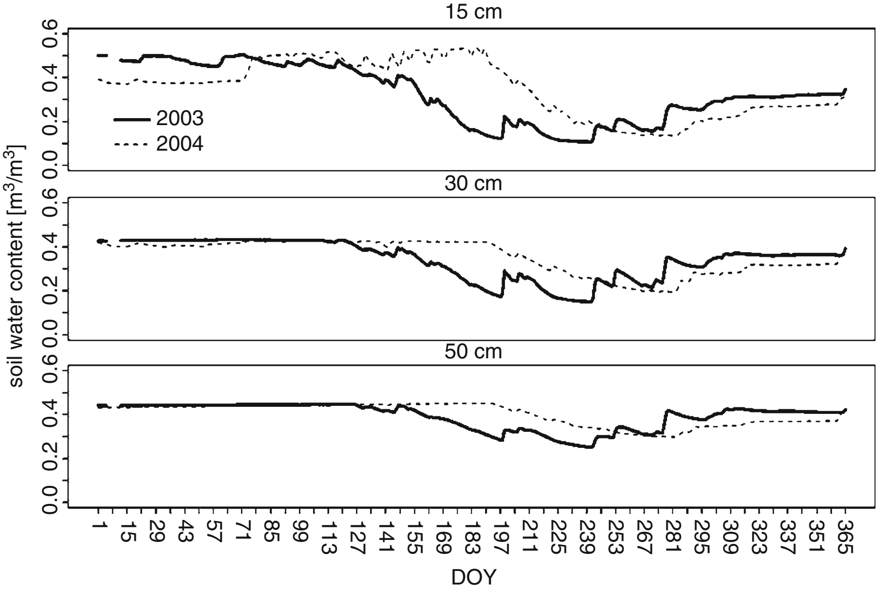


Fig. 7.3 Soil water content in three different depths in 2003 and 2004 in Commendewald near Fürstenfeld (DOY = Day of the Year)

soil water was available to satisfy the very high water demand of the trees (see Fig. 7.3). The sap flow of both tree species followed the atmospheric demand, which is determined by air temperature, radiation, wind speed and air humidity. Thus the diurnal sap flow pattern of both investigated tree species during this specific month was very similar (see Fig. 7.4a). At the end of June and the first half of July 2003 the trees restricted their water consumption and partly closed their stomata. After wetting of the soil following some precipitation in the second half of July, the sap flow of the trees was controlled by the atmospheric demand again. During the very hot and dry August of 2003 both spruce and birch had problems to extract water from the soil. Especially spruce trees suffered from the drought. The sap flow in spruce trees was reduced to such an extent, that the sap flow of some trees could hardly be measured. In addition, a significant midday depression of the sap flow resulted from that water deficit (see Fig. 7.4b). Although the birch trees suffered from the drought too, they could maintain a typical daily transpiration pattern (see Fig. 7.4b). During the summer months of 2004 no drought stress could be observed. All sample trees transpired according to the prevailing weather conditions. On cloudy and rainy days, which occurred frequently in 2004, the daily course of the sap flow did not necessarily follow the common pattern. The peak of sap flow was recorded in different times of the day. This is why the averaged maximum values in Fig. 7.4c and d are much lower than in Fig. 7.4a and b. However, there is hardly a difference in the daily course of all three tree species in June and August 2004.

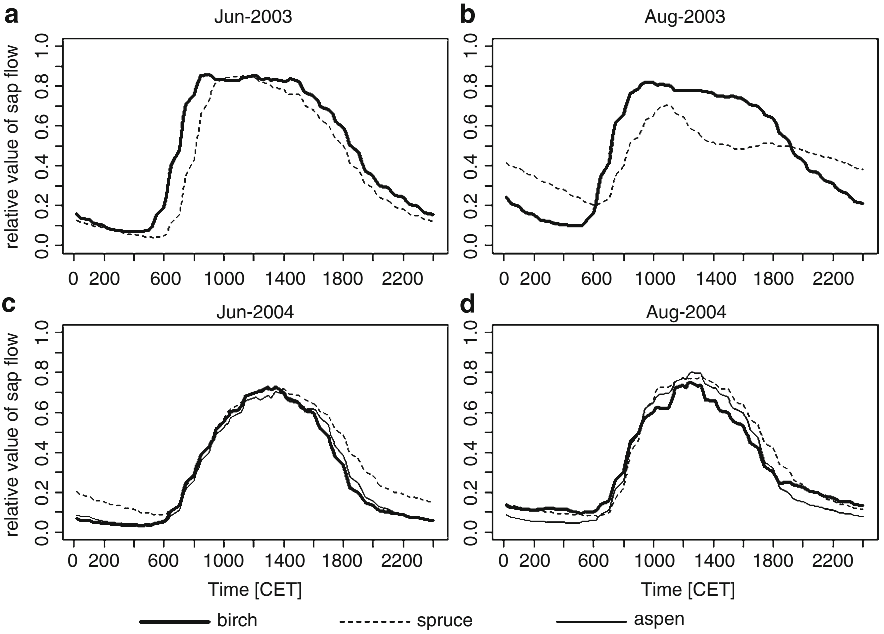


Fig. 7.4 Mean diurnal variation of sap flow (relative values) in birch, spruce and aspen trees: (**a**) June 2003, (**b**) August 2003, (**c**) June 2004 and (**d**) August 2004. The specified time is the Central European Time (CET)

The Kreisbach Study

Within the framework of the Austrian Special Research Programme on Forest Ecosystem Restoration, soil hydrological studies were done by Schume et al. (2003, 2004, 2005) and Jost et al. (2004). Water demand of Norway spruce and European beech in a low elevation site was compared. The study plot "Kreisbach" is located in Lower Austria at an elevation of 470 m. Kazda and Englisch (2005) classify this area as "medium risk" for Norway spruce which corresponds to the classification done by Niedermair et al. (2007). The mean annual precipitation of this area is 850 mm and the mean annual temperature is 8.4°C; occasionally summer drought can occur. The soil properties (high clay content, seasonal water logging) are similar to that of the study plot in Fürstenfeld. However, mature stands (pure stand of spruce and beech, mixed spruce/beech stand) were investigated at the Kreisbach study plot. For monitoring the volumetric water content a comprehensive TDR-measurement system was used. Additionally, precipitation, canopy throughfall, stemflow, global radiation, air temperature, wind speed and atmospheric humidity were continuously recorded (Schume et al. 2003). Figure 7.5 summaries the seasonal cumulative changes of the soil water storage in relation to the climatic water balance in the year 2000 and 2001 (Schume et al. 2004). The soil water storage is the stored water in the considered soil profile and was obtained by converting the

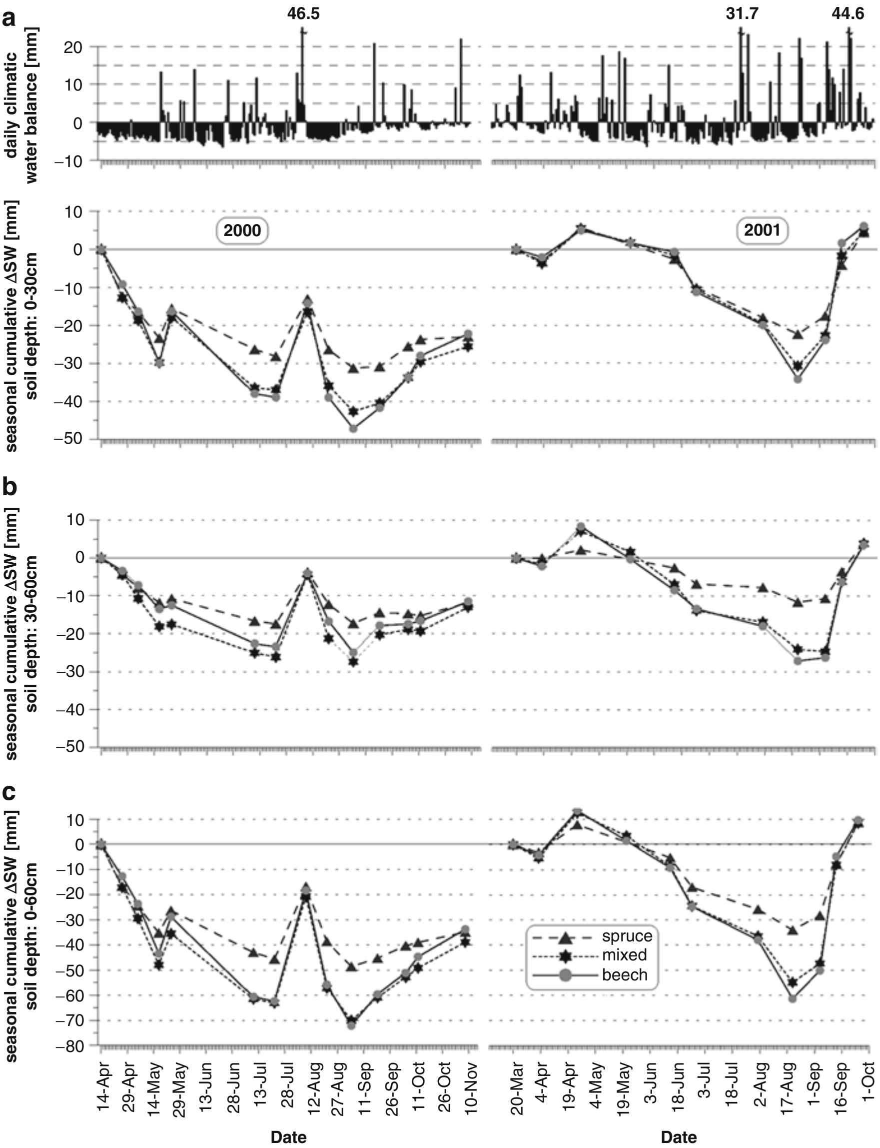


Fig. 7.5 Seasonal cumulative changes of soil water storage in relation to the climatic water balance for 2000 and 2001 by stands and soil layers in Kreisbach (Schume et al. 2004)

measured volumetric soil water content to a depth of water. The climatic water balance was computed from the difference between the measured precipitation and the potential evapotranspiration calculated using a modified Penman approach, Smith (1988). In the mixed stand the desiccation of the soil was more pronounced and affected deeper soil layers. During desiccation periods the average daily water consumption rate from the uppermost 60 cm of soil in the pure beech stand as well as in the mixed stand were about 45% higher than in pure spruce stands. In periods with a high evaporative demand differences of up to 84% could be found.

Root studies in this study plot, carried out by Schmid (2002), showed that spruce fine roots were restricted to the uppermost 30 cm of mineral soil in the mixed stand, whereas in the pure spruce stand 35% of the total fine-root mass were found between 30 and 60 cm depth. In general, fine roots of beech were more uniformly distributed over depth, and a small percentage reached a depth of 85 cm in the mixed as well as in the pure beech stand. That means that in a mixed stand the belowground competition leads to a very shallow root formation of spruce trees by shifting the roots towards the upper soil horizons. Thus the potential to use water reserves in deeper parts of the soil is restricted for spruce (Chapter 2, this volume). Consequently, in drought stress periods spruce growing in a mixed stand is very vulnerable, whereas beech can benefit from the root penetration into deeper soil horizons. Surprisingly, these results from Schmid (2002) are in contradiction to other traditional opinions (e.g. Mayer 1984) that spruce would benefit from an admixture of deeper rooting broadleaf trees.

Conclusions and Implications for Forest Management

Increasing drought stress, a very probable consequence of climate change, will heavily affect spruce dominated low elevation forests. The rough classification of risk areas, based on long term meteorological data, can provide only a hint for forest management. Local site factors and especially the soil water regime play a crucial role. Both studies highlighted the importance of the soil texture. It is well known that root system on Norway spruce is unable to penetrate heavy clay rich soil horizons (Oltchev et al. 2002). Though in drought situations a redristribution of the fine roots of Norway spruce to deeper mineral horizons can occur (Persson et al. 1995), other tree species, like birch and aspen are able to exploit deeper soil horizons more efficiently (Gartner 1997). Hence in drought stress periods these tree species can compensate the decline of available soil water by using deeper soil water reservoirs for maintaining their transpiration. Another reason for the drought stress of Norway spruce is the inefficient absorption of soil water by the fine roots due to the high negative water potential (Burk 2006). The sensitivity of Norway spruce to water stress was also shown by Ciencala et al. (1997) and Lagergren and Lindroth (2002). Also in mixed stands Norway spruce seems to be endangered by drought stress due to the belowground competition. However, it must be emphasized that there is only little information available about the belowground competition of tree roots. Therefore, the findings of the study site “Kreisbach” cannot be directly transferred to other site conditions and/or tree species.

The management of spruce dominated forests in low elevation should observe the following guidelines. First of all, the local site conditions should be analyzed thoroughly (site mapping). On heavy, clay rich or on dry and shallow soils the amount of spruce should be significantly reduced. Highly unstable Norway spruce stands should be converted into more stable mixed stands (Oleskog and Löf 2005; Spiecker et al. 2004). On such sites deciduous and more drought tolerant tree species, like oak and hornbeam are future forest management options (Lexer et al. 2007). After large scale disturbances in spruce monocultures on such sites the natural regeneration of pioneer tree species, like birch and trembling aspen play an important role in managing the unfavourable site conditions in the early regeneration phase. On more favourable sites, appropriate thinning strategies can help to improve the water supply in spruce forests (Laurent et al. 2003). A reduction of the rotation period might also help to lower the management risk of spruce forests.

References

Auer I, Böhm R, Schöner W (2001) ALOCLIM-Austrian long-term climate 1767-2000. – Multiple instrumental climate series from Central Europe. Österr Beitr Meteorol Geophys 25:1–147

Bolte A, Ibisch P, Menzel A, Rothe A (2008) Was Klimahüllen uns verschweigen. AFZ/Wald 63:800–803

Burk D (2006) Physiologische, anatomische und chemische Aspekte der Regulation der Wasseraufnahme bei Rotbuche, Kiefer und Birke auf unterschiedlich wasserversorgten Standorten. Dissertation (in German with English abstract), Georg-August University, Göttingen, 123 pp

Čermák J, Kučera J, Nadezhdina N (2004) Sap flow measurements with two thermodynamic methods, flow integration within trees and scaling up from sample trees to entire forest stands. Trees 18:529–546

Ciais Ph, Reichstein M, Viovy N, Granier A, Ogée J, Allard V, Aubinet M, Buchmann N, Bernhofer C, Carrara A, Chevallier F, De Noblet N, Friend AD, Friedlingstein P, Grünwald T, Heinesch B, Keronen P, Knohl A, Krinner G, Loustau D, Manca G, Matteucci G, Miglietta F, Ourcival JM, Papale D, Pilegaard K, Rambal S, Seufert G, Soussana JF, Sanz MJ, Schulze ED, Vesala T, Valentini R (2005) Europe-wide reduction in primary productivity caused by the heat and drought in 2003. Nature 437:529–533

Ciencala E, Kucera J, Lindroth A, Čermák J, Grelle A, Halldin S (1997) Canopy transpiration from boreal forest in Sweden during a dry year. Agric For Meteorol 86:157–167

Dobbertin M (2005) Tree growth as indicator of tree vitality and tree reaction to environmental stress: a review. Eur J For Res 124:319–333

Dubrovsky M, Nemesova I, Kalvova J (2005) Uncertainties in climate change scenarios for the Czech Republic. Clim Res 29:139–156

Eybl J, Godina R, Lalk P, Lorenz P, Müller G, Weilguni V (2005) Das Trockenjahr 2003 in Österreich (in German with English abstract). Mitt Hydrogr Dienst Österr 83:1–38

FAO (2006) World reference base for soil resources 2006 – A framework for international classification, correlation and communication. Intern. World Soil Resources Reports 103, FAO, Rome, 132 pp

Gartner K (1997) Wasserhaushalt ausgewählter Standorte im Osten Österreichs (in German with English abstract). In: Müller F (ed.) Silviculture on the lower limit of closed forests. BFW-Berichte 95, pp 31–43

Gartner K, Leitgeb E, Nadezhdina N, Englisch M, Čermak J (2009) Sap flow of birch and Norway spruce during the European heat and drought in summer 2003. For Ecol Manag 258:590–599

Granier A, Reichstein M, Bréda N, Janssens IA, Falge E, Ciais P, Grünwald T, Aubinet M, Berbigier P, Bernhofer C, Buchmann N, Facini O, Grassi G, Heinesch B, Ilvesniemi H, Keronen P, Knohl A, Köstner B, Lagergren F, Lindroth A, Longdoz B, Loustau D, Mateus J, Montagnani L, Nys C, Moors E, Papale D, Pfeiffer M, Pilegaard K, Pita G, Pumpanen J, Rambal S, Rebmann C, Rodrigues A, Seufert G, Tenhunen J, Vesala T, Wang Q (2007) Evidence for soil water control on carbon and water dynamics in European forests during the extremely dry year 2003. Agric For Meteorol 143:123–145

Harlfinger O (1999) Klimahandbuch der Österreichischen Bodenschätzung. Band 1. Universitätsverlag Wagner, Innsbruck, 196 pp

Hlasny T, Balaz P (2008) Climatic water balance of Slovakia using FAO Penman-Monteith evapotranspiration. Geogr Cas 60:15–30

IPCC (2007) Intergovernmental panel on climate change: climate change 2007: the physical science basis. In: Solomon S, Qin D, Manning M, Marquis M, Averyt K, Tignor MMB, Miller HL, Chen Z (eds) Contribution of working group I to the fourth assessment report of the intergovernmental panel on climate change. Cambridge University Press, Cambridge, 1009 pp

Jost G, Schume H, Hager H (2004) Factors controlling soil water-recharge in a mixed European beech (Fagus sylvatica L.)-Norway spruce [*Picea abies* (L.) Karst.] stand. Eur J For Res 123:93–104

Kazda M, Englisch M (2005) Definition of the conversion process priority. In: Oleskog G, Löf M (eds.) The ecological and silvicultural bases for underplanting beech (Fagus sylvatica L.) below Norway spruce shelterwood (*Picea abies* L. Karst.). Schr Forstl Fak Univ Gött Niedersächs Forstl Vers Anst 139:13–19

Klimo E (2007) Main ecological problems of Czech forestry in former times and at present. Rad Šumar Inst Izvanr Broj 10:117–126

Klimo E, Kulhavy J (2006) Norway spruce monocultures and their transformation to close-to-nature forests from the point of view of soil changes in the Czech Republic. Ekologia 25:27–43

Kölling C (2007) Klimahüllen für 27 Waldbaumarten. AFZ/Wald 62:1242–1245

Kölling C, Zimmermann L (2007) Die Anfälligkeit der Wälder Deutschlands gegenüber dem Klimawandel. Gefahrst Reinhalt Luft 67:259–268

Kölling C, Zimmermann L, Borchert H (2009) Von der "kleinen Eiszeit" zur „Großen Heißzeit". Vergangenheit, Gegenwart und Zukunft des Fichtenanbaus in Deutschland. LWF aktuell 69:58–61

Lagergren F, Lindroth A (2002) Transpiration response to soil moisture in pine and spruce trees in Sweden. Agric For Meteorol 112:67–85

Laurent M, Antoine N, Joël G (2003) Effects of different thinning intensities on drought response in Norway spruce (Picea abies (L.) Karst.). For Ecol Manag 183:47–60

Lexer MJ, Hönninger K, Scheifinger H, Matulla Ch, Groll N, Kromp-Kolb H, Schadauer K, Starlinger F, Englisch M (2002) The sensitivity of Austrian forests to scenarios of climatic change: a large-scale risk assessment based on a modified gap model and forest inventory data. For Ecol Manag 162:53–72

Lexer MJ, Seidl R, Rammer W, Formayer H (2007) Niederösterreichs Wald im Klimawandel. Klimafolgenstudie für die Region Waldviertel. In: Amt der NÖ Landesregierung Abt. Umweltwirtschaft, Raumordnungsförderung (eds) Auswirkungen des Klimawandels in Niederösterreich (in German). St. Pölten, Austria, 356 pp

Lindner M, Maroschek M, Netherer S, Kremer A, Barbati A, Garcia-Gonzalo J, Seidl R, Delzon S, Corona P, Kolström M, Lexer MJ, Marchetti M (2010) Climate change impacts, adaptive capacity, and vulnerability of European forest ecosystems. For Ecol Manag 259:698–709

Loibl W, Züger J, Köstl M (2009) Reclip:More Kleinräumige Klimaszenarien für Österreich. Standort 33:94–100

Matulla C, Groll N, Kromp-Kolb H, Scheifinger H, Lexer MJ, Widmann M (2002) Climate change scenarios at Austrian National Forest Inventory sites. Clim Res 22:161–173

Mayer H (1980) Waldbau, 2nd edn. Gustav Fischer Verlag, Stuttgart, New York, 483 pp

Mayer H (1984) Waldbau auf soziologisch-ökologischer Grundlage, 3rd edn. Gustav Fischer Verlag, Stuttgart, New York, 513 pp

Menzel A, Fabian P (1999) Growing season extended in Europe. Nature 397:659

Nadezhdina N, Čermák J, Nadezhdin V (1998) Heat field deformation method for sap flow measurements. In: Čermák J, Nadezhdina N (eds) Proceedings of the 4th international workshop on measuring sap flow in intact plants. IUFRO Publications, Publishing House of Mendel University, Brno, Czech Republic, pp 72–92

Niedermair M, Lexer MJ, Plattner G, Formayer H, Seidl, R (2007) Klimawandel und Artenvielfalt – Wie klimafit sind Österreichs Wälder, Flüsse und Alpenlandschaften? Österreichische Bundesforste AG, 27 pp

Oleskog G, Löf M (2005) The ecological and silvicultural bases for underplanting beech (Fagus sylvatica L.) below Norway spruce shelterwood (Picea abies L. Karst.). Schriften aus der Forstlichen Fakultät der Universität Göttingen und der Niedersächsichen Forstlichen Versuchsanstalt, Band 139. J.D. Saurländer's Verlag, Frankfurt am Main, 94 pp

Oltchev A, Čermak J, Nadezhdina N, Tatarinov F, Tishenko A, Ibrom A, Gravenhorst G (2002) Transpiratiion of a mixed forest stand: field measurements and simulation using SVAT models. Boreal Environ Res 7:389–397

Persson H, Von Fircks Y, Majdi H, Nilsson LO (1995) Root distribution in a Norway spruce (Picea abies (L. Karst.) stand subjected to drought and ammonium-sulphate application. Plant Soil 168–169:161–165

Prskawetz M, Schadauer K (2000) Conditions for forest restoration in Austria. Analysis based on forest inventory data. In: Hasenauer H (ed) Forest ecosystem restoration. Proceedings of the international conference held in Vienna, Austria 10–12.4.2000. University of Agricultural Sciences, Vienna, pp 223–228

Rebetez M, Mayer H, Dupont O, Schindler D, Gartner K, Kropp JP, Menzel A (2006) Heat and drought 2003 in Europe: a climate synthesis. Ann For Sci 63:569–577

Schmid I (2002) The influence of soil type and interspecific competition on the fine root system of Norway spruce and European beech. Basic Appl Ecol 3:339–346

Schume H, Jost G, Katzensteiner K (2003) Spatio-temporal analysis of the soil water content in a mixed Norway spruce (Picea abies (L.) Karst.)-European beech (Fagus sylvatica L.) stand. Geoderma 112:273–287

Schume H, Jost G, Hager H (2004) Soil water depletion and recharge patterns in mixed and pure forest stands of European beech and Norway spruce. J Hydrol 289:258–274

Schume H, Hager H, Jost G (2005) Water and energy exchange above a mixed European Beech – Norway Spruce forest canopy: a comparison of eddy covariance against soil water depletion measurement. Theor Appl Clim 81:87–100

Seidl R, Rammer W, Jäger D, Lexer MJ (2008) Impact of bark beetle (*Ips typographus L.*) on timber production and carbon sequestration in different management strategies under climate change. For Ecol Manag 256:209–220

Seidl R, Schelhaas MJ, Lindner M, Lexer MJ (2009) Modelling bark beetle disturbances in a large scale forest scenario model to assess climate change impacts and evaluate adaptive management strategies. Reg Environ Change 9:101–119

Smith M (1988) Calculation procedures of modified Penman equation for computers and calculators. FAO, Land and Water Development Division, Rome

Spiecker H (2000) Growth of Norway Spruce (Picea abies [L.] Karst.) under changing environmental conditions in Europe. In: Klima E, Hager H, Kulhavy J (eds) Spruce monocultures in Central Europe – problems and prospects. EFI-Proceedings No. 33. European Forest Institute, Joensuu, Finland, pp 11–26

Spiecker H, Hansen J, Klimo, E, Skovsgaard JP, Sterba H, Teufel, Kv (2004) Norway Spruce conversion – options and consequences. EFI Research Report, No. 18, 269 pp

Tüxen R (1956) Die heutige potentielle natürliche Vegetation als Gegenstand der Vegetationskartierung. Angew Pflanzensoziol 13:5–42

Wermelinger B, Seifert M (1998) Analysis of the temperature dependent development of the spruce bark beetle *Ips typographus (L.) (Col., Scolytidae)*. J Appl Entomol 122:185–191

Chapter 8
Observed Climate Change in Croatia and Its Impact on the Hydrology of Lowlands

Boris Vrbek, Ivan Pilaš, and Nikola Pernar

Introduction

Amongst the existing forest types in Europe, the specific biological function of water is nowhere more distinctive than in the lowland forest ecosystems. Consequently, no existing forest types are more shaped by historic hydrologic conditions, altered by manmade deviations from the natural water regime and vulnerable to consequences of global warming. The lowermost parts of the lowland forests, the wetlands, present the transitional zones between terrestrial and aquatic ecosystems, i.e. ponds, lakes, streams and rivers. Nowadays, due to excessive urbanization and the spread of agriculture, the total area of lowland forests and wetlands has been significantly reduced (Wenger et al. 2004). According to the mapped distribution of vegetation in Europe (European Environmental Agency 2006), one of the largest complexes of alluvial floodplain wetlands and lowland forests in Europe are situated in the South-East Europe, in the Sava and Drava River basins, tributaries to the Danube River. The Sava River is a unique example of a river where the floodplains are still intact, supporting both flood alleviation and biodiversity. Continental lowland forests in Croatia cover an area of more than 200,000 ha with a significant role in forestry production, flood reduction, recharging groundwater reserves, as well as filtering pollutants. The natural forest vegetation consists of mainly hardwood tree species such as the common oak (*Quercus robur* L.), field ash (*Fraxinus excelsior* Vahl.),

B. Vrbek (✉)
Croatian Forest Research Institute, Trnjanska 35, HR-10000 Zagreb, Croatia
e-mail: borisv@sumins.hr

I. Pilaš
Croatian Forest Research Institute, Cvjetno naselje 41, HR-10450 Jastrebarsko, Coatia
e-mail: ivanp@sumins.hr

N. Pernar
Department of Silviculture, Faculty of Forestry, University of Zagreb, HR-10002 Zagreb, Coatia
e-mail: nikola.pernar-mdg@zg.tel.hr

M. Bredemeier et al. (eds.), *Forest Management and the Water Cycle: An Ecosystem-Based Approach*, Ecological Studies 212, DOI 10.1007/978-90-481-9834-4_8,

common hornbeam (*Carpinus betulus* L.) and softwood species such as black alder (*Alnus glutinosa* (L.) Geartn.), poplars (*Populus* sp.) and willows (*Salix* sp.). For the biological diversity of lowland forests and attached wetlands is strongly important specific combination of hydrology and soils. The development of these forests in history was determined by regular seasonal alternations of groundwater tables (hydro period) and periodic flooding (Vašiček 1985; Rauš et al. 1996) which provided sufficient availability of soil water. Hydromorphic soils are one of the phenomena in the lowlands: they record through their morphology the long-term spatial and temporal distribution and circulation of water. Hydromorphic soils are strongly affected by water saturation (water logging) and because of their physical properties they strongly induce processes of water movement through soils, i.e. hydrodynamics and hydro periods.

The present state of forest and water relations, particularly soil water status i.e. groundwater tables, are strongly under the influence of intensive forest exploitation and numerous hydro-ameliorative activities in the past. During the nineteenth century almost 100,000 ha of virgin oak forests in Croatia were cut down because of demand for high quality wood. The extensive reduction of forest areas caused the creation of more moist conditions with prolonged water logging of soils in the remaining lowland forest areas (Prpić 1996). During the twentieth century the natural hydrographic network and groundwater regime were even more disrupted by intensive hydro technical water manipulative activities: river canalisation, flood manipulation which utilised this area for flood retention purposes, the construction of internal forest drainage networks and the discontinuance of surface runoff by forest roads and forest management (cutting and thinning) (Prpić 1996).

In the first decade of twenty-first century, the effect of regional progress of climate change has become clearly evident especially on the hydrology of lowlands in Croatia (Pilaš 2008). Across the north to south European gradient, existing IPCC scenarios predict forceful development of droughts and water scarcity in the region of South-Eastern Europe and the Mediterranean, which can also be currently evident from regional long-term trends of water balance components; an increased temperature trend, a decrease in precipitation and increasing evaporation (Zaninović and Gajić-Čapka 2000). Alternations in the groundwater regime in lowland forest are often perceived as a primary trigger in the decline of lowland forests, particularly the pedunculate oak (Prpić 1996; Pilaš et al. 2007). Pedunculate or common oak, according to the forest crown condition assessment is considered one of the most damaged tree species in Croatia and Europe (Fischer et al. 2008).

To reduce negative impacts of anthropogenic alternations in the groundwater regime from the past and to attenuate future very possible prolongation of droughts and water scarcity in the lowlands, various forest managerial and engineering practices could be considered. Basically, there are potentially two main contrasted measures of soil water balance manipulation: drainage of soils (through drainage ditches and channels) which contribute to decreasing the groundwater tables and forest management, i.e. (thinning) which contribute to increased soil water content and increase in the groundwater level. Through these water table management practices, the amount of soil water required for the survival of lowland forests could be maintained to some degree to attenuate the negative hydrologic trends and natural hazards such as droughts.

This study aims to explore the various possibilities of soil water management in lowland forests characterised by shallow groundwater tables to achieve long-term stability of the ecosystems. As the first step, an assessment procedure of the reconstruction of the historic state of waterlogging prior to anthropogenic activities is provided. The historic state of water logging is assumed as the optimal environmental condition for growth of these, close to virgin forests, and is also a streamline according to which all human practices should be adopted. As a second step, by comparing the historic state of water logging with the current groundwater status one can determine the degree of hydrologic changes on particular forest sites. This hydrologic changes, as third, are related to main impacts on the hydrology of lowlands (hydromeliorative works, stand structure disturbance) to ascertain the possibilities of improvement of soil water status by management of those impacts.

Environmental System of Lowland Forests

Climate

During the twentieth century, annual amounts of precipitation decrease throughout region, which conforms to the long-term desiccation trend in the Mediterranean. A decrease in the amount of precipitation is more marked in the northern Adriatic, on the Dalmatian islands and in Eastern Slavonia than in the highlands and north-western part of Croatia. According to measurements taken in the last 100 years at the Osijek meteorological station in the Eastern Slavonian continental lowlands (Zaninović and Gajić-Čapka 2000), an increase in the annual temperature amounting to 0.1°C in the past 100 years and 0.2°C during the warm season is evident. Evidence of a decline in the 100 yearly patterns of precipitation, −105 mm/100 years and −66 mm/100 years during the warm seasons, is even more significant. Respectively, the loss of soil water (potential evapotranspiration) 98 mm/100 years and 74 mm/100 years proves that scenarios of water scarcity is very certain in the near future. Droughts and heat waves are intensifying across the region, especially in last 2 decades where significant events appear in 1993, 2000 and most prominently in 2003. According to some indicative scenarios (UNDP 2008), for the period 2070–2100 in respect to the base period (1961–1990), a possible increase of precipitation (16.5 mm) could be expected only in winter season. The other seasons will encounter a decrease in precipitation with a possible maximum of –75.6 mm in summer. Also, a temperature rise during all seasons is expected with maximal amplitude of 4.6°C during winter.

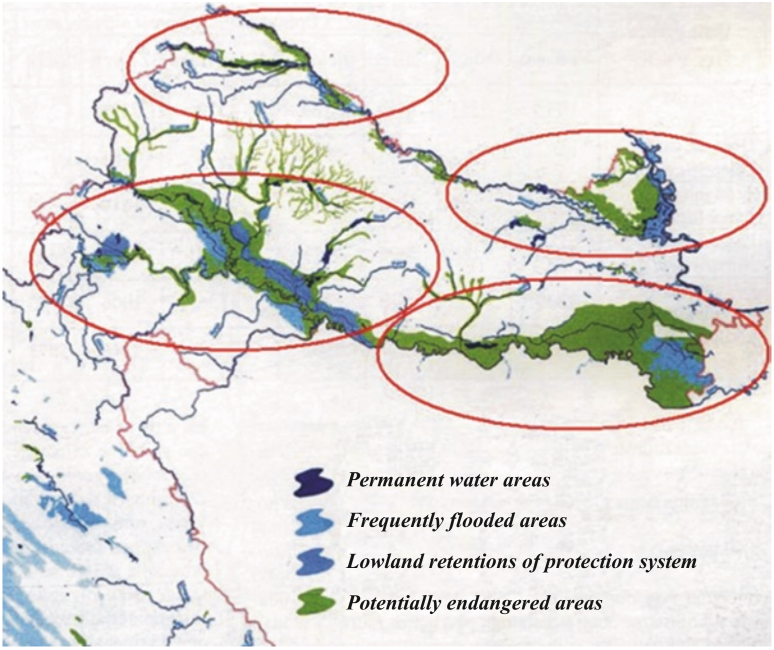


Fig. 8.1 Periodically and permanently flooded lowland areas in Croatia (According to Prpić and Milković 2005)

Geology, Soils and Vegetation

The main forest soil types of lowlands follow the topographic gradient, and as such, pseudogley and pseudogley-gley (stagnosols, albeluvisols) are dominant on river terraces and various gleysoils (hypogley, epigley, amphygley) are present in mezzo-elevations and depressions (Fig. 8.1). The parent materials consist mainly of a strip of alluvial and swamp sediments along rivers bordered by loess sediments on river terraces. Alluvial accumulation of finest textural classes after floods in the lowest parts of the terrain was the reason behind the formation of impermeable soil layers on which gleysol soil types mostly occur. The parent material on terraces have a somewhat coarser textural class and more suitable water relations which are associated with planosol soil types.

The appearance of the specific forest associations in lowlands is mostly determined by micro topography and soil physical properties, which, at a large degree, influence runoff, distribution and water movement into the soil. At the higher parts of the terrain, on the river terraces, the forest type of the common oak and hornbeam (*Carpino betuli – quercetum roboris*), at times also in combination with the common beech (*Carpino betuli – quercetum roboris fagetosum*) is predominant in the entire lowland forest region. In the amplitude between the river terraces and micro-depressions the forests of common oak with great green weed (*Genisto elatae*) is continuant with association varieties *Genisto elatae – Quercetum roboris caricetosum brizoides* in higher and

Genisto elatae – Quercetum roboris caricetosum remotae in lower positions. In the lowest positions of the micro topography, i.e. microdepresions, the forest associations of black alder (*Frangulo alnetum glutinosae*) and field ash (*Leucoio – Fraxinetum angustifoliae*) is prevailing.

The Observed Changes of Groundwater Tables

Information pertaining to groundwater tables was obtained from the series of groundwater level measured on permanent monitoring stations in lowland forest complexes in Croatia coalesced into the FORHIS (Forestry Hydropedological Information System) (Mayer 1995; Pilaš et al. 2005). FORHIS presents a network consisting of 120 piezometric stations established throughout the Croatian lowland forest region (Fig. 8.1). Trained personnel from forest offices carry out groundwater level measurements twice weekly. One piezometric station usually consists of four piezometers installed at numerical soil depths (0.5, 1.0, 2.0, and 7.0 m) with the aim of obtaining more detailed information regarding the groundwater status and hydrodynamics which can be influenced by irregular soil textural properties and the impermeability of a particular soil layer. The groundwater regime in this region has pluvio-thermal dynamics that is characterised by very high groundwater tables during the winter–spring period, which makes soil oversaturated. In late spring, the groundwater level decreases until late summer when it drops below the root zone and can be measured only in deep, 7.0 m piezometers. Apart from this seasonal pattern, strong fluctuations in the groundwater tables and periodic soil inundation in summer as a result of heavy rainfall are also very common.

The results of processing the groundwater monitoring data gathered from 56 piezometric stations in six forest complexes in central Croatia (Fig. 8.2a) for the period 1991–2003 can serve as an indicative example of relations between vegetation types (Fig. 8.2b), soil types (Fig. 8.2c) and average water table position in soils. In the presented data, there is evidence of groundwater decline between two subsequent periods (1991–1996, 1997–2002) and also of a strong groundwater drop caused by dry spell in 2003.

The groundwater decrease in 2003 as a reflection of extreme drought and heat wave occurs with equal intensity in all the respective forest complexes of central Croatia (Fig. 8.2). The problem of declining of groundwater tables and changes in the hydroperiod is even more prominent in the eastern part of Croatia (Eastern Slavonia) with semi-arid East European climate than in western part with semi-humid central European continental climate. The groundwater tables in Eastern Slavonian forests show much stronger affection by weather anomalies (Fig. 8.3) than in central Croatia (Fig. 8.4). The intense drop of groundwater as a result of the drought in 2000 continues through 5 subsequent years, with extreme minimal values in 2003, until 2005. After the drought episode, seasons with an extremely high amount of precipitation proceed (2005, 2006) causing extensive groundwater discharge. There is evidence that groundwater tables in Eastern Slavonia from 2000 nowadays are in a state of unbalance in comparison to regular annual variations of the hydro period. Prolongation of droughts

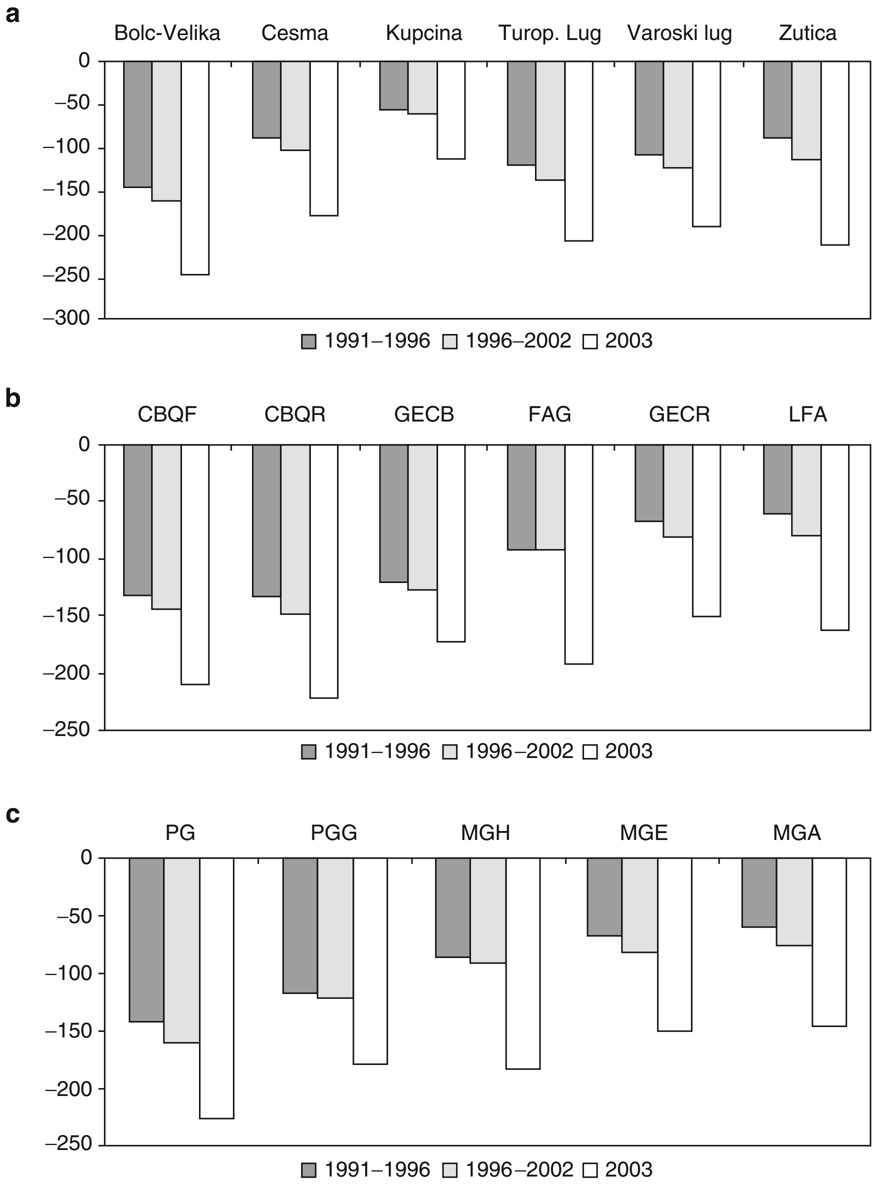


Fig. 8.2 (**a**) Average groundwater tables in two time sequences (1991–1996, 1997–2002) and in extremely dry 2003 in six forest complexes in the lowland area, Bolč-Velika, Česma, Kupčina, Turopoljski lug, Varoški lug, Žutica (**b**) Average groundwater tables in two time sequences (1991–1996, 1997–2002) and in extremely dry 2003 in various forest associations: forest association of common oak and hornbeam (*Carpino betuli – quercetum roboris – CBQR*), forest association of common oak and hornbeam with common beech (*Carpino betuli – quercetum roboris fagetosum – CBQF*), forests of common oak with great green weed (*Genisto elatae*) with association varieties *Genisto elatae – Quercetum roboris caricetosum brizoides – GECB* and *Genisto elatae – Quercetum roboris caricetosum remotae – GECR*, forest associations of black alder (*Frangulo alnetum glutinosae – FAG*) and field ash (*Leucoio – Fraxinetum angustifoliae – LFA*) (**c**) Average groundwater tables in two time sequences and extremely dry 2003 in respect to main forest soil types *pseudogley – PG*, *pseudogley-gley PGG*, *hypogley – MGH, epigley – MGE, amphygley – MGA*

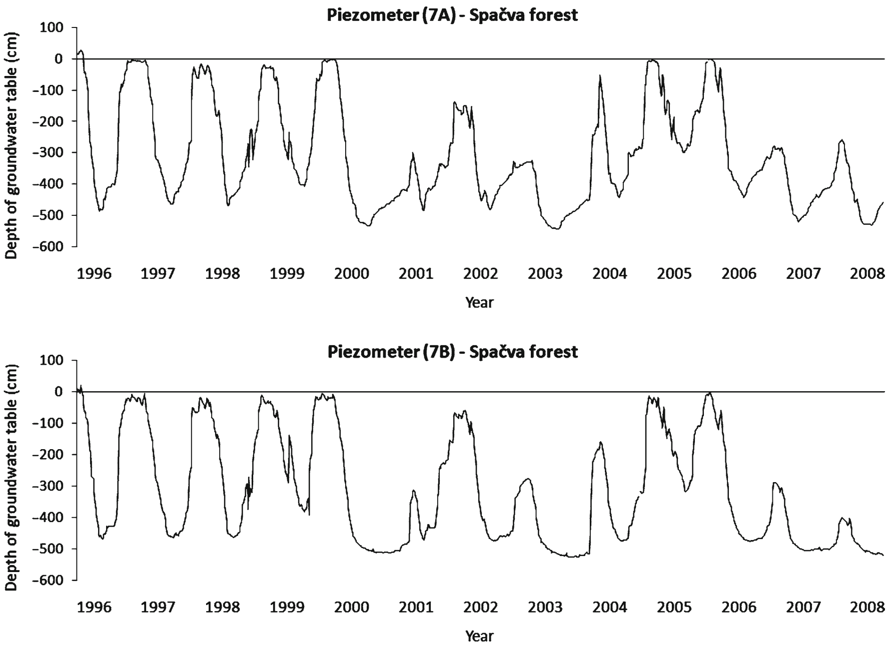


Fig. 8.3 The hydro-periods of groundwater tables in Eastern Slavonic lowlands in semiarid South-East European climate. The ground-water table decline during the drought episode 2000–2004 followed by excessive discharge in 2005, 2006 is obvious

during subsequent years (the decline of groundwater tables below the rooting zone) and sub arid environmental conditions of that region (evapotranspiration of oak forests exceeding the precipitation amount in the vegetation season) make the forest of the Eastern Slavonian area very vulnerable to droughts and dieback. The recently gathered field observation simply potential relationship between forest dieback and hydrological anomalies. The results of permanent crown condition monitoring on ICP Level 1 plots in Croatia (Fischer et al. 2008) also confirm that after the year 2003 up to the present a permanent decrease of health status of *Quercus robur* e exists.

The Reconstruction of the Historic State of Groundwater Tables

One of the main components of the lowland forests are hydromorphic (hydric) soils which are created by oxidation–reduction chemical reactions that occur when a soil is anaerobic and chemically reduced under water logging conditions (Hurt and Carlisle 2001). The hydrology of hydromorphic soils, the depth of groundwater levels and water table fluctuations are routinely estimated by soil scientists from a soil's morphology, mainly the soil colour (He et al. 2003). Formation of morphology of soils under differing water logging conditions is determined by the mobility

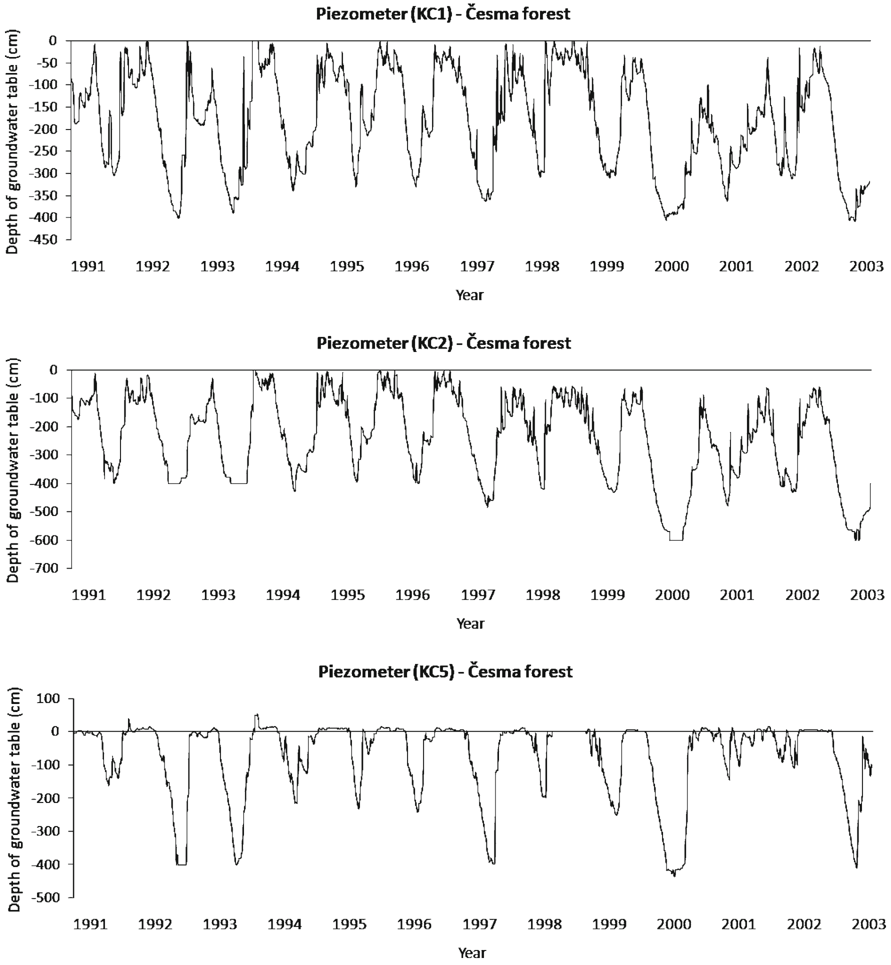


Fig. 8.4 Typical hydroperiods of groundwater tables in semi-humid continental climate of central Croatian lowlands. Relatively higher groundwater tables than in Eastern Slavonia is characteristic with no evidence of prolonged groundwater decline in consecutive years

status of iron (hydro) oxides. Redoximorphic features, redox concentrations, redox depletions and a reduced matrix are formed by reduction, movement and oxidation mostly of iron compounds (Schwertmann and Fanning 1976; Richardson and Daniels 1993; Vepraskas 2001; Vepraskas and Faulkner 2001; Fiedler and Sommer 2004). Soil stratigraphy and the degree of saturation (Richardson and Daniels 1993), along with soil organic matter (Schulze et al. 1993) and types of Fe mineral (Schwertmann 1993) have a significant influence on the formation of redoximorphic features. Soil redoximorphic features are to a large extent considered relict indicators (Vepraskas 2001) of soil hydrology. They reflect conditions regarding the water regime that existed at some point in history when soil morphology was formed.

Tandarich and Elledge (1996) assume that the current mapped hydric soils in the USA are a direct reflection of pre-European settlement wetlands that demonstrate temporal scale from which soils originate in more or less unchanged morphological conditions. These past conditions can also be considered as the most suitable for the growth and development of lowland forests because they reflect a time in history where no hydro-technical influences existed and forest exploitation was negligible, i.e. the conditions that were characteristic for the lowland close to natural forests. The property of soil morphology that stays permanently unchanged, irrelative of hydromeliorative changes, can provide information about previous unaltered conditions at the specific site in respect to the soil hydrology in the past.

To estimate the degree of groundwater shift on a particular forest sites, soil morphology or redoximorphic features as relict water logging conditions were compared with the current state of soil water status obtained by in situ measurements. The most common way of obtaining information regarding present temporal fluctuations in water levels (hydro period) and water logging duration of wetlands is to gather groundwater series of sufficient length on previously installed piezometers and water table wells (Richardson et al. 2001).

Assessment of Morphology of Hydromorphic Soils

To assess soil morphology, soil samples were taken from auger holes at the 56 respective plots from forest complexes in Central Croatia (Bolč-Velika, Česma, Kupčina, Turopoljski lug, Varoški lug, Žutica). Soil sampling was performed in three replications in close proximity to the piezometric stations. The main auger hole was located as close as possible to the piezometers, while two additional holes were situated at opposite sides at a 5 m distance. The 10 cm deep soil layers were examined and samples were taken to a total depth of 200 cm. A total of 60 samples were taken from each piezometric location for further soil colour assessment. In order to estimate the stand basal area (BA), the simple Bitterlich method of estimation was used for each location. The geographic co-ordinates of piezometric locations were measured by GPS. Gathered soil samples were air dried in the laboratory prior to assessment of morphological variables. Two basic features as a consequence of water logging could be visually distinguished on every soil sample: (a) a soil matrix which characterises areas with depleted iron (hydro) oxides, i.e. redox depletions and (b) iron mottles consisting of secondary oxidised iron newly termed redox concentrations (Vepraskas 2001). The redoximorphic features of every sample were assessed on an average basis of five structural aggregates that were split in half. Therefore, only the interior of the soil aggregates was assessed so as to avoid uncertain assessment of the surface area that is exposed to air and consequently more intensive oxidation of iron occurs. The colour of the matrix on soil samples was estimated by Munsell soil color charts under constant light conditions in the laboratory. Redox depletions on soil samples were described accordingly by three variables of Munsell color space: hue (H), value (V) and croma (C).

The colour of iron mottles on samples showed large variations and it was difficult to attribute them by use of Munsell charts. The occurrence of secondary iron minerals is very dependent on the pedo-environment, i.e. the oxidation rate or pore volume (Schwertmann 1993). Therefore, only iron mottles in abundance, i.e. redox concentrations, were assessed using the Munsell chart for estimating the percentage of mottles and coarse fragments. The largest number of samples (75.6%) was attributed to gley Munsell colour charts (45.2% gley1 and 30.4% gley2). The reddest Hue 10YR was assessed on 14.9% samples mostly in surface or subsurface layers, followed by more yellow 2.5Y (6.5%) and 5Y (3.0%) hues. The assessed Munsell values show that the dominant number of soil samples are lightly coloured (value of 7) and range from 3 (dark) to 8 (very light). Estimated redox concentrations in respect to other morphological variables have the largest variations along a profile and are quite abundant on the majority of soil samples with an average (median) of 30%. Hue 10YR occurs mostly in the surface soil layers as a consequence of soil organic matter (Shulze et al. 1993). Hues 2.5YR, 5Y occur in layers under surface horizons out of reach of organic matter where iron is not completely depleted from interior of soil aggregates. Gley colours GLEY1, GLEY2 as a result of intense iron depletion can be determined at a relatively shallow depth, somewhere directly underneath the surface horizon. Chroma of 1 or less determined on all considered samples is strong evidence of saturated conditions of soils (Hurt and Carlisle 2001). Soil morphology variable values and redcon show systematic vertical pattern as a result of different soil types, hydrology and position in the landscape that can be assessed by extraction of similar groups by cluster analysis.

First the reference soil groups planosol, albeluvisol and stagnosol (IUSS-WG WRB 2006) can be recognised in cluster groups in which differences in Munsell value vary along a profile between 1° and 2° (Fig. 8.5a). Respective vertical distribution of value can generally be attributed to soils on higher positions in the landscape, i.e. on river terraces with a perched groundwater table where vegetation is characteristic for common oak and common hornbeam forest. The gleysols characterise a gradual pattern of Munsell value, which ranges between 2° and 4°. This is characteristic for lowest landscape positions with groundwater discharge and common oak and narrow-leaved ash forest vegetation. Some cluster groups of Munsell value are distinguished because the soil colour pattern is influenced by lithological discontinuity, i.e. a regional occurrence of heavy clayey horizons as a result of alluvial deposition in the past (Fig. 8.5b). In planosols, albeluvisols and stagnosols an almost continuous increasing trend of redox concentrations with depth is apparent (Fig. 8.5f). For gleysols, a strong increase of redox concentrations in the upper half of profile (Fig. 8.5e and g) and a strong decrease of this feature in the lower half is significant.

Calculation of Water Logging Status from Groundwater Data

In order to calculate the yearly average soil water logging rate (abbreviated WLR) for the 13-year period under consideration (1991–2003), a synthesising of raw groundwater data from various pipes of one piezometric station was required.

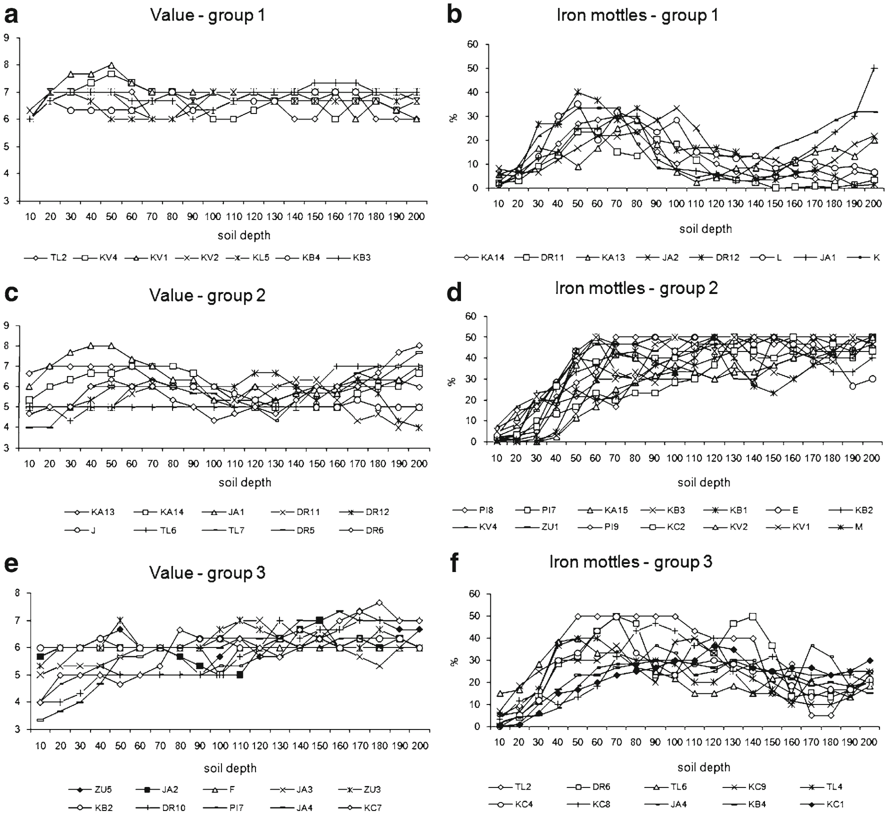


Fig. 8.5 The distribution of Munsel value and percentage of iron mottles in various soil groups

A single groundwater series was created by first taking measurements in shallow pipes when they were within reach of groundwater. For soil depths of 0–50 cm, measurements determined in the first subsurface piezometric pipe of 0.5 m depth were taken into account. For a groundwater drop below a soil depth of 50 cm, data from a second pipe was taken into account, for a groundwater drop below 100 cm data from third pipe was taken respectively until a depth of 200 cm. Further calculation of the soil water logging rate was provided by averaging measurements on a monthly basis in order to eliminate the possible effect of temporal variation caused by different periods of measurements. As a result of these calculations, a unique monthly groundwater series for the 13-year period from 56 piezometric locations was obtained. From respective monthly averages, yearly average soil water logging duration (WLR) was calculated for soil layers with a profile depth of 10–200 cm according to the formula:

$$WLR = Nx/Nt$$

Nx – the number of monthly average values above a particular soil depth
Nt – the total number of months in the 13-year period

The mean WLR and calculations from the transformed 13 years groundwater for respective 1,120 soil layers was 59.6% and the median WLR 63.4%. It is assumed that the WLR calculated from the data series during this period represent the average water logging conditions on the studied sites very well.

Prediction of the Average Water Logging Rate (WLR) from Soil Morphology

For the prediction of the soil water logging rate from soil morphological variables estimated on 3,360 soil samples, the generalized linear model (GLM) procedure was employed using independent morphological variables: REDCON, V, H in a particular depth or layer of soil. Soil morphological variables, hue, value and croma present a quantitative representation of soil colour (Blavet et al 1999) but some of the ambiguity in quantifications of the studied soils occurs. Munsell value and croma already present a numerical code and can be directly included in further data analysis. Munsel hues are spread around a circle in the Munsell colour cylinder and usually present a numerical code convenient for further calculations (Schwertmann 1993; Blavet et al 1999). However, due to the occurrence of numerous samples with neutral hues expressed with gley1 1/N, a large number of samples could not be associated to a unique value in the hue circle. This issue was solved by transforming values of hue according to the assumption that the redness degree declines in the following order: 10YR – 2,5YR – 5Y – GLEY. So as to quantify the hues according to the redness degree, each of the hue charts were attributed a particular percentage: 10YR – 100%; 2.5YR – 67%; 5Y – 33% and 0% for GLEY. Testing the significance of independent variables (REDCON, V, H, and S_layer) was performed by Wald statistics (Table 8.1). Parameter estimates of the WLR model are presented in Table 8.2.

Unexplained variability of WLR was attributed to two main anthropogenic effects: (1) Hydro technical constructions and (2) disturbance of stand structure. The impact of these two factors was assessed by calculating additional variables, which comprise residuals between measured WLR and GLM predicted values. For the assessment of effect of hydro-technical activities, the distance of plots and nearby hydrotechnical object was first measured. Hydro-technical constructions are considered to be either constructed canals through forests or old riverbeds that are not in function after river canalisation. For this purpose hydrographic network was

Table 8.1 The Wald test of significance of independent variables in prediction of waterlogging rate (WLR)

	Degree of freedom	Wald	p
Intercept	1	5154.328	0.000000
REDCON	1	45.779	0.000000
V	1	75.563	0.000000
H	1	0.566	0.451816
S_layer	19	454.219	0.000000

Table 8.2 Parameter estimates of the WLR model

	Level of effect	Estimate	Standard error	Wald stat	p
Intercept		4.76296	0.066342	5154.328	0.000000
REDCON		−0.00451	0.000666	45.779	0.000000
V		−0.09334	0.010738	75.563	0.000000
H		−0.00029	0.000386	0.566	0.451816
S_layer	10	−1.29187	0.122149	111.855	0.000000
S_layer	20	−0.78906	0.080471	96.148	0.000000
S_layer	30	−0.48392	0.061246	62.429	0.000000
S_layer	40	−0.22303	0.048878	20.822	0.000005
S_layer	50	−0.09820	0.044394	4.893	0.026961
S_layer	60	−0.03348	0.041830	0.641	0.423494
S_layer	70	0.01690	0.040231	0.176	0.674518
S_layer	80	0.05710	0.038797	2.166	0.141055
S_layer	90	0.09004	0.037386	5.800	0.016028
S_layer	100	0.12065	0.036508	10.921	0.000951
S_layer	110	0.14614	0.035653	16.802	0.000041
S_layer	120	0.17848	0.035090	25.870	0.000000
S_layer	130	0.20479	0.034484	35.269	0.000000
S_layer	140	0.22747	0.033969	44.842	0.000000
S_layer	150	0.25828	0.033414	59.748	0.000000
S_layer	160	0.28316	0.033033	73.478	0.000000
S_layer	170	0.30780	0.032573	89.294	0.000000
S_layer	180	0.32395	0.032266	100.801	0.000000
S_layer	190	0.33614	0.031735	112.191	0.000000
Scale		17.28907	0.365298	2240.000	0.000000

digitised from relevant topographic maps of scale 1:25,000 using GIS (Neteler and Mitasova 2003). Variable D_HIDRO was calculated as the closest difference between piezometric locations and the nearby hydro-technical construction. Because of the strong influence of microtopography on water movement in the lowlands, the distances in elevation were not taken into account because it could not be easily determined from the existing digital terrain model based on topography 1:25,000 maps. For the purpose of spatial analysis, a GIS database containing information on forest complexes under consideration was built. Stand basal area (BA) measured on the sample plots presents a factor of disturbance to the soil water balance due to a possible increase of water logging after reducing the number of trees as a result of forest decline or silvicultural practice such as thinning or forest regeneration.

Between BA and resGLM significant but not very high correlations were determined especially on soil depth around 100 cm (r = 0.47). This confirms that disturbance of stand structure at some degree causes variations in the WLR particularly regarding an increase in the water logging duration (Fig. 8.6). The linear relationship between D_HIDRO and WLR residuals was very small and insignificant. Through more detailed exploration of the nature of this relationship, it can be observed that channels and dry riverbeds create a so-called “cone of depression” in the groundwater table (Richardson et al. 2001). That is, they influence the decrease of water logging of lower layers in soils

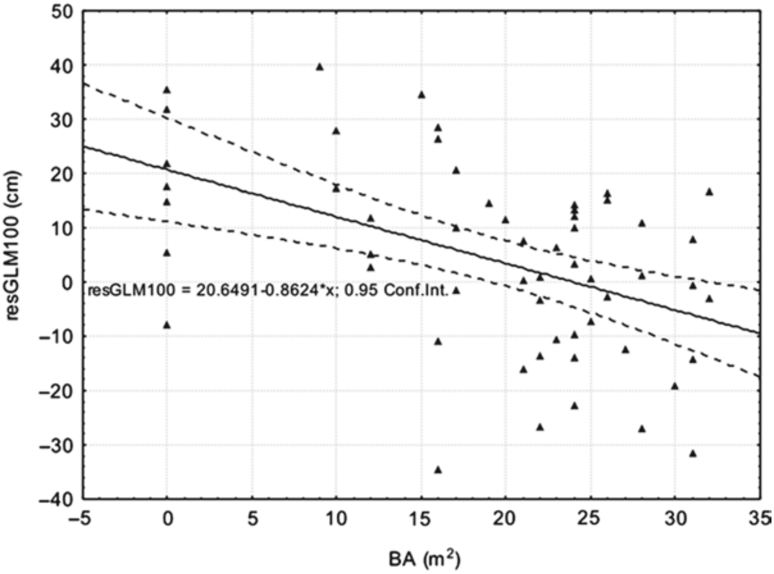


Fig. 8.6 Response of waterlogging (resGLM100) to decrease of stand basal area (BA)

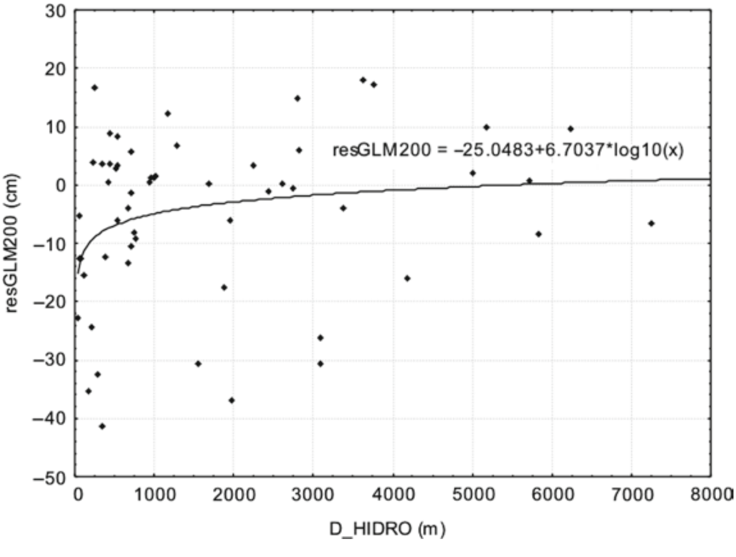


Fig. 8.7 The decline of waterlogging (resGLM200) in respect to the distance to hydro-meliorative constructions (large canals or dry riverbeds) (D_HIDRO) – “the cone of depression”

on sites closer to the constructions (Fig. 8.7). This relationship can be explained through an appropriate log function from which it is approximated that a decrease of the water logging duration takes place on sites under 200 m in distance. The nature of changes in water logging can be more clearly assessed from respective GIS layers. In evaluation of the aforementioned main sources of changes of WLR, soil stratigraphy (Richardson and Daniels 1993) should also be considered because it strongly regulates water movement in soils. Therefore, so as to assess prior spatial changes of water logging, soil colour of redox depletions was used to provide insight into soil stratigraphy of two diverse cases. From overlays of spatially interpolated Munsell variablesdiscontinuous impermeable heavy clayey layer at a soil depth of 120–140 cm can be clearly visually distinguished in the case of Pokupsko Basin (Fig. 8.8a) in relation to Cesma Forest (Fig. 8.8b). This characterises soils with more homogeneous stratigraphy and lighter texture. Spatially interpolated water logging residuals of these two forest cases shows that shortage of water saturation is present in areas close to large canals and old riverbeds and is clearly obvious in Cesma Forest (Fig. 8.10). In Pokupsko Basin (Fig. 8.9) the shorter WLR than predicted, present in upper soil layers, is the possible result of a smaller drainage network of ditches inside the forest whose influence was not examined in a study of this scale. Also, a simultaneous increase of the WLR in upper soil layers and a shortage of the WLR in lower soil layers occur on some sites close to the Kupcina Canal on which the forest stand is removed. The canal intersects the natural groundwater flow direction

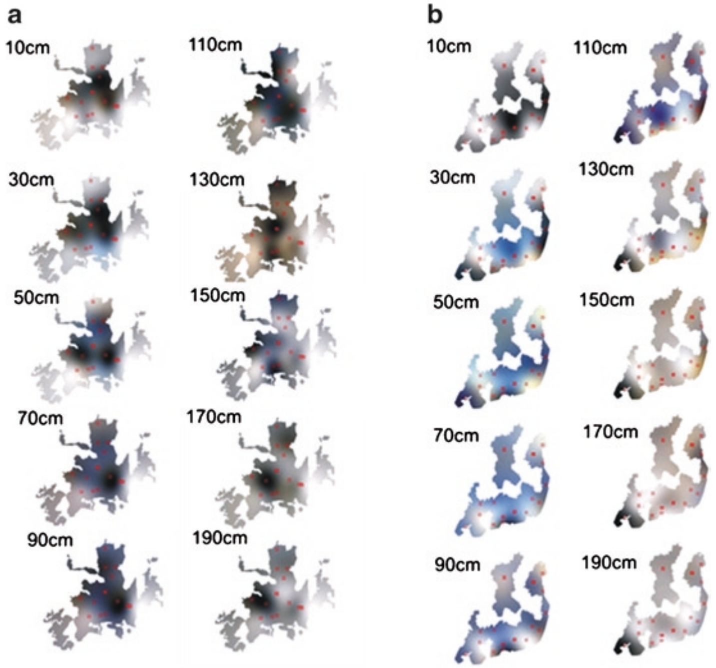


Fig. 8.8 Synthesised pseudo-colour layers from soil colour variables for various depth of (**a**) Pokupsko Basin and (**b**) Česma Forest

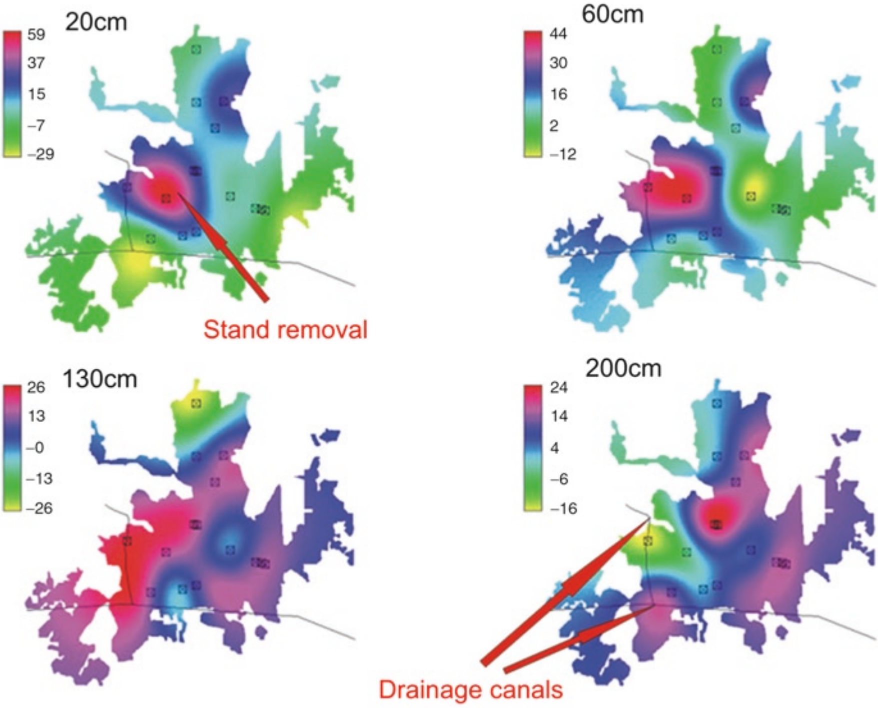


Fig. 8.9 The alternation of the waterlogging rate from historic groundwater conditions in Pokupsko Basin

from nearby hill slopes and causes groundwater table depression. By determining a corresponding increase of the WLR above the impermeable layer and the depression below it, the GLM procedure shows a satisfactory degree of sensitivity in which, even on the same site, evidence of respective causal factors on soil water can be distinguished.

The Possibilities of Water Table Management to Mitigate Water Scarcity Impact

Up to the present, the possibilities of water table management as part of the climate change adaptation strategies on forests have been very poorly evaluated. The recently proposed large-scale silvicultural strategies of European forests to climatic change (Kellomäki and Leinonen 2005) neglect the problem of groundwater decline in some specific regional environmental conditions as in lowland forests in South-Eastern Europe. According to the authors, for the continental region, “decreased precipitation during summer, in combination with increased evapotranspiration due to the high temperatures will result in increased drought stress in all but those sites that have access

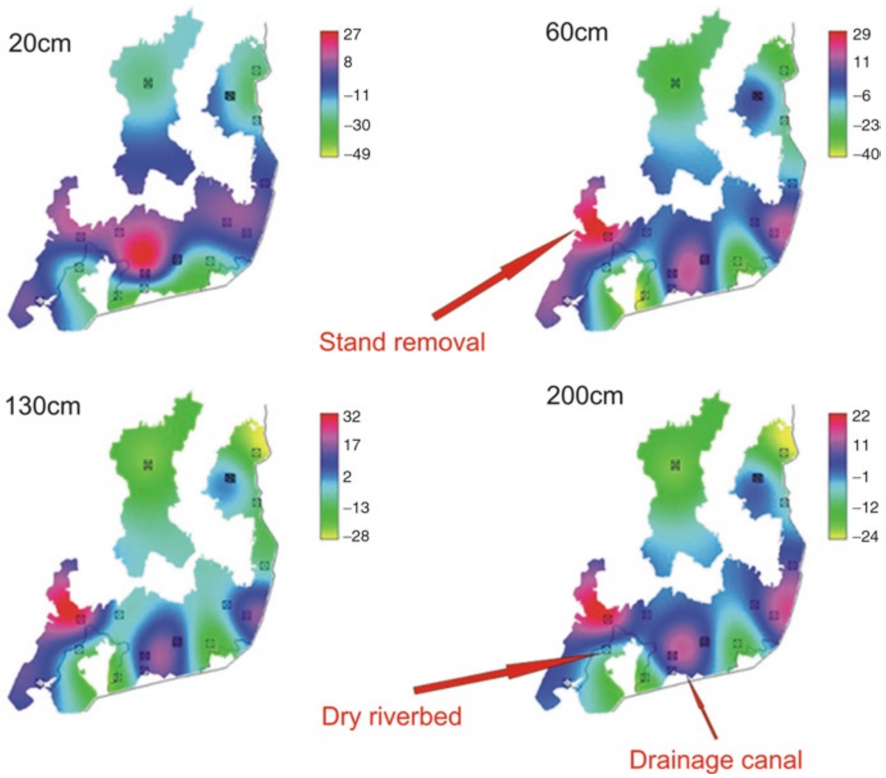


Fig. 8.10 The alternation of the waterlogging rate from historic groundwater conditions in Česma Forest

to groundwater supply from elsewhere". Thus, groundwater reserves as an important supply of water for forests are not considered as a factor of forest vulnerability, as in lowland forests, although they are even more subject to changes over time under the climate change scenario than precipitation. Forests with roots adapted to high groundwater levels should be hypothetically more exposed to changes in the water regime and groundwater decline. The presented assessment procedure can contribute to a better understanding of the effectiveness and limitations of the two main possibilities of manipulation of the soil water balance in lowland; through adjustment of the interception and evapotranspiration through stand canopy attenuation and to use the landscape to retain runoff and overall loss of water from the system. The effectiveness of water table manipulation as an impact management strategy also depends on early recognition of drought. Therefore, permanent measurements of groundwater tables on existing the piezometric network together with other ancillary drought indices is extremely important regarding future uncertainty due to climate change. There are also some other factors, such as tree defoliators, which affect stand canopy and thus can cause some changes in forest water balance but their performance is still unknown and scarcely evaluated.

Forest Canopy Treatment

Common oak forests are known for their water requirements (Čermak et al. 1982; Čermak and Prax 2001; Pietsch et al. 2003), therefore by managing the stand structure one can contribute to lessening the stand water demand and to retention of additional water reserves in soil. According to the presented assessment results, the reduction of forest stand below 15 m^2/ha of basal area can contribute to a prolonged yearly water logging duration of approximately 20% especially on heavy impermeable soils. One of the important parts of precipitation that is altered by forest canopy is interception loss. Interception in common oak forests is related to stand canopy density and also depends on the stand age. Krejči and Vrbek (1995) concluded that interception is lowest in young oak stands (2.3%), significantly larger in middle-aged (24.9%) and in old stands (15.6%). Thisadditional water excess can be used during the drought to partly eliminate its negative impact. However, for this measure to be most effective, it should be synchronised with drought incidence. According to Vogt and Somma (2000) and Roša et al. (2009) the drought in South-Eastern Europe and the Mediterranean has recurring characteristics. Roša et al. (2009) showed that drought in Croatia is a periodic event with relative significant occurrence of 10–11 years which can be explained by increased solar activity cycle. By having insight in the previous historic drought events, this periodicity can help in tempering future silvicultural actions.

Runoff Retention in the Landscape

Historic water tables, prior hydro technical and forest managerial influences were much more related to the landscape gradient or micro topography than today. It can also be concluded that in history, prior to the spread of agriculture and hydro regulations when forests were still intact and covered significantly larger parts of the watersheds, the extent of flooding was significantly lower than nowadays. In the historic floods, the extent of flooded areas were much larger than today because of non existent defence systems, therefore the elevation extent and duration of floods were much lower than today (Vujasinović 1971). All of above mentioned is confirmed through the fact that, due to the necessity for water, most historic human settlements were established very close to the present watercourses of Danube, Sava and Drava Rivers.

Furthermore, the proposed assessment procedure of the historic state of soil water logging based on relict soil indicators could highlight some hotspots, i.e. forest areas with significant water table decline. Gaps in water table depth inside forest complexes determine the origins responsible for groundwater loss or exfiltration (flow out of the soil), i.e. mostly large channels and dry old riverbeds. At the landscape level, to constrain the runoff, possibilities of micro topography should be applied; construction of forest roads to create depression storage, closing drainage ditches and reconstruction of old natural watercourses to stop exfiltration closer to a channel. These measures could partly sustain the realisation of very certain regional climate change scenario in Croatia which predicts a strong decline in the amount of precipitation during the spring, summer and autumn seasons and a possible excess only during the winter season.

Management of Forest Defoliators

Forest defoliators are also one of the unknown, concealed eradicating factors of soil water balance through their impact on reduction of leaf area, i.e. forest canopy. One of the main pedunculate oak forest defoliators is the gypsy moth (*Lymantria dispar*), usually perceived as an important part in the factorial complex of oak forests decline (Thomas 2008). The gypsy moth gradation in Croatian lowlands regularly occurred in the last century in 1948–1950, 1962–1966, 1970–1975, 1982–1984 and 1992–1994 with a clear 10–11 years cycle (Pernek and Pilaš 2005) and was repelled by regular treatments of forest protection service. According to the latest results (Pernek et al. 2008), gradations of gypsy moth in Croatia are synchronised with drought episodes, i.e. the peak of outbreaks usually lags a year or 2 behind the onset of the drought period. Hypothetically, gypsy moth defoliation during the drought causes the lowering of the stand water requirements because of less dense canopy and thus a reduction of the soil water uptake. Some recent observations partly confirm this theory, particularly regarding the point that after mayor gypsy moth gradations the overall decline of oak forests is not as intense as expected. Nevertheless, this hypothesis of abiotic–biotic interactions on oak forest decline and the influence of defoliators on stand water balance should draw focus on future research. Due to the strong significance of its periodicity in Croatian lowland forests (of 10–11 years) and significant relationship of this population density to weather extremes, i.e. the increase of population density is related to higher winter temperatures and the collapse of outbreaks is related to significantly higher precipitation in spring (Pernek et al. 2008), the gypsy moth population density could serve as the bioindicator of future periodic extreme weather events.

Drought Indices

To ascertain information about anomalies of groundwater tables that could have a negative impact on forests, the necessity of a permanent collection of groundwater data is of premium importance. For this purpose, the existing network of piezometric monitoring stations in lowland forests is a necessary tool for monitoring the state of groundwater resources in lowlands in present, very uncertain environmental conditions where the realisation of negative impacts of climate change is increasingly obvious. At a small scale quite satisfactory methodologies also exist for monitoring tree or stand water uptake through measurements of tree stem diameter changes for accurate drought recognition (Pilaš et al. 2007). Also indices for recognition of the onset of drought should be developed and implemented (meteorological indices, seasonal groundwater decline, tree stem diameter changes...).

Conclusion

The European climate gradient of the area of South-Eastern Europe is strongly affected by predicted scenarios of climate change. The predicted changes in climate across the area will most probably cause strong increases of temperatures together with a decrease of precipitation in all seasons but winter. The current hydrologic conditions, highly altered in the past decade through the occurrence of episodes of severe droughts, are becoming more and more unfavourable for the existence of lowland forests consisting of water demanding tree species such as the common oak which strongly depends on additional water supplies from shallow groundwater tables. One option in adaptive forest management strategy in respect to climate change should be groundwater table management, i.e. impact management strategy to retain as much available water in soils as possible through tempering forest canopy treatment, runoff retention inside the landscape, sustaining defoliators and maintaining the monitoring support for timely drought recognition. For the determination of thresholds inside which water table management is functional, the assessment procedure based on historic relict indicators of hydromorphic soils was established according to which two main factors of disturbance of natural groundwater regime, forest management and hydro technical activities, were assessed. Comprehensive timely performance of thinning prior to episodes of drought, measures to reconstruct the natural water regime and care of defoliators can partly sustain negative hydrologic processes in the region strongly threatened by future droughts and water scarcity.

References

Blavet D, Mathe E, Leprun JC (1999) Relations between soil color and water logging duration in representative hillside of the West African granito-gneisic bedrock. Catena 39:187–210

Čermak J, Prax A (2001) Water balance of a Southern Moravian floodplain forest under natural and modified soil water regimes and its ecological consequences. Ann For Sci 58:15–29

Čermak J, Ulehla J, Kučera J, Penka M (1982) Sap flow rate and transpiration dynamics in the full-grown Oak (Quercus robur L.) in floodplain forest exposed to seasonal floods as related to potential evapotranspiration and tree dimensions. Biol Plantarum 24:446–460

European Environment Agency (2006) European forest types. Technical report No 9/2006. http://www.eea.europa.eu/publications/technical_report_2006_9. Accessed 15 June 2009

Fiedler S, Sommer M (2004) Water and redox conditions in wetland soils – their influence on pedogenic oxides and morphology. Soil Sci Soc Am J 68:326–335

Fischer R, Lorenz M, Köhl M, Becher G, Granke O, Christou A (2008) The condition of forests in Europe. Institute for world forestry Executive Report. http://www.icp-forests.org/pdf/ER2008.pdf. Accessed 18 June 2009

He X, Vepraskas MJ, Lindbo DL, Skaggs RW (2003) A method to predict soil saturation frequency and duration from soil color. Soil Sci Soc Am J 67:961–969

Hurt GW, Carlisle VW (2001) Delineating hydric soils. In: Richardson JL, Vepraskas MJ (eds) Wetland soils, genesis, hydrology, landscapes, and classification. CRC Press LLC, Boca Raton, FL

International Union of Soil Sciences Working Group WRB. (2006) World reference base for soil resources 2006. World Soil Resources Reports No. 103. FAO, Rome

Kellomäki S, Leinonen S (2005) Final Report of the Project "Silvicultural Response Strategies to Climatic Change in Management of European Forests" funded by the European Union under the Contract EVK2-2000-00723 (SilviStrat) http://www.efi.int/files/attachments/projects/silvistrat Accessed 10 April 2005

Krejči V, Vrbek B (1995) Distribution of precipitation in the community of pedunculate oak and common hornbeam at the area of Česma basin influenced by the age and species of trees. Šumarski List 9–10:317–322

Mayer B (1995) Scope and meaning of the ground and surface water monitoring for the lowland forests in Croatia. Šumarski List 11–12:383–389

Neteler M, Mitasova H (2003) Open source GIS: a GRASS GIS approach. Kluwer, Dordrecht, The Netherlands

Pernek M, Pilaš I (2005) Gradations of gypsy moth – *Lymantria dispar* L. (*Lep., Lymantriidae*) in Croatia. Šumarski List 5–6:263–270

Pernek M, Pilaš I, Vrbek B, Benko M, Hrašovec B, Milković J (2008) Forecasting the impact of the Gypsy moth on lowland hardwood forests by analyzing the cyclical pattern of population and climate data series. For Ecol Manag 255(5–6):1740–1748

Pietsch AS, Hasenauer H, Kučera J, Čermak J (2003) Modeling effects of hydrological changes on the carbon and nitrogen balance of oak in floodplains. Tree Physiol 23:735–746

Pilaš I (2008) The climate change and hydrology of lowland forests in Croatia. Climate Change III in South-Eastern European Countries: Causes, Impacts, Solutions, 18th and 19th September 2008, Graz, Austria, Joanneum Research. http://www.joanneum.at/climate/Workshop%20Graz/Presentations.html Accessed 5 May 2009

Pilaš I, Vrbek B, Medak D (2005) Application of groundwater monitoring in management of pedunculate oak forests in Croatia. Pp. 275–284. In: Afgan NH, Bogdan Ž, Duić N, Guzović Z (eds) Sustainable development of energy, water and environment systems, vol 2. Faculty of mechanical engineering and naval architecture, Zagreb

Pilaš I, Lukić N, Vrbek B, Dubravac T, Roth V (2007) The effect of groundwater decrease on short and long term variations of radial growth and dieback of mature pedunculate oak (Quercus robur L.) stand. Ekol Bratisl 26(2):122–131

Prpić B (1996) Degradation of pedunculate oak forests. In: Moguš M, Serdarušić A (eds) Pedunculate oak (*Quercus robur* L.) in Croatia. Croatian academy of science and art, Zagreb, pp 459–463

Prpić B, Milković I (2005) The range of floodplain forests today and in the past, Monography: floodplain forests in Croatia. Academia of forest sciences, Zagreb city, city office for agriculture and forestry, Zagreb, pp 37–39

Rauš Đ, Seletković Z, Mayer B, Medvedović J, Raguž D (1996) Forest associations and synecological condition of pedunculate oak. In: Moguš M, Serdarušić A (eds) Pedunculate oak (*Quercus robur* L.) in Croatia. Croatian Academy of Science and Art, Zagreb, pp 385–408

Richardson JL, Arndt JL, Montgomery JL (2001) Hydrology of wetland and related soils. In: Richardson JL, Vepraskas MJ (eds) Wetland soils, genesis, hydrology, landscapes, and classification. CRC Press LLC, Boca Raton, FL

Richardson JL, Daniels RB (1993) Stratigraphic and hydraulic influences on soil color development. In: Bingham JM, Ciolkosz EJ (eds) Soil color. Special Publication no. 31. Soil Science Society of America, Madison, WI

Roša D, Pilaš I, Roša J, Vršnak B, Maričić D, Hržina D (2010) The relationship between solar activity and soil water balance. Sun and Geosphere (in press)

Schulze DG, Nagel JL, Van Scoyoc GE, Henderson TL, Baumgardner MF (1993) Significance of organic matter in determining soil colors. In: Bingham JM, Ciolkosz EJ (eds) Soil color. Special Publication no. 31. Soil Science Society of America, Madison, WI

Schwertmann U (1993) Relations between iron oxides. In: Bingham JM, Ciolkosz EJ (eds) Soil color Special Publication no 31. Soil Science Society of America, Madison, WI

Schwertmann U, Fanning DS (1976) Iron-manganese concentrations in hydrosequence of soils in loess in Bavaria. Soil Sci Soc Am J 41:1013–1018

Tandarich JP, Elledge AL (1996) Determining the extent of presettlement wetlands from hydric soil acreages: a comparison of SSURGO and STATSGO estimates. Hey & Associates, Inc, Chicago, IL

Thomas FM (2008) Recent advances in cause-effect research on oak decline in Europe. CAB Rev: Perspect Agric Vet Sci Nutr Nat Resour 37(3):1–12

UNDP (2008) A Climate for Change. Climate change and its impacts on society and economy of Croatia. Human Development Report Croatia 2008. United Nations Development Programme http://www.undp.hr/show.jsp?page=103395. Accessed 7 April 2009

Vašiček F (1985) Natural conditions of floodplain forests. In: Penka M, Vyskot M, Klimo E, Vašiček F (eds) Developments in agricultural and managed-forest ecology 15A. Floodplain forest ecosystem. I. Before water management measures. Elsevier, Amsterdam, The Netherlands, pp 13–29

Vepraskas MJ (2001) Morphological features of seasonally reduced soils. In: Richardson JL, Vepraskas MJ (eds) Wetland soils, genesis, hydrology, landscapes, and classification. CRC Press LLC, Boca Raton, FL

Vepraskas MJ, Faulkner SP (2001) Redox chemistry of hydric soils. In: Richardson JL, Vepraskas MJ (eds) Wetland soils, genesis, hydrology, landscapes, and classification. CRC Press LLC, Boca Raton, FL

Vogt JV, Somma F (2000) Drought and drought mitigation in Europe. Advances in natural and technological hazards research. Kluwer, Dordrecht, The Netherlands

Vujasinović B (1971) Savjetovanje o Posavini. Poljoprivredni Fakultet, Zagreb

Wenger EL, Zinke A, Gutzweiler KA (2004) Present situation of the European floodplain forests. For Ecol Manag 33–34:5–12

Zaninović K, Gajić-Čapka M (2000) Changes in components of the water balance in the Croatian Lowlands. Theor Appl Clim 65(1–2):111–117

Chapter 9
The Case of Yatir Forest

Gabriel Schiller

Introduction

Aims of Forest Establishment

The forest plantations of about 3,000 ha, planted from 1964 onward, are located 30 km east of the town of Be'er-Sheba, in the hilly region at the northern edge of the Israeli Negev desert (31°21′ N and 35°04′ E, Alt. 600–700 m a.s.l.). The establishment of the plantations, in spite of disapproval from the academic community, was the will and achievement of Josef Weitz, then director of the Land Development Authority of the JNF (Jewish National Fund). Planting of the Yatir forest in the then desolate area had several aims: to mark the border of Israel, to supply ample work for immigrants, and last but not least, to become (in the future) the most important recreation center in the Negev, because of its proximity to, and accessibility from the larger urban centers of the Negev (Aronson and Lovinger 1976).

Ecological Background

The prevailing climatic conditions at Yatir lie between Xero-thermo-Mediterranean and sub-desertic (UNESCO-FAO 1963), i.e., at the transition between the driest sub-humid Mediterranean climate and a semi-arid climate. The average maximum air temperature in the hottest month is 32.3°C, the average air temperature in the coldest month is 6.9°C, and the yearly average air humidity at 1400 h is 45%. Heavy snowfall is not frequent, but when it does occur, severe damage may be inflicted on the forest as a result of canopy breakage or tree bending and uprooting.

G. Schiller (✉)
Department of Natural Resources, Agricultural Research Organization, the Volcani Centre,
P.O. Box 6, Bet Dagan, 50250, Israel
e-mail: vcgabi@volcani.agri.gov.il

M. Bredemeier et al. (eds.), *Forest Management and the Water Cycle: An Ecosystem-Based Approach*, Ecological Studies 212, DOI 10.1007/978-90-481-9834-4_9,

Average annual rainfall (P) at Yatir for the period 1964–2005 was 283 ± 89 mm year^{-1}; the probability of receiving annual rainfall higher than 300 mm year^{-1} is less than 31%, whereas that of receiving annual rainfall less than 200 mm year^{-1} is 12%; in other words, there is a chance for a very droughty winter once in every 8 years. Annual evaporation from a class-A pan is ~2,500 mm year^{-1}; the calculated potential evapotranspiration (PET) is ~1,600 mm, hence the Aridity Index (P/PET) = ~0.18 (Israel Meteorological Service, personal communication). The underground water table lies at a depth of 300 m (Israel Geological Service, personal communication).

Studies on rainfall and runoff in the region performed from 2001 to 2005 recorded no runoff from the forested catchments' area, even under the most severe rain-storm event of 100 mm in 24 h (Safriel 2005). Shachnovich et al. (2008) concluded that soil and bedrock properties contribute to within-site small-scale water redistribution by overland or below-ground flow. Merzer (2007) concluded that even in winters with high amounts of rainfall, the soil wetting front never exceeds 2.5 m in depth. Still, as a result of lithological properties of the bedrock formation, a small amount of deep percolation may occur. This means that all of the rainfall becomes part of the forest site's internal water balance. In general, and in accordance with the topography, the region is covered by 30- to 50-cm thick Leptosols or Regosols, covering a bedrock formation of Turonian era chalk and limestone covered with a compact and hard calcrete crust (Dan and Raz 1970; Schiller 1972). General landscape features of the hilly area have been summarized by Aronson and Lovinger (1976). As a result of soil-bedrock properties and rainfall patterns, the physiological activity of the tree roots is concentrated in the upper soil layers where the moisture content is highly dependent on the annual distribution of rain spells and depth of rainfall (Schiller and Cohen 1998).

The Species Planted

The Yatir forest was planted mainly with Aleppo pine (*Pinus halepensis* Mill.), a drought-resistant (Oppenheimer 1967; Schiller 2000) low-elevation Mediterranean tree species that has a circum-Mediterranean distribution, and grows naturally under sub- to thermo-Mediterranean climatic conditions. Hence, the forest plantations in the semi-arid region of the northern Negev (UNESCO-FAO 1963) are alien ecosystems to the region (Danin 1970).

The species is the only native pine species in Israel and Jordan. It is very easy to handle in nurseries and plantations. As a consequence, it has been planted extensively under Mediterranean climatic conditions since the beginning of forestry activity in the region in the early twentieth century, and elsewhere in the world under similar climatic conditions. Within its range-wide area of distribution, the species shows genetic variability related to eco-geographic regions and it is therefore divided into several genetic groups (Grünwald et al. 1986; Schiller et al. 1986; Korol et al. 2002).

Aleppo pine has a relatively shallow root system with a few main roots penetrating into deeper soil-filled crevices in the bedrock (Oppenheimer 1955, 1957). In shallow soils, growth of Aleppo pine is related to the lithological properties of the bedrock, i.e., the number of crevices and the water-holding capacity of the soil–bedrock complex (Schiller 1982). Soil water content decreases soon after the last winter rainfalls, and Aleppo pine root tips are covered with a corky lamella (metacutis) to prevent their desiccation during the dry season (Leshem 1967). As a consequence of its ecological and physiological features, Aleppo pine became the most planted tree species in the Yatir area, although this area is several tenths of a kilometer to the south of the nearest autochthonous Aleppo pine population.

Research Aims at Yatir

The uniqueness of the Yatir forest plantations growing in a semi-arid area (<300 mm average annual rainfall) presents an excellent opportunity to study interactions between environmental factors and a tree species that originated in more humid ecological environments. Therefore, in the last decade, the Yatir forest has become home to studies on Aleppo pine population genetics (Schiller et al. 2004) and on eco-physiological properties that enable survival and growth of the species in such an environment (e.g., Schiller and Cohen 1998; Grünzweig et al. 2003; Atzmon et al. 2004; Klein et al. 2005; Safriel 2005; Maseyk 2006; Maseyk et al. 2008a, b; Schiller and Atzmon 2009).

Climate trends observed over Israel in the last 30 years have shown east-ward and south-ward increasing air temperature and aridity (Kafle and Brunis 2009). Forecasts of increasing aridity in the East Mediterranean region in the next decades (IPCC 2007) and the occurrence of droughty winters (Paz 2004) make information on the annual course of water-balance partitioning among the forest components indispensable to the preparation of ecosystem-management plans. Such plans should be aimed at enhancing the sustainability of the forest, which might be jeopardized as a result of harsh competition for water among the vegetative components of this man-made ecosystem.

A globally widespread increase in reported tree mortality and forest decline has been attributed to more frequent drought and heat events (Allen et al. 2009; Van Mantgem and Stephenson 2007; Chapter 3, this volume). According to the physiological characters of broadleaf or conifers species, mortality was attributed to prolonged embolism or carbohydrate starvation (McDowell et al. 2008; Adams et al. 2009; Allen et al. 2009; Breshears et al. 2009), raising the question: What is the carrying capacity, or in other words, the stocking density in different age classes that will enable the forest to survive at annual rainfall amounts that are 30% less of the present average annual rainfall of 283 ± 89 mm year^{-1}? Several more questions arise with respect to better forest management because the present tree generation, planted before 1975, i.e., between 28 and 38 years of age in 2003, has passed the stage of maximum yearly height and diameter growth rates (Roehle 1991). None of the problems related to natural regeneration have been solved (Yavlovich 2008) and in the face of the growing social demands from the forest, its sustainability has become one of the greatest challenges for forest managers.

The aim of this chapter is to summarize the results of research performed in Yatir forest on:

1. Partitioning of water balance in a rain-fed semi-arid Aleppo pine forest. Of interest is the annual course of transpiration in relation to the annual course of rainfall, soil water content, ecosystem evaporation and potential evapotranspiration as influenced by the ambient meteorological parameters measured by an eddy covariance system on a flux tower erected in the geographic centre of the Yatir forest.
2. The eco-physiological adaptation of a Mediterranean pine species to semi-arid climatic conditions, i.e., relations of the daily, seasonal and annual course of photosynthesis, respiration and phenology to semi-arid climatic conditions.
3. Changes in the genetic constituency of the population driven by the harsh ecological conditions that enables the species to cope and survive in that area.

Materials and Methods

Research Sites and Trees

Three plots of 0.25 ha each were selected in 2003 within the Yatir forest according to slope orientation:

1. Plot situated on a south-east-facing slope (azimuth 113°, inclination of 45%)
2. Plot situated on a north-west-facing slope (azimuth 290°, inclination of 32%)
3. Plot situated on a plateau with about 3% inclination to the north-west

These plots are within the circular "foot-print" area of the meteorological tower which has a radius of 1,000 m. The tower was erected at the geographic centre of the Yatir forest on a plateau.

The Aleppo pine trees growing within that "foot-print" area were 34 years of age in 2003, with a few trees being up to 4 years younger resulting from replacement plantings. Average stocking density was ~300 trees ha^{-1}; tree height averaged 11.2 m (±2.3 m SD), and DBH (diameter at breast height) averaged 16.9 cm (±4.8 cm SD), resulting in a basal area (G) of ~7 m^2 ha^{-1}; leaf area index (LAI) in 2003 was estimated at approximately 1.6 (M. Sprinstin, personal communication) and tree's canopy projection area covered about ~50% of the area of the plot's forest floor.

During the hydrological years Oct 2003–Sept 2004 and Oct 2004–Sept 2005, three sets of equipment were deployed, one at each plot. During the hydrological year Oct 2005–Sept 2006, only one set was deployed in the forest stand surrounding the meteorological tower on the plateau. In all cases hydrological year refers to 1 Oct–30 Sept of the following year.

The rainfall data were collected from the standard station located at the forest administration building, 1.5 km south-west of the meteorological tower. Rainfall during the hydrological years 2003–2004, 2004–2005 and 2005–2006 amounted to 231, 374.5 and 224 mm year^{-1}, respectively.

Instrumentation and Measurements

Sap Velocity Measurements and Hourly Sap Flux Calculations

Sap flow velocity (V, cm h^{-1}) was determined by means of the heat-pulse method (Cohen 1994), which uses the convective heat velocity as a tracer for sap velocity, using the same configurations as in earlier studies (Schiller and Cohen 1995, 1998). Three measurement systems were used, one in each plot. Each system was comprised of a battery-powered data logger (21X, Campbell Scientific, Logan, UT, USA) connected to a custom multiplexer (model TJB 818, Ariel, Tel Aviv, Israel) and eight heaters and probes (produced in-house), 60 mm in length, each bearing six thermistors 8 mm apart along the axis; this means that sap velocity was measured in the xylem to a depth of 48 mm. Heaters and probes were inserted radially into the trunks, with the probe 15 mm above the heater, at 1.30 m above ground as in previous studies (Schiller and Cohen 1995; 1998). The system enabled measurements of sap flow velocity in the trunk's sapwood xylem of eight Aleppo pine trees per plot by cycling through the group every hour and measuring HPV (Heat Pulse Velocity) in each for 7.5 min per cycle.

Sap flow velocity measurements during the 2003–2004 hydrological year were begun on 26 Jan 2004 (day 118 of the hydrological year), when the instrumentation started to function, and taken until 16 June 2004 (day 258), i.e., 140 days. During the 2004–2005 hydrological year, measurements were taken from 3 Nov 2004 (day 34), before the start of rainfall, to 18 June 2005 (day 261), i.e., 227 days. In the 2005–2006 hydrological year, measurements were taken from 28 Oct 2005 (day 28), before rainfall started, to 4 June 2006 (day 247), i.e., 219 days. The interruption in measurements from June to October was due to the very low sap flow velocities during those months, less than 0.040 m h^{-1}. These data are considered unreliable due to the inherent constraints of the heat-pulse technique (Schiller and Cohen 1995).

In trees, hourly sap flux is considered to be the hourly transpiration rate (T_{tree}, l h^{-1}), which was computed from the hourly sap velocity measured by each of the six thermistors per probe inserted into each tree at breast height, the physical properties of the wood, the conducting xylem sap wood area (Schiller and Cohen 1995) and the calibration coefficient. The calibration coefficient, which is the slope of the linear regression between the hourly or daily gravimetrically determined water loss and the calculated sap flux, was 0.565 (Schiller and Cohen 1998), similar to that reported by Swanson and Whitfield (1981). The following calculations were performed using the computed hourly transpiration in each tree:

1. Summation of the hourly T rates to daily (24 h) transpiration rate (T_{tree}, l day^{-1}).
2. Summation of the hourly T rates to daytime (0600 to 1900 h) transpiration rate (T_{tree}, l day^{-1}).
3. Average hourly or daily T rates of the eight trees per plot (T_{tree}, l h^{-1} or T_{tree}, l day^{-1}).

Total transpiration by the Aleppo pine forest stand, i.e., forest canopy transpiration (T_{canopy}, mm day^{-1}), was calculated by multiplying the average hourly or daily T rate of the eight trees per plot (l h^{-1} $tree^{-1}$ or l day^{-1} $tree^{-1}$) by the stocking density (300 trees ha^{-1}) using Eq. (9.1):

$$T_{canopy}\ (\text{mm h}^{-1} \text{ or mm day}^{-1}) = \text{average hourly or daily T rate* } 300/10000, \quad (9.1)$$

where division by 10000 is the transformation of liters per ha into mm.

Soil Volumetric Water Content

Soil volumetric water content (SWC, m^3m^{-3}) was measured by two systems that were integrated at the research site (Raz Yaseef 2008). The first consisted of three reflectometry sensors (CS616, Campbell Scientific) positioned vertically in the ground and measuring the average water content for a soil depth of 0–30 cm. A specific calibration equation was prepared in the laboratory for these sensors to fit the dense soil at the Yatir site. In addition, their high sensitivity to soil temperatures demanded a linked soil temperature measurement, which was carried out at depths of 1, 5, 15 and 30 cm within close proximity to the CS616 sensors (HOBO H8 loggers, Onset Computers, USA). An integral average of these soil temperatures was then used with the factory-defined temperature correction equation. Since 2005, a second system based on TDR sensors (TRIME, IMKO, Germany) was integrated, measuring θ at different depths in three pits (pit 1: 5, 15, 30, 50, 70 and 125 cm; pit 2: 5, 15 and 30 cm; pit 3: 5, 15, 30, 50 and 70 cm).

The Meteorological Tower

An instrumented tower was erected in the geographic centre of the Yatir forest (31° 20′ N; 35° 3′ E) (E. Rotenberg, personal communication) which stretches 8 km in the east-west direction, and 4 km in the south-north direction. The tower is part of the Carboeuroflux network (http://www.carboeurope.org/), and measurements follow the European methodology using Unitus software (University of Tuscia, Italy). Rainfall data presented here were taken from the standard station located at the KKL forest house, 1.5 km south-west of the meteorological tower.

Evapotranspiration

Evapotranspiration at Yatir was computed (S. Cohen, personal communication) according to the procedure recommended by Allen et al. (1998). This procedure simulates the evapotranspiration of a well-irrigated 12-cm tall grass surface according to the Penman–Monteith equation (Monteith 1965) applied to ambient climate data measured by the meteorological tower. The climate data used to solve the equations were derived from half-hourly averages of global radiation, temperature, relative humidity and wind speed measurements. For more information on the procedure see Allen et al. (1998). The climate data were all measured at a height of 15 m, or approximately 4 m above the tree tops. Soil heat flux was assumed to be negligible for the daily averages. The calculations were also done using wind speed computed for two meter height above the zero plane displacement (as done in agricultural canopies). This resulted in very low wind speeds and unacceptably high aerodynamic resistance. Therefore, the presented data were calculated using tower wind speed as that for two meters height in the Allen et al. (1998) procedure.

Water-Balance Partitioning in the Forest

Daily, Seasonal and Annual Evapotranspiration in Relation to Meteorological Parameters, Soil Water Availability and Topography

A major component in forest water-balance partitioning is the evapotranspiration (ET), i.e., evaporation from the soil surface (E) and water used by the vegetation (in this case transpiration by the forest trees) (T). Transpiration by the Aleppo pine trees was studied during the hydrological years (1992–1993, 2003–2004, 2004–2005 and 2005–2006, using the heat-pulse system (Cohen 1994). The measured sap velocity, translated to transpiration (mm day^{-1}) (Schiller and Cohen 1995) was related to the annual rainfall (mm year^{-1}), soil water content at 30 cm depth (SWC$_{-0.30}$, m^{3}m^{-3}), vapour pressure deficit (VPD, kPa) and potential evapotranspiration (PET, mm day^{-1}) resulting from ambient meteorological data measured by the flux tower.

Measurements in 1992–1993

At several periods in the hydrological year 1992–1993, 24-h measurements of the hourly sap flow velocity were performed continuously over several days, in the oldest section of the Aleppo pine forest planted in winter 1964–1965. The forest then had a stocking density of ~600 trees ha^{-1}, with a basal area (G) of ~8.9 m^2 ha^{-1}, and LAI of ~2.0. Figure 9.1 shows the annual course of rainfall, average daily canopy transpiration rate (T$_{canopy}$), and the relation between daily T rates and PET. Transpiration was high during the wet season, i.e., December to March, culminated in March and declined steeply in April–May; in June, daily T rates were between 0.15 and 0.25 mm day^{-1} and T/PET ratios were ~0.03. Summation of the daily

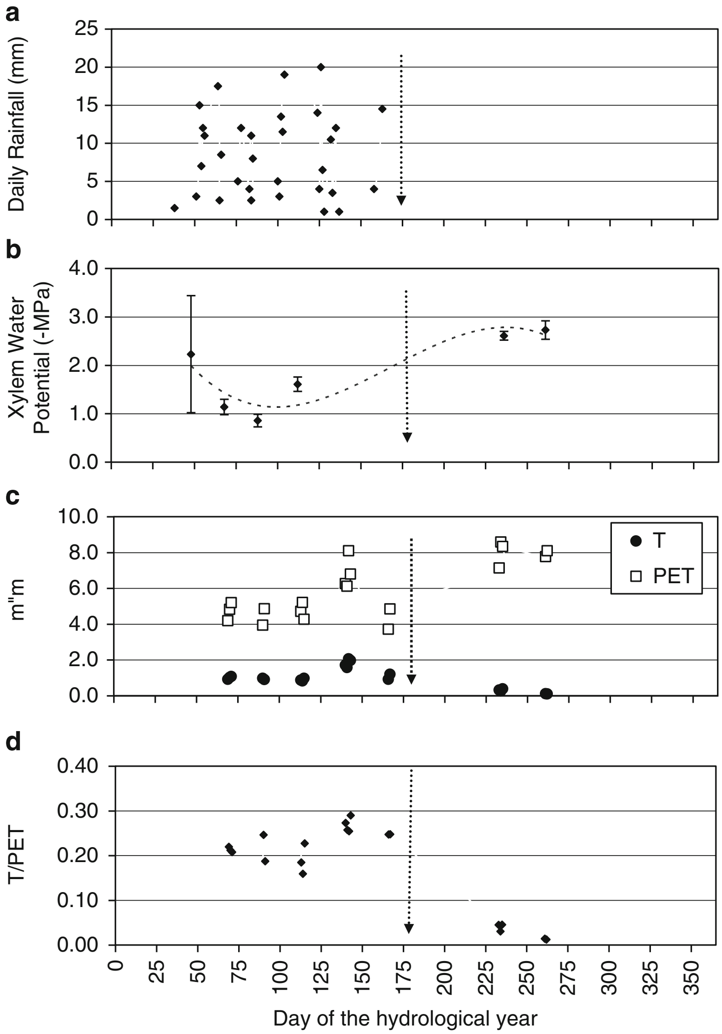


Fig. 9.1 Annual courses of: daily rainfall (**a**), predawn xylem water potential in MPa (**b**), transpiration-(T) and potential evapotranspiration-(PET) rates in mm (**c**), and relations of T to PET (**d**), during the 1992–1993 hydrological year at Yatir. (*Dashed arrows* point the end of the rainy season at that year)

T rates during the hydrological year gave 210 mm, ~80% of the 262 mm annual rainfall. The result led to the conclusion that there is an urgent need to reduce stocking density by thinning in order to alleviate competition for water (Schiller and Cohen 1998). Thinning soon after to a stocking density of ~300 trees ha^{-1} probably saved the forest from total destruction during the hydrological years 1998–1999 and 1999–2000, in which annual rainfall was only 144 and 150 mm, respectively; many non-thinned forests in the southern region of Israel declined in those 2 years.

Measurements During 2003–2006

Transpiration in Relation to SWC, ET and PET

Figure 9.2 clearly depicts the dependence of T_{canopy} (mm day^{-1}) on the daily SWC (m^3 m^{-3}), which depends solely on the annual distribution of storm events and rainfall depth of each storm (Fig. 9.3). Daily T_{canopy} declined sharply when SWC at a depth of 0.30 m ($SWC_{-0.30}$) declined to less than ~0.20 m^3 m^{-3}, which happened shortly after the last significant rain storm. When $SWC_{-0.30}$ was reduced to ~0.10 m^3 m^{-3}, which occurred toward the end of May–beginning of June, sap flow velocities were equal to or less than 0.040 m h^{-1}; such velocities measured using the heat-pulse technique are considered unreliable because of constraints inherent to the method (Schiller and Cohen 1995*)*. Using that limit of sap flow velocity measurements and the area of the conducting xylem of each tree, we calculated the daily transpiration of each tree for the rest of the hydrological year (from end of May to October). This amount was then added to the sum of measured forest canopy transpiration. The overall "water use" (Table 9.1), i.e., measured and estimated transpiration amounts, was 148, 181 and 135 mm year^{-1}, representing ~65%, ~48% and ~60%, respectively, of the annual rainfall in the hydrological years 2003–2004, 2004–2005, 2005–2006.

Summed daily T_{canopy} during the measurement periods in the 3 hydrological years constituted ~59%, ~51%, and ~70%, respectively, of the summed ecosystem evapotranspiration (ET, mm day^{-1}) during the daytime (0600 to 1900 h); furthermore, summed daily T_{canopy} constituted ~14%, ~20% and ~15%, respectively, of the summed PET (mm day^{-1}).

In this semi-arid area, the importance of rain-storm intensity, in addition to rainstorm amounts, is revealed in Fig. 9.3, which shows that the hydrological years 2003–2004 and 2005–2006 had similar rain amounts of 231 and 224 mm, respectively, but very different storm intensities; hence, they had very different impacts on the soil profile water content resulting in very different daily T rates and different T/PET (Fig. 9.4) ratios.

In the hydrological year 2003–2004 and 2004–2005, as long as $SWC_{-0.30}$ was higher than 0.20 m^3 m^{-3}, the average T/PET ratio was 0.36 and 0.42, respectively; with the decline in $SWC_{-0.30}$, the average T/PET ratio declined to between 0.17 and 0.05. In the hydrological year 2005–2006, as a result of the low $SWC_{-0.30}$, the average T/PET ratio was only 0.14 for most of the period. These results corresponded well with those measured earlier, in which the T/PET ratio in Feb 1993 was between 0.25 and 0.30. These data suggest that excluding the period during which $SWC_{-0.30}$ is higher than 0.20 m^3 m^{-3} and the T/PET ratio is maximal at between 0.3 and 0.4; canopy transpiration is strongly limited by insufficient water availability throughout the hydrological year. In comparison, the T/PET ratio of Aleppo pine forests on Mt. Carmel in the wet season (average annual rainfall of 550 mm, stocking density 1,600 trees ha^{-1}, basal area (G) of 19.5 m^2 ha^{-1}) was between 0.40 and 0.60 (Schiller and Cohen 1995), which is twice the T/PET ratio at Yatir during periods of water availability.

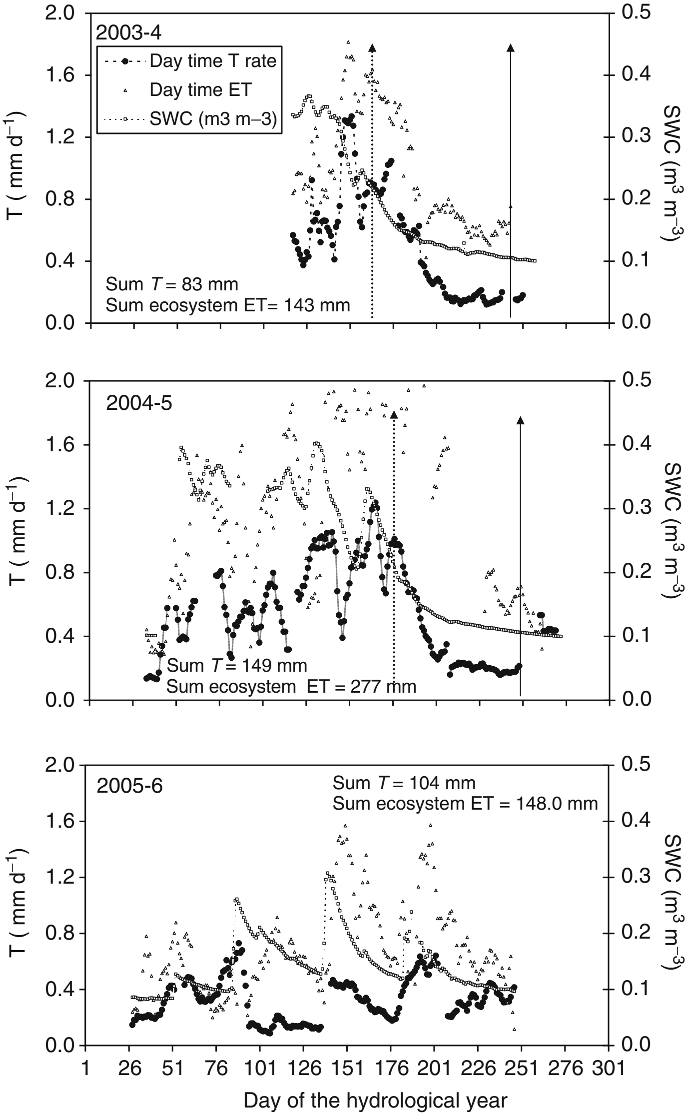


Fig. 9.2 Annual courses during the hydrological years of 2003–2004, 2004–2005, 2005–2006, of daily transpiration rates (T, mm), daily ecosystem evapotranspiration rates (ET, mm) and soil water content (SWC, m^3m^{-3}). (*Dashed arrows* indicate date on which SWC reached 0.20 m^3 m^{-3}; *solid arrows* indicate date on which SWC reached 0.10 m^3 m^{-3})

There is a high correlation between a tree's basal area and its transpiration during the wet period (Schiller and Cohen 1998); hence, the difference between the two sites (Yatir and Mt. Carmel) in T/PET ratio may result from the difference in their basal areas, which in Yatir is about half of that in Mt. Carmel forest. The early decline of the T/PET ratio (Fig. 9.3) suggests that canopy transpiration was limited by insufficient $SWC_{-0.30}$.

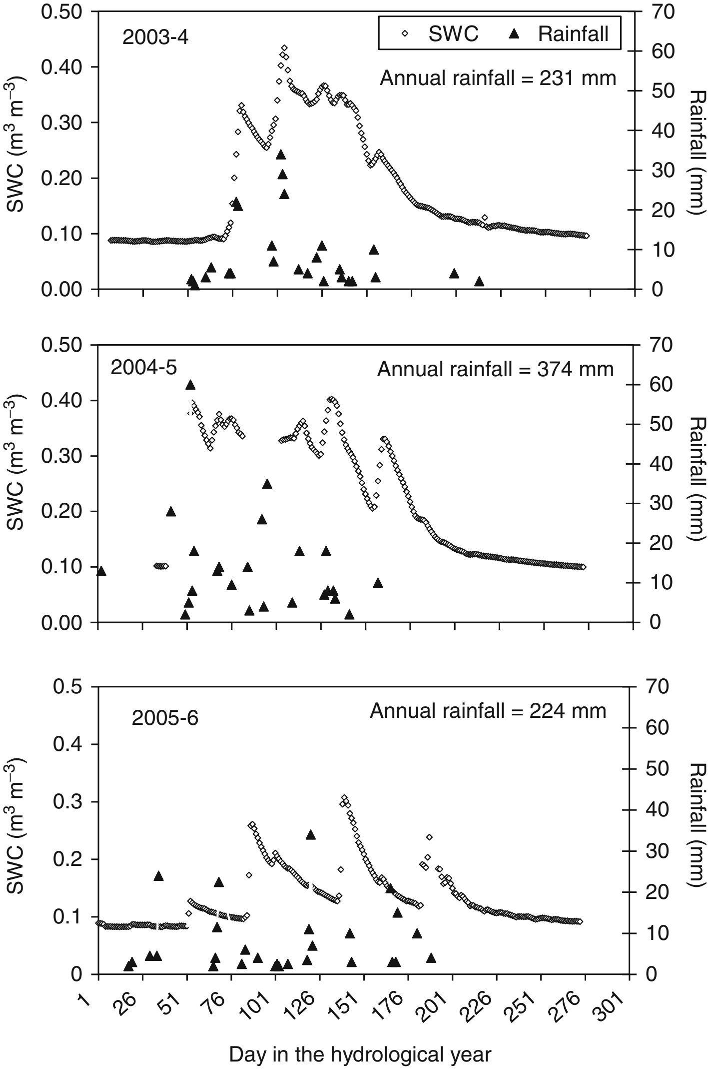


Fig. 9.3 Annual course of soil water content (SWC, $m^3\ m^{-3}$) as determined by rainfall events distribution and amounts (mm) during the hydrological years 2003–2004, 2004–2005, 2005–2006

Slope Orientation Influence on Transpiration

The ecological factors of slope aspect and inclination are known to influence vegetation type, growth and survival in our region (Zohary 1962) and elsewhere. The Yatir forest stretches across a hilly region, i.e., most of the forest covers slopes of different aspects. Analysis of variance (ANOVA) was implemented on a set of light hour's transpiration data (l day^{-1}) from no less than five trees per plot in each

Table 9.1 Measured and estimated transpiration rates (mm year^{-1}) by 30 *Pinus halepensis* trees per 1,000 m^2 at Yatir during 3 hydrological years

Hydrological year (1 Oct–30 Sep)	2003–2004	2004–2005	2005–2006
Period of HPV measurements	26 Feb 04–14 June 04	27 Oct 04–18 June 05	28 Oct 05–04 June 06
Number of days	140	227	219
Annual rainfall (mm)	231.0	373.5	224.0
Measured transpiration during the period (mm)	83.0	141.0	104.0
Estimated transpiration for the rest of the hydrological year (mm)	65.5	35.7	30.0
Sum total transpiration by Aleppo pine (mm)	148.5	176.7	134.0
Total transpiration in % of annual rainfall	64.3	47.3	59.8

of the three plots during 3 successive days simultaneously, in the hydrological years 2003–2004 and 2004–2005. The number of days for which these conditions were fulfilled was only 30 and 33 in the respective hydrological years; the total amount of observations included in the analysis was 778. The sources of variance included in the analysis were: year, plot, day in the year, [year × plot] and [tree (plot)]; each of these parameters had a significant influence on the average daily canopy transpiration rate ($F < 0.0001$), with an R^2 of 0.757. ANOVA results (Table 9.2) showed that average daily T_{canopy} (mm day^{-1}) differed significantly among the slopes during the 30 days in hydrological year 2003–2004, and that they did not differ significantly during the 33 days in hydrological year 2004–2005. The same trees transpired higher quantities in 2004–2005 than in 2003–2004, probably because of the higher rainfall amount in the winter of 2004–2005 (374 mm) than in that of 2003–2004 (231 mm). The differences among slope aspects in the annual courses of daily T_{canopy} rates were more pronounced in 2003–2004 than in 2004–2005. Summation of the transpiration amounts according to slope aspects (Table 9.3) shows that in less rainy years, slope aspect exerts a larger influence on the daily and periodic transpiration rate than in more rainy years.

Division of Evapotranspiration into Transpiration and Ecosystem Evaporation

In addition to the results of the forest T_{canopy} measurements (mm day^{-1}) and their relation to ET measured by the flux tower, fine-scale partitioning of the ET flux in the forest was calculated from measurements made with a portable infra-red gas analyzer (Raz Yaseef 2008). Forest soil evaporation (E) was measured during 24 h at several points within a horizontal forest site. The daily high spatial variation of E (SD of up to ±47%) was best correlated with photo-synthetically active radiation (PAR) and SWC at each point of measurement. According to Raz Yaseef (2008), seasonal patterns of

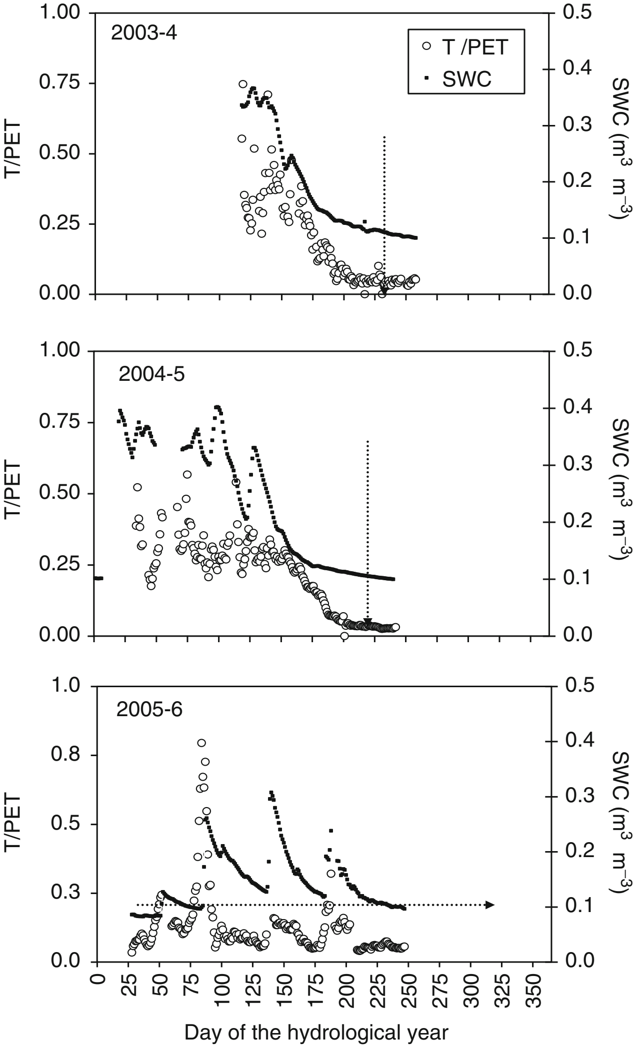


Fig. 9.4 Annual courses of T/PET ratios as determined by the soil water content (SWC, m^3m^{-3}) during the hydrological years 2003–2004, 2004–2005, 2005–2006 (The *dashed arrows* indicate the date on which SWC reached 0.10 m^3 m^{-3})

E and T differed. E peaked twice, during early and late winter (up to 0.80 mm day^{-1}), and T peaked in spring (up to 1.20 mm day^{-1}). Low E and T were measured in the summer while $SWC_{-0.30}$ was below 0.10 m^3m^{-3}. E was correlated to environmental parameters on a seasonal time scale: with $SWC_{-0.30}$ during the wetting and drying seasons (rainy and after the rainy season), and with net radiation during the winter and dry summer. The ET component of the hydrological equation could be divided into E of 36% and T of 45%. Using rainfall, differences in SWC, soil ET and percolation

Table 9.2 Differences among slope orientations in average daily transpiration rates (l day^{-1}tree^{-1}) of the average *Pinus halepensis* tree during the hydrological years of 2003–2004 and 2004–2005 at Yatir, using ANOVA

Hydrological year and plot	Average daily *T* rate (l day^{-1} tree^{-1})	SD
Year 2003–2004		
South-east facing slope	14.8	1.0
North- west facing slope	8.8	0.7
Plateau area (around tower)	16.8	0.6
Year 2004–2005		
South-east facing slope	18.5	0.9
North-west facing slope	20.2	1.1
Plateau area (around tower)	20.7	0.6

and Aleppo pine T, a complete annual ecosystem hydrological budget was constructed for the Yatir forest (Raz Yaseef 2008). The results indicated that precipitation was supplemented by vapour adsorption of about 5% on average in late summer. Average rainfall interception at the horizontal site was 11%, soil storage runoff and deep drainage were estimated to be on the order of 6%, and ET was 81%.

Physiological Activity Under Semi-arid Climate

The properties that enable Aleppo pine to withstand stressful environmental conditions in the summer (drought and ozone) under Mediterranean climatic conditions (>350–400 mm average annual rainfall) were reviewed by Barnes et al. (2000), Schiller (2000) and Inclán et al. (2005). Hence, the genetic and physiological properties that enable this species to cope with even more pronounced drought conditions, i.e. <300 mm average annual rainfall in semi-arid environments, are of interest in relation to the predicted climate change.

Carbon sequestration by plants is their only means of growth and/or maintenance. At Yatir, the period during which ecological conditions, i.e., SWC, are sufficient for high rates of physiological activity is short, on average between December and April (Figs. 9.1–9.3), i.e., atmospheric CO_2 uptake is highest during the wet season. Still, at the age of 35 years (2005), this Aleppo pine forest (planting density 1,800 per ha^{-1}, concurrent stocking density 300 trees ha^{-1}) contained 6.5 ± 1.2 t C ha^{-1}, i.e., net ecosystem productivity (NEP) of 2.1 t C ha^{-1} year^{-1}; it continues to accumulate 0.13–0.24 t C ha^{-1} year^{-1}. This amount of carbon sequestration is similar to that sequestered on average by the European forests (Grünzweig et al. 2003). This relatively high NEP requires adaptation of physiological and phenological processes to the given ecological conditions. Hence, the daily, seasonal and annual net photosynthetic rate, transpiration rate, intracellular CO_2 concentration, foliage, stem and soil respiration, were measured using a portable infra-red gas analyzer. In addition, concurrent year needle growth and nitrogen contents were measured (Maseyk 2006).

Table 9.3 Differences among slope orientations in: the average daily canopy transpiration rates (mm day^{-1}); estimated total of daily e transpiration (T, mm); average daytime T rates in percent of daily T rate; Total transpiration as percent of ET and PET

	Sites			
	Plateau	North-west facing slope	South east facing slope	Global averages
Hydrological year 2004				
Measurement period: 26 Jan 04–16 June 04				
Rainfall (mm): 231.0				
Number of days in the period: 140				
Average daily T (mm day^{-1}) (±SD)	0.592 ± 0.355	0.351 ± 0.150	0.820 ± 0.691	0.588 ± 0.357
Estimated total of daytime transpiration (T, mm)	82.9	49.1	114.9	82.4
Average daytime T rate in percent of daily T rate (%)	84.7	73.1	84.9	82.0
Total transpiration in percent of daytime ET (%)	58.1	34.4	80.5	57.8
Total transpiration in percent of total PET (%)	9.3	7.7	13.3	9.1
Hydrological year: 2005				
Measurement period: 3 Nov 04-18 June 05				
Rainfall (mm): 373.5				
Number of days in the period: 227				
Average daily T (mm day^{-1}) (±SD.)	0.621 ± 0.336	0.792 ± 0.286	0.744 ± 0.319	0.665 ± 0.315
Estimated total of daytime transpiration (T, mm)	141.1	179.8	168.9	148.7
Average daytime T rate in percent of total T rate (%)	85.7	68.0	84.1	80.9
Total transpiration in percent of total ET (%)	48.5	61.8	58.1	53.8
Total transpiration in percent of total PET (%)	18.0	22.9	21.6	18.9

Pine needle phenology involves spring flushing and growth through the stressful summer period. According to Maseyk et al. (2008a), "Peak photosynthesis was in late winter, when high soil moisture, mild temperatures and low leaf VPD allowed high rates associated with high water and nitrogen use efficiency. Self-sustained new needle growth through the dry and hot summer maximized photosynthesis in the following wet season, without straining carbon storage. Low rates of water loss were associated with increasing sensitivity of stomatal conductance to soil moisture below relative extractable water amount (REW) of 0.4, (equivalent to SWC of 0.20 $m^3\ m^{-3}$) and decreased stomatal conductance to VPD below REW of approx. 0.2. While most of the physiological parameters and responses measured were typical of temperate pines, the photosynthesis-phenological phasing contributed to high productivity under warm-dry conditions. This contrasts with reported effects of short-term periodical droughts that could lead to different predictions of the effect of warming and drying climate on pine forest productivity".

Efficient energy dissipation is necessary to protect the photosystem from damage under summer drought conditions (Martinez-Ferri et al. 2000). According to Maseyk (2006) "The capacity for photosynthetic activity through summer was facilitated by effective photo protection mechanisms. Both pre- and post-photochemical electron transport energy dissipation was employed to prevent any chronic reduction of photo system efficiency under the conditions of high light and low stomatal conductance. The relative importance of various protection mechanisms depended on the nature and extent of the drought stress, but pigment bed thermal dissipation was seen to be more important than sustaining electron flow through photo respiration". Maseyk et al. (2008b) concluded that "Autotrophic respiration in an evergreen pine forest displayed significant acclimation to warm and dry summer conditions. This was dominated by reductions in normalized respiration (R_{20}) and accompanied by reductions in temperature sensitivity (Q_{10}) leading to low foliage (photosynthesis driven) and stem (phenology driven) respiration rates over summer. These responses underline a relatively high annual-scale carbon-use efficiency, which, combined with high levels of primary productivity, contribute to high ecosystem productivity in this seasonally dry forest".

Results From a Provenance Trial at Yatir

Provenance trials are a common way of analysing the adaptiveness of different seed sources (ecotypes/genotypes) to a given set of ecological conditions. A review on among-provenance variation of Aleppo pine in Israel at the seed and seedling stage (Schiller and Waisel 1989) showed the existence of significant differences in heterozygosity (by isoenzyme analysis) and eco-physiological and growth parameters. Significant differences exist among circum-Mediterranean provenances of Aleppo pine as well (Bariteau et al. 2000; Schiller 2000; Korol et al. 2002). Therefore, provenance trials were established within Yatir forest and at Bet Dagan (in the coastal plain, average annual rainfall of ~550 mm) to examine adaptation of

Israeli and overseas Aleppo pine provenances (Morandini 1976) to semi-arid ecological conditions. Of the 22 provenances planted in 18 incomplete replications in the "Yatir-east" provenance trial in 1985, five provenances, namely, the three best provenances in terms of growth and survival rate and the two least developed provenances were analyzed for their genetic and eco-physiological properties (Schiller et al. 2001; Schiller and Atzmon 2009).

Habitat Effects on Adaptive Genetic Variation in *Pinus halepensis* (Aleppo Pine) Provenances

The genetic diversity and structure of the more southern Israeli autochthonous Aleppo pine populations (Mt. Carmel, Umm-Safa, Bet-Meir) and a 36-year-old Aleppo pine stand at Yatir (planted in 1964–1965) were analysed using isoenzyme starch gel electrophoresis (Schiller et al. 2004). Percentage of polymorphic loci, at the 1% criterion, was found to be 14.9 in the autochthonous populations and 17.8 at Yatir. From these results, it can be deduced that the environmental selection pressure in Yatir forest is such that only individuals with a higher amount of polymorphic loci are able to withstand it (Schiller et al. 2004). Hence, forest sites at Yatir subjected to recent eco-physiological research have undergone selection that favored the more adapted individuals.

Twenty years after planting of the provenance trial, average survival rate of the 22 planted provenances declined to 38 ± 21% with a range of between 0.0% and 92%. The heterozygosity of the surviving trees in the few provenances analyzed in comparison with the heterozygosity of the autochthonous populations that were the seed source was determined by means of isoenzyme (Table 9.4) and DNA analysis (Table 9.5). The two tables show that all of the genetic estimates were higher in the Yatir provenance trial than in the autochthonous populations. This phenomenon is probably due to the heavy selection pressure exerted by the ecological conditions in the area, which eliminated the less heterozygous individuals (Schiller et al. 2004).

Site Effects on Physiological Aspects

Predawn needle water potential (PNWP) was measured on trees of several provenances during the hydrological year 1998–1999 with only 144 and 250 mm annual rainfall at Yatir and Bet Dagan, respectively, using a pressure chamber (Scholander et al. 1965). The annual course of PNWP in four of the selected provenances growing at Yatir and at Bet Dagan (Fig. 9.5) revealed significant differences within and between the two sites. During most of the measurement period, PNWP values were higher than 2.5 (−MPa) in all trees at Yatir, with minor differences between the provenances; at Bet Dagan, PNWP values were lower than 2.5 (−MPa), with significant differences among provenances. These data indicate that the severe water stress throughout the whole of that hydrological year at Yatir

Table 9.4 Genetic variability parameters at the allelic level in few autochthonous *Pinus halepensis* populations (natural) and those planted at Yatir (After Schiller et al. 2004)

Population	N	Al/L	P%[a]	Hobs	Hexp.[b]
Telagh (Yatir)	5	1.6	52	0.216	0.221
Telagh (Natural)	–	–	–	–	–
Elea (Yatir)	26	1.9	60	0.220	0.214
Elea (Natural)	76	2.0	52	0.138	0.188
Otricoli (Yatir)	22	1.6	36	0.145	0.152
Otricoli (Natural)	–	–	–	–	–
Bet J'ann (Yatir)	16	1.5	40	0.160	0.133
Bet J'ann (Natural)	27	1.3	12	0.049	0.054
Mt. Carmel (Yatir)	18	1.7	48	0.144	0.153
Mt. Carmel (Natural)	238	1.8	44	0.116	0.152

N = Mean sample size per locus
Al/L = Mean number of alleles per locus
P% = Percentage of polymorphic loci
Hobs = observed heterozygosity
Hexp = expected level of heterozygosity
[a] A locus is considered polymorphic if the frequency, of the most common allele, does not exceed 0.95
[b] Unbiased estimate (see Nei 1978).

Table 9.5 Genetic diversity parameters at the nucleotide level in autochthonous *Pinus halepensis* populations and those planted at Yatir (After Schiller et al. 2004)

Population	Site	N	na	h	I
Yirka	Natural	31	2.0	0.316	0.476
Bet J'ann	Natural	20	1.8	0.264	0.405
Mt. Carmel	Natural	27	1.9	0.303	0.452
Bet J'ann	Yatir	14	2.0	0.431	0.619
Mt. Carmel	Yatir	14	2.0	0.406	0.589

na = Observed number of alleles per RAPD locus
h = Nei's (1973) gene diversity
I = Shannon's Information Index (Lewontin 1972).

(water potentials >1.8 –MPa) caused stomata closure, leading to carbon starvation and associated increases in susceptibility to other disturbances (Breshears et al. 2009). On the other hand, at Bet Dagan, gas exchange took place on at least 100 days, when water potentials were <1.8 (–MPa) (Melzack et al. 1985).

Isotopic carbon ratio, $^{13}C/^{12}C = \delta^{13}C$ ($^{0}/_{00}$), is another parameter used to evaluate drought tolerance among provenances within species (Bariteau et al. 2000). This ratio was measured in 1-year-old needles picked from 12 Aleppo pine trees per each of several provenances growing at Yatir and Bet Dagan. $\delta^{13}C$ values in these needles, picked in the summers of 1998 and 1999 (Fig. 9.6), were significantly less negative at Yatir than at Bet Dagan, suggesting enrichment in ^{13}C, which indicates

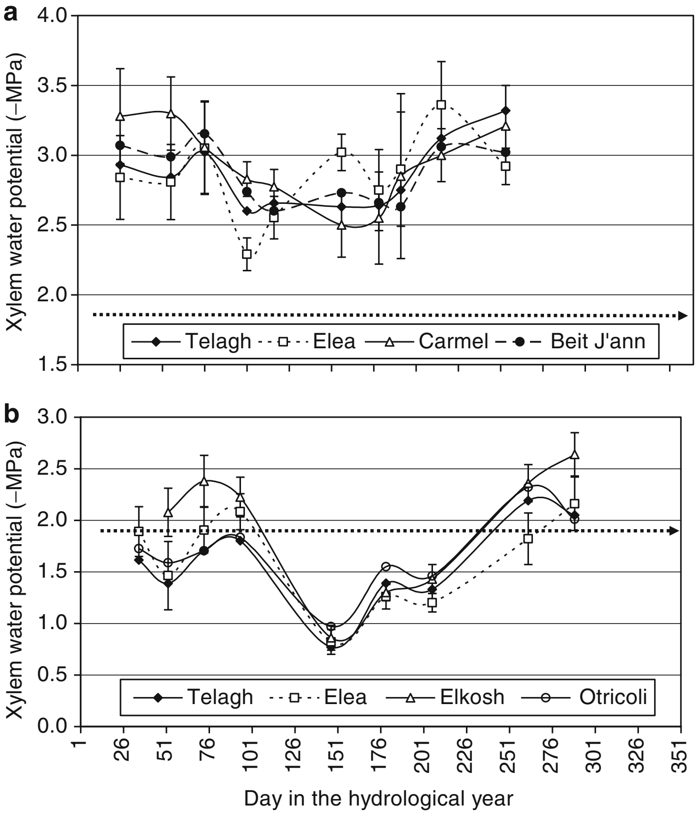


Fig. 9.5 Annual course of predawn xylem water potentials (–MPa) in trees of different populations grown at Yatir (**a**) and Bet Dagan (**b**) during the hydrological year of 1998–1999. (*Dashed arrows* indicate xylem water potential of −1.8 MPa according to Melzack et al. 1985)

greater water-use efficiency (WUE). Significant differences among provenances within sites (Yatir or Bet Dagan) were revealed as well.

Rates of transpiration and photosynthesis under dry site conditions may provide information on drought tolerance (Table 9.6). Using a portable infra-red gas analyser, the gas-exchange rates of Aleppo pine trees of different provenances grown at Yatir and Bet Dagan were measured from Jan to Apr 1999 (Atzmon et al. 2004). As noted earlier, that winter was the driest on record. At both sites, transpiration and photosynthetic rates were strongly affected ($P < 0.0001$) by date of measurement, photon flux density (μmol m^{-2} s^{-1}) and air and needle temperatures (°C). Transpiration rates (mmol H_2O m^{-2} s^{-1}) were measured between 0800 and 10 00 h local time (GMT+2) (Table 9.3). Within sites, among provenances, differences in photosynthetic rates (μmol CO_2 m^{-2} s^{-1}) were nearly significant at Bet Dagan ($P < 0.071$) and highly significant at Yatir ($P < 0.0001$). The differences in photosynvthat WUE (mmol CO_2 mmol H_2O^{-1}) also differed significantly among provenances

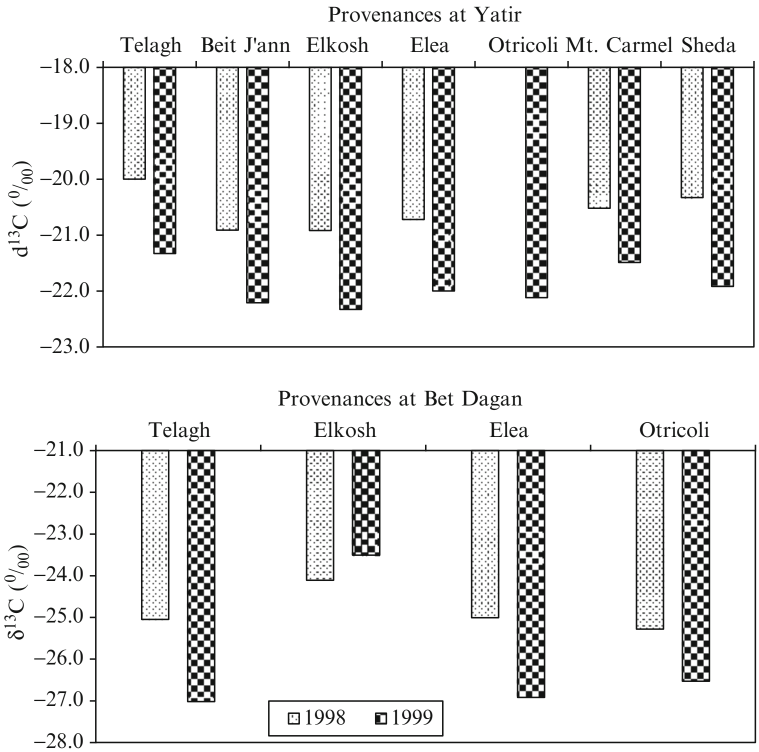


Fig. 9.6 Average Isotopic carbon ratio ($\delta^{13}C$ ($^0/_{00}$) in needles of 12 trees per population of *Pinus halepensis* growing at Yatir and Bet Dagan

Table 9.6 Photosynthesis, transpiration and water-use efficiency of several provenances grown at Yatir and Bet Dagan (within trail plots provenances not followed by the same letter are significantly different at the 0.05 level according to Tukey's HSD). (Significance level: **P<0.01; ***P<0.001)

Provenances at		Transpiration mmol $m^{-2} s^{-1}$	Photosynthesis μmol $m^{-2} s^{-1}$	Water use efficiency
Bet Dagan				
Otricoli	(Italy)	0.610a	3.054a	4.451b
Telagh	(Algeria)	0.619a	3.215a	4.782a
Elea	(Greece)	0.597a	2.931a	4.719a
Mt. Carmel	(Israel)	0.616a	2.808a	4.015b
Prob>*F*		0.348	0.071	0.037
Significance		n.s.	n.s.	**
Yatir				
Telagh	(Algeria)	0.389a	1.220b	2.106b
Elea	(Greece)	0.413a	1.668a	3.077a
Mt. Carmel	(Israel)	0.408a	1.217b	2.240b
Bet J'ann	(Israel)	0.401a	1.401ab	2.770ab
Prob>F		0.658	0.001	0.001
Significance		n.s.	***	***

within each site: $P < 0.037$ at Bet Dagan and $P < 0.001$ at Yatir. Moreover, WUE differed significantly ($P < 0.001$) between sites. At Yatir, the Greek provenance Elea showed the highest rates in all parameters tested, suggesting that this provenance is physiologically the most efficient and therefore most adapted to the arid region. At Bet Dagan, the Algerian provenance Telagh showed the highest rates although it did not differ significantly in WUE from the Greek provenance.

Conclusions

The afforestation project at Yatir started in 1964–1965 was aimed mainly at alleviating the influences of the harsh environmental conditions at the edge of the Negev desert. This forest plantation became a focal point of studies on the influences of harsh ecological conditions on a typical Mediterranean forest tree species planted under semi-arid conditions. In spite of previous eco-physiological research efforts on Aleppo pine (Schiller 2000), the research conducted recently at Yatir has resulted in a better understanding of its genetic and physiological capability to cope with drought (e.g., Schiller and Cohen 1998; Grünzweig et al. 2003; Atzmon et al. 2004; Schiller et al. 2004; Klein et al. 2005; Safriel 2005; Maseyk 2006; Maseyk et al. 2008a, 2008b; Raz Yaseef 2009).

The ecological conditions prevailing at Yatir are challenging in terms of implementing skilful management practice to sustain the present tree generation and future regeneration (Yavlovich 2008). The water-balance partitioning in the forest among its components suggests a precise forest management which aims at alleviating competition, even in more droughty years.

In general, differences in drought adaptation of species, and of provenances within species, probably involve mechanisms which in our case have been largely uninvestigated such as: root/shoot ratios, rooting depth and root properties, xylem properties, osmotic adjustment, hormonal responses (e.g. ABA). Therefore, developing drought-resistant species and provenances will necessitate long-term field trials. However, these differences in drought adaptation afford the opportunity to use the harsh conditions as the driving force for selection among and within the introduced plant material for more heterozygous and drought-resistant plants, in order to enhance the sustainability of newer afforestation projects (Safriel et al. 1994). In our case, it seems that the genetic variation within *Pinus halepensis* is large enough (Tognetti et al. 1997; Schiller et al. 2004) to enable selection, which in turn may significantly improve the selected genetic material's ability to survive under extreme periodical droughts. Hence, the presented results clearly emphasize the need for broad selection programmes among and within the species to be introduced for planting forests under harsh environmental conditions, and the necessity of establishing seed orchards or forest plantations in which those selected genotypes will be kept is becoming clear: its urgency lies in the fact that progress can only be measured in years.

References

Adams HD, Guardiola-Claamonte M, Barron-Gafford G, Villegas JC, Breshears DD, Zou CB, Troch PA, Huxman TE (2009) Temperature sensitivity of drought-induced tree mortality portends increased regional die-of under global-change type drought. PNAS 106:7063–7066

Allen RG, Pereira LS, Raes D, Smith ML (1998) Crop evapotranspiration – guidelines for computing crop water requirements. FAO, Rome

Allen CD, Macalady AK, Chenchouni H, Bachelet D, McDowell N, Vennetier M, Kitzberger Th, Rigling A, Breshears DD, Hogg EH, Gonzalez P, Fensham R, Zhang Z, Castro J, Demidova N, Lim JH, Allard G, Running SW, Semerci A, Cobb N (2009) A global overview of drought and heat-induced tree mortality reveals emerging climate change risks for forests. For Ecol Manage. doi:10.1016/j.foreco.2009.09.0001

Aronson S, Lovinger R (1976) Yatir forest. In memory of Joseph Weitz. The Jewish National Fund, Land Development Authority, the Forest Department, Israel. (Hebrew with English summary)

Atzmon N, Moshe Y, Schiller G (2004) Eco-physiological response to severe drought in *Pinus halepensis* Mill. trees of two provenances. Plant Ecol 171:15–22

Bariteau M, Huc R, Vendramin GG (2000) Adaptation and selection of Mediterranean *Pinus* and *Cedrus* for sustainable afforestation of marginal lands (MPC). Final Report to the Commission of the European Communities, Directorate General for Agriculture, DGVI FII3, Contract no FAIR CT95-0097, 181 pp

Barnes J, Gimeno B, Davison A, Dizengremel P, Gerant D, Bussotti F, Velissariou D (2000) Air pollution impacts on Pine forests in the Mediterranean Basin. In: Ne'eman G, Trabaud L (eds) Biography and management of *Pinus halepensis* and *P. brutia* forest ecosystems in the Mediterranean Basin. Backhuys, Leiden, pp 391–404

Breshears DD, Myers OB, Barnes FJ, Zou CB, Allen CD, McDowell NG, Pockman W (2009) Tree die-off in response to global change-type drought: mortality insights from a decade of plant water potential measurements. Front Ecol Environ. doi:10.1890/080016

Cohen Y (1994) Thermoelectric methods for measurement of sap flow in plants. In: Stenhill G (ed) Advances in bioclimatology, vol 3. Springer, Heidelberg, Germany, pp 63–88

Dan J, Raz Z (1970) The soil association map of Israel. Ministry of Agriculture, Department of scientific publications, 145 pp and two maps 1: 250,000 (In Hebrew)

Danin A (1970) Phytosoziological-ecological monograph of the northern Negev, Israel. Ph.D. thesis, the Hebrew University of Jerusalem, Israel (in Hebrew with English summary)

Grünwald C, Schiller G, Conkle MT (1986) Isoenzyme variation among native stands and plantations of Aleppo pine in Israel. Israel J Bot 35:161–174

Grünzweig JM, Lin T, Rotenberg E, Schwartz A, Yakir D (2003) Carbon sequestration in arid-land forest. Global Change Biol 9:791–799

Inclán R, Gimeno BS, Dizengremel P, Sanchez M (2005) Compensation processes of Aleppo pine (*Pinus halepensis* mill.) to ozone exposure and drought stress. Environ Pollut 137:517–524

IPCC (Intergovernmental Panel on Climate Change) (2007) Climate Change 2007: The Physical Science Basis; Summary for Policy Makers, 21 pp. http://www.ipcc.ch

Kafle HK, Brunis HJ (2009) Climatic trends in Israel 1970-2002: warmer and increasing aridity inland. Climate Change 96:63–77

Klein T, Hemming D, Lin T, Grünzweig JM, Maseyk K, Rotenberg E, Yakir D (2005) Association between tree-ring and needle $\delta^{13}C$ and leaf gas exchange in *Pinus halepensis* under semi-arid conditions. Oecologia 144:45–54

Korol L, Shklar G, Schiller G (2002) Diversity among circum-Mediterranean populations of Aleppo pine and differentiation from Brutia pine in their isoenzymes: Additional results. Silvae Genet 51:35–41

Leshem B (1967) Adaptation of pine roots to an arid environment. Ph.D. thesis, the Hebrew University of Jerusalem, Israel (In Hebrew), 87 pp

Lewontin RC (1972) The apportionment of human diversity Evol Biol 6:381–398

Martinez-Ferri E, Balaguer L, Valladares F, Chico JM, Manrique E (2000) Energy dissipation in drought-avoiding and drought-tolerant tree species at midday during Mediterranean summer. Tree Physiol 20:131–138

Maseyk, S K (2006) Eco-physiological and phonological aspects of *Pinus halepensis* in an arid-Mediterranean environment. Ph.D. thesis, the Scientific Council of the Weizmann Institute of Science, Rehovot, Israel, 142 pp

Maseyk KS, Lin T, Rotenberg E, Grunzweig JM, Schwarz A, Yakir D (2008a) Physiology-phenology interaction in productive semi-arid pine forest. New Phytol. doi:10:1111/j.1469-8137.2008.02391.x

Maseyk KS, Grunzweig JM, Rotenberg E, Yakir D (2008b) Respiration acclimation contribution to high carbon-use efficiency in seasonally dry pine forest. Global Change Biol 14:1553–1567

McDowell N, Pockman WT, Allen CD, Breshears DD, Cobb N, Kolb Th, Plaut J, Sperry J, West A, Williams DG, Yepez EA (2008) Mechanisms of plant survival and mortality during drought: why do some plants survive while others succumb to drought? New Phytol. doi:10.1111/j.1469-8137.2008.02436.x

Melzack RN, Bravdo B, Riov J (1985) The effect of water stress on photosynthesis and related parameters in *Pinus halepensis* Mill. Physiol Plant 64:295–300

Merzer T (2007) The effects of different vegetative cover on the local hydrological balance of a semi-arid afforestation. M. Sc. thesis, Ben-Gurion University of the Negev, Israel

Monteith JL (1965) Evaporation and environment. In: Fogg GE (ed) The state and movement of water in living organisms. Cambridge University Press, Cambridge, pp 205–234

Morandini R (1976) Mediterranean conifers. In: Forest genetic resources, Information No 5. Forestry Occasional Paper 1976/1. FAO, Rome, Italy, pp 12–18

Nei M (1978) Estimation of average heterozygosity and genetic distance from small number of individuals. Genetics 89:583–590

Oppenheimer HR (1955) Pénétration active des raciness de buissons méditerranéens dans les calcaires. Bull Res Counc 5D:219–222

Oppenheimer HR (1957) Further observations on root penetration into rock and their structure. Bull Res Counc 6D:18–26

Oppenheimer HR (1967) Mechanisms of drought resistance in conifers of the Mediterranean zone and the arid west of the USA. Part I: Physiological and anatomical investigations. Final Report on project No. A10-FS 7, Grant No. FG-Is-119. The Hebrew University of Jerusalem, Rehovot, Israel

Paz S (2004) Desertification in the Eastern Mediterranean Basin: possible climatic explanation. In: Kliot N (ed) Studies in natural resources and environmental management 2(2): 7–17 (In Hebrew with English summary)

Raz Yaseef N (2008) Partitioning the evapotranspiration flux of the Yatir semi-arid forest. Ph.D. thesis, the Scientific Council of the Weizmann Institute of Science, Rehovot, Israel, 124 pp

Raz Yaseef N (2009) Ecohydrology of a semi-arid forest: partitioning among water balance components and its implications on predicted precipitation changes. Ecohydrology. doi:10.1002/eco.65

Roehle H (1991) Yield tables for Aleppo pine (Pinus halepensis) in Israel. Chair of Forest Yield Sciences, University of Munich, Federal Republic of Germany

Safriel NU, Volis S, Kark S (1994) Core and peripheral populations and global climate change. Israel J Plant Sci 42:331–345

Safriel N U (2005) Monitoring and Evaluation of Watersheds in the Middle-East. Final Scientific Report-Israel 2001–2005, Submitted to the USDA Forest Service, 191 pp

Schiller G (1972) Decisive ecological factors affecting growth of Aleppo pine in the southern Judean hills. Leafl For Div Agric Res Organ, Ilanot No 44

Schiller G (1982) Significance of bedrock as a site factor for Aleppo pine. For Ecol Manage 4:213–223

Schiller G (2000) Ecophysiology of *Pinus halepensis* Mill. and *P. brutia* Ten. In: Ne'eman G, Trabaud L (eds) Biography and management of Pinus halepensis and P. brutia forest ecosystems in the Mediterranean Basin. Backhuys, Leiden, pp 51–65

Schiller G, Cohen Y (1995) Water regime of a pine forest under a Mediterranean climate. Agric For Meteorol 74:181–193

Schiller G, Cohen Y (1998) Water balance of a *Pinus halepensis* Mill. afforestations in an arid region. For Ecol Manage 105:121–128

Schiller G, Conkle MT, Grunwald C (1986) Local differentiation among Mediterranean populations of Aleppo pine in their isozymes. Silvae Genet 35:11–19

Schiller G, Korol L, Atzmon N (2001) In: Final Report of the FORADAPT Project on: Global, physiological and molecular responses to climatic stresses of three Mediterranean conifers (pines of the halepensis-brutia section, maritime pine and Mediterranean cedars. The INCO-DC Program, Contract no. ERBIC-18CT-970-200

Schiller G, Korol L, Shklar G (2004) Habitat effects on adaptive genetic variation in *Pinus halepensis* Mill. provenances. Forest Genetics 11:325–335

Schiller G, Waisel Y (1989) Among-provenance variation in *Pinus halepensis* Mill. in Israel. For Ecol Manage 28:141–151

Schiller G, Atzmon N (2009) Performance of Aleppo pine (*Pinus halepensis*) provenances grown at the edge of the Negev desert: a review. J Arid Environ 73:1051–1057

Scholander PF, Hammel HT, Bradstreet ED, Hemminsen EA (1965) Sap pressure in vascular plants. Science 149:339–346

Shachnovich Y, Berliner PB, Bar P (2008) Rainfall interception and spatial distribution of throughfall in a pine forest planted in an arid zone. J Hydrol 349:168–177

Swanson RH, Whitfield DWA (1981) A numerical analysis of heat pulse velocity theory and practice. J Exp Bot 32:321–339

Tognetti R, Michelozzi M, Giovannelli A (1997) Geographic variation in water relations, hydraulic architecture and terpene composition of Aleppo pine seedlings from Italian provenances. Tree Physiol 17:241–250

UNESCO-FAO (1963) Bioclimatic map of the Mediterranean Zone, Arid Zone Research, Ecological investigations in the Mediterranean zone. FAO, Rome

Van Mantgem PJ, Stephenson NL (2007) Apparent climatically induced increase of tree mortality rates in temperate forests. Ecol Lett 10:909–916

Yavlovich H (2008) Natural regeneration of *Pinus halepensis* forest in semi-arid regions. M.Sc thesis, The Hebrew University of Jerusalem, Israel

Zohary M (1962) Plant life of Palestine, Israel and Jordan. The Ronald Press Co, New York

Chapter 10
How Mediterranean Deciduous Trees Cope with Long Summer Drought? The Case of *Quercus pyrenaica* Forests in Western Spain

Gerardo Moreno, Juan F. Gallardo, and María Ángeles Vicente

Introduction

Most of the climate change models predict an increase of aridity of the Mediterranean basin and that drought episodes could be more frequent in the present century due to climate change (IPCC 2007). Similar changes are also expected in temperate regions in the Northern Europe (Bréda et al. 2006) and South-Eastern Europe (Chapter 8, this volume), with direct consequences on forest health. A massive dieback of oaks forest (both evergreen and deciduous) has been observed some summers in last decades (Peñuelas et al. 2001; Corcuera et al. 2004), and several authors have shown how some mesic Meditterranean trees could be replaced for other more xeric species under the current scenario of climate change in Mediterranean forests (Martínez-Vilalta et al. 2002; Peñuelas and Boada 2003).

The functional characteristics of Mediterranean deciduous trees provide an interesting model for investigating adaptative mechanisms to drought, useful to understand future changes of northern forests in a scenario of climate change (Baldocchi and Liukang 2007). The genus *Quercus* comprises 531 tree and shrub species distributed among contrasting phytoclimates in the Northern Hemisphere, from temperate and subtropical deciduous forests to dry Mediterranean evergreen woodlands (Corcuera et al. 2002). The range of variation in the ecophysiology of Mediterranean oaks is comparable to that observed between other deciduous and evergreen species in the Mediterranean basin (Duhme and Hinckley 1992), in the Californian chaparral (Blake-Jacobson ME 1987) and in Tropical dry forests Sobrado (1986). Corcuera et al. (2002) identified three

---

G. Moreno (✉)
Forestry School, University of Extremadura, Plasencia 10600, Spain
e-mail: gmoreno@unex.es

J.F. Gallardo and M.Á. Vicente
CSIC, I.R.N.A., Salamanca 37071, Spain

M. Bredemeier et al. (eds.), *Forest Management and the Water Cycle: An Ecosystem-Based Approach*, Ecological Studies 212, DOI 10.1007/978-90-481-9834-4_10,

phytoclimatic groups of *Quercus*, which were characterized by their contrasting ecophysiological response to water stress. They found that Mediterranean deciduous oaks perform better than temperate deciduous oaks under water-stress conditions, the last having difficulties to thrive in dry soils. Valladares et al. (2004) predicted that Mediterranean deciduous oaks will be replaced by evergreens, and the former species will extend in areas currently occupied by temperate deciduous oaks.

The main characteristics defining Mediterranean ecosystems generally are the scarcity and irregularity of rainfall (P) and potential evapotranspiration (E_P) values, the latter usually higher than the former in an annual basis. Mountainous areas, where Mediterranean deciduous oaks are mostly located, exhibit similar or higher P than E_P values, but a much higher evaporative demand than rainfall during summer, in addition to long and cold winters. These climate conditions determine that deciduous oaks have a short growing season (around 120 days year^{-1}), mostly coincident with the period without rain. Deciduous leaves can be as costly to produce as evergreen leaves (Merino et al. 1982) and in a relatively short time, should maintain a high photosynthetic activity throughout summer to obtain a positive annual C balance. This pattern clearly contrasts with that of Mediterranean evergreen species, which are photosynthetically active all year. Evergreen oaks gain significant amounts of C in winter period, and water stress usually cause stomatal closure early in the growing season (Rodá et al. 1999).

Under these circumstances, Mediterranean deciduous trees surely exhibit a high dependence on soil water reserves (SWR) accumulated on soil along wet season. This reserve however could be limited in many sites because of the frequent shallowness of Mediterranean soils. The investment on very deep root systems is a morphological traits frequently associated to Mediterranean evergreen species (Canadell et al. 1996; Moreno et al. 2005). However this trait was scarcely studied in Mediterranean deciduous trees yet. We hypothesised that Mediterranean deciduous oaks have also very deep root systems and use bedrock water reserve; this powerful root system entails a resource use allowing an increment in the aboveground production.

Apart of the use of SWR, Mediterranean deciduous forest could exhibit a set of anatomical and physiological traits compatible with long summer drought. In this article we also analyse the diurnal and seasonal dynamic of stomata conductance on one hand and, on the other hand, a set of potential of morphological/physiological traits that could contribute to increasing the water use efficiency (WUE). Stomata regulation is considered the most immediate mechanism of the optimisation of C-gain/water-loss in plants (Rambal 1993) and we hypothesised that deciduous trees exhibit a non-conservative pattern of water transpiration (stomata activity), contrary to their neighbour or co-existing evergreen trees (Damesin et al. 1998). The reduction of leaf area index (LAI) is a third mechanism associated to optimisation of WUE and, presumably, Mediterranean deciduous oaks have low LAI compared to congeneric species, either temperate deciduous or Mediterranean evergreen oaks.

Given the wealth of information that exists on water dynamics of temperate coniferous and deciduous broadleaves (e.g., Bréda et al. 2006), and Mediterranean evergreen trees and forests (e.g., Rodá et al. 1999; Ciais et al. 2003), corresponding knowledge is scarce about Mediterranean deciduous species. The understanding of these southern deciduous forests is needed for modelling water budgets and water yield, planning forest conservation and/or restoration programmes, and understanding the future dynamic of both Mediterranean and temperate forests under current scenarios of climate change.

In this article we analyse how a Mediterranean deciduous oak, *Quercus pyrenaica*, with a short vegetative period coincident with summer drought, cope with water deficit in that period. We revised published data on temporal dynamic of soil moisture and physiological status of tree leaves of several forest stands of Central–Western Spain and discuss the significance of soil water reserve (SWR) and deep rooting system on the maintenance of tree transpiration and physiological activity of the trees along summer drought. Accordingly, the following three questions are addressed: (a) Are trees water-limited during summer drought? (b) What is the tree water source during summer drought? and (c) Which adaptative mechanisms exhibit this species to face water deficit?

Study Species

Quercus pyrenaica is a slow-growing Mediterranean deciduous species, distributed throughout mountainous sub-humid Mediterranean areas of the southwestern region of Europe, especially in the western regions of the Iberian Peninsula. It has a short growing season (from the end of May till mid October), which may determine its distribution. The species is well adapted to survive maintaining photosynthetic production in spite of the long summer water deficit (Gallego et al. 1994; Rico et al. 1996) and, therefore, it often occupies transitional areas from sub-humid to semi-arid conditions (Hernández-Santana et al. 2008a). Despite its distribution and interesting ecology, *Q. pyrenaica* has been poorly studied in comparison with other Mediterranean *Quercus* species (Silla and Escudero 2006), probably because of its low aboveground productivity (Gallardo 2000).

A common strategy to thrive on water-limited conditions is the reduction of the transpiration area (Rambal 1993); then, leaf area index (LAI) usually decreases with water deficit (Grier and Running 1977; Ogaya and Peñuelas 2007). Hernández-Santana et al. (2008a) have reported an average value of 2.4 $m^2\ m^{-2}$ for *Q. pyrenaica* stands located in Central–Western Spain. This values can be qualified as low or very low compared with LAI values reported for other congeneric species: 5.8 for *Q. petraea* (Davi et al. 2009), 5.0 for *Q. robur* (Chapter 4, this volume), 3.4 for *Q. cerris* (Llorens and Domingo 2007), 3.3 and 2.9 $m^2\ m^{-2}$ for *Q. pubescens* (Čermák et al. 1998; Damesin et al. 1998, respectively).

In a comparative study of 17 *Quercus* species (Corcuera et al. 2004), *Q. pyrenaica* shows values for some water-related parameters closer to Mediterranean evergreen oaks than those of deciduous ones. For instance, *Q. pyrenaica* showed very low leaf water potential (Ψ_l) at the turgor-loss point ($\Psi_{tlp} < -3.2$ MPa), similar to other Mediterranean deciduous oaks (*Q. cerris* and *Q. frainetto*) and clearly lower than temperate deciduous ones (around −2.4 MPa). Osmoregulation is a strategy for drought tolerance, and *Q. pyrenaica* showed a low osmotic potential (π) at full turgor ($\pi_o < -2.6$ MPa), again similar to other Mediterranean deciduous oaks and clearly lower than temperate deciduous ones (around −2 MPa). Finally, *Q. pyrenaica* showed a high maximum bulk modulus of elasticity ($\varepsilon_{max} \approx 18$ MPa), the highest among the deciduous oaks. More inelastic tissues will result in a more rapid decrease of turgor with loss of water; this will generate a steeper soil to leaf water potential gradient and, consequently, less leaf dehydration. This may improve water uptake from dry soils, avoiding severe water deficit in leaf tissues, what is important for species with deep rooting as oaks. Their capacity of exploiting deep soil layers improves soil–water extraction, maintaining the soil to leaf water potential gradient without leaves becoming too much dehydrated (Aranda et al. 2004).

Study Area

Data used in this article are based on a long-term monitoring carried out in four *Q. pyrenaica* stands located in the *Sierra de Gata* (40° 2′ 40″ N, 3° 0′ 50″ W, CW of Spain). Four stands are located in a smoothly hilly area along a rainfall gradient: 720, 872, 1,245 and 1,580 mm a^{-1} for S_{dry}, $S_{m\text{-}dry}$, $S_{m\text{-}wet}$ and S_{wet}, respectively. The climate is sub-humid Mediterranean according to the Emberger's climogram, with relatively high amount of annual rainfall and moderate temperatures (annual mean about 12°C). Soils are acid, predominantly *Cambisols* with a typical *A*-humic, *B*-weathering, and C-horizons (AhBwC profile) developed over granites, gneiss, schists, and greywackes.

From litterfall data (Gallardo et al. 1998) we have estimated LAI values ranging from 1.8 and 2.6 $m^2\ m^{-2}$ in the four *Q. pyrenaica* stands. These values are similar to those reported for other *Q. pyrenaica* stands (Hernández-Santana et al. 2008a).

Other characteristics of the four stands are given in Table 10.1.

Temporal Dynamic of Soil Moisture. Estimation of Stand Evapotranspiration

Soil water content (SWC), in terms of both extreme and mean values, increased gradually with soil depth throughout growing season (June–September). The minimum SWC was found at the surface, as expected, owing to a more marked drying process

Table 10.1 Characteristics of the four experimental *Q. pyrenaica* plots ('*Sierra de Gata*' Moutains, Spain)

		Fuenteguinaldo	Villasrubias	El Payo	Navasfrías
Characteristics	Units	(S_{dry})	($S_{m\text{-}dry}$)	($S_{m\text{-}wet}$)	(S_{wet})
Geology (Paleozoic)		Granite	Schist	Granite	Schist
Altitude	m a.s.l.	870	900	940	960
Slope	%	2–5%	10–15%	5%	5–15%
Mean annual temperature	°C	13.3	Not available	Not available	11.4
Mean annual precipitation	mm	720	872	1,245	1,580
Soil units (ISSS-ISRIC-FAO 1998)		Humic Cambisol	Humic Cambisol	Humic Cambisol	Humic Cambisol
Soil depth	cm	–85	–55	–125	–80
Soil texture		Sandy-loam	Silt-loamy	Loamy	Silt-loamy
AWC[a] soil	$cm^3\ cm^{-3}$	0.16	0.21	0.13	0.18
AWC[a] bedrock	$cm^3\ cm^{-3}$	0.08	0.11	0.05	0.11
Dominant understory		Shrubs	Shrubs	Grasses	Ferns
Tree density	Tree ha^{-1}	738	1,043	406	820
LAI[b]	$m^2\ m^{-2}$	2.6	2.0	1.9	1.8
Tree DBH[c]	mm	165	110	254	152
Tree height	m	12.0	8.5	17.0	13.0
Tree age[d]	Years	80	60	80	60
Basal area	$m^2\ ha^{-1}$	21.2	13.5	19.9	15.6
Last thinning	Years ago	35	25	50	35
DBH increment	mm $year^{-1}$	3.2	2.5	0.8	2.4
Biomass[e]	Mg ha^{-1}	98	63.8	131	64.5
Production[e]	Mg ha^{-1} $year^{-1}$	4.19	2.85	3.52	2.6

[a] Available water content
[b] Leaf Area index
[c] Diameter at breast height
[d] Dominant tree height
[e] Aboveground tree biomass

at the soil surface (Fig. 10.1). Increases in SWC at depth, with a certain delay over the more superficial horizons, were noticeable from the beginning autumn, i.e., the soil wetting front advanced very rapidly (data not shown). The pattern of soil profile dryness, similar throughout years and plots, showed a gradual change of the relative importance of the water at different depths (Fig. 10.1). Comparing SWC values of consecutive dates, it observed that at the beginning of the growing season, soil dry mostly in the uppermost soil layers, while dryness is a slow process at the deep layers. Over time, the deep soil water acquires higher relative importance; that is, SWC values decreases mostly at deep layers, indicating that only deep soil water is being consumed by plants.

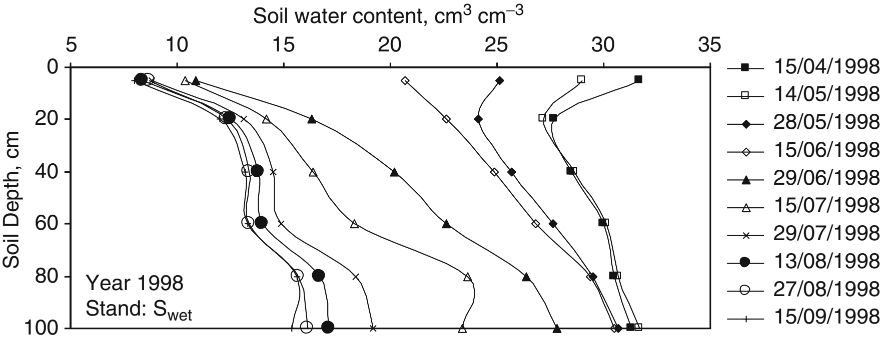


Fig. 10.1 Temporal evolution of soil drying in *Quercus pyrenaica* forests of Central–Western Spain. More examples, including soil re-watering period, in Moreno et al. (1996), Vicente et al. (2003)

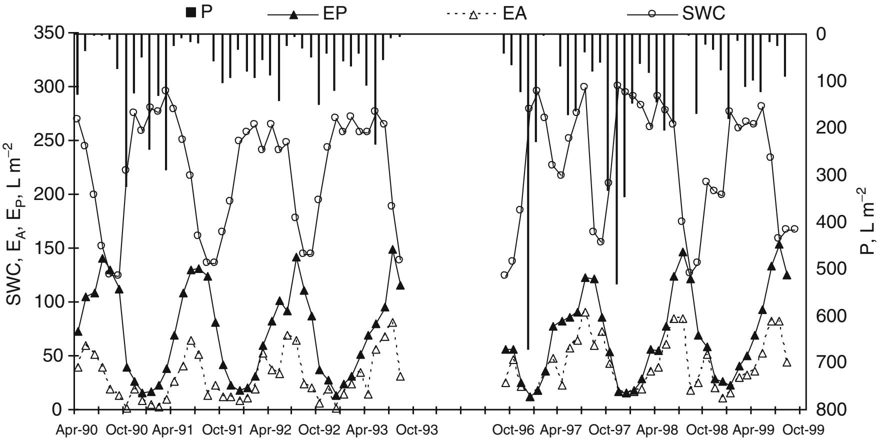


Fig. 10.2 Temporal course of soil water content (0–100 cm depth; SWC), precipitation (P), potential evapotranspiration (E_P) and actual evapotranspiration (E_A) in a *Quercus pyrenaica* forests of Central–Western Spain (Adapted from Moreno et al. (1996) and Vicente et al. (2003)

The seasonal variation of SWC within the first 100 cm depth closely mirrored the precipitation pattern, with alternating wet and dry periods occurring in all stands and years analysed (Fig. 10.2). Autumn recharge entailed a rapid soil moistening, obtaining frequently maximum values of SWC at the end of autumn. Although in dry autumn, complete soil recharge was reached later, it was reached every year. Then, SWC content remained more or less constant till mid spring, when soil begun to dry quickly. Some years, when trees leafed out (end May–early June) more than 50% of the available water content (AWC) had been consumed and, every year, AWC practically was exhausted halfway through the summer. Then the soil remained similarly dry during 30–50 days, with the exception of some abnormally wet summer. The situation of water deficit occurred earlier and lasted longer at the dry sites, but minimum SWC values are similar for each plot every year. On average, SWC declined from the bud burst to the end of the summer by 115 mm in S_{wet}, 111 mm in $S_{m\text{-}wet}$, 113 mm in $S_{m\text{-}dry}$, and 79 mm in S_{dry}, differences being more related with soil depth and texture, than rainfall amounts (Table 10.1).

From the temporal evolution of SWC, monthly actual evapotranspiration (E_A) of the four stands was calculated by applying a simple iterative model of monthly water balance (Moreno et al. 1996). The model is based on the relationship between deep drainage (water flowing down from the soil to the bedrock; Dp) and SWC, and on the limitation imposed by E_P (potential evapotranspiration). These authors found that P was significantly related to Dp but not to E_A. The higher P values in wet and cold season were, the higher Dp values were, without involving a significant change

Table 10.2 Mean values of precipitation (P), actual evapotranspiration (E_A), and ratios of actual and potential evapotranspiration (E_A/E_P) in four *Q. pyrenaica* stands located along a rainfall gradient (wet, moderately wet, moderately dry and dry sites). Data refer consecutive growing seasons (May–September)

		Experimental sites			
Growing season	Water flow	S_{wet}	$S_{m\text{-}wet}$	$S_{m\text{-}dry}$	S_{dry}
1990	P	207	184	179	137
	E_A	180	202	193	159
	E_A/E_P	0.29	0.32	0.31	0.25
1991	P	114	104	107	82
	E_A	205	178	213	154
	E_A/E_P	0.34	0.29	0.34	0.24
1992	P	347	272	244	173
	E_A	216	209	208	167
	E_A/E_P	0.37	0.35	0.35	0.27
1993	P	562	422	407	342
	E_A	261	256	278	239
	E_A/E_P	0.48	0.46	0.49	0.42
Average	P	308	246	234	183
	E_A	216	211	223	180
	E_A/E_P	0.37	0.36	0.37	0.30

in the water availability for plants. Rainfall interception was moderate and surface runoff negligible (15 and <0.5% of the rainfall, respectively). The limited SWR determined moderate to low E_A rates in the growing season (Table 10.2) and with scarce differences among plots. E_A differs significantly only among S_{dry} and the rest of stands, because of the lower P and SWC of the former site. Maximum E_A value was usually found in June, and August usually showed the lowest mean values. The E_A values (mm day^{-1}) for August ranged from 0.40 to 0.97 for S_{wet}, 0.74 to 0.87 for $S_{m\text{-}wet}$, 0.74 to 0.90 for $S_{m\text{-}dry}$, and 0.37 to 0.80 for S_{dry}. According to the low E_A values, it should seem that *Q. pyrenaica* trees experienced a strong water deficit, with mean E_A/E_P ratios ranging, on average, from 0.30 to 0.37 in the growing season (0.14–0.28 in August).

Daily and Seasonal Evolution of Stomata Activity. Estimating Tree Transpiration

Studies carried out in two extremes of the rainfall gradient (S_{wet} and S_{dry}) showed that trees maintained very favourable water conditions throughout summer, with predawn leaf water potential ($\Psi_{l\text{-}pd}$) very high at two sites in two consecutive summers (Fig. 10.3). Only 1 day of the 18 monitored, the $\Psi_{l\text{-}pd}$ was < -1 MPa (Gallego et al. 1994). Similarly, Hernández-Santana et al. (2008a) found $\Psi_{l\text{-}pd} < -1$ MPa only 1 day from a total of 24 days monitored (2004–2006) in different *Q. pyrenaica* stands, while evergreen-oak forests frequently reach $\Psi_{l\text{-}pd}$ values around −4 MPa (Savé et al. 1999).

$\Psi_{l\text{-}pd}$ correlated better with the seasonal dynamics of SWC of the deepest layers (60–100 cm soil depth; Gallego et al. 1994). Differences between wet and dry sites were only found at the end of the driest summer. Leaf water potential (Ψ_l) decreased quickly during the day with the minimum value early in the afternoon, but it remained the whole time above the turgor-loss point (−3.2 MPa; Corcuera et al. 2002). In the afternoon, Ψ_l increased also quickly, and at the sunset values similar to $\Psi_{l\text{-}pd}$ has been reached (Fig. 10.3).

Despite the rapid decrease of Ψ_l, stomatal conductance (g_s) was high in all monitored days (Fig. 10.4). Maximum mean values of gs (around 250 mmol m^{-2} s^{-1}) and mean daily values varied very little along summer (Fig. 10.4), in spite of the fact that SWR was depleted. The diurnal behaviour of stomatal activity (g_s) indicated the absence of limiting factors most of the summers; g_s increased in the morning and reached a maximum about 3 h before the daily minimum in Ψ_l occurred. Maximum g_s values were reached at wet site (S_{wet}), but daily curves showed a more quick and acute decreased of g_s along day at S_{wet}, defining a stronger effect of water deficit at S_{wet} than at S_{dry}.

Stomata conductance behaviour with respect to atmospheric humidity may be linear or nonlinear, depending on the type of control mechanism. Rico et al. (1996) showed for S_{wet} and S_{dry} stands that stomata conductance of *Q pyrenaica* remained constant at low and moderate vapour pressure deficit (VPD) and strongly decreased

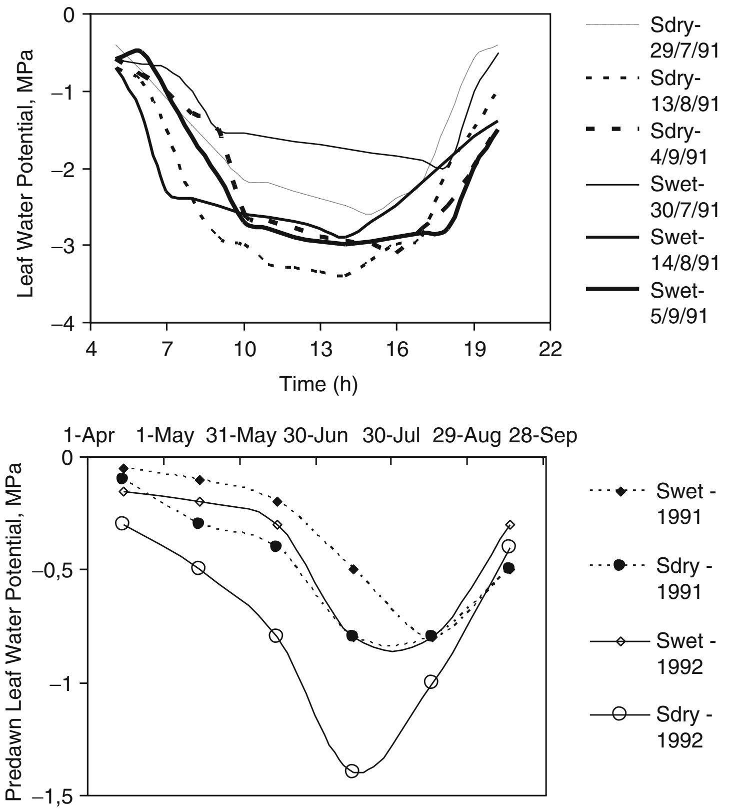


Fig. 10.3 Daily curve of leaf water potential (Ψ_l) in some selected days of summer 1991 (*top*) and seasonal evolution of predawn leaf water potential ($\Psi_{l\text{-pd}}$) along summers 1991 and 1992 (*bottom*) at two *Quercus pyrenaica* stands (dry and wet; Adapted from Gallego et al. 1994)

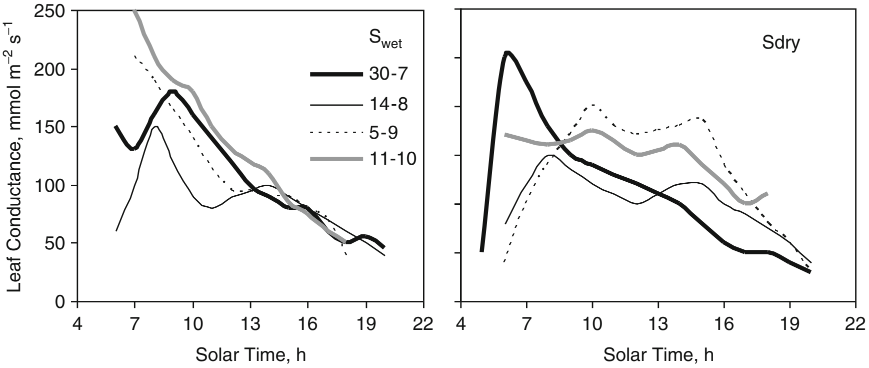


Fig. 10.4 Daily curves of stomatal conductance (g_s) at at two *Quercus pyrenaica* stands (dry and wet) along summer 1991 (Adapted from Gallego et al. 1994)

after a VPD threshold (2.4 kPa). This response is interpreted in terms of high sensitivity to drought and weak functional adaptative strategy of trees, similarly to other deciduous *Quercus* (Chambers et al. 1985). This decrease of g_s with VPD was more attenuated, but began earlier (lower VPD), at the drier site. Here g_s showed a more linear tendency, typical of species adapted to aridity, with a conservative strategy. g_s showed also little response to Ψ_l, remaining high in certain range of Ψ_l. After a threshold (Ψ_l around −2 MPa) a rapid decrease in g_s occurred as potential continued to decline. Winkel and Rambal (1990) detected similar response in other deciduous oaks. Differences between dry and wet sites were again detected, although less acute than for g_s response to VPD. A better response of g_s to Ψ_l was detected in dry site, with a high threshold and a less pronounced trend than in wet site (Rico et al. 1996).

From seasonal and diurnal curves of g_s values, VPD and LAI data, E_A was estimated for S_{dry} and S_{wet} stands. Estimation also involved the calculation of the decoupling coefficient (Ω) of Jarvis and McNaughton (1986), which is a measure of the coupling between conditions at the canopy surface and in the free air stream. For the study stands and days, we have estimated Ω values moderately high (0.34 on average), as a result of large stomata and low wind velocities. This value indicated a rather low stomata control of transpiration (which grows progressively weaker as Ω approaches 1.0) if compared with other broadleaf forests (Jarvis and McNaughton 1986; Goldberg and Bernhofer 2008).

Values of E_A, so estimated, do not show a clear seasonal trend or clear differences between sites; values ranged 2–4 mm day^{-1} (Fig. 10.5). Recently, Hernández-Santana et al. (2008b) also reported values of daily E_A above 2 mm for most of the summer days in similar *Q. pyrenaica* forest stands. E_A values here estimated are threefold higher than those estimated from soil water balance. Although both approaches of E_A estimation have a certain level of uncertainty, the high differences must be, a least partially, explained because of the existence of an additional source of water for tree transpiration that is not accounted in the soil water balance.

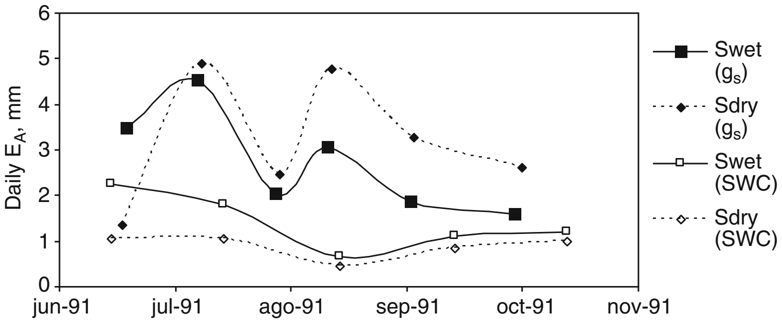


Fig. 10.5 Mean values of daily actual evapotranspiration (E_A) in two (dry and wet sites) estimated from stomata conductance (g_s) and from time course of soil water content (SWC)

Lessons Learn

Quercus pyrenaica Oak Is Only Slightly Water-Limited During Summer Drought

In response to progressive decreases in soil water, no substantial changes were found in Ψ_l or in g_s, with a tendency to use up water reserves from progressively deeper soil layers, thereby avoiding marked stomata closure. *Q. pyrenaica* seems well adapted to surviving summer drought period while maintaining photosynthetic activity and it seems to have a non-conservative strategy of water consumption (sense Jones 1992), according to patterns described by other authors for different deciduous oaks (Mediavilla and Escudero 2003; Manes et al. 2006). Ψ_l and g_s of *Q. pyrenaica* remained high when SWR had been completely depleted. Tree transpiration (E_A) only decreased slightly during summer in dry and wet stands of '*Sierra de Gata*' mountains and other stands of Central–Western Spain (Hernández-Santana et al. 2008b). However, stand E_A estimated from the soil water balance gave values much lower than those estimated from the leaf conductance, in spite of the fact that the latter only included tree transpiration and the former included soil evaporation (presumably negligible) and understorey (shrubs and grasses) transpiration, besides the tree transpiration. Irrespective of possible imprecision on E_A estimates, differences among both approaches allow speculating that trees take up water from an extra water reserve.

Quercus pyrenaica Oaks Depend on Thick Porous-Weathered Bedrocks

The ability to take up water stored in the upper few meters of the weathered bedrock during the pronounced dry season is likely the key feature allowing *Q. pyrenaica* to thrive under dry summer conditions in the shallow, rocky soils of study sites, as other authors have shown for other Mediterranean-type ecosystems (Sternberg et al. 1996; Witty et al. 2003; Schiller et al. 2007). Although weathered bedrock has much lower available-water capacities in the four *Q. pyrenaica* stands than those of soils (Table 10.1), the weathered rock zone is much thicker (several meters) than the soil (< -1 m). From our data, we estimated that a hydraulically active bedrock of around −2.2 and −4.8 m depth (on schist and granite soils, respectively, derived of old weathering during the last interglacial periods; Gallardo and Molina 1979) can provide the water needed by *Q. pyrenaica* trees during summer in the study area. Witty et al. (2003) reported that weathered bedrock supplied at least 70% of the water used by the *Pinus jeffreyi* during the growing season in Sierra Nevada mountains, USA.

The scarce information available on the *Quercus pyrenaica* rooting system indicates a rapid decrease of root length density with depth (Schneider et al. 2001) and also

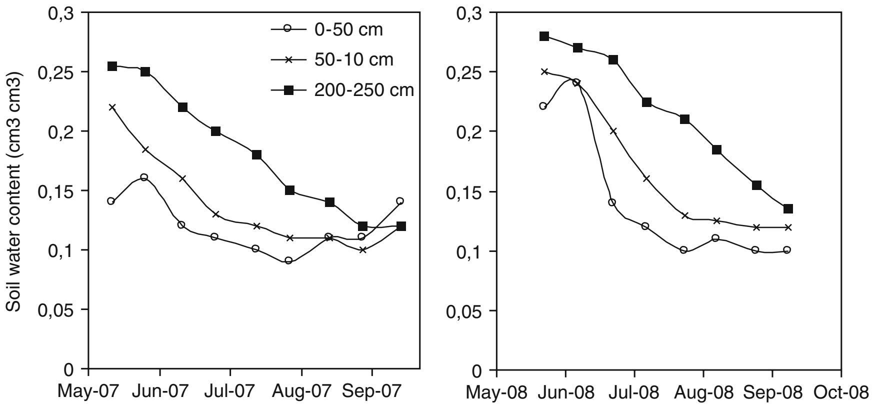


Fig. 10.6 Variation in SWC at three depths from June to September 2006 and 2007 in a *Q. pyrenaica* stands in Central–Western Spain (Adapted from Hernández-Santana et al. 2008b)

that most of the root is located in the first −50 cm of the soil (Hernández-Santana et al. 2008a). However, deep rooting has been suggested to be a key trait to overcome water depletion in the upper soil layers during the dry season in Mediterranean ecosystems (Rambal 1984; David et al. 2004; Kurz-Besson et al. 2006; Querejeta et al. 2007). Indeed, Cubera and Moreno (2007) and Hernández-Santana et al. (2008b) have shown that *Q. ilex* and *Q. pyrenaica* stands of Central–Western Spain use soil water located deeper than −2.5 m depth (Fig. 10.6).

A complete re-moistening of thick bedrocks in wet season needs of high rates of water infiltration and non-torrential rainfall regimes. However, rainfall regime seems to be changing and less frequent and more torrential rainfall events are predicted for the near future in Mediterranean Basin (IPCC 2007). Through a decadal analysis of the rainfall over the last 55 years, Hernández-Santana et al. (2008a) have shown a decrease of mean annual rainfall of 17.7%. This trend was more marked in summers (decrease of 26.1%). This trend could reduce adequate bedrock re-watering in the forthcoming years and water stress experienced by *Q. pyrenaica* would be more pronounced. For instance, Mediavilla and Escudero (2003) found $\Psi_{\text{l-pd}}$ values as low as −3.2 MPa where thick weathered bedrock was not present.

Quercus pyrenaica Is a Well-Performed Deciduous Oak to Cope with Increasing Summer Drought

Morphological (e.g., low LAI) and physiological (e.g., low Ψ_{tlp} and $\pi_{o,}$ and high ε_{max}) differences between *Q. pyrenaica* and other deciduous oaks will be surely crucial under this global change scenario. E_A values here reported and those reported by Hernández-Santana et al. (2008a) for *Q. pyrenaica* can be qualified as moderately low. E_A rates always showed an upper limit well below E_p, indicating a control of transpiration by stomata (Rico et al. 1996; Hernández-Santana et al. 2008b). The fact that transpiration becomes constant above a certain threshold of VPD, with a gradual stomata closure as VPD increases, protects xylem from catastrophic cavitations (David et al. 2004; Chapter 3, this volume).

Stomata activity and its response to atmospheric conditions varied clearly between sites (Rico et al. 1996). With favourable water availability, wet sites reached higher conductance values, although stomata functioning fell off sharply starting with a threshold value, mainly VPD and Ψ_1. Under dry conditions, more homogeneous conductance values are found, with a less pronounced but more immediate and constant response to environmental variability (Rico et al. 1996). This could be interpreted as a functional adaptation, implying a more conservative strategy of *Q. pyrenaica* growing in drier sites. This possible genetic differentiation will require further studies given its potential interest for future forest dynamic under increasing water deficit in Mediterranean and temperate forests in Europe.

References

Aranda I, Gil L, Pardos JA (2004) Osmotic adjustment in two temperate oak species [*Quercus pyrenaica* Willd and *Quercus petraea* (Matt.) Liebl] of the Iberian Peninsula in response to drought. Invest Agrar: Sist Recur For 13:339–345

Baldocchi DD, Liukang X (2007) What limits evaporation from Mediterranean oak woodlands – the supply of moisture in the soil, physiological control by plants or the demand by the atmosphere? Adv Water Resour 30:2113–2122

Blake-Jacobson ME (1987) Stomatal conductance and water relations of shrubs growing at the chaparral-desert ecotone in California and Arizona. In: Tenhunen JD, Catarino EM, Lange OL, Oechel WC (eds) Plant responses to stress. Functional analysis in Mediterranean ecosystems, vol G15, Nato ASI Ser. Springer Verlag, Berlin, pp 223–245

Bréda N, Huc R, Garnier A, Dreyer E (2006) Temperate forest trees and stands under severe drought: a review of eco-physiological responses, adaptation processes and long-term consequences. Ann For Sci 63:625–644

Canadell J, Jackson RB, Ehleringer JR, Mooney HA, Sala OE, Schulze E-D (1996) Maximum rooting depth of vegetation types at the global scale. Oecologia 108:583–595

Čermák J, Nadezhdina N, Raschi A, Tognetti R (1998) Sap flow in *Quercus pubescens* and *Quercus cerris* stands in Italy. In 4th International workshop on measuring sap flow in intact plants. IUFRO Publications, Mendel University, Zidlochovice, Czech Republic

Chambers JL, Hinckley TM, Cox GS, Metcalf CL, Aslin RG (1985) Boundary-line analysis and models of leaf conductance for four oak-hickory forest species. For Sci 3:437–450

Ciais Ph, Reichstein M, Viovy N, Granier A, Ogeé J, Allard V, Aubinet M, Buchmann N, Chr B, Carrara A, Chevallier F, De Noblet N, Friend AD, Friedlingstein P, Grünwald T, Heinesch B, Keronen P, Knohl A, Krinner G, Loustau D, Manca G, Matteucci G, Miglietta F, Ourcival JM, Papale D, Pilegaard K, Rambal S, Seufert G, Soussana JF, Sanz MJ, Schulze ED, Vesala T, Valentini R (2003) Europe-wide reduction in primary productivity caused by the heat and drought in 2003. Nature 437:529–533

Corcuera L, Camarero J, Gil-Pelegrín E (2002) Functional groups in *Quercus* species derived from the analysis of pressure–volume curves. Trees 16:465–472

Corcuera L, Camarero J, Gil-Pelegrín E (2004) Effects of a severe drought on *Quercus ilex* radial growth and xylem anatomy. Trees 18:83–92

Cubera E, Moreno G (2007) Effect of single *Quercus ilex* trees upon spatial and seasonal changes in soil water content in Dehesas of central western Spain. Ann For Sci 64:355–364

Damesin C, Rambal S, Joffre R (1998) Cooccurrence of trees with different leaf habit: a functional approach on Mediterranean oaks. Acta Oecol 18:195–204

Davi H, Barbaroux C, Francois C, Dufrêne E (2009) The fundamental role of reserves and hydraulic constraints in predicting LAI and carbon allocation in forests. Agr For Meteorol 149:349–361

David TS, Ferreira MI, Cohen S, Pereira JS, David JS (2004) Constraints on transpiration from an evergreen oak tree in southern Portugal. Agric For Meteorol 122:193–205

Duhme F, Hinckley TM (1992) Daily and seasonal variation in water relations of macchia shrubs and trees in France (Montpellier) and Turkey (Antalya). Vegetatio 99–100:185–198

Gallardo JF (2000) Biogeochemistry of Mediterranean forest ecosystems: a case study. In: Bollag JM, Stotzky G (eds) Soil biochemistry. Marcel Dekker, New York, pp 423–460

Gallardo JF, Martín A, Santa-Regina I (1998) Nutrient cycling in deciduous forest ecosystems of the '*Sierra de Gata*' mountains: aboveground litter production and potential nutrient return. Ann Sci For 55:749–769

Gallardo JF, Molina E (1979) Relaciones entre procesos edáficos y superficies morfológicas de la Cuenca Duero, vol 1, Actas I Reunión Geológica. Salamanca, Spain, pp 211–223

Gallego HA, Rico M, Moreno G, Santa Regina I (1994) Leaf water potential and stomatal conductance in *Quercus pyrenaica* Willd forests: vertical gradients and response to environmental factors. Tree Physiol 14:1039–1047

Goldberg V, Bernhofer C (2008) Testing different decoupling coefficients with measurements and models of contrasting canopies and soil water conditions. Ann Geophys 26:1977–1992

Grier CC, Running SW (1977) Leaf area of mature northwestern coniferous forests: relation to the site water balance. Ecology 58:893–899

Hernández-Santana V, David TS, Martínez-Fernández J (2008a) Environmental and plant-based controls of water use in a Mediterranean oak stand. For Ecol Manag 255:3707–3715

Hernández-Santana V, Martínez-Fernández J, Morán C, Cano A (2008b) Response of *Quercus pyrenaica* (melojo oak) to soil water deficit: a case study in Spain. Eur J For Res 127:369–378

IPCC (2007) Climate change 2007: the physical scientific basis. Cambridge University Press, Cambridge, UK, 1009 pp

ISSS-ISRIC-FAO 1998. World Reference Base for Soil Resources. World Soil Resources, Reports No. 84. FAO UN, Rome, 1998

Jarvis PG, McNaughton KG (1986) Stomatal control of transpiration: scaling up from leaf to region. Adv Ecol Res 15:1–49

Jones HG (1992) Plants and microclimate. A quantitative approach to environmental plant physiology. Cambridge University Press, Cambridge

Kurz-Besson C, Otieno D, Lobo do Vale R, Siegwolf R, Schmidt M, Herd A, Nogueira C, David T, David J, Tenhunen J, Pereira J, Chaves M (2006) Hydraulic lift in cork oak trees in a savannah-type Mediterranean ecosystem and its contribution to the local water balance. Plant Soil 282:361–378

Llorens P, Domingo F (2007) Rainfall partitioning by vegetation under Mediterranean conditions. A review of studies in Europe. J Hydrol 335:37–54

Manes F, Vitale M, Donato E, Giannini M, Puppi G (2006) Different ability of three Mediterranean oak species to tolerate progressive water stress. Photosynthetica 44:387–393

Martínez-Vilalta J, Piñol J, Beven K (2002) A hydraulic model to predict drought-induced mortality in woody plants: an application to climate change in the Mediterranean. Ecol Model 155:127–147

Mediavilla S, Escudero A (2003) Stomata l responses to drought at a Mediterranean site: a comparative study of co-occurring woody species differing in leaf longevity. Tree Physiol 23:987–996

Merino J, Field C, Mooney HA (1982) Construction and maintenance costs of Mediterranean-climate evergreen and deciduous leaves. I. Growth and CO_2 exchange analysis. Oecologia 53:208–213

Moreno G, Gallardo JF, Ingelmo F, Cuadrado S, Hernández J (1996) Soil water budget in 4 *Quercus pyrenaica* forests across a rainfall gradient. Arid Soil Res Rehabil 10:65–84

Moreno G, Obrador JJ, Cubera E, Dupraz C (2005) Root distribution in dehesas of Central-Western Spain. Plant Soil 277:153–162

Ogaya R, Peñuelas J (2007) Leaf mass per area ratio in Quercus ilex leaves under a wide range of climatic conditions. The importance of low temperatures. Acta Oecol 31:168–173

Peñuelas J, Boada M (2003) A global change-induced biome shift in the Montseny mountains (NE Spain). Global Change Biol 9:131–140

Peñuelas J, Lloret F, Montoya R (2001) Severe drought effects on Mediterranean woody flora in Spain. For Sci 47:214–218

Querejeta JI, Egerton-Warburton LM, Allen MF (2007) Hydraulic lift may buffer rhizosphere hyphae against the negative effects of severe soil drying in a California oak savanna. Soil Biol Biochem 39:409–417

Rambal S (1984) Water balance and pattern of root water uptake by a *Quercus coccifera* L. evergreen scrub. Oecologia 62:18–25

Rambal S (1993) The differential role of mechanisms for drought resistance in a Mediterranean evergreen shrub: a simulation approach. Plant Cell Environ 16:35–44

Rico M, Gallego HA, Moreno G, Santa Regina I (1996) Stomata l response of *Quercus pyrenaica* to environmental factors in 2 sites differing in their annual rainfall (Sierra de Gata, Spain). Ann For Sci 53:221–234

Rodá F, Retana J, Gracia CA, Bellot J (1999) Ecology of Mediterranean evergreen oak forests. Springer, Berlin, p 373

Savé R, Castell C, Terradas J (1999) Gas exchange and water relations. In: Rodá F, Retana J, Gracia CA, Bellot J (eds) Ecology of Mediterranean evergreen oak forests, Ecological Studies, vol 137. Springer, Berlin, pp 135–144

Schiller G, Cohen S, Ungar ED, Moshe Y, Herr N (2007) Estimating water use of sclerophyllous species under East-Mediterranean climate III. Tabor oak forest sap flow distribution and transpiration. For Ecol Manag 238:147–155

Schneider K, Turrión MB, Grierson PF, Gallardo JF (2001) Phosphatase activity, microbial phosphorus, and fine root growth in forest soils in the '*Sierra de Gata*', western central Spain. Biol Fertil Soils 34:151–155

Silla F, Escudero A (2006) Coupling N cycling and N productivity in relation to seasonal stress in *Quercus pyrenaica* Willd. samplings. Plant Soil 282:301–311

Sobrado MA (1986) Aspects of tissue water relations and seasonal changes of leaf water potential components of evergreen and deciduous species coexisting in tropical dry forest. Oecologia 68:413–416

Sternberg PD, Anderson MA, Graham RC, Beyers JL Tice KR (1996) Root distribution and seasonal water status in weathered granitic bedrock under chaparral. Geoderma 72:89–98

Valladares F, Peñuelas J, De Luis-Calabuig E (2004) Impactos sobre los ecosistemas terrestres. In: Moreno JM (Coord) Evaluación Preliminar de los impactos en España del cambio climático (ECCE). Ministerio de Medio Ambiente, Madrid, pp 65–112

Vicente MA, Gallardo JF, Moreno G, González MI (2003) Comparison of soil water-contents as measured with a neutron probe and time domain reflectometry in a Mediterranean forest ('*Sierra de Gata*', Central Western Spain). Ann For Sci 60:185–193

Winkel T, Rambal S (1990) Stomata l conductance of some grapevines growing in the field under a Mediterranean environment. Agric For Meteorol 51:107–121

Witty JH, Graham RC, Hubbert KR, Doolittle JA, Wald JA (2003) Contributions of water supply from the weathered bedrock zone to forest soil quality. Geoderma 114:389–400

Section II
Forest Structures, Management and Water Fluxes

Chapter 11
Introduction

Viliam Pichler

The majority of forestry interventions and measures concern forest tree species composition, density, structure and other patterns and characteristics, which in turn influence leaf area index, canopy roughness, albedo, surface humus properties, soil edaphone associations, tree roots density, and many other properties. All of these characteristics are expected to have a profound impact on the water cycle in forests and their adjacent areas.

For instance, new findings in the field of trees ecophysiology have cast fresh light on the transpirational responsiveness of various tree species to soil moisture, which plays an important role in the runoff process and thus in choosing tree species featuring ability to affect runoff in the desired way. As an illustration, European beech (*Fagus sylvatica*) shows a high and almost instantaneous transpirational response to variations in soil water content or global irradiation (e.g. Čermák et al. 1993; Střelcová et al. 2006). This ability may facilitate a rapid regeneration of the water storage capacity in forest soils.

At the same time, forest effects cannot be confined to flood mitigation. Nowadays, it is widely accepted that an increase in forest cover leads to a change in the water balance of a hydrological catchment, namely to an increase in annual evapotranspiration and thus to a decrease in annual runoff (Badoux et al. 2006). Therefore, efforts are undertaken to reconcile production, regulation and socio-economic functions of forests with maintaining and/or increasing groundwater recharge and runoff in arid and semi-arid regions. Such forestry measures often lean on forest density reduction and a subsequent medium-term increase of the water yield from forests (Tryon 1972; Stogsdill et al. 1989; Stednick 1996).

Besides, alteration or a departure from natural forest cover, tree species composition and forest structures due to local, regional or global changes show a considerable impact on the water cycle. The evidence of the past vegetation cover

---

V. Pichler (✉)
Department of Natural Environment, Technical University in Zvolen, T.G. Masaryka 24,
960 53 Zvolen, Slovak Republic
e-mail: pichler@vsld.tuzvo.sk

M. Bredemeier et al. (eds.), *Forest Management and the Water Cycle: An Ecosystem-Based Approach*, Ecological Studies 212, DOI 10.1007/978-90-481-9834-4_11,

destruction by anthropogenic intervention has been documented by the Holocene erosion events. Three regional studies of Holocene erosion and alleviation in Greece (van Andel et al. 1990) showed that most recorded Holocene soil erosion events are spatially and temporally related to human interference in the landscape, i.e. introduction of farming, clearing land for farming, deforestation for timber and grazing. Sedimentation is the most common water pollution agent that may result from forestry activities (Ripl 1995). Especially sound surface humus cover has the capacity to prevent erosion, while its formation and quality can be influenced by appropriate forest management.

Thus, further research of forest management and water cycle interactions is not only plausible and interesting, but also necessary and highly beneficial. According to new legislation being prepared or implemented by some European countries, forest owners have to produce evidence of their forests' societal benefits in order to be eligible for subsidies. Of course, this may be complicated by uncertainties ensuing from the inherent spatio-temporal variability of forest ecosystems (Tužinský and Soroková 2001). For instance, to assess the validity of climate change modelling for local conditions, it shall be stressed that the General Circulation Model on which other models are based, provides outputs related to a minimum area of 60,000–100,000 km^2 irrespective of local or regional effects. For example, the Alps are represented along with the Carpathians as a plateau at a uniform elevation of about 1,000 m a.s.l. (Minďáš and Škvarenina 2003). With regard to the variability of soils, it is usually considerable. Therefore, spatial design of measurements, soil sampling schemes and sample support must account for it. Generally, field methods used for soil hydrological investigations suffer less from the highly localized character of soil samples used laboratory measurement (Green et al. 1986).

Building on recent scientific results and authors' research, the following chapters are supposed to bring new insights into the important issue of forest management control on the water cycle. In addition to providing their expertise, authors also hoped to encourage further and well focused research.

References

Badoux A, Jeisy M, Kienholz H et al (2006) Influence of storm damage on the runoff generation in two sub-catchments of the Sperbelgraben, Swiss Emmental. Eur J For Res 125:27–41

Čermák J, Matyssek R, Kučera J (1993) The causes of beech decline on heavy soils after sudden reduction of stand density. (In Czech, with English abstract). Lesnictví – Forestry 39:175–183

Green RE, Ahuja LR, Chong SK (1986) Hydraulic conductivity, diffusivity, and sorptivity of unsaturated soils: field methods. In: Klute A (ed) Methods of soil analysis, Part 1. Physical and mineralogical methods. ASA, Inc./SSSA, Madison

Minďáš J, Škvarenina J (eds) (2003) Forests and global climate change. (In Slovak, with English abstract). EFRA Zvolen, FRI Zvolen, Zvolen

Ripl W (1995) Management of water cycle and energy flow for ecosystem control: the energy-transport-reaction (ETR) model. Ecol Model 78:61–76

Stednick JD (1996) Monitoring the effects of timber harvest on annual water yield. J Hydrol 176:79–95

Stogsdill WR Jr, Wittwer RF, Hennessey TC et al (1989) Relationship between throughfall and stand density in a Pinus taeda plantation. Forest Ecol Manag 29:105–113

Střelcová K, Minďáš J, Škvarenina J (2006) Influence of tree transpiration on mass water balance of mixed mountain forests of the West Carpatians. Biologia 61, Suppl. 19:75–79

Tryon ChP (1972) Partial cutting and increased water yields – a new multiresource approach. J Soil Water Conserv 2:66–70

Tužinský L, Soroková M (2001) Hydropedological cycles in a Luvisol under a an oak forest. (In Slovak, with English abstract). Acta Facultatis Forestalis Zvolen 43:355–368

van Andel TH, Zangger E, Demitrack A (1990) Land use and soil erosion in prehistoric and historical Greece. J Field Archaeol 17:379–396

Chapter 12
Changes in Forest Cover and its Diversity

Viliam Pichler, Paulo Godinho-Ferreira, Tzvetan Zlatanov,
Magdaléna Pichlerová, and Juraj Gregor

Introduction

Current options for the management of European forests are profoundly affected by the interplay of natural history and land use. Today, forest cover in Europe ranges from 0.3% in Iceland to 72% in Finland, including a broad spectrum of boreal, temperate and Mediterranean forest types, as well as forest tundra and mountain formations. In total, 1,001.4 Mha of forests cover 44.3% of the European landscape, including Russia (FAO 2005, 2009).

At the same time, considerable differences between the potential and actual vegetation cover exist. This divergence has evolved since the commencement of the re-colonisation of the European continent by trees after the last glacial, owing to two major factors. Firstly, it has been affected by climate changes that took place following the end of the Tertiary period and particularly those occurring after the last ice age, i.e. during the last 15,000 years. Even though trees and their associations had been present in Europe before glaciation, these are not directly relevant to nowadays forests because the ecological continuity of soils and the vegetation was broken (Rackham 1990). Secondly, during the Holocene, prior to the beginning of the Neolithic period, the activity of the dwelling humans commenced to cause further differentiation of nowadays actual vegetation (Lang et al. 2003).

V. Pichler (✉) and J. Gregor
Department of Natural Environment, Technical University in Zvolen, 960 53 Zvolen, Slovakia
e-mail: pichler@vsld.tuzvo.sk

P. Godinho-Ferreira
Departamento de Ecologia, Recursos Naturais e Ambiente, Estação Florestal Nacional,
Quinta do Marquês, 2780–159 Oeiras, Portugal

T. Zlatanov
Forest Research Institute, 132 St. Kliment Ohridski` Blvd, 1756 Sofia, Bulgaria

M. Pichlerová
Department of Landscape Planning and Design, Technical University in Zvolen,
960 53 Zvolen, Slovakia

M. Bredemeier et al. (eds.), *Forest Management and the Water Cycle: An Ecosystem-Based Approach*, Ecological Studies 212, DOI 10.1007/978-90-481-9834-4_12,

While the first factor immediately affected both forest coverage and species composition, the anthropogenic influence was first largely confined to the forest coverage reduction and its fragmentation. Later on, with the onset of forestry, tree species composition of forests, as well as their structure and textural patterns started being altered or replaced purposefully. Inadvertently, such changes have had profound effects on the water balance of forest ecosystems and their adjacent areas.

Forest cover directly affects rates of transpiration, evaporation, soil freezing, as well as patterns of snow accumulation and snow melt (Hetherington 1987). Thus, changes in forest structure, including those induced by forest management can modify processes that control water balance in space and time. Studies in North America showed that percentage of rainfall intercepted and evaporated by the forest canopy ranges from about 12% to 48% of rainfall, depending on the climate, tree species, canopy structure and tree epiphytes (Kimmins 1997). Case study in different age forest stands in Russian taiga showed that evaporatranspiration was lower in old-growth forests, when compared to young stands: 530 mm in 60 year old stands versus 430 mm in 140 year old stands for pine forests, or 590 mm in 60 year old stands versus 430 mm in 140 year old stands for deciduous forests, respectively (Shiklomanov and Krestovsky 1988). In Australian mountain ash forest evaporation was also found to be lower in old-growth forest when compared to young regrowth. Consequently, water interception was lowest and runoff highest in an old-growth mountain ash stands (Vertessy et al. 2001).

Such effects result not only from distinct total precipitation and its spatio-temporal distribution, but also from different acclimation mechanisms of various tree species, their distinct water demand and responsiveness to draught and soil water supply. Therefore, this chapter provides an outline of the natural vegetation cover dynamics in Europe, as well as of the past, current and projected changes thereof due to the two main factors.

The Climax Forest Cover in Europe

Considering forest cover deviations from a steady-state presumes the existence of natural succession series. As the floral associations were asymptotically reaching their climax stages, these series became apparent. This approximation is admissible owing to the relative constancy of ecological conditions characteristic of a given site. Since the end of the last glaciation, however, the forest cover in Europe, its prevailing species compositions and structures have been highly dynamic due to varying climate. These changes have represented the driving force of the re-colonization of the European continent by trees, as seen in Table 12.1, which provides a brief overview of the climate, soils and vegetation development in the Temperate Zone of Europe during Holocene.

The scientific and conservation importance of such processes has recently been recognized by the inscription of Carpathian beech primeval forests on the World

Table 12.1 The development of climate, soils and vegetation in the Central Europe during Holocene (According to Plesník 1972; Krippel 1986; Bublinec and Pichler 2001)

Chronology (years BP)	Period	Climate changes	Soil types formation	Description of dominant vegetation
<1500	Subrecent	Short-term changes, extremities	Fluvisols	– Retreat of forests – Cultural and managed forests, monocultures
2650–1500	Subatlantic	Moister and colder	Orthic luvisols	– Retreat of steppes – Stands with beech, fir and spruce
3200–2650	Subboreal	Dryer and warmer	Chhernozems	– Steppes, hornbeam – Beechwoods and beech-fir mixed forests – Origin of forest vegetation stages
5950–3200	Epiatlantic	Dryer and warmer	Cambisols	– Expansion of beech and fir
7950–5950	Atlantic	Moist and warm	Luvisols	– Mixed oak forests – Spruce forests – The onset of beech and fir
9700–7950	Boreal	Continental, dry and warm (22°C)	Cambisols	– Steppes, forest-steppes – *Quercetum mixtum* (oak, lime, hazel, elm, succession of spruce)
10250–9700	Pre-boreal	Warming by 2–3°C (12–13°C)	Gleysols Podsols	– Birch – Pine incl. dwarf pine
>10250	Dryas	Average July temperature <10°C	Permafrost	– Barren sub-arctic tundra and alpine meadows

Heritage List. The inscription occurred on the merits of representing the best and outstanding example of the ongoing process of re-colonisation and development of forest ecosystems in Europe after the last ice age (Pichler et al. 2007). European beech (*Fagus sylvatica*) has been used as a model species in European palaeoecology, and its past distribution has been discussed in a number of papers (e.g. Magri et al. 2006). The species seems to be a markedly successful tree species owing to its phenotypic plasticity and evolutionary adaptability that may counteract a further contraction of the European beech range arising from climate change in the future (Bolte et al. 2007). Therefore, it is sometimes considered a tree species of choice and the back-bone of close-to-nature forestry in Central Europe.

Until now, successful forest management approaches have been able to draw on the fact that the long-term selection of species led to the formation of associations optimally adjusted to the respective environment. Such an adjustment meant both the stabilisation of tree species composition for a certain territory, as determined by latitude, continentality and altitude, and the development of ecotypes optimally adapted to the respective site conditions. Generally, the European continent can be divided into several regions with respect to the forest constituents:

Fig. 12.1 Birch (*Betula* sp.) megafossils at Drumbabót in Fljótshlíð, southern Iceland (Photo: Olafur Eggertsson)

The boreal region extends in Northern Europe, including coniferous forests composed of Norway spruce (*Picea abies*) and Scots pine (*Pinus sylvestris*) with pioneer deciduous species such as birch (*Betula* spp.) (Fig. 12.1).

The Atlantic region encompasses the islands (Ireland, United Kingdom) and the West Coast of the Europe. The natural forests in this part of Europe are now extremely rare due to deforestation and conversion. Their potential vegetation includes mainly certain deciduous species, e.g. European beech, oak (*Quercus* spp.), lime (*Tilia* spp.) and hazel (*Corylus avellana*).

The Continental region extends over the Central and Eastern Europe and it also includes high mountains areas dominated by supramontane forests composed of conifers, mainly Norway spruce, stone pine (*Pinus cembra*) and European larch (*Larix decidua*) in analogy with the Boreal region. However, a higher irradiance at lower latitudes allowed for the enrichment of local associations by silver fir (*Abies alba*) and beech, prevailingly at lower elevations. In the mountain zone, mixed spruce-fir-beech forests with admixtures of noble hardwoods such as sycamore (*Acer platanoides*), wych elm (*Ulmus glabra*) and ash (*Fraxinus excelsior*) constitute the bulk of the natural tree species associations. The lowlands host oak, hornbeam (*Carpinus betulus*) and other broadleaved species.

The Mediterranean region stretches along the southern border of the European continent. Its climate is warm and dry during most of the vegetation period. Its forest and woodlands are composed of oak species (*Quercus ilex*, *Q. rotundifolia*, *Q. coccifera*, *Q. suber)* and Mediterranean pines (*Pinus nigra*, *P. pinaster*, *P. halepensis*, *P. brutia*).

Natural forests of the respective regions feature several characteristic patterns. Their structure is usually determined by the presence of species and trees of various

Fig. 12.2 Optimum stage in a natural beech forest, Chorna Uholka, Ukraine (Photo: Viliam Pichler)

age, diameter, and height classes. In turn, the texture results from the spatial arrangements of areas, in which one specific developmental stage prevails. A natural sequence of such stages is known as the developmental cycle. Its dynamics has been mainly studied in the Boreal Zone of Europe, as well as in the Carpathians and in the Balkans. In the predominantly mixed temperate natural forests, three main developmental stages are commonly identified (Leibundgut 1993; Korpeľ 1995): the growth phase, optimum phase, and the final stage of disintegration (Fig. 12.2).

Recently, Brang (2005) reviewed the conceptual foundations for the transfer of knowledge from primeval to managed forests. One difficulty inherent to the concept of primeval forests as a knowledge source for European silviculture is the scarcity of primeval forest remnants. Today, natural forests represent only 1.7% of the total forest area in Europe, left in strict forest reserves and other protection categories in Europe (Parviainen et al. 2000). Only in very few sites is it possible to compile a complete ecological spectrum of tree species (Fig. 12.3) that would represent the respective ecological conditions.

Several European primeval forests deserve a special attention owing to their potential for the research of forest structures and biogeochemical cycles, including that of water, whose results could be applied in practical forest management. Perucica is the largest primeval forest (1,400 ha) that represents the mixed mountain systems biome (beech, fir, spruce). It is found in the the Sutjeska NP (Bosnia-Herzegovina). Other remnants are found in the Eastern and Western Carpathians, e.g. Chornohyrski Massif in the Carpathian Biosphere Reserve and Dobročský Primeval Forest in Slovakia (Pichler et al. 2007). The dynamics of the latter ecosystem has been studied for several decades (Korpeľ 1989). The best example of indigenous European oak forests is found in Muniellos (Spain). The reserve encloses 5,542 ha with an altitudinal range from 520 to 1,685 m a. s. l. European

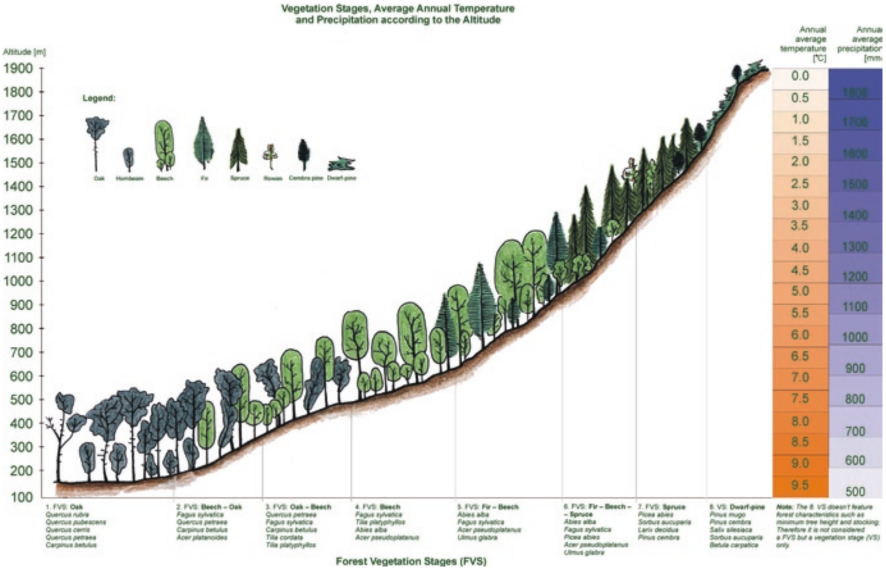


Fig. 12.3 Tree species composition changes due to varying average temperature and precipitation in the Western Carpathians (Illustration: Boris Beláček)

beech forests are represented in several forest preserves. The most important beech primeval forests series stretches from the Stužica (Slovak Republic) to Chorna Uholka reserve (4,734 ha) (Ukraine), considered the largest continuous area of beech primeval forest in Europe. The Danube Delta in Romania (biome of temperate grasslands) has 679,222 ha of the largest continuous marshland in Europe with alluvial forest elements composed of pedunculate oak (*Quercus robur*), white poplar (*Populus alba*), black poplar (*Populus nigra*) and other species. Pristine Komi Forest in the Russian Federation is the largest preserve representing boreal spruce forests of the Western Palaearctic Realm (IUCN 2004).

Both natural and anthropogenic changes of forest tree species composition play a considerable role in the water balance of forest ecosystems, owing to the differing ecophysiological plasticity and responses to water availability or scarcity, as shown in various investigations. In beech for instance, reduction in stomatal aperture in dry air was negatively correlated with the sum of photosynthetic photon flux density of the previous 3 weeks, but was positively correlated with the mean vapour pressure deficit of the previous month. These acclimation effects contribute significantly to explaining variances in stomatal sensitivity across the growing season (Kutsch et al. 2001). Zweifel et al. (2007) found that oak withdrew more water from the tissue storage pools and reduced leaf water potentials to lower levels, without risking serious damage or cavitation, than pine. Generally, the variability in driving forces of the water and energy fluxes appears to be primarily species related. On the other hand, site-dependent characteristics and climatic differences will however generate rather different total annual fluxes (Dolman et al. 2003).

Zoogenic and Anthropogenic Effects on European Forests

Distribution of the potential natural vegetation in Europe (Bohn et al. 2004) illustrates changes in the forest cover and its composition and structure, when compared with syntheses of current inventories (e.g. Schuck et al. 2003). The development of current European forests started following the last ice age. From its beginning, the process was determined by the recolonisation of barren territories by a multitude of tree species and influenced by soil edaphone, large herbivores and humans during various periods of time. Although the cyclical model (Korpeľ 1989), in which lowlands of the Continental Europe would have been covered by closed deciduous forest, and the wood-pasture model (Vera 2000) seem to be mutually exclusive, the natural vegetation probably consisted of a mosaic of closed forests, grassland and scrub, resembling nowadays agroforestry areas (in a broader sense), mainly scattered on karst plateaus featuring clay soils. Rare opportunities to study the impact of large herbivores still exist in certain areas, most importantly within the Białowieza Primeval Forest (Poland) that represents lowland forests biome with its floral and faunal diversity that includes a free-roaming population of wisent (*Bison bonasus*). However, existing populations of elk (*Alces alces*) and red deer (*Cervus elaphus*) continue to affect forest regeneration patterns across the boreal and temperate forests in Europe.

Parallel to the former influences, ecological disturbances including fire, floods, pests infections, and storms formed forest gaps. Mainly the intermediate wind disturbances play a more important role in the beech forest dynamics than it was previously thought (Nagel and Diaci 2006; Kenderes et al. 2008), whereas large scale disturbances due to windthrow, insect pests and fire determine the dynamics of boreal forests with a more homogeneous tree species composition.

Besides the large herbivores, the contribution of macroedaphone in the cyclical model of forests development has recently been recognized. Ponge (2003) suggested that humus forms, shaped and stabilized by interactions between plants, microbes, animals and their physical environment, could be the driving force which explains most of the variation in terrestrial ecosystems. Various organisms affiliated either with mull, moder or moor, such as enchytraeids and earthworms, affect the hydrological response of forest soils in that some of the organisms create continuous macropores, while others do not.

As for the anthropogenic influence, Mesolithic and even Paleolithic men availed of an array of tools including axes. But their numbers were small and activities limited to temporary clearings around their homes (Sims 1973). The initial large scale deforestation in Europe occurred due to early tillage of soils gained from forest steppes (Krippel 1986; Plesník 2004) and subsequently on land cleared of forest – often by burning (Bradshaw 2004) – some 4000–3000 BC. It was the start of agricultural acquisition and use which lasted through the Bronze Age with its large-scale clearing of woods, which changed the water balance and regional climate (Bastian and Bernhardt 1993). About half of the elm population disappeared from Europe around 4000 BC, a decline associated with a sudden increase in agricultural

Fig. 12.4 Deforested, heavily grazed uplands of The Burren, County Claire, Ireland (Photo: Edward P. Farrell)

weeds supported by archeological evidence from Neolithic settlements (Rackham 1990). However, in analogy to the impact of herbivores discussed by Bradshaw and Mitchell (1999), grazing of livestock and sheep may have been of major importance owing to the limits on forest regeneration, most importantly on geographically or orographically isolated land such as islands (Iceland, Ireland, United Kingdom) (Fig. 12.4) or karst (karstic plateaux in Croatia, Bosnia-Hercegovina, Greece, Cyprus and others). Nevertheless, quantitative deforestation trajectories for the time span 5000 BC–AD 1000 remain unknown due to the lack of data. On the other hand, species such as hornbeam, hazel and beech were among those whose post-glacial spread was partly linked with prehistoric anthropogenic activity (Behre 1998, Bradshaw 2004). Changes that took place during the last 1,000 years are better documented. The integrated development during this period made wide use of all the other potentials and resources the natural landscape had to offer beyond purely agricultural pursuits, both in extensive and intensive forms (development of ore-mining areas, multipurpose and purely extensive use of timber, wood and its derivatives) (Bastian and Bernhardt 1993). During the period between approximately 1,000 and 1,800, the forest cover in most European countries was reduced by tens of percent. The reason leading to the encroachment of deforested areas on the remaining forests were multiple. In some extended European mountain ranges, the rapid spread of livestock and sheep grazing in the sixteenth and seventeenth centuries caused and unprecedented destruction of the treeline. In the Carpathians, 50–60% of the treeline was lowered during the sixteenth to eighteenth centuries following the Valachian colonization by several hundred meters, causing an increase in the frequency of floods (Plesník 1972; Korpeľ 1989; Coldea 2003). Similar, but more dramatic situation occurred in the Alps and Pyrenees, where long tracts of the treeline ecotone entirely disappeared. Apart from the spread of subalpine and alpine meadows with a high diversity of flowering species, contributions to cultural and lifestyle diversity, such developments resulted in an increased runoff and erosion. In such areas, forest amelioration practices may provide a remedy for degraded land and its water cycle. In the boreal region, and mainly in Finland, slash and burn agriculture led to the reduction of the forest cover. Also, sectors of the northern treeline in Eurasia were pushed southward by indigenous peoples who used the scarce wood resources and changed land cover through semi-domestic reindeer grazing. In other parts of Europe, a rapid development of mining activities, metallurgy, leather tanning, cork production, tar production, shipbuilding and other industries starting in the fifteenth century caused an enormous demand for timber, wood, charcoals, potassium, tar, tannin and other products, which peaked in the early nineteenth century.

Forest Cover Transition

The “wood-famine”, a contemporary term, and perceived environmental damage were so intense that many European governments gradually enacted legislations on the prohibition of decreasing the total forest cover or stipulating obligatory afforestation of degraded areas throughout the eighteenth to nineteenth centuries (Binder 1962). As a result, the origins of forest management and forestry science date back to the eighteenth century due to the pressures resulting from an extensive use of forests. From the very beginning of the nineteenth century, higher education and forestry research establishments were founded in several European countries. This was a turning point, from which on the original tree species composition of European forests has been subject to a purposeful alteration and experiment.

Until then, the exploitation of forests and their regeneration resulted in altering forest coverage, its spatial patterns and the forest type, as determined by the successive application of methods aimed at the establishment of a new crop, such as coppicing, suckering, pollarding, shredding and others. Nevertheless, the overall tree species composition was rarely subject to a planned change, although some practices favored certain species, such as Scots pine against Norway spruce in the slash and burn agriculture (Pitkänen et al. 2003). However, the sanctioned afforestation meant the beginning of forest conversions in terms of their species composition in some parts of Europe. During the last two centuries, Norway spruce was the most favored tree species in the Central Europe due to economic reasons (Diaci 2002), i.e. rapid growth, high yield and good workability. Its share on the tree species composition rose by tens of percent, mainly at the cost of fir and beech. The latter species had been utilized mainly as firewood by the end of nineteenth century. Only later technological developments enabled its use for many other purposes, a fact that was not lost on the most important economic treatise of its time (Eucken 1950). As a consequence, spruce stands of unsuitable provenance and ecotypes grow in many sites across Central Europe, leaving these man-made ecosystems exposed to adverse effects of drought, bark beetle, pest infections and degraded soil conditions induced by monocultures and air pollution (Fig. 12.5).

Fig. 12.5 Only larch trees have remained standing after the 2003 windthrow in the High Tatras, Slovakia. The winds destroyed 12,000 ha of forest in approximately 20 min (Photo: Magdaléna Pichlerová)

As a result of these contemporary policies, forests recovery and a slow expansion began. For instance, forest cover change in the Orawa region of Poland showed a forest coverage increase from 25% to 40% over past 180 years. Changes in forest area were largely related to a decline in agriculture (Kozak 2003). These examples illustrate that forest transitions have often occurred during two, sometimes overlapping, circumstances. Firstly, it occurred when the economic development created enough non-farm jobs to pull farmers off of the land, thereby inducing the spontaneous regeneration of forests in old fields. As one of the consequences, secondary successions takes place in areas in which traditional grazing regimes were abandoned (e.g. large sections of the Spanish dehesas) (Bradshaw 2004). Secondly, it went on due to afforestation aimed to overcome the scarcity of forest resources and to mitigate environmental problems (Rudel et al. 2005). This trend will likely continue in the future. It is because starting from 2013, more subsidies will be spent on rural development compared to agricultural production, according to the EU Common agricultural policy. The role of forestry is hence changing from being predominantly productive, to take account of non-consumptive service orientated needs which arise from social and environmental demands and policy objectives. In this concept, forests are regarded as contributing towards the character and quality of rural landscapes with multiple impacts and demands on forest management (Simpson et al. 2009) and the supply of renewable resources, also including water for recreational facilities.

Until now, many of the newly established forests have been plantations. These restored forests improved ecosystem services, but did not match the composition and structure of the original forest cover. Therefore, these forests require adaptive management as dynamic, resilient systems that can withstand stresses of climate change, habitat fragmentation, and other anthropogenic effects (Chazdon 2008). In addition, particular demand will be put on designing attractive, diverse and sustainable forest landscapes (Bell and Apostol 2008).

To account for all these trends, various attempts to elaborate scenarios for the development of European forests have been made. Because the burning of fossil fuels leads to problems in applying the traditional concept of sustainable forestry, in which site factors are assumed steady-state (Kauppi 1996), the scenarios are accompanied by a set of assumptions and uncertainties resulting from the ongoing natural, environmental and socio-economic processes. Firstly, climate modeling results show an increase in annual temperature in Europe of 0.1–0.4°C per decade over the twenty-first century based on a range of scenarios. This increase will be likely accompanied by a widespread rise in precipitation in the north, small decreases in the south, and ambiguous changes in central Europe, while the seasonality of precipitation will change, including intense precipitation events, prolonged dry seasons and heatwaves throughout Europe (Alcamo et al. 2007).

Projected impacts on European forests have been assessed using the Holdridge Life Zone model (Holdridge 1947; Holdridge et al. 1971) and Forest Gap models (Risch et al. 2005) that provide tools for interpreting and evaluating spatial changes across regions and the temporal dynamics of forest sites due to climate change. Although potential distributional changes of species across climate scenarios are obscured by a strong variability in equally justifiable projections (Thuiller 2004), the global climate change will seriously affect the extent of forests, their composition and structure according to the regions (Raev et al. 1997; Maracchi et al. 2005). Table 12.2 compiles the impacts of global changes on forest characteristics as projected by several sources.

These changes along with inherent uncertainties must be considered when assessing projections of the future European forests development. A prognosis by Nabuurs et al. (2003) mostly shows a continuous build-up of growing stock and a continuing difference between increment and felling with a possible ratio between the felling and gross increment up to 0.62 for the multi-functional scenario. The total gross increment is foreseen to vary between 4.8 m^3 ha^{-1} $year^{-1}$ and 5.0 m^3 ha^{-1} $year^{-1}$, the latter value being achieved through the largest proportion of total fellings coming form thinnings, and a set-aside policy aimed at increasing the area of forests under various levels of protection. Protected forests area (i.e. those with the protection of forest ecosystems as their primary role) increased from 5.0% in 1990 to 9.1% in 2005, although the main rise occurred from 1990 till 2000 (FAO 2001, 2005) and, for the time being, the increasing trend seems to have leveled off. The multi-functional scenario is reflected und upheld by various studies and documents, e.g. Kankaanpää and Carter (2004) and the European Forest-Based Technology Platform, whose strategic objectives include the commercialization of the soft forest values, growing of trees enabling "tailor-made" wood supply, forests for multiple needs, advancing knowledge on forest ecosystems and adapting forestry to climate change.

Also, the relationship between the productivity and diversity of tree species in forests, which is supposed to further their ecosystem functions and services

Table 12.2 Impact of global (mainly climate) and regional change and some other global changes on forest characteristics (Sykes and Prentice 1995; Bradshaw et al. 2000; Bradshaw 2004; Minďáš and Škvarenina 2003; Maracchi et al. 2005)

Biogeogr. region	Forest cover and productivity	Species composition	Structure
Boreal, Subboreal	– Northward shift, contraction in the south – Reduction in peat formation in Fennoscandian and northern Baltic peatlands – Incr. productivity compromised by disturbances and pests	– Relative increase in Norway spruce compared, decline in Scots pine (abandonement of "slash and burn" cultivation), establishment and increase of European beech – Shift of deciduous species from central Europe – Expansion of birch (*Betula* spp.)	– Structural richness enhanced by immigrating deciduous species
Atlantic	– Slight increase in forest cover due to spontaneous re-growth and afforestation on abandoned agricultural land – Increase in drought-induced mortality	– Loss of the strong dominance of broadleaved deciduous trees, colonization by Mediterranean broadl. evergreen trees (parts of Britain, W coast of France, N and NW coasts of Spain)	– Increased uniformity, reduced differentiation across various sites
Continental (including the Alpine region and mountain areas)	– Upward shift of the tree line – Slight increase in forest cover due to spontaneous re-growth and afforestation on abandoned agricultural land	– Retreat of Norway spruce and silver fir, expansion of European beech – Replacement of temperate deciduous broadleaved species by Mediterranean species – Northward expansion of *Quercus* spp. (*Q. frainetto, Q. pubescens, Q. rubra*)	– Decrease in structural diversity
Submediterranean, Mediterranean	– Increase in shrub-domi-nated landscapes-Expansion of some thermophilous tree species (e.g., *Quercus pyrenaica*)	– Replacement of European beech by hornbeam (*Carpinus betulus*), esp. in more continental areas of eastern Europe, including the Balkan peninsula – Disappearance of some *Quercus* spp. (e.g. *Q. frainetto*) from coastal regions	– Increased uniformity (mainly Central Spain, southern France, the Adriatic coast)

(Pretzsch 2005), will perhaps be relaxed in the future. It is because the allocation of wood by short-term rotation plantations may help reduce the pressure on the sustainable, ecologically oriented forest management as an integration strategy, which should be implemented on the predominantly forested area in the future (Schmidt and Gerold 2008). Today, tree plantations cover an area of approximately 137 Mha (ca. 13 Mha in Europe), which equals 3.5% of the worldwide forest area with conifers of the genera *Pinus*, *Larix* and *Picea*, as well as broadleaves like *Eucalyptus* and *Populus* as some of the most represented species (Bemmann et al. 2008). To assure expected yield of short term rotation plantations, precipitation should reach at least 300 mm during the vegetation period, or there must be a shallow groundwater capillary fringe in the rooting zone (Röhle et al. 2008).

Overall, the increasing forested areas of Europe will likely contribute to multiple forest services, timber production being just one of them (Kankaanpää and Carter 2004). Implications and possible ways to maintain or improve the desired interactions between forests and the water cycle will be discussed in more detail through the following chapters.

Acknowledgments Parts of this work have been funded by the APVV project No. 0468/06 and VEGA 1/0723/08 related to FP0601.

References

Alcamo J, Moreno JM, Nováky B, Bindi M, Corobov R, Devoy RJN, Giannakopoulos C, Martin E, Olesen JE, Shvidenko A (2007) Europe. In: Parry ML, Canziani OF, Palutikof JP, van der Linden PJ, Hanson CE (eds) Climate change 2007: impacts, adaptation and vulnerability. Contribution of working group ii to the fourth assessment report of the intergovernmental panel on climate change. Cambridge University Press, Cambridge

Bastian O, Bernhardt A (1993) Anthropogenic landscape changes in Central Europe and the role of bioindication. Lands Ecol 8:139–151

Behre K-E (1988) The role of man in European vegetation history. In: Huntley B, Webb T (eds.) Vegetation History. Kluwer Academic Publishers, Dordrecht-Boston-London

Bell S, Apostol D (2008) Designing sustainable forest landscapes. Taylor & Francis, London

Bemmann A, Pretzsch J, Schulte A (2008) Baumplantagen weltweit – eine Übersicht. Schweiz Z Forstwes 159:124–132

Binder R (1962) Settlers in the upper Hron Valley. (In Slovak). Stredoslovenské vyd. Banská Bystrica, Central Slovakia

Bohn U, Gollub G, Hettwer C, Neuhäuslová Z, Raus T, Schlüter H, Weber H (2004) Map of the natural vegetation of Europe. Federal Agency for Nature Conservation, Bonn

Bolte A, Czajkowski T, Kompa T (2007) The north-eastern distribution range of European beech – a review. Forestry 80:413–429

Brang P (2005) Virgin Forests as a knowledge source for Central European silviculture: Reality or Myth? For Snow Landsc Res 79(1/2):19–31

Bradshaw RHW (2004) Past anthropogenic influence on genetic structure and diversity within European forests. For Ecol Manag 197:203–212

Bradshaw RHW, Holmqvist BH, Cowling SA, Sykes MT (2000) The effects of climate change on the distribution and management of Picea abies in southern Scandinavia. Can J Forest Res 30:1992–1998

Bradshaw RHW, Mitchell FJG (1999) The palaeoecological approach to reconstructing former grazing-vegetation interactions. For Ecol Manag 120:3–12

Bublinec E, Pichler V (eds) (2001) Slovak primeval forests – diversity and conservation. IFE SAS, Zvolen

Chazdon RL (2008) Beyond deforestation: restoring forests and ecosystem services on degraded lands. Science 320:1458–1460

Coldea G (2003) The alpine flora and vegetation of the South-Eastern Carpathians. In: Nagy L, Grabherr G, Körner C, Thompson DBA (eds) Alpine Biodiversity in Europe. Springer, Berlin

Diaci J (2002) Regeneration dynamics in a Norway spruce plantation on silver fir-beech forest site in the Slovenian Alps. For Ecol Manag 161:27–38

Dolman AJ, Moors EJ, Grunwald T, Berbigier P, Bernhofer C (2003) Factors controlling forest atmosphere exchange of water, energy, and carbon. In: Valentini R (ed) Fluxes of carbon water and energy of European forests. Springer-Verlag, New York

Eucken W (1950) The Foundations of economics: History and theory in the analysis of economic reality. William Hodge, London

FAO (2001) Global Forest Resources Assessment (2001) FAO forestry paper 140. Food and Agriculture Organization, Rome

FAO (2005). Global Forest Resources Assessment (2005) FAO forestry paper 147. Food and Agriculture Organization, Rome

FAO (2009) State of world’s forests. Food and Agriculture Organization, Rome

Hetherington ED (1987) The importance of forests in the hydrological regime. In: Healy MC, Wallace RR (eds) Canadian aquatic resources. Department of Fisheries and Oceans, Ottawa, Ontario

Holdridge LR (1947) Determination of world plant formations from simple climatic data’. Science 105:367–368

Holdridge LR, Grenke WC, Hatheway WH, Liang T, Tosi JA (1971) Forest environments in tropical life zones: a pilot study. Pergamon Press, Oxford

IUCN (2004) Evaluation of nominations of natural and mixed properties to the World Heritage List. Report to the World Heritage Committee 28th Session, June/July 2004 – Suzhou, China. WHC-WHC-04/28.COM/INF.14B

Kankaanpää S, Carter TR (2004) An overview of forest policies affecting land use in Europe. Finnish Environment Institute, Helsinki

Kauppi PE (1996) What is changing in the global environment? In: Korpilahti E, Mikkelä H, Salonen T (eds) Caring for the forest: research in a changing world. IUFRO Congress Report, Tampere, Finland

Kenderes K, Mihók B, Standovár T (2008) Thirty years of gap dynamics in a central European beech forest reserve. Forestry 81:111–123

Kimmins JP (1997) Forest ecology: a foundation for sustainable management, 2nd edn. Prentice Hall, Upper Saddle River, NJ

Korpel’ Š (1989) Pralesy Slovenska (Primeval forests of Slovakia, in Slovak). Veda, Bratislava

Korpel’ Š (1995) Die Urwälder der Westkarpaten. Fischer, Stuttgart/Jena

Kozak J (2003) Forest cover changes in the Western Carpathians over the past 180 years: a case study from the Orawa region in Poland. Mt Res Dev 23:369–375

Krippel E (1986) Postglaciálny vývoj vegetácie Slovenska (Postglacial development of vegetation in Slovakia, in Slovak). Veda, Bratislava

Kutsch WL, Herbst M, Vanselow R, Hummelshøj P, Jensen NO, Kappen L (2001) Stomatal acclimation influences water and carbon fluxes of a beech canopy in northern Germany. Basic Appl Ecol 2:265–281

Lang H, Bork HR, Mäckel N, Preston R, Wunderlich J, Dikau R (2003) Changes in sediment flux and storage within a fluvial system – some examples from the Rhine catchment. Hydrol Process 17:3321–3334

Leibundgut H (1993) Europäische Urwälder. Haupt, Bern/Stuttgart

Magri G, Vendramin G, Comps B, Dupanloup I, Geburek T, Gömöry D, Latalowa M, Litt T, Paule L, Roure JM, Tantau I, van der Knaap WO, Petit RJ, de Beaulieu JL (2006) A new scenario for the Quaternary history of European beech populations: palaeobotanical evidence and genetic consequences. New Phytol 117:199–221

Maracchi G, Sirotenko O, Bindi M (2005) Impacts of present and future climate variability on agriculture and forestry in the temperate regions: Europe. Clim Change 70:117–135

Mindas J, Skvarenina J (eds) (2003) Forests and global climate change (in Slovak, with English abstract). EFRA Zvolen/FRI Zvolen, Zvolen, Slovakia

Nabuurs GJ, Päivinen R, Pussinen A, Schelhaas, MJ (2003) Development of European forests until 2050 – a projection of forests and forest management in thirty countries. EFI Research Report 15. European Forest Institute, Joensuu, Finland

Nagel TA, Diaci J (2006) Intermediate wind disturbance in an old-growth beech-fir forest in Southeastern Slovenia. Can J For Res 36:629–638

Parviainen J, Bücking W, Schuck A, Päivinen R (2000) Strict forest reserves in Europe: efforts to enhance biodiversity and research on forests left for free development in Europe (EU-COST-Action E4). Forestry 73:107–118

Pichler V, Hamor F, Vološčuk I, Sukharyuk D (2007) Outstanding universal value of the ecological processes in the Primeval Beech Forests of the Carpathians and their management as World Heritage Sites. Acta Ecol, Veda, Bratislava

Pitkänen A, Huttunen P, Jungner H, Merilainen J, Tolonen K (2003) Holocene fire history of middle boreal pine forest sites in eastern Finland. Ann Bot Fenn 40:15–33

Plesník P (2004) Všeobecná biogeografia (General Biogeography, in Slovak). Comenius University, Bratislava

Plesník P (1972) Obere Waldgrenze in den Gebirgen Europas von den Pyrenäen bis zum Kaukasus. In: Troll C (ed) Geoecology of the high-mountain regions of Eurasia. Franz Steiner Verlag, Wiesbaden

Ponge JF (2003) Humus forms in terrestrial ecosystems: a framework to biodiversity. Soil Biol Biochem 35:935–945

Pretzsch H (2005) Diversity and productivity in forests: Evidence from long-term experimental plots. In: Scherer-Lorenzen M, Körner Ch, Schulze E-D (eds) Forest diversity and function: temperate and boreal systems, vol 176, Ecol Stud. Springer Verlag, Berlin

Rackham O (1990) Trees and woodland in the British landscape. Phoenix Press, London

Risch AC, Heiri C, Bugmann H (2005) Simulating structural forest patterns with a forest gap model: a model evaluation. Ecol Model 181:161–172

Raev I, Asan U, Grozev O (1997) Accumulation of CO_2 in the above-ground biomass of the forests in Turkey and Bulgaria in the recent decades. XI World Forestry Congress, Turkey. pp. 131–138

Röhle H, Böcker L, Feger K-H, Petzold R, Wolf H, Wael A (2008) Anlage und Ertragsaussichten von Kurzumtriebsplantagen in Ostdeutschland. Schweiz Z Forstwes 159:133–139

Rudel TK, Coomes OT, Moran E, Achard F, Angelsen A, Xu J, Lambin E (2005) Forest transitions: towards a global understanding of land use change. Global Environ Change 15:23–31

Schmidt P, Gerold D (2008) Kurzumtriebsplantagen – Ergänzung oder Widerspruch zur nachhaltigen Waldwirtschaft? Schweiz Z Forstwes 159:152–157

Schuck A, Päivinen R, Häme T, Van Brusselen J, Kennedy P, Folving S (2003) Compilation of a European forest map from Portugal to the Ural mountains based on earth observation data and forest statistics. Forest Policy Econ 5:187–202

Shiklomanov IA, Krestovsky OI (1988) The influence of forests and forest reclamation practice on streamflow and water balance. In: Reynolds ERC, Thompson FB (eds) Forests, climate, and hydrology: regional impacts. United Nations University, Tokyo

Simpson M, Pichler V, Martin S, Brouwer R (2009) Integrating forest recreation and nature tourism into the rural economy. In: Bell S, Simpson M, Tyrväinen L, Sievänen T, Pröbstl U (eds) European Forest Recreation and Tourism. Taylor & Francis, London/New York

Sims RE (1973) The anthropogenic factor in East Anglian vegetational history: an approach using APF techniques. In: Birds HJB, West RG (eds) Quaternary plant ecology. Blackwell Scientific Publications, London

Sykes MT, Prentice IC (1995) Boreal forest futures: modelling the controls on tree species range limits and transient responses to climate change. Water Air Soil Poll 82:415–428

Thuiller W (2004) Patterns and uncertainties of species' range shifts under climate change. Glob Change Biol 10:2020–2027

Vera FWM (2000) Grazing ecology and forest history. CAB International, Oxford

Vertessy RA, Watson FGR, O'Sullivan SK (2001) Factors determining relations between stand age and catchment water balance in mountain ash forests. Forest Ecol Manag 143:13–26

Zweifel R, Steppe K, Sterck F-J (2007) Stomatal regulation by microclimate and tree water relations: interpreting ecophysiological field data with a hydraulic plant model. J Exp Bot 58:2113–2131

Chapter 13
Afforestation Strategies with Respect to Forest–Water Interactions

Yannis Raftoyannis, Michael Bredemeier, Rasa Buozyte, Norbert Lamersdorf, Andreas Mavrogiakoumos, Edda Oddsdóttir, and Ivaylo Velichkov

Introduction

Afforestation and reforestation are two forms of direct human-induced conversions of non-forest to forest land through planting, seeding, and/or anthropogenic dispersal of natural seed sources. The two terms are distinguished by how long the

---

Y. Raftoyannis (✉)
Department of Forestry and Environmental Management, TEI Lamias, Karpenisi 36100, Greece
e-mail: rafto@teilam.gr

M. Bredemeier
Forest Ecosystems Research Centre, University of Goettingen, Buesgenweg 2, D-37077 Goettingen, Germany
e-mail: mbredem@gwdg.de

R. Buozyte
Department of Ecology, Lithuania Forest Research Institute, Liepu 1, Girionys, LT-53101 Kaunas distr, Lithuania
e-mail: r.buozyte@mi.lt

N. Lamersdorf
Department of Pedology of the Temperate Regions, Buesgen Institute, University of Goettingen, Buesgenweg 2, D-37077 Goettingen, Germany
e-mail: nlamers@gwdg.de

A. Mavrogiakoumos
Forestry Department, Cyprus Forestry College, 4841 Prodromos, Limassol, Cyprus
e-mail: amavrogiakoumos@fd.moa.gov.cy

E. Oddsdóttir
Icelandic Forest Research, Mogilsa, IS-116 Reykjavik, Iceland
e-mail: edda@skogur.is

I. Velichkov
Department of Silviculture, Institute of Forestry, Bulgarian Academy of Sciences,
132 St. Kl. Ohridski blvd., 1756, Sofia, Bulgaria
e-mail: ivo_velichkov@abv.bg

M. Bredemeier et al. (eds.), *Forest Management and the Water Cycle: An Ecosystem-Based Approach*, Ecological Studies 212, DOI 10.1007/978-90-481-9834-4_13,

non-forest condition has prevailed and for the remainder of this chapter, afforestation is used to imply both afforestation or reforestation (Nabuurs et al. 2007).

Afforestation is a major environmental goal worldwide (Savill et al. 1997) and especially so in areas with a long history of human settlement, such as the Mediterranean basin. Intense exploitation of these areas has altered the original composition of species and the physical environment (Blondel and Aronson 1999). Under such circumstances, planting is particularly necessary to recover the vegetation and to ensure the success of regeneration.

Afforestation and reforestation are included under Article 3.3 and in Articles 6 and 12 of the Kyoto Protocol and are the dominant mitigation options in specific regions such as Europe. To date, carbon sequestration has rarely been the primary driver of afforestation, but future changes in carbon valuation could result in large increases in the rates of afforestation. Accumulation of carbon in biomass after afforestation varies greatly by tree species and site, and ranges globally between 1 and 35 Mg CO_2 $ha^{-1}year^{-1}$ (Richards and Stokes 2004).

Afforestation may result in a more balanced regional water cycle by reducing run-off, flooding, and by increasing the control of groundwater recharge and watersheds protection. However, massive afforestation of grasslands may reduce water flow into other ecosystems and rivers, and affect aquifer water level and recharge, thus leading to substantial losses in stream flow (Jackson et al. 2005).

Effects of Afforestation on Water Yields and Quality

Water Yields

The flow of water out of a catchment is termed water yield and ultimately determines the amount of water that will be available for downstream use, such as urban supply, irrigation, industries or maintaining river flow (Zhang et al. 2007). Water yield is altered through changes in transpiration, interception and evaporation which all tend to increase when grasslands or shrublands are afforested. Forests use more water and there is a consistent difference in evapotranspiration between forests and grasses or short crops (Zhang et al. 2001). Furthermore, forests tend to produce less surface run off, groundwater recharge and streamflow than shallow rooted forms of vegetation (Farley et al. 2005; Johnson 1998; van Dijk and Keenan 2007).

Following afforestation, flow reduction has been demonstrated (Farley et al. 2005; Jackson et al. 2005; Lane et al. 2005) and the average loss of streamflow due to afforestation of grasslands and shrublands has been estimated to be one third to three-quarters (Farley et al. 2005). The response to afforestation can be seen very rapidly and it increases with plantation age with maximum runoff reductions to be expected between 15 and 20 years after planting (Cornish and Vertessy 2001;

Farley et al. 2005; Jackson et al. 2005; Roberts et al. 2001), due to the increase in canopy and forest floor interception as the forest grows (Putuhena and Cordery 2000). However, the water yield may increase again when the forest matures (Putuhena and Cordery 2000; Roberts et al. 2001).

The runoff is reduced by afforestation across a broad range of climates but the effects of afforestation on water yield may be more severe in drier regions than wetter ones (Farley et al. 2005; Jackson et al. 2005). In drier regions and during dry seasons, low flows are particularly important for river flow regime as certain minimum flow is always needed to ensure water supplies. Forest growth leads to an increase in the rates of interception and transpiration which results in a decreased recharge to the soils and decreased, low flows (Johnson 1998).

Even though the majority of results indicate that afforestation reduces runoff, an analysis on water resources in Australia did not suggest that the runoff from cleared areas was more than from forested catchments and the authors speculated that it may be related to scale differences, since their study area is a thousand times larger than in other studies (Van Dijk et al. 2007). Similarily, Mannerkoski et al. (2005) showed that clear-cutting of mixed *Picea abies* Karst., *Pinus sylvestris* L. and *Betula* sp. forests did not have any significant effect on the average depth of groundwater in Finland.

Changes in streamflow are associated with several factors, such as natural vegetation type, canopy cover and plantation species (Cornish and Vertessy 2001; Farley et al. 2005). As said earlier, the conversion from grassland to forest normally reduces the water yield (see e.g. Buytaert et al. 2007; Calder 2007; Farley et al. 2005; Scott and Lesch 1997; van Dijk and Keenan 2007; Wattenbach et al. 2007) even though the impacts seem to vary depending on climate and tree species used. Generally, eucalypts reduce the runoff more than other tree species (Bren and Hopmans 2007; Bruijnzeel 2004; Farley et al. 2005; Scott and Lesch 1997) and conversion from a pine dominated forest to oak forest increased both runoff and groundwater recharge (Wattenbach et al. 2007). In Europe, the most marked changes to flows are likely to occur in eucalyptus plantations in Southern Europe and coniferous plantations in poorly drained soils in Northern Europe (Robinson et al. 2003). Other forest types, such as European mixed broadleaves and open forests in the Mediterranean, are likely to have a relatively small effect on peak and low flows. Furthermore, even though forest cutting leads to short-term increase in peak and base flow at local scale, it may not be detectable at the larger catchment scale (Robinson et al. 2003). It is likely that differences in growth patterns between species play a role on water yields and this might be useful for planning afforestation projects (Farley et al. 2005).

The modelling of afforestation impacts on groundwater resources showed that afforestation leads to a reduction in groundwater recharge (Allen and Chapman 2001). In the UK, long term recharge rates were reduced by 48% beneath oak and by 75% beneath pine compared with grassland (Calder 2007). Studies have also shown that afforestation of abandoned land with spruce can lead to a reduction of the total run-off and groundwater replenishment (Meuser 1990) and in the Netherlands, the water recharge declined from approximately 485 mm/year in arable land to 172 mm in an 18-year old oak stand and approximately 100 mm in a 14-year old spruce stand (Van der Salm et al. 2006). Ladekarl et al. (2005) reported that in Denmark the estimated annual recharge for oak stands is 390 mm, while for heathland it reaches 733 mm.

Afforestation and Changes in Water Yields in Bulgaria

According to the World Resources Institute (2008) the total forested area in South Eastern Europe (SEE) was approximately 24 Mha by the end of 2000 and 1.5 Mha of that area was covered by forest plantations. Plantations consisted to a high percentage of Bulgaria's forested area (26.3%) followed by that of Albania (10.3%) while the average share of plantations in other SEE countries varied from 0.1% in Slovenia to 3.3% in Greece and FYR Macedonia (Fig. 13.1).

More than 85% of the forests in Bulgaria grow on steep slopes that were degraded by clear cuttings, overgrazing, deliberate fires and the torrential character of summer rainfalls (Raev and Jelyazkov 1980). *Pinus silvestris* L. and *P. nigra* Arn. were the main tree species used in afforestations until the 1980s when they were replaced by deciduous species (oaks, beech, limes, ash etc.). Large areas in the lower mountain belt (600–1,000 m a.s.l.) were covered by coniferous monocultures that now are in the phase of intensive height growth and their impact on water yields is definitely negative, especially in the drier sites (Raev and Jelyazkov 1980; Nedyalkov and Raev 1988). The deciduous understorey emerging in most plantations during the aggregation phase of their development additionally increases the total evapotranspiration from the area and decreases the plantations' stability on dry and poor sites (Zlatanov and Hinkov 2005; Zlatanov 2006).

Fig. 13.1 Afforestations with Austrian pine (*Pinus nigra* Arn.) replaced the native deciduous tree vegetation in part of the territory of Hisar State Forestry District, Central Bulgaria (Photo: Tzvetan Zlatanov)

Furthermore, the natural colonization of abandoned agricultural land by shrubs and pioneer tree species also contributes to water yield reduction. As a result, the total water yield of Bulgaria has decreased by 20% in the last 25 years, possibly due to climate and land use change that includes the vast afforestation programme (Bulgarian Ministry of Environment and Water 2008).

Water Quality

Generally, a well developed tree cover has positive effects on water quality (Calder 2007) and can prevent surface erosion and shallow landsliding, especially by reducing surface runoff (Bruijnzeel 2004; Sidle et al. 2006). Water quality can be improved by afforestation as nutrient, pesticide and erosion runoff from crop production are reduced (Pattanayak et al. 2005) and tree belts have been successfully used to intercept and absorb surface runoff before it reaches streams (Ellis et al. 2006). Furthermore, the build-up of litter, the undergrowth and the increased surface roughness provided by the tree roots can improve protection of the soil under a forest stand (Waterloo et al. 2007) (Fig. 13.2).

Afforestation is seen as an effective way to reduce leaching of nitrate to groundwater and improve the quality of water leaving the root zone (Hansen et al. 2007; Iversen et al. 1998). In afforested arable land in Denmark, the lowest concentration and leaching rates were observed at stand ages between 5 and 20 years, but in older forests nitrate leaching had increased (Hansen et al. 2007). Increased nitrate leaching with stand age has also been observed in old spruce forest ecosystems in Germany (Rothe and Mellert 2004). Other factors that

Fig. 13.2 In Iceland, forests cover less than 1.5% of the country but recently large afforestation programs have been initiated. The research project ForStreams aims to investigate the impact of afforestation on water quality (Photo: Kjartan Kjartansson)

influence N-leaching include harvesting regime, rotation period and choice of tree species (Rothe et al. 2002; Rothe and Mellert 2004). Coniferous forests intercept more atmospheric pollutants than deciduous forests at comparable sites under the same climatic conditions (De Schrijver et al. 2007). The evergreen foliage, the aerodynamic roughness of the canopy and the usually high leaf area index of conifers result in higher dry deposition rates in conifer stands than in deciduous stands. The higher capacity for interception of atmospheric nitrogen (N) and sulphur (S) compounds is generally reflected by larger throughfall deposition of N and S. In areas with elevated atmospheric N pollution, the N canopy uptake is lower than the N input from deposition, which results in elevated N throughfall deposition (De Schrijver et al. 2007). Numerous studies have found that throughfall deposition was up to twice as high in conifers as in deciduous forests (e.g. v. Wilpert et al. 2000; Rothe et al. 2002; Gundersen et al. 2006; De Schrijver et al. 2007, 2008). The difference between the coniferous and deciduous throughfall deposition increases with the deposition load and is negligible in remote, relatively unpolluted areas (Rothe et al. 2002; De Schrijver et al. 2007).

A number of reviews (Gundersen et al. 2006; De Schrijver et al. 2007) revealed relatively high outputs of nitrate from the soil under conifers when compared to deciduous species with the same site conditions. Comparisons of adjacent pairs of spruce and beech stands with the same soil properties and the same level of N emissions showed that nitrate leaching was generally higher below the rooting zone of spruce stands compared to beech (Fig. 13.3) (Rothe et al. 2002; Rothe and Mellert 2004; v. Wilpert 2008). However, because broadleaves are usually more abundant on fertile and N-rich soils, the monitoring of nitrate leaching at 104 sites across Europe showed that deciduous trees had higher N leaching than conifers at similar input (Kristensen et al. 2004). Nitrification is influenced by litter composition of tree species and is usually greater under beech than under spruce

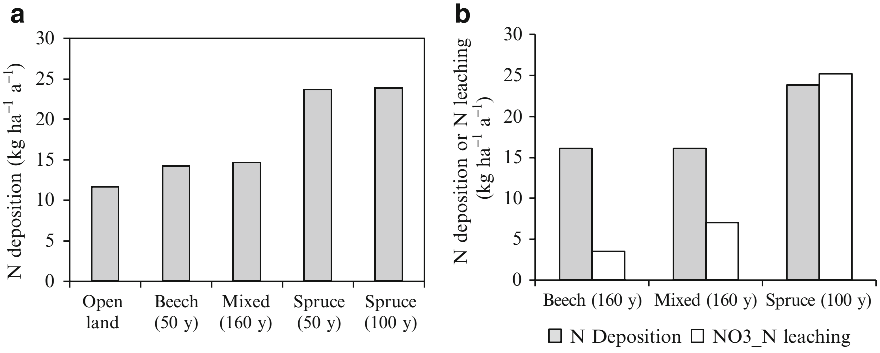


Fig. 13.3 Mean nitrogen (NO_3-N + NH_4-N) deposition (kg N ha^{-1} a^{-1}) between 1992 and 2002 in Conventwald in Southwest Germany in open land, beech stand, mixed stand (beech/spruce/fir) and spruce stand of different ages (**a**), and N deposition and nitrate nitrogen leaching (kg N ha^{-1} a^{-1}) at a depth of 120 cm in old beech, mixed and spruce stands for the period 1995–1997 (**b**) (Modified from v. Wilpert et al. (2008)

(Augusto et al. 2002). The forest floor C to N (C:N) ratio is a good proxy for N availability and there is usually an inverse relationship between C:N and nitrate leaching. Several studies showed a threshold between leaching and retention of nitrate, with leaching occurring at C:N ratios in the organic layer below 25 (MacDonald et al. 2002; Kristensen et al. 2004; Gundersen et al. 2006) or 23 (Dise et al. 2009; Ross et al. 2009). The onset of nitrification at C:N below 23–25 seems to control nitrate leaching (Ross et al. 2009).

While nitrate leaching under spruce and beech has been studied intensively, fewer data are available for other tree species or in mixed stands (Berger et al. 2009). Lower nitrate concentrations were measured in fir stands compared to spruce stands (Rothe and Mellert, 2004; Heitz and Rehfuess, 1999). But Horváth et al (2009) reported higher concentrations under Douglas fir, followed by Scots pine and oak in Northwest Germany. Among broad-leaved species (*Acer pseudoplatanus*, *Fagus sylvatica*, *Fraxinus excelsior* and *Quercus robur*), low nitrate concentrations were reported and no clear difference could be observed between tree species (Rothe and Mellert 2004). High concentrations of nitrate have been measured in soil water from forests with N-fixing species (Binkley et al. 1992; Robertson et al. 2000; Williard et al. 2005).

In Central and Northwest Europe, where deciduous species have been replaced by plantations of conifers over the last century, trends to return to the original deciduous species may, by means of the lower N input in deciduous forests, decrease the relatively widespread elevated nitrate levels in seepage water. Weis et al. (2006) showed that the removal of spruce trees combined with shelterwood regeneration of beech lead to a decreased nitrogen input due to lower interception of dry deposition of N and a higher water infiltration into the soil. Müller et al. (2002) also reported that the conversion of a pure pine stand into a beech forest in the Northeastern German lowlands resulted in an increase in drainage from the pine to the beech stand.

Energy Wood Plantations in Central Europe and Their Hydrological Constraints

A particular motivation for afforestations can be the production of wood as an energy source. Energy wood has received increasing attention in the recent past, after it had lost most of its significance in industrialized Europe during the “coal and oil age” of the twentieth century, and mostly served as a “luxury and hobby” fuel for firesides in the wealthier households. It is now appreciated as a potential carbon dioxide neutral energy carrier, suitable to partly substitute non-renewable energy sources. This aspect was strengthened in the course of the climate change and energy security debates. It is expected that additional demand for energy wood, including the developing demand of biogas facilities, may reach the range of millions of tons per year (Deutmeyer 2007). Although by now related industries are still in their developing stages, the phenomenon of competition for wood can already be observed in numerous European countries. Large combines such as "RWE Innogy" (Germany) begin to establish themselves as important electric power suppliers. This company has created a fuel-wood plantations project in Germany, which will cover about 10,000 ha and provide energy wood for use in biomass-fired plants with combined heat-and-power generation (RWE 2008).

Plantations of fast growing tree species in very short rotations (2–5 years) on marginal or set-aside agricultural land will probably develop throughout Europe in order to comply with the strongly rising demand, thus complementing the established sources of wood biomass from forests. Most of these cultivations will probably be in the form of short rotation coppices (SRCs). The aspiration is that these plantations will provide an ecologically, economically and socially sensible perspective for European farmers in line with the development of the common agricultural policy (CAP). Several integrated and interdisciplinary projects considering the diverse aspects of the issue have recently started. The "DENDROM" project in Germany compares the net returns, energy budgets and ecological impacts of poplar and salix clones and black locust, which are the major SRCs in central Europe, with those of relevant annual crops like maize, wheat or rye (Knur 2008). However, the data basis on how fast growing tree plantations will affect the water cycle and budgets at the stand, landscape and regional scales is at present still quite narrow.

In the German DENDROM project, first results seem to indicate seepage water flow reductions of approximately 15–30% for the SRCs compared with annual agricultural crops in side by side comparisons. The water budget characteristics of both poplar and black locust SRCs resembled those of young Scots pine stands. Model calculations also indicated a potential influence of the rotation period on overall leaching rates (Knur et al. 2008).

In a European synopsis, the results to date indicate that high biomass production by energy crops entails a related consumption of large amounts of water, as would be expected from general physiological considerations (Anonymous 2001). Energy crops grow very rapidly and have a high surface area during the growing season. The large leaf area means that the foliage may intercept 20–30% of rainfall, which then evaporates directly from the leaves and does not replenish soil moisture. Deep rooted energy wood crops grown on soils with large available water content will cause substantial reductions in the amount of water percolating below the root zone. Soil water deficits of up to 250 mm may build up over the growing season (Anonymous 2001). In wetter locations, this deficit would usually be replenished over the winter season. In this constellation, the higher water use could be regarded as a contribution to peak flow mitigation, presenting a positive by-product of enhanced biomass gains in the shape of seepage and discharge reduction. However, in drier environments, there may be insufficient rainfall during the winter months to rewet the soil to field capacity, which may lead to a lack of water and no deep percolation in the subsequent hydrologic year (Fig. 13.4).

Even in the humid environment of Sweden, Lindroth and Bath (1999) found tight relationships of SRC growth and water availability (Fig. 13.5), confirming the generally high transpirational demand of the fast growing tree species. Yields of

Fig. 13.4 Re-sprouting short rotation coppice (SRC) with different poplar clones (*left* vs *right side*), planted in 1997 and partly harvested in 2008 on former agricultural land in central Germany (site Georgenhof of the NOVALIS project; for further project information and results see Lamersdorf et al. 2008; Kroiher et al. 2008; Schulz et al. 2008 (Photo: N. Lamersdorf)

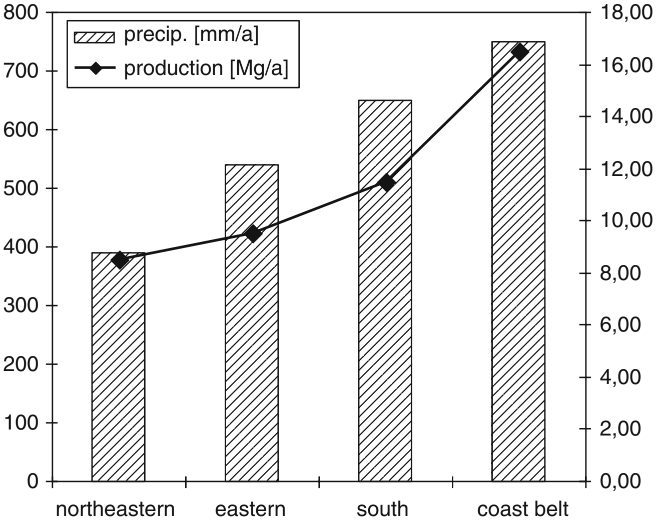


Fig. 13.5 Stemwood production of willow short rotation coppices along a precipitation gradient in southern Sweden (Based on data from Lindroth and Bath 1999)

SRCs remain far below the potential levels when water availability is limiting, and water may often be the most important of the limiting factors. Therefore, site selection criteria for SRCs should focus strongly on local hydrology in order to find sites of maximum potential water utilisation. This includes localities where more water than the amount delivered by precipitation is available, i.e. soils with groundwater connection or stagnant water. The tapping of groundwater might, obviously, conflict with other stakes of land users, but stagnant water sites are not so well suited for other cultivations, so that SRC plantations for energy could become a promising option there. However, it has to be considered that stagnant water itself can be a strong soil ecological stressor, which may substantially limit growth and vitality of plantations, owing to poor soil aeration, soil compaction and nutritional deficiencies. Furthermore, it has to be considered that shallow soils of stagnant water may dry out very strongly when intense and prolonged drought periods occur. Under such conditions, productivity of SRCs would remain far below optimum, in spite of copious water availability at most times.

Other special sites with a potential for energy wood production comprise those resulting from human land surface modifications, such as roadside slopes, embankments, green areas around industrial facilities or airports, and landfill caps. In a study in southern England, Nixon et al. (2001) estimated a potential production of about 20 Mg ha^{-1} annually, which however in many instances will be reduced as a consequence of water limitation. To maintain maximum productivity, irrigation employing the landfill leachate could be applied, since such leachate is frequently available, and would in such a constellation not constitute an additional contamination of the site. Nixon et al. (2001) conclude that there is a potential to achieve good biomass yields from species such as willow and poplar on restored landfill caps. These species are also known for their ability to tolerate temporal water logging on compacted soil and transient moderate drought periods. Likewise, Bungart and Huttl (2001) see such a potential for sites in post-mining landscapes.

In conclusion, energy wood plantation can be an economically attractive and hydrologically feasible option in the more humid environments of Europe, while the high transpirational demand of such afforestations would immediately bring about water problems in the more arid environments. Many studies have demonstrated the tremendous reduction of seepage and runoff following afforestation in dry areas, even if the species employed are not particularly fast growing (e.g., Schiller and Cohen 1998). Correspondingly, no energy wood plantations have been reported so far from arid European environments. The preference here would be clearly on water conservation over energy wood production. Trade-offs between wood production and water yields have to be generally considered in the more arid environments, such as southeastern Europe.

Water Constraints on Afforestation in the Mediterranean Region

In the Mediterranean Basin, vegetation is expected to be subject to increased future evapotranspiration leading to increased drought, which has been projected to induce forest dieback and accelerate tree mortality (Körner et al. 2005; Alcamo et al. 2007; Fischlin et al. 2007). The ability of newly planted seedlings to maintain photosynthesis under water stress will be related to their water status (Kubiske and

Abrams 1992), nursery culture (Seiler and Johnson 1988), stored carbohydrates (Guehl et al. 1993), and root expansion (Burdett 1990).

Afforestation failures can often be the result of inappropriate species selection. Differences in plant mortality during the first years after planting can be related to species not suitable for the site, the use of native genetic material often increases the chances of afforestation success. In Greece, a species comparison showed that *Fraxinus ornus* L. was the best adapted species to the dry conditions of the planting site, having lower mortality rates than *Acer pseudoplatanus* L. and *Castanea sativa* Miller. seedlings (Radoglou and Raftoyannis 2001). Also, the superiority of *Quercus frainetto* Ten. seedlings was probably related to the fact that it was the only native species and could tolerate low water potentials during the mid-summer drought (Fig. 13.6) (Raftoyannis et al. 2006).

Modification of nursery regimes can influence the performance of planting stock (Duryea 1984). Hardening has been related to improved field performance at dry sites (van den Driessche 1991). During the last phase of container nursery culture, known as the hardening phase, the growing medium is allowed to dry to induce a moderate moisture stress level in the seedlings (Brisette et al. 1991). However, water shortages induced by hardening can cause premature budset and there is a danger of inhibiting frost hardening (Duryea 1984). In *Pinus halepensis* Mill., hardening significantly affected nursery morphology but had little effect on the measured aspects of physiology or field performance showing that it behaves as a very plastic species under varying water availability (Royo et al. 2001).

Deep containers should be used for Mediterranean species that develop a deep tap root when planted in dry conditions. The relative allocation of resources to roots or shoots has been considered a key factor for seedling performance in the field (Leyva and Fernandez-Ales 1998) and one of the main

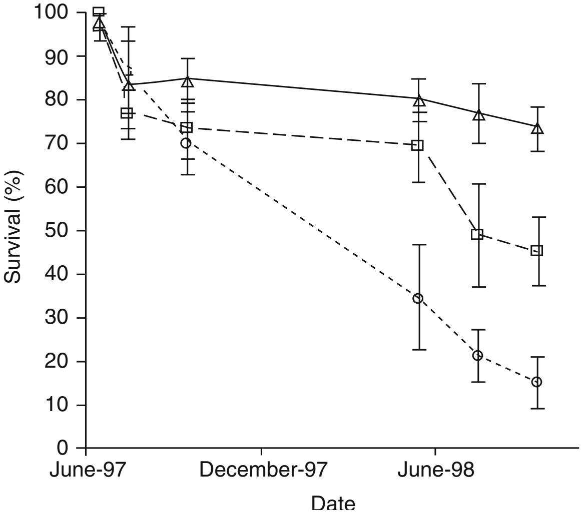


Fig. 13.6 Seedling survival during 1997 and 1998, for *Acer pseudoplatanus* (O), *Castanea sativa* (□) and *Quercus frainetto* (Δ)

strategies of Mediterranean species is to develop a deep tap root during the early stages of plant development (Tsakaldimi et al. 2005). Deep containers produce seedlings with a longer tap root that can quickly reach the deeper soil horizons and thus contribute to a better water status under drought stress conditions (Chirino et al. 2008).

Mycorrhization of seedlings in the nursery is another afforestation strategy that can improve the outcome of stand establishment through better water uptake or plant adaptation to water stress (Duan et al. 1996; Morte et al. 2001). In Mediterranean conditions, mycorrhizal symbionts are able to improve plant growth and adaptation not only during the periods of greatest symbiotic activity in spring and autumn, but also during summer drought (Querejeta et al. 1998; Garcıa et al. 2000).

Seedlings can be exposed to drying conditions at several stages during the process of lifting, handling and transportation before planting. McKay (1997) reviewed the effect of stresses between lifting and planting on nursery stock quality and performance and noted the importance of desiccation as one of the most common potential dangers in nursery operations. Desiccation adversely affects the physiological parameters and survival of many tree species (Fig. 13.7) (Insley and Buckley 1985; Radoglou and Raftoyannis 2002). Broadleaved tree species can be prone to desiccation even during the winter period, when there are no leaves, the main transpiring apparatus of a plant (Radoglou and Raftoyannis 2001). The effect of desiccation on plant condition depends on the length and intensity of exposure, as well as the species sensitivity to water stress (Kozlowski 1985).

Transplanting stress affects seedling water status (Grossnickle 1988; Kaushal and Aussenac 1989) and may result in a decline or a complete cessation of the CO_2 assimilation capacity (Guehl et al. 1989; Kozlowski et al. 1991) that can lead to limited early growth or even seedling mortality (Kozlowski and Davies 1975). Physiological indicators can be used to forecast the resistance of seedlings to drought before planting. Root electrolyte leakage is considered a valuable method of predicting seedling field performance (McKay 1997; McKay and White 1997)

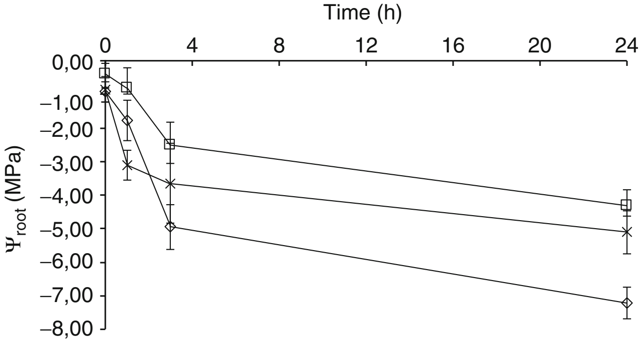


Fig. 13.7 Effect of desiccation on fine root water potential (Ψroot) of *Acer pseudoplatanus* (◊), *Castanea sativa* (□) and *Fraxinus ornus* (x) seedlings

and root growth potential is a useful method for measuring seedling quality before planting out, particularly when planting is taking place on drought-prone sites (Ritchie 1984). Needle water potential at the time of transplanting was a reliable predictor of the ability of bareroot *Pinus nigra* ssp. *Laricio* var. *Corsicana* seedlings to regenerate new roots and of seedling mortality after planting (Girard et al. 1997). Shoot water potential may offer a useful and rapid measure of seedling physiological quality and is used as a plant quality indicator in afforestation practice (Webb and von Althen 1980). In the USA, many nurseries routinely measured shoot water status with a pressure chamber, they did not lift when stress exceeded 1.5 MPa and did not permit stress to exceed –0.5 MPa when grading and packing (Ritchie 1984).

Site preparation before afforestation, including both vegetation clearing and soil preparation, may be of paramount importance to enhance the outcome of planted seedlings (Flemming et al. 1996). Vegetation clearing has been reported to diminish competition for water and nutrients, thus increasing the survival rates of seedlings (Harvey et al. 1996; Ashby 1997). Soil preparation by ploughing may reduce runoff losses, increase the total water stored in the profile and decrease the penetration resistance of the soil (Varelides and Kritikos 1995; Querejeta et al. 2001).

As opposed to the traditional afforestation techniques on shrub-cleared ground, the benefits of using shrubs as nurse plants for afforestation has been demonstrated in Spain (Castro et al. 2004; Gómez et al. 2004). Shade from neighbours can protect plants from photo-inhibition and extreme temperatures reducing plant transpiration and improving the water status of the understory species (Pugnaire et al. 1996). Additionally, nurse plants can enhance water availability via hydraulic lift (Horton and Hart 1998).The use of shrubs and perennial tussock grasses as nurse plants has been proposed as an afforestation technique that offers both economic and ecological advantages for restoring degraded semi-arid ecosystems (Castro et al. 2002; Maestre et al. 2003).

Watering can boost afforestation success in a Mediterranean environment (Rey-Benayas 1998). Irrigation at the end of the dry period is more effective than irrigation at the beginning of summer or in the period of higher drought stress, when the high temperatures provoke high evaporation (Jimenez et al. 2007). However, irrigation is an expensive option and efforts to increase the success of afforestation must focus on management techniques that increase water sources for plants, such as the use of small runoff collection areas, mulches, water-retaining polymers, or tree shelters (Navarro et al. 2005). In arid and semiarid regions, dew and fog precipitations, although relatively low compared to rain, are a constant and stable water source (Kidron 2005; Agam and Berliner 2006). Dew harvesting may yield moisture that could alleviate the water stress of planted seedlings during establishment (del Campo et al. 2006).

Laboratory experiments with *Pinus halepensis* Mill. showed, that during drought, survivorship and growth can be increased by adding hydrogels (Huttermann et al. 1999). However, the addition of hydrophilic acrylic copolymer had a slightly negative effect at the very beginning, but not thereafter (Barbera et al. 2005). Further research on the most suitable doses for application in the field could well lead to different conclusions but the good results obtained without hydrophilic acrylic copolymer, and specifically in some combinations of treatments means that extra investment in similar substances and labour could be unnecessary.

The use of organic amendments in site preparation has yielded good results by promoting the survival and growth of seedlings (Roldan et al. 1996; Querejeta et al. 2001). The treatments which reduce evaporation, such as the application of forest debris, and especially the placement of boulders in the planting bed around the seedling, proved the most effective treatments for plant development (Jimenez et al. 2007). Addition of composted solid urban wastes, despite the increase in nutrient content in the soil, did not promote seedling performance due to the competition with undergrowth plants. The application of anaerobically stabilized biosolids in the planting hole had negative effects on seedling survival. At the beginning of the dry period, the rapid desiccation of these biosolid patches generates cavities within the planting hole, followed by surface cracks near the seedling root collar. These may negatively affect local soil water storage and root contact with the soil, and thus seedling establishment (Fuentesa et al. 2007).

Conclusions

This chapter considers the relationships of afforestation and the water cycle in a broad approach, covering various eco-hydrological constellations and forest management approaches across Europe.

From a review of the current state of literature it already emerges very clearly that water consumption of forests is usually higher than that of alternative vegetation and land use types, owing both to the well known high specific surface area and interception evaporation, and to the more intense rooting and transpirational demand of forests. However, the evaluation also shows that in specific constellations these traits can have desired and favourable effects, such as improved peak flow control in wetter environments. Peak flow formation depends strongly on the build-up of surface- and near-surface flow, which is minimal under forest compared to other vegetation forms, owing to the presence of a mulch or litter layer, and high porosity and infiltration capacity in general. Likewise, a clear tendency shows of reductions in ground water yield as a consequence of afforestations.

We then consider some special cases of afforestation efforts across Europe, starting out with energy wood plantations, which gain quite some momentum currently on the background of the fossil energy crisis and political processes such as the Kyoto protocol. Economic returns and even ecological effects arising from short rotation coppices for wood energy can be attractive, but it shows that these highly productive woodlands apparently rely on very good water supply to maintain the desired high biomass production rates, and hence are only an option in the more humid environments of Europe.

Likewise, it has clearly emerged that in other afforestation efforts in Europe, benefits are often counterbalanced by drawbacks and problems. In south-eastern

Europe, major afforestation efforts were undertaken mostly motivated by the desire for soil protection, for instance in torrential precipitation environments such as in Bulgaria. While the desired soil protection effects can be achieved in most instances, usually more or less severe reductions in runoff and water yields have to be faced in such a constellation.

The feasibility of afforestations in the dry Mediterranean environment is another concern in our context. Carefully selected tree species and their combinations, together with specially adapted site preparation and planting techniques are offering perspectives for successful afforestation efforts even under harsh and adverse drought regimes.

Overall, the following general picture emerges: in the moister environments of Europe with ample precipitation supply, afforestation usually offers attractive ecological perspectives, such as better peak flow control combined with biodiversity support, and attractive economic returns, for instance from energy wood grown in short rotations. However, where water supply gets more limited, a situation of distinct trade-off may emerge very quickly: while afforestation may still provide desired ecological benefits such as soil protection and habitat provision, the enhanced water use will entail decreases in water yield, groundwater recharge rates, surface water flow and recharge of reservoirs. Careful consideration of the trade-off situation at catchment-, landscape - and regional scales has to be applied under such circumstances.

References

Agam N, Berliner PR (2006) Dew formation and water vapor adsorption in semi-arid environments. A review. J Arid Environ 65:572–590

Alcamo J, Moreno JM, Nováky B, Bindi M, Corobov R, Devoy RJN, Giannakopoulos C, Martin E, Olesen JE, Shvidenko A (2007) Europe. In: Parry ML, Canziani OF, Palutikof JP, van der Linden PJ, Hanson CE (eds) Climate change 2007: impacts, adaptation and vulnerability, Contribution of working group II to the fourth assessment report of the intergovernmental panel on climate change. Cambridge University Press, Cambridge

Allen A, Chapman D (2001) Impact of afforestation on groundwater resources and quality. Hydrogeol J 9:390–400

Anonymous (2001) Review of the effects of energy crops on hydrology, Final project report UK. http://www.defra.gov.uk/farm/crops/industrial/research/reports/nf0416.pdf

Ashby WC (1997) Soil ripping and herbicides enhance tree and shrub restoration on stripmines. Restor Ecol 5:169–177

Augusto L, Ranger J, Binkley D, Rothe A (2002) Impact of several common tree species of European temperate forests on soil fertility. Ann For Sci 59:233–253

Barbera GG, Martinez-Fernandez F, Alvarez-Rogel J, Albaladejo J, Castillo V (2005) Short- and intermediate-term effects of site and plant preparation techniques on reforestation of a Mediterranean semiarid ecosystem with Pinus halepensis Mill. New For 29:177–198

Berger TW, Inselsbacher E, Mutsch F, Pfeffer, M (2009) Nutrient cycling and soil leaching in eighteen pure and mixed stands of beech (Fagus sylvatica) and spruce (Picea abies). Forest Ecology and Management 258 (11):2578–2592

Binkley D, Sollins P, Bell R, Sachs D, Myrold D (1992) Biogeochemistry of adjacent conifer and alder-conifer stands. Ecology 73:2022–2033

Blondel J, Aronson J (1999) Biology and wildlife of the Mediterranean region. Oxford University Press, Oxford

Bren L, Hopmans P (2007) Paired catchments observations on the water yield of mature eucalypt and immature radiata pine plantations in Victoria, Australia. J Hydrol 336:416–429

Brisette JC, Barnett JP, Landis TD (1991) Container seedlings. In: Duryea ML, Dougherty PM (eds) Forest regeneration manual. Kluwer, Dordrecht, The Netherlands/Boston/London

Bruijnzeel LA (2004) Hydrological functions of tropical forests: not seeing the soil for the trees? Agric Ecosyst Environ 104:185–228

Bulgarian Ministry of Environment and Water (2008) Program concerning the necessary measures in the circumstances of trend to drought. http://www.moew.government.bg/index_e.html. Accessed 09 Sep 2010

Bungart R, Huttl RF (2001) Production of biomass for energy in post-mining landscapes and nutrient dynamics. Biomass Bioenergy 20:181–187

Burdett AN (1990) Physiological processes in plantation establishment and the development of specifications for forest planting stock. Can J For Res 20:415–427

Buytaert W, Iniguez V, De Bievre B (2007) The effects of afforestation and cultivation on water yield in the Andean paramo. For Ecol Manage 251:22–30

Calder IR (2007) Forests and water-ensuring forest benefits outweigh water costs. For Ecol Manage 251:110–120

Castro J, Zamora R, Hodar JA, Gomez JM (2002) The use of shrubs as nurse plants: a new technique for reforestation in Mediterranean mountains. Restor Ecol 10:297–305

Castro J, Zamora R, Hodar JA, Gomez JM, Gomez-Aparicio L (2004) Benefits of using shrubs as nurse plants for reforestation in Mediterranean mountains: a 4-year study. Restor Ecol 12:352–358

Chirino E, Vilagrosa A, Hernandez EI, Matos A, Vallejo VR (2008) Effects of a deep container on morpho-functional characteristics and root colonization in *Quercus suber* L. seedlings for reforestation in Mediterranean climate. For Ecol Manag 256:779–785

Cornish PM, Vertessy RA (2001) Forest age-induced changes in evapotranspiration and water yield in a eucalypt forest. J Hydrol 242:43–63

De Schrijver A, Geudens G, Augusto L, Staelens J, Mertens J, Wuyts K, Gielis L, Verheyen K (2007) The effect of forest type on throughfall deposition and seepage flux: a review. Oecologia 153:663–674

De Schrijver A, Staelens J, Wuyts K, Van Hoydonck G, Janssen N, Mertens J, Gielis L, Geudens G, Augusto L, Verheyen K (2008) Effect of vegetation type on throughfall deposition and seepage flux. Environ Pollut 153:295–303

Del Campo AD, Navarro RM, Aguilella A, Gonzalez E (2006) Effect of tree shelter design on water condensation and run-off and its potential benefit for reforestation establishment in semiarid climates. For Ecol Manag 235:107–115

Deutmeyer M (2007) Strategies for supply of raw material in the energy industry. FAO, Geneva, http://www.unece.org/trade/timber/workshops/2007/wmw/presentations/ 07_Choren.pdf

Dise NB, Rothwell JJ, Gauci V, van der Salm C, de Vries W (2009) Predicting dissolved inorganic nitrogen leaching in European forests using two independent databases. Sci Total Environ 407:1798–1808

Duan X, Neuman DS, Reiber JM, Green CD, Saxton AM, Auge RM (1996) Mycorrhizal influence onhydraulic and hormonal factors involvedinthe control of stomatal conductance during drought. J Exp Bot 47:1541–1550

Duryea ML (1984) Nursery cultural practices: impacts on seedling quality. In: Duryea ML, Landis TD (eds) Forest nursery manual: production of bareroot seedlings. Martinus Nijhoff/Dr. W. Junk Publishers, The Hague, Boston, Lancaster

Ellis TW, Leguedois S, Hairsine PB et al (2006) Capture of overland flow by a tree belt on a pastured hillslope in south-eastern Australia. Aust J Soil Res 44:117–125

Farley KA, Jobbagy EG, Jackson RB (2005) Effects of afforestation on water yield: a global synthesis with implications for policy. Glob Change Biol 11:1565–1576

Fischlin A, Midgley GF, Price JT, Leemans R, Gopal B, Turley C, Rounsevell MDA, Dube OP, Tarazona J, Velichko AA (2007) Ecosystems, their properties, goods, and services. In: Parry ML, Canziani OF, Palutikof JP, van der Linden PJ, Hanson CE (eds) Climate change 2007: impacts, adaptation and vulnerability, Contribution of working group II to the fourth assessment report of the intergovernmental panel on climate change. Cambridge University Press, Cambridge

Flemming RL, Black TA, Adams RS (1996) Site preparation effects on Douglas-fir and lodgepole pine water relations following planting in a pine-grass dominated clearcut. For Ecol Manag 83:47–60

Fuentesa D, Valdecantosa A, Cortina J, Vallejo VR (2007) Seedling performance in sewage sludge-amended degraded Mediterranean woodlands. Ecol Eng 31:281–291

Garcıa C, Hernandez T, Roldan A, Albaladejo J, Castillo V (2000) Organic amendment and mycorrhizal inoculation as a practice in afforestation of soils with *Pinus halepensis* Mill: effect on their microbial activity. Soil Biol Biochem 32:1173–1181

Girard S, Clement A, Cochard H, Boulet-Gercourt B, Guehl JM (1997) Effects of desiccation on post-planting stress in bareroot Corsican pine seedlings. Tree Physiol 17:429–435

Gomez-Aparicio L, Zamora R, Gomez JM, Hodar JA, Castro J, Baraza E (2004) Applying plant facilitation to forest restoration: a meta-analysis of the use of shrubs as nurse plants. Ecol Appl 14:1128–1138

Grossnickle SC (1988) Planting stress in newly planted jack pine and white spruce. 1. Factors influencing water uptake. Tree Physiol 4:71–83

Guehl JM, Aussenac G, Kaushal P (1989) The effects of transplanting stress on photosynthesis, stomatal conductance and leaf water potential in *Cedrus atlantica*: role of root regeneration. Ann Sci For 46S:464–468

Guehl JM, Clement A, Kaushal P, Aussenac G (1993) Planting stress, water status and non-structural carbohydrate concentrations in Corsican pine seedlings. Tree Physiol 12:173–183

Gundersen P, Schmidt IK, Raulund-Rasmussen K (2006) Leaching of nitrate from temperate forests – effects of air pollution and forest management. Environ Rev 14:1–57

Hansen K, Rosenqvist L, Vesterdal L et al (2007) Nitrate leaching from three afforestation chronosequences on former arable land in Denmark. Glob Change Biol 13:1250–1264

Harvey AE, Page-Dumroese DS, Jurgensen MF, Graham RT, Tonn JR (1996) Site preparation alters biomass, root and ectomycorrhizal development of outplanted western white pine and Douglas-fir. New For 11:255–270

Heitz R, Rehfuess KE (1999) Reconversion of Norway spruce (*Picea abies* L.) stands into mixed forests: effects on soil properties and nutrient fluxes. In: Olsthoorn AFM (ed) Management of mixed-species forest: silviculture and economics. Institute for Forestry and Nature Research, Wageningen

Horton JL, Hart SC (1998) Hydraulic lift: a potentially important ecosystem process. Trends Ecol Evol 13:232–235

Horváth B, Meiwes KJ, Meesenburg H (2009) Die Bedeutung von Baumart und Bestandesalter für die Nitrat-versickerung unter Wald in der Region Weser-Ems. Forstarchiv 80:35–41

Huttermann A, Zommorodi M, Reise K (1999) Addition of hydrogels to soil for prolonging the survival of *Pinus halepensis* seedlings subjected to drought. Soil Till Res 50:295–304

Insley H, Buckley GP (1985) The influence of desiccation and root pruning at the survival and growth of broadleaved seedlings. J Hortic Sci 60:377–387

Iversen TM, Grant R, Nielsen K (1998) Nitrogen enrichment of European inland and marine waters with special attention to Danish policy measures. Environ Pollut 102:771–780

Jackson RB, Jobbagy EG, Avissar R, Roy SB, Barrett DJ, Cook CW, Farley KA, le Maitre DC, McCarl BA, Murray BC (2005) Trading water for carbon with biological sequestration. Science 310:1944–1947

Jimenez MN, Fernandez E, Ondono E, Ripoll MA, Navarro FB, Gallego E, De Simon E, Lallena AM (2007) Influence of different post-planting treatments on the development in Holm oak afforestation. Trees 21:443–455

Johnson R (1998) The forest cycle and low river flows. A review of UK and international studies. For Ecol Manage 109:1–7

Kaushal P, Aussenac G (1989) Transplanting shock in Corsican Pine and Cedar of Atlas seedlings: internal water deficits, growth and root regeneration. For Ecol Manage 27:29–40

Kidron GJ (2005) Angle and aspect dependent dew and fog precipitation in the Negev desert. J Hydrol 301:66–74

Knur L (2008) Unpublished manuscript. http://www.dendrom.de/content/index_ 1.cfm?id_bereich = 51&id_nr=407&CFID=104718&CFTOKEN=19452341

Körner C, Sarris D, Christodoulakis D (2005) Long-term increase in climatic dryness in the East Mediterranean as evidenced for the island of Samos. Reg Environ Change 5:27–36

Kozlowski TT (1985) Tree growth in response to environmental stresses. J Arboric 11:97–111

Kozlowski TT, Davies WJ (1975) Control of water balance in transplanted trees. Arboriculture 1:1–10

Kozlowski TT, Kramer PJ, Pallardy SG (1991) The physiological ecology of woody plants. Academic, Toronto

Kristensen HL, Gundersen P, Callesen I, Reinds GJ (2004) Relationships between soil nitrate concentrations and environmental factors. Ecosystems 7:180–192

Kroiher F, Bielefeldt J, Bolte A, Schulter M (2008) Die Phytodiversität in Energieholzbestän¬den – erste Ergebnisse im Rahmen des Projekts NOVALIS. Arch Forstwesen Landschaftsökol 42:158–165

Kubiske ME, Abrams MD (1992) Photosynthesis, water relations, and leaf morphology of xeric versus mesic *Quercus rubra* ecotypes in central Pennsylvania in relation to moisture stress. Can J For Res 22:1402–1407

Ladekarl UL, Rasmussen KR, Christensen S, Jensen KH, Hansen B (2005) J Hydrol 300:76–99

Lamersdorf N, Bielefeldt J, Bolte A, Busch G, Dohrenbusch A, Kroiher F, Schulz U, Stoll B (2008) Das Projekt NOVALIS – zur naturverträglichen Produktion von Energieholz in der Landwirtschaft. Arch Forstwesen Landschaftsökol 42:138–141

Lane PNJ, Best AE, Hickel K et al (2005) The response of flow duration curves to afforestation. J Hydrol 310:253–265

Leyva MJ, Fernandez-Ales R (1998) Variability in seedlings water status during drought within a *Quercus ilex* subsp. *ballota* population, and its relation to seedling morphology. For Ecol Manage 111:147–156

Lindroth A, Bath A (1999) Assessment of regional willow coppice yield in Sweden on basis of water availability. For Ecol Manage 121:57–65

MacDonald JA, Dise NB, Matzner E, Armbruster M, Gundersen P, Forsius M (2002) Nitrogen input together with ecosystem nitrogen enrichment predict nitrate leaching from European forests. Glob Chang Biol 8:1028–1033

Maestre FT, Cortina J, Bautista S, Bellot J (2003) Does *Pinus halepensis* facilitate the establishment of shrubs in Mediterranean semi-arid afforestations? For Ecol Manage 176:147–160

Mannerkoski H, Finer L, Piirainen S et al (2005) Effect of clear-cutting and site preparation on the level and quality of groundwater in some headwater catchments in eastern Finland. For Ecol Manage 220:107–117

McKay HM (1997) A review of the effect of stresses between lifting and planting on nursery stock quality and performance. New For 13:369–399

McKay HM, White IMS (1997) Fine root electrolyte leakage and moisture content: indices of Sitka spruce and Douglas-fir seedling performance after desiccation. New For 13:139–162

Meuser A (1990) Effects of afforestation on run-off characteristics. Agric For Meteorol 50:125–138

Morte A, Diaz G, Rodrıguez P, Alarcon JJY, Sanchez-Blanco MJ (2001) Growth and water relations in mycorrhizal and nonmycorrhizal *Pinus halepensis* plants in response to drought. Biol Plant 44:263–267

Müller J, Beck W, Hornschuch F, Steiner A (2002) Quantifizierung der ökologischen Wirkungen aufwachsender Kiefern-Buchen-Mischbestände im nordeutschen Tiefland. Beiträge Forstwirtschaft Landschaftsökol 36:125–131

Nabuurs GJ, Masera O, Andrasko K, Benitez-Ponce P, Boer R, Dutschke M, Elsiddig E, Ford-Robertson J, Frumhoff P, Karjalainen T, Krankina O, Kurz WA, Matsumoto M, Oyhantcabal W, Ravindranath NH, Sanz Sanchez MJ, Zhang X (2007) Forestry. In: Metz B, Davidson OR, Bosch PR, Dave R, Meyer LA (eds) Climate change 2007: mitigation, Contribution of working group III to the fourth assessment report of the intergovernmental panel on climate change. Cambridge University Press, Cambridge

Navarro RM, Fragueiro B, Ceacero C, Del Campo A, De Prado R (2005) Establishment of *Quercus ilex* L. subsp. *ballota* [Desf.] Samp. using different weed control strategies in Southern Spain. Ecol Eng 25:332–342

Nedyalkov S, Raev I (1988) Hydrological role of forest ecosystems. Zemizdat, Sofia (in Bulgarian)

Nixon DJ, Stephens W, Tyrrel SF, Brierley EDR (2001) The potential for short rotation energy forestry on restored landfill caps. Bioresour Technol 77:237–245

Pattanayak SK, McCarl BA, Sommer AJ et al (2005) Water quality co-effects of greenhouse gas mitigation in US agriculture. Clim Change 71:341–372

Pugnaire FI, Haase P, Puigdefábregas J (1996) Facilitation between higher plant species in a semiarid environment. Ecology 77:1420–1426

Putuhena WM, Cordery I (2000) Some hydrological effects of changing forest cover from eucalypts to Pinus radiata. Agric For Meteorol 100:59–72

Querejeta JI, Roldan A, Albaladejo J, Castillo V (1998) The role of mycorrhizae, site preparation and organic amendment in the afforestation of a semi-arid Mediterranean site with *Pinus halepensis*. For Sci 44:203–211

Querejeta JI, Roldan A, Albadalejo J, Castillo V (2001) Soil water availability improved by site preparation in a *Pinus halepensis* afforestation under semiarid climate. For Ecol Manag 149:115–128

Radoglou K, Raftoyannis Y (2001) Effects of desiccation and freezing on vitality and field performance of broadleaved tree species. Ann For Sci 58:59–68

Radoglou K, Raftoyannis Y (2002) The impact of storage, desiccation and planting date on seedling quality and survival of woody plant species. Forestry 75:179–190

Raev I, Jelyazkov P (1980) Ecological effect of afforestation with conifers in the zone of deciduous forests. In: Marinov M (ed) Reproduction and preservation of the forest ecosystems. Sofia, Zemizdat, pp 129–146 (in Bulgarian)

Raftoyannis Y, Radoglou K, Halivopoulos G (2006) Ecophysiology and survival of *Acer pseudoplatanus* L. Castanea sativa Miller. and Quercus frainetto Ten. seedlings on a reforestation site in northern Greece. New For 31:151–163

Rey-Benayas JM (1998) Growth and mortality in *Quercus ilex* L. seedlings after irrigation and artificial shading in Mediterranean set-aside agricultural lands. Ann Sci For 55:801–807

Richards KR, Stokes C (2004) A review of forest carbon sequestration cost studies: a dozen years of research. Clim Change 63:1–48

Ritchie GA (1984) Assessing seedling quality. In: Duryea ML, Landis TD (eds) Forest nursery manual: production of bareroot seedlings. Martinus Nijhoff/Dr. W. Junk Publishers, The Hague/Boston/Lancaster

Roberts S, Vertessy R, Grayson R (2001) Transpiration from *Eucalyptus sieberi* (L. Johnson) forests of different age. For Ecol Manage 143:153–161

Robertson SMC, Hornung M, Kennedy VH (2000) Water chemistry of throughfall and soil water under four tree species at Gisburn, northwest England, before and after felling. For Ecol Manage 129:101–117

Robinson M, Cognard-Plancq AL, Cosandey C et al (2003) Studies of the impact of forests on peak flows and baseflows: a European perspective. For Ecol Manage 186:85–97

Roldan A, Querejeta I, Albaladejo J, Castillo V (1996) Survival and growth of *Pinus halepensis* Miller seedlings in a semiarid environment after forest soil transfer, terracing and organic amendments. Ann Sci For 53:1099–1112

Ross DS, Wemple BC, Jamison AE, Fredriksen G, Shanley JB, Lawrence GB, Bailey SW, Campbell JL (2009) A cross-site comparison of factors influencing soil nitrification rates in Northeastern USA forested watersheds. Ecosystems 12:158–178

Rothe A, Mellert KH (2004) Effects of forest management on nitrate concentrations in seepage water of forests in southern Bavaria, Germany. Water Air Soil Pollut 156:337–355

Rothe A, Huber C, Kreutzer K et al (2002) Deposition and soil leaching in stands of Norway spruce and European Beech: Results from the Hoglwald research in comparison with other European case studies. Plant Soil 240:33–45

Royo A, Gil L, Pardos JA (2001) Effect of water stress conditioning on morphology, physiology and field performance of *Pinus halepensis* Mill. seedlings. New For 21:127–140

RWE (2008) RWE Innogy creates fuel-wood plantations for energy. http://www.rwe.com/generator.aspx/konzern/verantwortung/aktuelles/ 2008/0328/language=en/id=623060/page.html. Accessed 09 Sep 2010

Savill P, Evans J, Auclair D, Falck J (1997) Plantation silviculture in Europe. Oxford University Press, Oxford

Schiller G, Cohen Y (1998) Water balance of *Pinus halepensis* Mill. afforestation in an arid region. For Ecol Manage 105:121–128

Schulz U, Brauner O, Grub H, Neuenfeldt N (2008) Vorläufige Aussagen zu Energieholzflä¬chen aus tierökologischer Sicht. Arch Forstwesen Landschaftsökol 42:83–87

Scott DF, Lesch W (1997) Streamflow responses to afforestation with *Eucalyptus grandis* and *Pinus patula* and to felling in the Mokobulaan experimental catchments, South Africa. J Hydrol 199:360–377

Seiler JR, Johnson JD (1988) Physiological and morphological responses of three half-sib families of loblolly pine to water stress conditioning. For Sci 34:487–495

Sidle RC, Ziegler AD, Negishi JN et al (2006) Erosion processes in steep terrain – truths, myths, and uncertainties related to forest management in Southeast Asia. For Ecol Manage 224:199–225

Tsakaldimi M, Zagas T, Tsitsoni T, Ganatsas P (2005) Root morphology, stem growth and field performance of seedlings of two Mediterranean evergreen oaks species raised in different container types. Plant Soil 278:85–93

v. Wilpert K (2008) Waldbauliche Steuerungsmöglichkeiten des Stoffhaushalts von Waldökosystemen am Beispiel von Buchen- und Fichtenvarianten der Conventwald-Fallstudie. Schriftenreihe Freiburger Forstl. Forschung 40

v. Wilpert K, Zirlewagen D, Kohler M (2000) To what extent can silviculture enhance sustainability of forest sites under the immission regime in Central Europe? Water Air Soil Pollut 122:105–120

Van den Driessche R (1991) Influence of container nursery regimes on drought resistance of seedlings following planting. I. Survival and growth. Can J For Res 21:555–565

Van der Salm C, Van der Gon HD, Wieggers R, Bleeker A, Van den Toorn A (2006) The effect of afforestation on water recharge and nitrogen leaching in The Netherlands. For Ecol Manage 221:170–182

Van Dijk AIJM, Keenan RJ (2007) Planted forests and water in perspective. For Ecol Manage 251:1–9

Van Dijk AUM, Hairsine PB, Pena Arancibia J et al (2007) Reforestation, water availability and stream salinity: a multi-scale analysis in the Murray-Darling Basin, Australia. For Ecol Manage 251:94–109

Varelides C, Kritikos T (1995) Effect of site preparation intensity and fertilization on *Pinus pinaster* survival and height growth on three sites in northern Greece. For Ecol Manage 73:111–115

Waterloo MJ, Schellekens J, Bruijnzeel LA et al (2007) Changes in catchment runoff after harvesting and burning of a *Pinus caribaea* plantation in Viti Levu, Fiji. For Ecol Manage 251:31–44

Wattenbach M, Zebisch M, Hattermann F, Gottschalk P, Goemann H, Kreins P, Badeck F, Lasch P, Suckow F, Wechsung F (2007) Hydrological impact assessment of afforestation and change in tree-species composition: a regional case study for the Federal State of Brandenburg (Germany). J Hydrolog 346:1–17

Webb DP, von Althen FW (1980) Storage of hardwood planting stock: effects of various storage regimes and packaging methods on root growth and physiological quality. N Z J For Sci 10:83–96

Weis W, Rotter V, Göttlein A (2006) Water and element fluxes during the regeneration of Norway spruce with European beech: effects of shelterwood-cut and clear-cut. For Ecol Manage 224:304–317

Williard KWJ, Dewalle DR, Edwards PJ (2005) Influence of bedrock geology and tree species composition on stream nitrate concentrations in Mid-Appalachian forested watersheds. Water Air Soil Pollut 160:55–76

World Resources Institute (2008) http://www.wri.org/. Accessed 10 Jan 2009

World Resources Institute (2008) Forest Landscapes Initiative. http://www.wri.org/project/global-forest-watch. Accessed 09 Sep 2010

Zhang L, Dawes WR, Walker GR (2001) Response of mean annual evapotranspiration to vegetation changes at catchment scale. Water Resour Res 37:701–708

Zhang L, Vertessy R, Walker G et al (2007) Afforestation in a catchment context. Understanding the impacts on water yield and salinity. Industry report 01/07. CSIRO Land and water science report number 01/07

Zlatanov T (2006) Perspectives for sustainable management of the forests in Lesnovska river basin. J Balkan Ecol 9:125–130

Zlatanov T, Hinkov G (2005) An ecological approach for transformation of Austrian pine (*Pinus nigra* Arn.) plantations on the south slopes of the Middle Balkan Range. Ecol Eng Environ Prot 4:58–62 (in Bulgarian)

Chapter 14
Tree Species' Tolerance to Water Stress, Salinity and Fire

Martin Lukac, Margus Pensa, and Gabriel Schiller

Introduction

Predicted scenarios of global climate change on the European continent and adjacent geographic regions by 2100 include a rise of average air temperature by about 2°C, shifting geographical and seasonal rainfall patterns, a concurrent intensification of rainstorms and heavy rain events (mm h^{-1}), as well as a rise of the sea level. These changes may profoundly affect forest vegetation by shifting species' climatic boundaries and hence by forcing substantial changes in species composition of forest communities as they adapt to the new environmental conditions. These conditions will probably include increased frequency and length of heat and drought spells, leading to subsequent rise of fire risk. At the same time, intensification of rainfall events may raise the probability of severe flooding of lowlands, heavy erosion and disruption of normal annual cycle in seasonally flooded forests. The rise of the sea level may severely affect coastal forest areas by exposure to salt spray during severe weather evens, infiltration of salt water into the underground water table and by the salt contamination of fresh-water coastal marshes. It is of importance to consider available climate change scenarios and to use the forewarning they provide to prepare measures to reduce the severity of their impact on forest ecosystems. One has to bear in mind that a forest planted today is likely to exist in

---

M. Lukac (✉)
School of Agriculture, Policy and Development, University of Reading,
Whiteknights, PO Box 237, Reading, RG6 6AR
e-mail: m.lukac@reading.ac.uk

M. Pensa
Institute of Ecology, Tallinn Universiy, Uus-Sadama 5, 10120, Tallinn, Estonia
e-mail: margus@ecoviro.johvi.ee

G. Shiller
Department of Agronomy and Natural Resources, Forestry Section, Agricultural
Research Organization, The Volcani Center, P. O. Box 6, Bet Dagan, 50250, Israel
e-mail: vcgabi@volcani.agri.gov.il

M. Bredemeier et al. (eds.), *Forest Management and the Water Cycle: An Ecosystem-Based Approach*, Ecological Studies 212, DOI 10.1007/978-90-481-9834-4_14,

a different climate when it reaches maturity. As discussed in Chapter 13, forest area in Europe is expanding due to afforestation of marginal land, often with allochtonous tree species. This process brings a host of interrelated problems and further highlights the need for considering the impacts of changing environment on European forests. Future conditions, as well as future requirements for forest ecosystem services, need to be taken into account when selecting tree species for afforestation. It is likely that future management practices will need to be adapted and improved to ensure forest sustainability. Therefore, we think it useful to explore the existing knowledge on forest tree species, whether indigenous to the Mediterranean basin or from other homologous climatic zones, which have evolved the necessary features to withstand the new climatic and environmental conditions.

Water Stress

Drought

Firstly, since there is no precise and universally accepted definition of drought, it is often difficult to ascertain whether a drought has occurred, what was its length and severity. It is considered to be a slow-onset natural hazard, often referred to as a 'creeping phenomenon' (Gilette 1950). In relation to forest stands, it is usually not a one-off event that causes severe damage, but a cumulative effect of several years' below average rainfall that induces dieback of resident tree species. Moreover, it is fair to assume that any definition of drought will be region specific, since different regions have vastly different rainfall amounts and rainfall distribution throughout the growing season. On this basis, a drought is, unlike aridity, an aberration from the norm and as such must also be distinguished from seasonal aridity (Wilhite and Buchanan-Smith 2005). According to Chapter 11 of the IPCC 2007 Assessment Report (Christensen et al. 2007), it is very likely that European territory will suffer geographical shifts and alterations of its precipitation patterns. It is forecasted that the southern half of Europe, including the Mediterranean area will suffer a significant reduction in rainfall, while northern half will see an increase in the amount of precipitation inside 2100 (Fig. 14.1).

However, the temporal pattern of precipitation has been predicted to change too, resulting in wet winters and dry springs and early summers in northern Europe (Fig. 14.2, Kont et al. (2003)). Although annual rainfall will increase in northern Europe, vegetation may still suffer from drought in summer at the height of the growth period. As a consequence of spring and early-summer drought, plants may reduce their leaf area, which in turn may cause decline in growth throughout the growing season (Nilsen et al. 1996). For example, R. Jalkanen and M. Pensa (2005) reconstructed leaf dry weight dynamics of Scots pine (*Pinus sylvestris*) in northern Lapland by using the well defined relationship between leaf size and leaf trace diameter (Pensa et al. 2004). The reconstruction indicates that the years with higher dry mass of a single leave tend to coincide with high precipitation in May and June (Fig. 14.3).

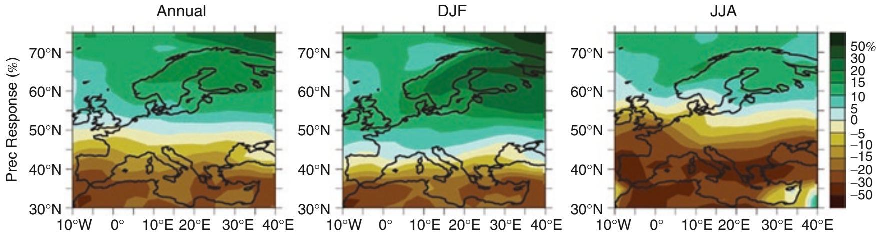


Fig. 14.1 Precipitation changes over Europe from the MMD-A1B simulations, DJF and JJA fractional change in precipitation between 1980–1999 and 2080–2099, averaged over 21 models (Christensen et al. 2007)

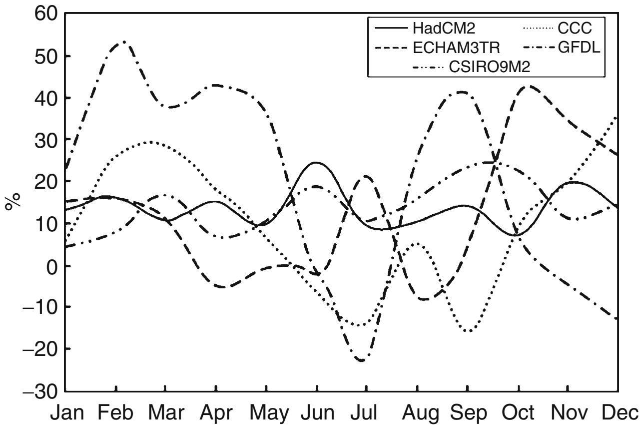


Fig. 14.2 Percentage change in monthly mean precipitation in Estonia for the year 2100 calculated by five alternative General Circulation Models; HadCM2, ECHAM3TR, CCC, GFDL and CSIRO9M2 (Kont et al. 2003)

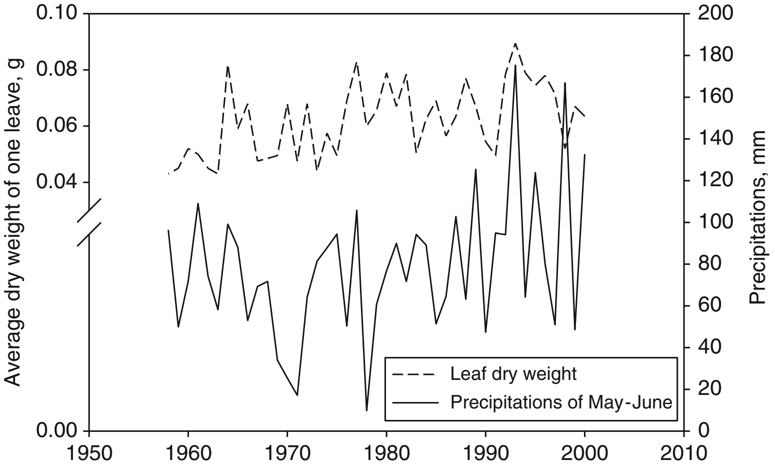


Fig. 14.3 Changes in average needle dry weight attached to the topmost shoot of Scots pine stem measured in Laanila, northern Lapland. The variation in the dry weight may indicate the effect of spring precipitation on leaf size (M. Pensa, 2005)

Since there is very little, or nothing, that a forest manager can do to prevent a drought from occurring and since the means of irrigation of forest stands are very limited or non-existent, a choice of tree species or provenances suitable for the forecasted climatic conditions appears to be the only viable course of action. One can utilize non-native tree species already adapted to dry conditions, or if this is not possible due to, for example, a ban on planting foreign stock, the most promising line of approach is a selection of drought resistant varieties of existing species, as suggested already by Maximov and Yapp (1929). Tree species choice is especially important in the case of large-scale afforestation programmes (see Chapter 12), where future impacts on water yields, soil protection and biodiversity conservation must be taken into account.

As with other stress factors, biotic or climatic, trees respond to drought stress by reducing the annual radial growth. Tree ring analysis can therefore be used as a tool for assessing whether tree species in a particular forest or area are suffering from drought stress (Pichler and Oberhuber 2007). Since the constraint water scarcity puts on carbon fixation is well established (Ciais et al. 2005), an analysis of annual growth rings during drought years can identify tree species which are sensitive to drought in a particular location. However, different tree species respond to climatic influences in varying fashion, it is important to possess a reliable database characterizing site conditions since similar climatological conditions yield divergent growth response patterns (Neuwirth et al. 2006).

Species coexisting in a given forest do not experience the same level of water stress, trees evolved several alternative strategies to cope with drought. For example, Mediterranean species have been classified as drought-tolerant, when they are able to maintain photosynthetic activity under severe water stress as certain evergreen oaks do, and drought avoiders, when they either shed leaves or close stomata during the dry periods, as certain pines do. Further, species composition of a forest stand directly influences its water balance. Trees influence the amount of water in the soil through crown interception and evapotranspiration. Schume and Hager (2004) have shown that evapotranspiration in a mixed Norway spruce (*Picea abies* Karst.) and European beech (*Fagus sylvatica* L.) stand was disproportionately greater than in single species stands. The increase in evapotranspiration in the mixed stand was exclusively attributable to beech, which in a mixture deepened and intensified its fine-root system, while spruce rooted more shallowly. During prolonged drought, beech could thus gain a competitive advantage since the mixed stand extracted a higher percentage of water from deeper soil layers than the pure stands.

In general, drought tolerance is associated with high water use efficiency (Ferrio et al. 2003), leading to the development of forest ecosystems able to persist under constant and significant lack of water. However, as Breda and Badeau (2008) point out in their analysis, extreme drought events surpassing the tolerance threshold can substantially increase tree species' probability to decline. This is valid both for arid environments with regular occurrence of drought and for localities not normally exposed to lack of water during the growing season. Kozlov and Niemelä (2003) have found that the vitality of Scots pine populations at the Arctic northern tree line is greatly affected by lack of late summer precipitation, even though water availability during the rest of the year is not a limiting factor. Tree vitality reduced by a drought event usually leads to a higher susceptibility to pests and is therefore very difficult to single out as the primary cause of forest decline. The issue is further complicated by the fact that trees affected by a severe drought do not decline immediately, sometimes taking decades to die (Pedersen 1998). Drought stress can change the development trajectory of a forest ecosystem in several ways, whether directly through different species' mortality, by changing the viability of reproductive structures and seeds (Ogaya and Penuelas 2007) or by reductions in biotic associations (e.g. avian seed dispersers, ectomycorrhizas and nurse plants; (Mueller et al. 2005)).

This far, drought resistance research has been leaf-centric, seeking to establish the mechanisms behind water loss reduction during transpiration. Drought tolerance is not achieved by a single combination of trait values and that these combinations can differ at different levels of analysis, i.e., in individual responses to drought versus interspecific comparisons of drought tolerance (Valladares and Sanchez-Gomez 2006). Recent developments in plant genetics, perhaps unsurprisingly, indicate that root biology does play a very significant role in drought tolerance and drought adaptation at the root level might have significant role to play when dealing with effects of climate change on trees (Pennisi 2008).

Alongside choosing suitable tree species tolerant of drought conditions in certain areas, forest managers have developed a host of afforestation strategies aimed at helping tree survival and at ensuring minimal impacts on water yields and water quality from afforested catchments (see Chapter 13 for more details).

Hypoxia

Moving across the European climate gradient from the very dry environment to the very wet, both the amount of rainfall and its intensity are forecasted to increase in the Central and Northern parts of Europe in the latter half of the twenty-first century. Together with increasing probability of substantial surface runoff and higher risk of soil erosion (see Chapter 15), this weather pattern is likely to bring an increased incidence of flooding in lowland areas. In addition, forecasted rise in sea level might result in severe and prolonged flooding of coastal areas by sea water. The increased risk of these events highlights the need to find and utilize tree species capable of withstanding long periods of flooding, especially in areas already considered flood-prone at present. Restricted gas exchange between the rhizosphere and aerated environment during a flooding event reduces the concentration of oxygen and elevates the concentration of carbon dioxide in the root zone. Air within soil pores is replaced by water and any remaining oxygen is quickly consumed by root and soil microbial respiration (Kreuzwieser et al. 2003), a process which gives rise to a condition commonly defined as hypoxia. This induces severe stress onto all aerobic organisms, including tree roots, since functions such as mitochondrial respiration, oxidation and oxygenation processes cannot take place (Vartapetian and Jackson 1997).

Tree responses to submersion vary among species, the most commonly observed reaction is a reduction in shoot growth probably accompanied by decline in root growth. Other responses can include injury, inhibition of seed germination, changes in vegetative and reproductive growth, changes in plant anatomy, and promotion of early senescence and mortality (Kozlowski 1997). There are several adaptations to submerging, riparian tree species frequently exposed to flooding such as willows, poplars and alders often feature some or all of them. For example, hypertrophied lenticels develop around the stems of *Salix* spp. and *Populus* spp., generally where the stomata once occurred. During flooding, lenticels provide a gas exchange pathway between the living cells in the bark and ambient air, substituting normal gas exchange within the soil. Aerenchyma tissues are extensive intercellular air spaces that form continuous passageways allowing the diffusion of oxygen from the aerial portions of the plants to the roots. Several species form adventitious roots after the death of the original roots due to flooding. These roots are produced as an extension of the surviving portion of the original root system and on the submerged portions of stems. These roots are generally negatively geotropic, thicker and possess larger intercellular spaces than original roots and can tolerate higher CO_2 concentrations and maintain respiration, despite the lack of oxygen. *Salix* spp., *Populus* spp., *Alnus glutinosa* and *Fraxinus excelsior* have all been reported to form adventitious roots during flooding, together with several other European tree and shrub species. In addition, there are thought to exist numerous metabolic adaptations to flooding, most of which still need to be investigated further (Armstrong et al. 1994; Drew 1997). Choosing tree species for a specific site should be accompanied by a tailored forest management plan. Trees have been shown to alter soil conditions in the rooting zone, both in the humus and in the mineral layers (see Chapter 16). Under suitable management, forest tree can contribute to the removal of excess water from the soil profile, thus contributing to soil amelioration.

In conditions of hypoxia, the ability of roots to supply shoots with water may decrease due to restricted hydraulic conductivity of roots (Sellin 2001). Consequently, as the atmospheric evapotranspirational demand (AED) increases, trees suffering from root hypoxia cannot compensate for transpirational water losses from shoots, which is reflected by lower shoot water potential of trees in waterlogged sites as compared with the trees on drier sites. For example, A. Sellin and M. Pensa (1997) compared the shoot water potential of Norway spruce trees growing on ombrotrophic peat with that of spruce trees growing on free-draining soils in Estonia. They found that spruces in the ombrotrophic peatland had lower values of shoot water potential at the same AED than the trees on the non-flooded soil (Fig. 14.4). This effect leads to lower leaf conductance and consequently to lower rate of photosynthesis as shown by Sellin (2001) in Norway spruce. Paradoxically, this proves that hypoxia caused by excess of water at the root level may lead to a water deficit at the shoot level.

Recently, there are indications of a shift in the approach to flood protection of inhabited areas – away from constricting the flood water in the present river bed and towards creating water overflow and retention zones often located in areas of lowland forests (Buijse et al. 2002). If the periods of flooding are relatively short

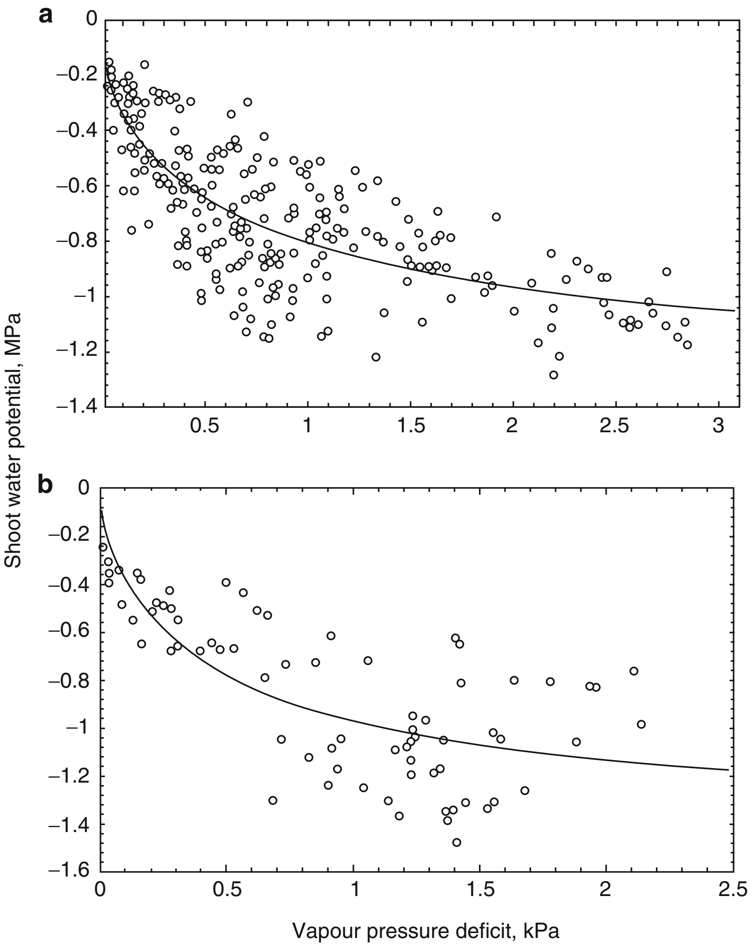


Fig. 14.4 The comparison of Norway spruce shoot water potential response to atmospheric evapotranspirational demand (measured by vapour pressure deficit) in two contrasting habitat in central Estonia. (**a**) Trees on fresh soils; (**b**) Trees on waterlogged ombrotrophic peatland. (M. Pensa, 1997)

and happen predominantly during winter or early spring when the trees are dormant, the risk of damage to the trees should be relatively low. However, prolonged and frequent flooding during the growing season can cause severe damage to the tree stands, especially if unsuitable species used to afforest such areas (Fig. 14.5).

Increased precipitation, together with rising winter temperatures, in many areas in northern Europe present an increasing risk of peatland encroachment, with adverse consequences for forest growth and regeneration (Crawford et al. 2003). Together with the ground water level, water nutrient and oxygen concentrations determine the impact of bog environment on trees. For example, the measurements of tree basal area in an Estonian peatland fed by precipitation only

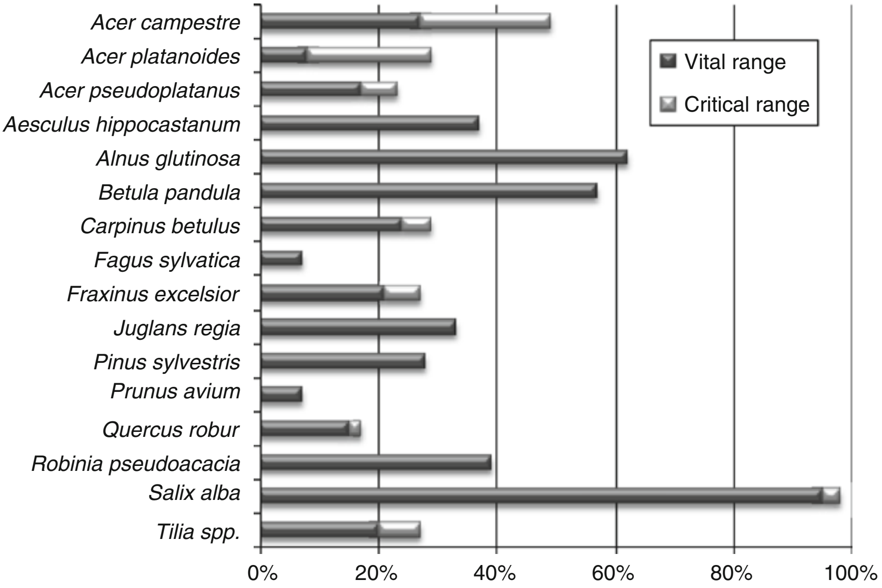


Fig. 14.5 Proportion of flooding during the growing season without negative impact on tree vitality (vital range) and with observed tree mortality (critical range) (Adapted from Glenz et al. 2006, data from Späth 2002)

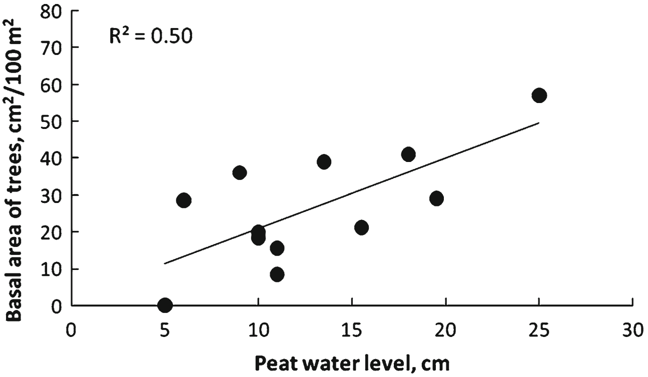


Fig. 14.6 Relationship between the depth of peat water table and the basal area of Scots pine (*Pinus sylvestris*) trees in a raised bog (Selisoo, Estonia). The trees were measured within plots of 100 m^2 and the level of peat water was recorded in tree replicates (M. Pensa, 2008)

(hence lacking in nutrients; ombrotrophic bog) indicated that the water table depth determines Scots pine (*Pinus sylvestris*) growth rates (Fig. 14.6). Scots pine is among the few tree species that grow in nutrient-poor peatlands, as well as on nutrient rich soils, and shows corresponding variability in its morphological traits (Pensa et al. 2007).

Ultimately, forecasting of the impact of flooding on tree species, success of floodplain forest restoration projects, planning of retention basins and even on the estimates of economic losses in forestry due to flooding will only be improved through better knowledge of the flooding stress response of individual tree and shrub species. More systematic investigation is called for in this area, starting with physiological responses and up to community interactions and forest survival in stands exposed to flooding. (Glenz et al. 2006).

Salinity

Soil salinity has several origins: intrinsic salt content of the soils' parent material, redistribution of salt by water flow from higher ground into the valleys that lack natural drainage and, most commonly, high groundwater tables which enable capillary rise. In coastal areas, soil salinity can also be the result of sea water spray by high winds or sea water penetration into underground aquifers due to sea level rise or excess fresh water extraction. In arid areas, soil salinity is mainly the result of high evaporation rate, a process during which an upward movement of soil water leads to salt deposits on or close to the surface. Global climate change scenarios predicted for the Mediterranean region include rising temperature and decline of rainfall, combined with rising sea levels. All these factors are likely to increase the area affected by soil salinity.

Apart from the Mediterranean, areas affected by salinity are found in many arid regions of the world and are thought to cover over 397 million hectares worldwide, which is an area three times larger than that used for agriculture (FAO 2000, www.fao.org). Saline soil is an area where soils contain high concentrations of various mineral salts in both as cations (Na^+, Ca^{2+}, Mg^{2+}, and K^+) and anions (Cl^-, SO_4^{2-}, HCO_3^-, CO_3^{2-}, and NO_3^-; Tanji,(2002)). A common response of trees to salt toxicity is a reduction in productivity. Several mechanisms explaining negative effects of excess soil salt content on plant physiology have been proposed, such as decrease in plant water potential resulting in alteration of water use efficiency (Glenn and Brown 1998), or disturbance in ion homeostasis and toxicity (Monteverdi et al. 2008). The salinity tolerance varies widely both among and within species, the drop in productivity is especially dramatic in not-salt tolerant species when exposed even to minor increases of soil salinity (Brugnoli and Lauteri 1991). Generally, plants adapted to saline conditions are characterised by higher water use efficiency (WUE, unit of biomass produced per unit of water utilised) than salt sensitive plants. Monteverdi et al. (2008) suggest that high WUE could be looked at as one of desirable traits in trees to be planted in areas affected by salinity. High WUE implies that fewer salts are taken up by the plant to produce a unit biomass, which subsequently reduces the energy costs of compartmentation and extrusion of salts.

Reforestation is one of the most practical and effective strategies to solve the problem of soil salinity. Trees phyto-remediate the soil by lowering saline water table, utilising underground water and by decreasing the rate of water evaporation from soil surface (Barrett-Lennard 2002). However, a lack of information or, perhaps, a lack of salt-tolerant tree species is an important barrier to wide application of afforestation. Screening for salt tolerant genotypes has been investigated in many woody plant species such as Thai neem (*Azadirachta siamensis* Val.), olive (*Olea europaea* L.), eucalyptus (*Eucalyptus* sp.), acacia (*Accacia* sp.), pine (*Pinus* sp.), toothbrush tree (*Salvadora persica* L.) and mulberry (*Morus* sp.) (Cha-um and Kirdmanee 2008). The quantification of salt tolerance is not straightforward as it is confounded by several other factors related to both site and tree species. Aspects such as soil fertility, soil physical conditions, distribution of salt in the soil profile, climate, stage of tree development, variety, and rootstock are all thought to affect salt tolerance (Kozlowski 1997).

A substantial amount of research still needs to be carried out before salt tolerant tree genotypes can be reliably used on saline soils. To date, most of the available knowledge has been acquired through field trials of various species and their varieties and hybrids. For example, *Populus euphratica* Oliv. has long been recognised as one of the most salt-tolerant poplar species (Chen et al. 2002), together with *Populus deltoides* Marshall (Singh 1998) or *Populus alba* L. (Imada et al. 2009). In Australia, *Eucalyptus camaldulensis* has been hybridised with *Eucalyptus grandis* and *Eucalyptus globulus* and subjected to field trials to combine the salt tolerance of the former with growth rate, stem form and wood properties of the latter two species (Dale and Dieters 2007). Similarly, several varieties of olive (*Olea europaea* L.), a major crop tree grown almost exclusively in the Mediterranean appear to be salt tolerant and merit further study, especially to assess long-term acclimation to salt stress (Chartzoulakis 2005).

At present, advances in plant genomics have presented a new opportunity to assess salt tolerance in trees. The genes responsible for the variability of salt tolerance could be valuable resources in breeding programs, but they are often difficult to identify since the responses identified in salt-stressed plants are often common to other stresses. Here, comparative genomics based on the comparison of genotypes differing in pheno-typical behaviour looks like a promising approach to identify genes that control the heritable genetic variation of salt tolerance (Beritognolo et al. 2008).

Fire

There is no species or vegetation that can withstand a furious fire. However, since not all fires are lethal, plant species have developed various adaptations and strategies to tolerate and to survive fire, both at individual and population level. There is therefore a need to distinguish between the survival of trees at the time or immediately after the fire event and the survival of the forest community over a longer time frame. Forest fires resulting from volcanic activity or due to meteorological phenomena have always been one of the main natural forces significantly influencing tree species morphological and anatomical traits, regeneration modes and life cycles (Bond and Keeley 2005). Not all fire events are the same, the frequency, intensity, duration and behaviour of each forest fire is governed by an interplay of several factors, such as climate (e.g. air temperature, relative humidity, wind velocity), fuel load, topography (e.g. slope incline) and several others. The heat load imposed on trees during a fire event is the main stress factor determining the level of disturbance and the survival of trees.

In addition to the characteristics of the fire itself, several anatomical, morphological and life cycle features contribute to tree survival. Persisting fire individually or at the population level, trees have evolved several strategies which are not mutually exclusive. These could be grouped into the following three categories:

1. *Living tissue protection and enhancing survival during the fire*, such as bark thickening (e.g., *Quercus suber* L.), reduced bark heat conductance (Vines 1968), trunk circumference and height growth rate, sapwood to heartwood area ratios (Gignoux et al. 1997; Wilson and Witkowski 2003; Nefabas and Gambiza 2007) or underground storage organs and contractile roots (Bond and van Wilgen 1996)
2. *Stimulating regrowth after the fire*, such as epicormic resprouting from pre-formed preventitious buds hidden within bark (e.g. *Pinus canariensis* Sweet ex *Spreng*), enhanced flowering, drought tolerance and short leaf life-span
3. *Post-fire seed dispersal and propagule persistence*, such as cone serotiny and heat or smoke stimulated germination

Perhaps surprisingly, the co-occurrence of these persistence features differs among the Mediterranean zones in the Northern Hemisphere, i.e. between California and the Mediterranean basin (Pausas et al. 2006). Propagule persistence and resprouting are negatively correlated in the Mediterranean Flora, hence they probably evolved under a parallel selection scenario. Resprouting capability is thought to have evolved in the Tertiary as the response to frequent fire disturbances (Pausas and Verdu 2005). Having said that, there are tree species which possess features from all three categories of fire resistance. For example *Pinus canariensis* Sweet ex Spreng, a pine endemic to the Canary Islands, is thus protected with serotinous cones, has sprouting ability and a thick bark (Climent et al. 2004).

Sprouting ability is common both in Angiosperms and Gymnosperms at the seedling stage and with maturation that ability declines. Within the Gymnosperms, only very few species such as *Pinus canariensis* Sweet ex Spreng or *Sequoia sempervirens* D. Don do not lose this feature. Commonly, sprouting ability is altered with age, plant size and severity of injury. Various trees display a complexity of resprouting behaviour; there are root collar sprouters; lignotubers sprouters (e.g., *Eucalyptus* sp.).; sprouters from roots and from layered branches (Bond and Midgley 2003).

Seeding after fire means the production and development of a vegetative body (cone or fruit) that has the capability to protect the seeds from fire (serotiny), thus implying late liberation of seeds into the post-fire ecological conditions (bradychory). In the Northern hemisphere this feature has been developed mainly in Gymnosperms in the *Pinus* genus, whereas in the southern hemisphere, this feature has been developed in closed-fruited angiosperms. Since none of the pines native to the Mediterranean basin are able to resprout, their post-fire regeneration depends on their seedbank. The most common lowland Mediterranean pines, such as *P. halepensis* Mill., *P. brutia* Ten. And *P. pinaster* Ait, thus rely on the canopy seed bank being protected in the serotinous cones (Pausas et al. 2004).

Apart from volcanic activity, during the Holocene there have been no phenomena, other than dry lightning storms, that can act as an ignition source in the Mediterranean basin and Europe. However, since mankind acquired fire as a tool several hundred thousand years ago (James 1989), and with the world-wide spread of human population, purposefully set fires, runaway fires, arson fires, fires due to negligence and many other reasons, forest fires became frequent to such a degree that fire became a significant ecological factor in environments not naturally prone to fire. Tree populations and forest ecosystems adapted to a particular fire regime can offer us a reference point and act as a source of fire-adapted provenances. Any information originating from these localities, at ecological, managerial or genetic levels, is likely to be used to plant and maintain more fire resistant forests.

Forests communities' species composition might change profoundly with time as a result of climate change, but will also be influenced by local topography. South facing slopes will become dryer and hotter than north facing slopes. Hence, vulnerability to fire will differ among topographic orientations. Current Global Circulation Models predict a rise in temperatures and drought occurrence, leading to the development of climatic conditions even more enhancing the risk of fire in today's Mediterranean basin and wider geographic regions in the south and south-east Europe. The eventual new climatic conditions will affect the forest vegetation both in the short and in the long term as the result of (a) weakening and dieback of species genetically not adapted to drought and/or higher temperatures, (b) resulting changes in the species composition of today's forest communities by shifting species' distribution area boundaries in relation to the new environmental conditions, and (c) changing soil properties resulting from severe and frequent fire events may affect belowground functionality and nutrient cycle (Neary et al. 1999).

Taking into account future scenarios of forest fires and increasing forest vulnerability to fire, it is of importance to take preparatory measures and include fire prevention in forest management plans. If the predictions related to climate change are correct, it will be increasingly useful to explore the existing knowledge on fire resistance in forest tree species, whether indigenous to the Mediterranean Basin or of other homologous climatic zones in the world. The utilisation of tree species which have developed the necessary features to withstand, slow down or even prevent the spread of fire (e.g., *Cupressus sempervirens* L. var *pyramidalis* (Italian cypress); *Tamarix* sp.) not normally used in forestry is likely to become the norm over an expanding area not just in the Mediterranean. Hand in hand with species selection, it will also be necessary to adapt forest management techniques to cope with fire and its damages to the forest (see Chapter 17).

Conclusion

As illustrated in this chapter, stress factors related to water scarcity, soil salinity and forest fires are ever present natural phenomena. Tree species have evolved a variety of physiological, structural and life-cycle mechanisms to deal with these factors. To deal with adverse impacts of climate change on forests, we need to design management and policy approaches which would make use of this bank of adaptations to ensure future stability and sustainability of European forests. For example, presented findings on fire and water stress resistance in trees may help to make optimum choices with regards to tree species choice in areas prone to soil erosion or intended as water overflow and retention zones during periods of flooding.

References

Armstrong W, Brandle R, Jackson MB (1994) Mechanisms of flood tolerance in plants. Acta Botanica Neerl 43:307–358

Barrett-Lennard EG (2002) Restoration of saline land through revegetation. Agric Water Manage 53:213–226

Beritognolo I, Sabatti M, Brosché M et al (2008) Functional genomics to discover genes for salt tolerance in annual and perennial plants. In: Abdelly C, Öztürk M, Ashraf M et al (eds) Biosaline agriculture and high salinity tolerance. Birkhäuser Verlag, Switzerland

Bond WJ, van Wilgen BW (1996) Fire and plants. Chapman & Hall, London

Bond WJ, Midgley JJ (2003) The evolutionary ecology of sprouting in woody plants. Int J Plant Sci 164:S103–S114

Bond WJ, Keeley JE (2005) Fire as a global 'herbivore': the ecology and evolution of flammable ecosystems. Trends Ecol Evol 20:387–394

Breda N, Badeau V (2008) Forest tree responses to extreme drought and some biotic events: Towards a selection according to hazard tolerance? CR Geosci 340:651–662

Brugnoli E, Lauteri M (1991) Effects of salinity on stomatal conductance, photosynthetic capacity, and carbon isotope discrimination of salt-tolerant (*Gossypium hirsutum L.*) and salt-sensitive *(Phaseolus-vulgaris L)* C3 non-halophytes. Plant Physiol 95:628–635

Buijse AD, Coops H, Staras M et al (2002) Restoration strategies for river floodplains along large lowland rivers in Europe. Freshw Biol 47:889–907

Cha-um S, Kirdmanee C (2008) Assessment of salt tolerance in eucalyptus, rain tree and thai neem under laboratory and the field conditions. Pak J Bot 40:2041–2051

Chartzoulakis KS (2005) Salinity and olive: growth, salt tolerance, photosynthesis and yield. Agric Water Manage 78:108–121

Chen SL, Lia JK, Fritz E et al (2002) Sodium and chloride distribution in roots and transport in three poplar genotypes under increasing NaCl stress. Forest Ecol Manag 168:217–230

Christensen JH, Hewitson B, Busuioc A et al (2007) Regional climate projections. In: Solomon SD, Qin D, Manning M et al (eds) Climate change 2007: the physical science basis. Contribution of working group I to the fourth assessment report of the intergovernmental panel on climate-change. Cambridge University Press, Cambridge, United Kingdom and New York, NY

Ciais P, Reichstein M, Viovy N et al (2005) Europe-wide reduction in primary productivity caused by the heat and drought in 2003. Nature 437:529–533

Climent J, Tapias R, Pardos JA et al (2004) Fire adaptations in the Canary Islands pine (Pinus canariensis). Plant Ecol 171:185–196

Crawford RMM, Jeffree CE, Rees WG (2003) Paludification and forest retreat in Northern Oceanic Environments. Ann Bot 91:213–226

Dale G, Dieters M (2007) Economic returns from environmental problems: breeding salt- and drought-tolerant eucalypts for salinity abatement and commercial forestry. Ecol Eng 31:175–182

Drew MC (1997) Oxygen deficiency and root metabolism: injury and acclimation under hypoxia and anoxia. Ann Rev Plant Physiol Plant Mol Biol 48:223–250

FAO (2000) The state of food and agriculture 2000: Lessons from the past 50 years. Agricultural and Developmental Economics Working Papers 32. UN FAO Economic and Social Development Department

Ferrio JP, Florit A, Vega A et al (2003) Delta C-13 and tree-ring width reflect different drought responses in Quercus ilex and Pinus halepensis. Oecologia 137:512–518

Gignoux J, Clobert J, Menaut JC (1997) Alternative fire resistance strategies in savanna trees. Oecologia 110:576–583

Gilette HP (1950) A creeping drought under way. Water Sewage Works 97:104–105

Glenn EP, Brown JJ (1998) Effects of soil salt levels on the growth and water use efficiency of Atriplex canescens (Chenopodiaceae) varieties in drying soil. Am J Bot 85:10–16

Glenz C, Schlaepfer R, Iorgulescu I et al (2006) Flooding tolerance of Central European tree and shrub species. Forest Ecol Manag 235:1–13

Imada S, Yamanaka N, Tamai S (2009) Effects of salinity on the growth, Na partitioning, and Na dynamics of a salt-tolerant tree, *Populus alba* L. J Arid Environ 73:245–251

James SR (1989) Hominid use of fire in the lower and middle pleistocene: a review of the evidence. Curr Anthropol 30:1–26

Kont A, Jaagus J, Aunap R (2003) Climate change scenarios and the effect of sea-level rise for Estonia. Global Planet Change 36:1–15

Kozlov MV, Niemelä P (2003) Drought is more stressful for northern populations of Scots pine than low summer temperatures. Silva Fenn 37:175–180

Kozlowski TT (1997). Responses of woody plants to flooding and salinity. Tree Physiology Monographs 1

Kreuzwieser J, Buchholz J, Rennenberg H (2003) Emission of methane and nitrous oxide by Australian mangrove ecosystems. Plant Biol 5:423–431

Maximov NA, Yapp RH (1929) The plant in relation to water; a study of the physiological basis of drought resistance. G. Allen & Unwin Ltd., London

Monteverdi CM, Lauteri M, Valentini R (2008) Biodiversity of plant species and adaptation to drought ans salt conditions. Selection of species for sustainable reforestation activity to combat desertification. In: Abdelly C, Öztürk M, Ashraf M et al (eds) Functional genomics to discover genes for salt tolerance in annual and perennial plants. Birkhäuser Verlag, Switzerland

Mueller RC, Scudder CM, Porter ME et al (2005) Differential tree mortality in response to severe drought: evidence for long-term vegetation shifts. J Ecol 93:1085–1093

Neary DG, Klopatek CC, DeBano LF et al (1999) Fire effects on belowground sustainability: a review and synthesis. Forest Ecol Manag 122:51–71

Nefabas LL, Gambiza J (2007) Fire-tolerance mechanisms of common woody plant species in a semiarid savanna in south-western Zimbabwe. Afr J Ecol 45:550–556

Neuwirth B, Schweingruberb FH, Winigera M (2006) Spatial patterns of central European pointer years from 1901 to 1971. Dendrochronologia 24:79–89

Nilsen ET, Orcutt DM, Hale MG (1996) The physiology of plants under stress. Wiley, New York

Ogaya R, Penuelas J (2007) Species-specific drought effects on flower and fruit production in a Mediterranean holm oak forest. Forestry 80:351–357

Pausas JG, Blade C, Valdecantos A et al (2004) Pines and oaks in the restoration of Mediterranean landscapes of Spain: new perspectives for an old practice – a review. Plant Ecol 171:209–220

Pausas JG, Verdu M (2005) Plant persistence traits in fire-prone ecosystems of the Mediterranean basin: a phylogenetic approach. Oikos 109:196–202

Pausas JG, Keeley JE, Verdu M (2006) Inferring differential evolutionary processes of plant persistence traits in Northern Hemisphere Mediterranean fire-prone ecosystems. J Ecol 94:31–39

Pedersen BS (1998) The role of stress in the mortality of midwestern oaks as indicated by growth prior to death. Ecology 79:79–93

Pennisi E (2008) Plant genetics: getting to the root of drought responses. Science 320:173–173

Pensa M, Aalto T, Jalkanen R (2004) Variation in needle-trace diameter in respect of needle morphology in five conifer species. Trees-Struct Funct 18:307–311

Pensa M, Jalkanen R, Liblik V (2007) Variation in Scots pine needle longevity and nutrient conservation in different habitats and latitudes. Can J Forest Res-Revue Canadienne De Recherche Forestiere 37:1599–1604

Pichler P, Oberhuber W (2007) Radial growth response of coniferous forest trees in an inner Alpine environment to heat-wave in 2003. Forest Ecol Manag 242:688–699

Schume HG, Hager H (2004) Soil water depletion and recharge patterns in mixed and pure forest stands of European beech and Norway spruce. J Hydrol 289:258–274

Sellin A (2001) Hydraulic and stomatal adjustment of Norway spruce trees to environmental stress. Tree Physiol 21:879–888

Singh B (1998) Biomass production and nutrient dynamics in three clones of Populus deltoides planted on Indogangetic plains. Plant Soil 203:15–26

Späth V (2002) Hochwassertoleranz von Waldba¨umen in der Rheinaue. AFZ Der Wald 15:807–810

Tanji KK (2002) Salinity in the soil environment. In: Läuchli A, Lüttge U (eds) Salinity: environment-plant-molecules. Kluwer, Dordrecht

Valladares F, Sanchez-Gomez D (2006) Ecophysiological traits associated with drought in Mediterranean tree seedlings: individual responses versus interspecific trends in eleven species. Plant Biol 8:688–697

Vartapetian BB, Jackson MB (1997) Plant adaptations to anaerobic stress. Ann Bot 79:3–20

Vines RG (1968) Heat transfer through bark, and the resistance of trees to fire. Aust J Bot 16:499–514

Wilhite DA, Buchanan-Smith M (2005) Drought as hazard: understanding the natural and social context. In: Wilhite DA (ed) Drought and water crises: science, technology, and management issues. CRC Press, Boca Raton, FL

Wilson BG, Witkowski ETF (2003) Seed banks, bark thickness and change in age and size structure (1978–1999) of the African savanna tree, Burkea africana. Plant Ecol 167:151–162

Chapter 15
Overland Flow, Soil Erosion and Stream Water Quality in Forest Under Different Perturbations and Climate Conditions

Meni Ben-Hur, Cristina Fernandez, Sakari Sarkkola,
and Juan Carlos Santamarta Cerezal

Introduction

Surface runoff and soil erosion are, in general, natural phenomena associated with hydrological, geological, and ecological processes. Such surface runoff and soil loss could confer environmental benefits by supplying water and nutrients to the streams (Andreassian 2004; Bruijnzeel 2004), but, on the other hand, sometimes these runoff and soil erosion processes could cause widespread land and water deterioration problems, in terms of water loss, soil degradation, transporting of pollutants, and environmental hazards (Ben-Hur and Agassi 1997; Jones et al. 1998; Jackson et al. 2005; Calder 2007; Van Dijk and Keenan 2007). For example, surface runoff could lead to downstream flooding, which, in turn, could destroy or damage infrastructures and threaten humans and animals (Waterloopkundig Laboratorium 1994; Mendel 1996). The annual cost of flood damage in Europe over the 5 years from 1995 until 1999, was 2000 million euro in average (Swiss Re, various dates), and it was shown that although flood damage costs varied widely from year to year, they were steadily increasing over time (Munich Re 2000).

M. Ben-Hur (✉)
Institute of Soil, Water and Environmental Sciences, ARO, Volcani Center,
Bet-Dagan 50-250, Israel
e-mail: meni@volcani.agri.gov.il

C. Fernandez
Centro de Investigaciones Forestales y Ambientales de Lourizan, Conselleria de Medio
Ambiente, Xunta de Galicia, 36080, Spain
e-mail: Cfernandez.cifal@siam-cma.org

S. Sarkkola
Finish Forest Research Institute, Vantaa Research Center, FI-01301 Vanta, Finnland
e-mail: sakari.sarkkola@metla.fi

J.C.S. Cerezal
Universidad de La laguna, Escuela Técnica Superior de Ingeniería Civil e Industrial,
Avda. Astrofísico Francisco Sánchez s/n, La Laguna, Canary Islands, Spain
e-mail: jcsanta@ull.es

M. Bredemeier et al. (eds.), *Forest Management and the Water Cycle: An Ecosystem-Based Approach*, Ecological Studies 212, DOI 10.1007/978-90-481-9834-4_15,

In addition, some pollutants, such as toxic heavy metals, nutrients and organic pollutants, could be transported with the surface runoff and eroded sediments, and could pollute good water resources and the ecosystem (Allen and Chapman 2001).

Runoff and soil erosion are affected by various factors, such as landscape, land use, and soil and bedrock properties. The major land use in the European Union is forestry, which accounts for ~37% of the total land area (FAO 2001), of which ~80% comprises exploitable or managed forests. Thus, most of the forest lands in Europe are exposed to intensive perturbations, such as clear cutting, land drainage, site preparation, coppice sprout thinning, and attacks by beetles. It is well known that forest cover significantly affects the hydrology of the watershed, and especially the runoff and soil erosion (Rothacher 1970; Troendle 1983; Hornbeck et al. 1993; McCulloch and Robinson 1993; Zhang et al. 2001; Farley et al. 2005; Sidle et al. 2006). These effects, however, are strongly related to the climatic conditions (Farley et al. 2005; Jackson et al. 2005). With respect to the water issue, watersheds can be divided into two main types: (1) those under water-limited conditions; and (2) those under non-water-limited conditions. This division can be applied by reference to the "aridity index" (AI), which is defined by Eq. (15.1) (Thornthwaite 1948).

$$AI = \frac{P}{ET_p} \tag{15.1}$$

where, P = annual precipitation, in millimeters; and ET_p = annual potential evapotranspiration, in millimeters. According to this index, watersheds with $AI \geq 1$ are considered to be watersheds under non-water-limited conditions, and those with $AI < 1$ are considered as being under water-limited conditions.

Recently, the effects of forest cover on the hydrological processes of watersheds have been studied intensively, and reviewed in a number of papers (e.g., Bosch and Hewlett 1982; Johnson 1998; Allen and Chapman 2001; Robinson et al. 2003; Calder 2007; Van Dijk and Keenan 2007). These studies, however, concentrated mainly on forest sustainability and management, water use by various tree species, and their negative impacts on water resources (e.g., Calder 1990; Kirby et al. 1991; McCulloch and Robinson 1993; Cosandey et al. 2002). Far fewer studies have addressed the perturbing interactions between forest stand, on the one hand, and runoff and soil erosion processes under various climatic conditions, on the other hand. Chapters 13 and 14 in the this book discussed the afforestation strategies based on a proper choice of tree species according to their water use and tolerant to fire, which can affect the hydrological processes of watersheds including the runoff and soil erosion.

The overall objective of this chapter is to describe and discuss the relationships between forest and climate conditions, and their effects on runoff, soil erosion and stream water quality, when subjected to various perturbations. The discussion addresses only water-driven soil erosion; not landslide. Three different climate types are discussed in this chapter: (1) a Mediterranean, semi-arid climate with annual precipitation of ~500 mm; (2) a semi-humid northern climate with annual precipitation of ~600 mm and freezing temperatures and snow conditions in the winter; and (3) a mild, humid climate with average annual rainfall of ~1,800 mm and no freezing temperatures or snow conditions.

Mechanisms and Processes of Overland Flow and Soil Erosion

Runoff and soil erosion processes on forest-cover watersheds depend on the type and kind of vegetation, and the physical and chemical characteristics of the catchment terrain. The forest terrain could be formed of deep (>1.5 m, below the main root zone depth of the trees) or shallow (<1.5 m) soil that covers the bedrock, or bedrock exposed to the atmosphere. The bedrock could have a very low permeability or it could be full of extensive cracks through which the water can flow freely. All of these parameters can significantly affect the hydrology and the soil erosion processes in the forest.

Runoff and Overland Flow Processes

When precipitation (rainfall and snow) reaches the forest terrain, a certain amount of the water penetrates the terrain surface and is absorbed into successively deeper layers of the soil and bedrock profile. Some of this water recharges the groundwater reservoir and raises the water table level; some of it, when it reaches an impermeable layer, flows laterally as subsurface runoff and can emerge as water springs that charge surface streams. That part of the precipitated water that does not penetrate into the terrain can accumulate on its surface or flow on it as surface runoff that can also charge the streams. Surface runoff in this chapter is defined as the part of the precipitation that does not infiltrate into the terrain, but flows on the surface as sheet flow or in small rills or channels. The surface runoff and the streams together are described as overland flow.

From the water budget point of view, and for relatively small upland catchments, the overland flow (*OF*) at the outlet from a catchment is described in Eq. (15.2), in which all the parameters are expressed in millimeters (water volume per unit surface area)

$$OF = TP - (ET + GR + SW + WA) \tag{15.2}$$

where, *TP* – total precipitation; *ET* – evapotranspiration (interception, transpiration, and evaporation from the catchment); *GR* – groundwater recharge; *SW* – water retention by the terrain; *WA* – water accumulation on the surface of the terrain.

Infiltration is the entry of precipitation water into the terrain; it is a dynamic process, and has an important role in the hydrological cycle when the water that does not penetrate into the terrain can accumulate on its surface or flow as surface runoff (Hillel 2004). Surface runoff occurs when the rainfall intensity exceeds the terrain infiltration rate and the terrain surface water-holding capacity. Several factors can reduce the infiltration rate during the wetting process. The initial infiltration capacity of dry soil is high because of the combined gravitational and matric potential gradients that draw water into the soil. As the water content in the soil increases and the wetting front moves to deeper levels, the matric potential gradient decreases, this, in turn, decreases the infiltration rate (Hillel 2004). For soils with a stable structure that does not deteriorate during wetting, a simplistic equation that describes the change of the infiltration rate with time during soil wetting is the Green and Ampt equation (Eq. [15.3]).

$$IR_t = K\frac{z_o + h_{wf} + z_f}{z_f} \tag{15.3}$$

where, IR_t is the infiltration rate at time *t; K* is soil hydraulic conductivity; z_o is the thickness of water layer on the soil surface; z_f is the depth of the water front; h_{wf} is soil water potential at the water front. It is important to emphasize here that use of Eq. (15.3) assumes that there is no change in soil structure during soil wetting.

For bare soils with low structural stability, the main factor that controls the soil *IR* under rainfall conditions is seal formation at the soil surface (Morin et al. 1981; Ben-Hur et al. 1985, 1987; Assouline and Mualem 1997; Ben-Hur 2008). This seal is thin, only a few millimeters deep, and is characterized by greater density, higher strength, finer pores, and lower saturated hydraulic conductivity than the underlying soil (McIntyre 1958; Chen et al. 1980; Gal et al. 1984; Onofiok and Singer 1984; West et al. 1992; Wakindiki and Ben-Hur 2002). Agassi et al. (1981) suggested that formation of a seal is a result of two complementary mechanisms: (1) physical disintegration of surface soil aggregates, caused by the impact energy of the raindrops; and (2) the physicochemical dispersion of soil clays, which can migrate and clog the pores in the seal layer.

Under seal formation at the soil surface, the change of *IR* with time during the rainstorm is described by Eq. (15.4) (Morin et al. 1981).

$$IR_t = IR_f + (IR_i \cdot IR_f) - \exp(\cdot\gamma \cdot p \cdot t) \tag{15.4}$$

in which IR_t is the *IR* at time t_i; IR_f is the final *IR*, which is fairly constant and close to the steady-state *IR*; IR_i is the initial *IR*; γ is a soil coefficient, related to the structural stability of the soil surface; *p* is the rainfall intensity; and *t* is the time from the beginning of the rainstorm. Under seal formation conditions the *IR* of the soil is controlled mainly by the low hydraulic conductivity of the seal layer (Morin et al. 1981). When the surface of the terrain is bedrock, the *IR* could be very low (~0) because the very low permeability of the bedrock. In this case, most of the rainwater will flow as surface runoff. In some cases, however, when the bedrock is very cracked, the *IR* could be very high, and most of the rainwater will infiltrate into the bedrock, and will flow downward as preferential flow, driven mainly by the gravitational potential.

Forest cover on slopes can affect the overland flow by means of three main mechanisms: (1) water use by vegetation (i.e., through interception and transpiration) – heavy water use by the vegetation decreases the amount available to flow as runoff and to recharge the streams, in accordance with Eq. (15.2), and could increase the infiltration rate of the terrain, in accordance with Eq. (15.3); (2) the vegetation and the litter on the forest floor can act as semipermeable barriers that reduce the surface runoff flow velocity and, consequently, could lead to increased infiltration of water and reduction of the runoff; (3) the plant canopy and the litter can protect the soil surface against the raindrop impact and increase the aggregate stability, thus, in turn, preventing seal formation and infiltration reduction, in accordance with Eq. (15.4).

Soil Erosion Processes

Soil erosion by water involves two main processes: (1) detachment of soil materials from the surface soil by raindrop impact and surface runoff shear; and (2) transport of the resulting sediment by raindrop splash and overland flow (Baver et al. 1972). Recent erosion models have separated the erosion process into two main components: rill and interrill erosion (Foster et al. 1982). Runoff from the soil surface may concentrate into small erodible channels known as rills. In rill erosion, the soil loss is mainly due to detachment of soil particles and their transport by flowing water. In contrast, in interrill erosion, soil detachment is caused essentially by raindrop impact, and soil transport is due to raindrop splash and runoff flow (Watson and Laflen 1986). Soil erosion in forests could be controlled by four main mechanisms: (1) the vegetation and the litter on the forest floor can decrease the amount and flow velocity of the surface runoff, as described above, and thereby decrease the erosive and transport capacity of the surface runoff, and, consequently, reduce soil loss; (2) the plant canopy and the litter can protect the soil surface against raindrop impact, and thereby reduce soil detachment during the rainstorm; (3) the vegetation and the litter on the forest floor can trap eroded sediments and limit their transportation, which, in turn, limits the soil loss; (4) plant-root activities, and decomposition of the forest floor litter can change the soil properties, as described below, and could thereby increase the aggregate stability and decrease soil detachment.

Mediterranean, Semiarid Climate

A narrow zone contiguous to the Mediterranean Sea is classified as a Mediterranean region. The Mediterranean semiarid climate is characterized by long, dry, hot summers and short, wet, mild winters. The annual rainfall in the Mediterranean semiarid region ranges mainly from 400 to 600 mm, and snow precipitation is rare (Hotzl 2008). Likewise, the rainfall in this region is highly variable, both during the

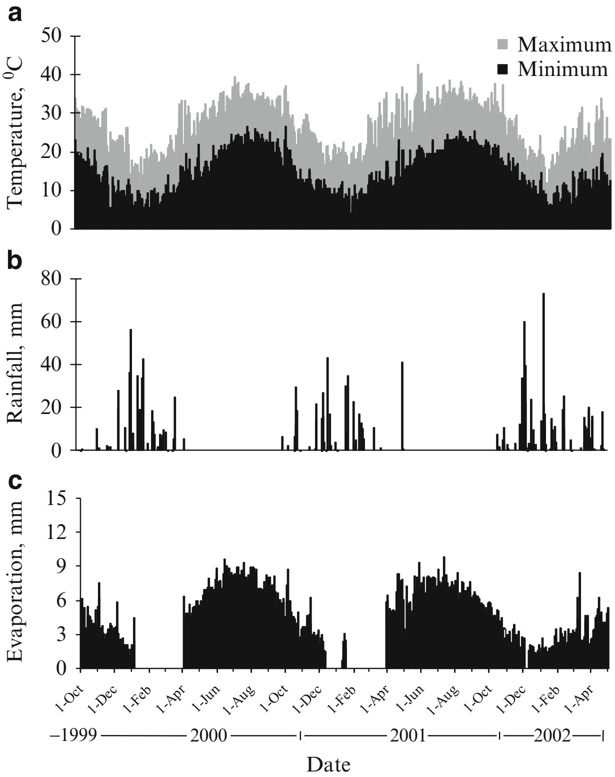


Fig. 15.1 Daily maximum/minimum temperatures, rainfall, and class "A" pan evaporation in semiarid region, Ramat Yishay, Israel

season and from year to year. Examples of daily maximum/minimum temperatures, rainfall, and class "A" pan evaporation in the semiarid region in Israel are presented in Fig. 15.1. In the Mediterranean semiarid region, water is a limited resource, and droughts, salinity and desertification are common features. In this region, the watersheds are water-limited, and the soils are characterized by relatively low organic matter content, high contents of expandable clay minerals, and high levels of salinity and sodicity (Singer 2007), all which decrease the stability of the soil structure (Ben-Hur 2008). The forests in the Mediterranean region can be divided into two main types. The first comprises open mixed forests, including spruce and mountain pine, which are common in the semiarid part of the Mediterranean region. These forests are not intensively managed for commercial purposes, but they are widely perceived as having a high conservation value and being beneficial for tourism. The second type comprises intensively managed eucalyptus plantations. The planting of these plantations has been promoted mainly in southern Europe as a means to develop areas where agricultural productivity is low, and there is high rural unemployment. However, these plantations are uncommon in the semiarid part of the Mediterranean region.

Runoff and Soil Erosion

The forested watersheds in the Mediterranean semiarid region are under water-limited conditions. However, in this region the total precipitation during the rainy season (winter) is, in general, higher than the total evapotranspiration, therefore it is expected that surface runoff would be formed. However, it is known that forests with close cover in the Mediterranean semiarid region are very effective in preventing runoff and soil erosion. In order to study the mechanisms that prevent runoff and soil loss under Mediterranean, semiarid conditions, experiments with runoff plots were conducted in the semiarid part of Israel. The runoff plots were established in September 2004 before the rainy season in two different locations: the Kaddorie site, characterized by sandy clay loam soil, and the Ramat-Hacovech site, with sandy soil. The saturated hydraulic conductivity of both of these soils was >100 mm/h, which was higher than the rainfall intensity (Morin et al. 1981). The treatments in the runoff plots were: bare soil, cover of annual plants, mulched soil, and cover of olive or citrus trees, which are very common in the Mediterranean region. The soil in the bare treatment was sprayed with herbicide to prevent germination and development of vegetation. The tree-covered runoff plots were below the tree crowns, floor litter was sparse and was removed from the soil surface before the rainy season, and no annuals were grown below the trees. In the mulched treatment, a 6-cm-thick layer of wood chips was spread on the soil surface, and it represented soil cover with a dense litter layer.

The cumulative surface runoff and soil loss from the runoff plots with the various soil covers were measured during the 2006–2007 rainy season, and the results are presented as functions of cumulative rainfall in Figs. 15.2 and 15.3. High surface runoff and soil loss were obtained in the bare soil treatment (Fig. 15.2); after 372 mm of rainfall the cumulative runoff was 67 mm, i.e., 18% of the seasonal rainfall, and the cumulative soil loss was 0.65 kg/m^2 (Fig. 15.2). In this treatment, the impact energy of the raindrops broke down the aggregates at the bare soil surface, which led to increases in soil detachment, seal formation, runoff and soil loss (Fig. 15.2). Similar results were obtained in the annual-covered soil treatment (Fig. 15.3), in which the soil surface was bare at the beginning of the rainy season, with the result that the impact energy of the raindrops broke down the aggregates and formed a seal. In this case, the subsequent development of the canopy of the annuals covered soil that already bore a seal, so that the effect of the canopy in preventing runoff and soil loss was negligible (Fig. 15.3). Similar results were obtained by Ben-Hur et al (1989) with a cotton canopy under sprinkler irrigation.

The surface runoff and soil loss in both experimental sites were the lowest in the mulch treatment (Figs. 15.2 and 15.3). In this treatment, the cumulative surface runoff and soil loss were 3.3 mm and 0.02 kg/m^2, respectively, in the Kaddori site (Fig. 15.2), and 3 mm and 0, respectively, in the Ramat-Hacovech site (Fig. 15.3). This low runoff and soil loss in the mulch treatment could be attributed to the covering effect of the wood chips that protected the soil surface against raindrop impact. However, in this treatment, the wood chips decomposed rapidly because of the high

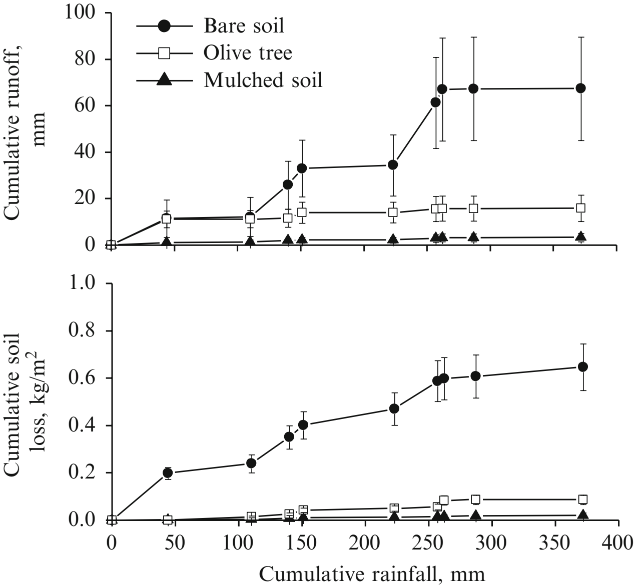


Fig. 15.2 Cumulative surface runoff and soil loss as functions of cumulative rainfall during 2006–2007 rainy season for various soil cover treatments in Kaddorie site, Israel. *Vertical bars* indicate standard deviations (Ben-Hur, unpublished data)

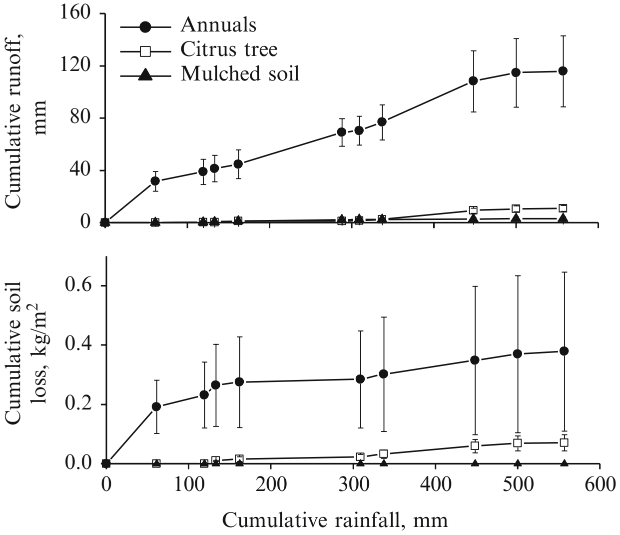


Fig. 15.3 Cumulative surface runoff and soil loss as functions of cumulative rainfall during 2006–2007 rainy season for various soil cover treatments in Ramat-Hacovech site, Israel. *Vertical bars* indicate standard deviations (Ben-Hur, unpublished data)

temperatures of the Mediterranean semiarid climate (Fig. 15.1). Therefore, the cover of the soil surfaces by the wood chips during the 2006–2007 rainy season, about 3 years after the mulch application, was limited and its effectiveness in protecting the soil surface against raindrop impact was low. Another mechanism that could decrease the runoff and the soil loss in the mulch treatment (Figs. 15.2 and 15.3) is augmentation of the organic matter content in the soil (Fig. 15.4a and b) as a result of decomposition of the wood chips. This increase in the organic matter content in the mulched soil enhanced the stability of the aggregates in this soil, which, in turn, increased the saturated hydraulic conductivity of its 0–5-cm layer (Fig. 15.4c).

In the olive and the citrus treatments, the cumulative, surface runoff values and the soil loss were low: 16 and 11 mm of runoff, and 0.09 and 0.07 kg/m^2 of soil loss, respectively (Figs. 15.2 and 15.3). In these treatments, however, the litter was sparse and was removed from the soil surface in the runoff plots, so that the organic matter content of the soils was relatively low. Therefore, the low values of the runoff and soil loss in the tree treatments could be attributed to two main factors: (1) the trees increased the water interception and transpiration, and thereby decreased the hydrological load on the soil; (2) the canopy of the trees decreased the impact energy of the rain drops and thereby, in turn, prevented the seal formation. However, the runoff and soil loss in the trees treatment were higher than in the mulch treatment (Figs. 15.2 and 15.3). Probably, in the case of the trees treatment, the water drops falling from the leaves of the olive and the citrus trees during the rainstorms, probably, caused some breakdown of the aggregates at the soil surface, so that partial seals were formed, leading to some surface runoff and soil loss (Figs. 15.2 and 15.3). It could be concluded from the results in Figs. 15.2–15.4 that the litter layer on the forest floor could be a very effective means of preventing surface runoff and soil erosion in deciduous forests: the litter can act as a protective layer against raindrop impact, and its decomposition can increase the aggregate stability in the soil layer. Therefore, under these conditions, seal formation and soil detachment would be avoided, most of the rainfall would infiltrate into the soil surface, and the surface runoff and soil loss would be prevented.

These covering effects of the tree canopy and of the forest floor litter in preventing runoff by increasing aggregate stability and absorbing the impact energy of the raindrops, and thereby preventing seal formation could be effective only when the soil layer is deep and has a high water storage capacity, or when the infiltrated water can move downward easily. Kutiel and Inbar (1993) measured runoff and soil loss from approximately 150-m^2 plots in a plantation of Aleppo and Brutia pine in Yoqneam forest, Israel, which lies in a Mediterranean semiarid climatic region. The soil in these plots was pale rendzina, less than 20 cm in depth, lying on a chalky hill slope with a 45% gradient. Kutiel and Inbar (1993) found that the total runoff and soil loss after 331 mm of rainfall during the 1989 rainy season (January–March) were negligible: 0.28 mm and 0.08 g/m^2, respectively. In addition, Inbar et al. (1998) measured surface runoff from a small (~1 km^2) upland catchment on Mount Carmel, Israel with Mediterranean semiarid, climatic and forest cover comprising a complex of pines (*Pinus halepensis*) and *Quercus calliprinos–Pistacia palastina*.

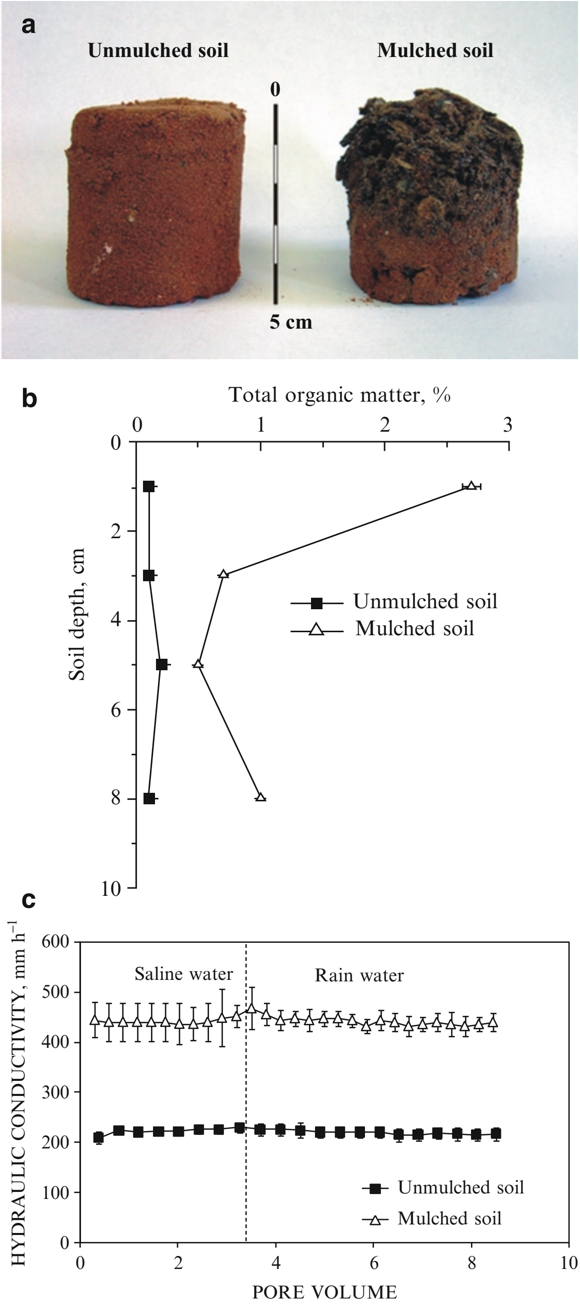


Fig. 15.4 Organic matter content in various depths of bare and mulched soil, 6 years after the mulched application (**a** and **b**) and their saturated hydraulic conductivity values as functions of leaching volume with saline and rain water (**c**). *Lateral* and *vertical bars* indicate standard deviations (Ben-Hur, unpublished data)

The chalk and limestone bedrock of the catchment was covered with shallow (~40 cm depth) terrarosa soil. No surface runoff from this catchment was observed after rainfall of 714 mm in the 1989–1990 winter and 501 mm in the 1990–1991 winter. The water storage capacity of the shallow soils in these two experimental sites – Yoqneam forest and Mount Carmel – was low and could not absorb all the rainwater. Therefore, the very low runoff observed in these two catchments was, most likely, because of the high water use of the trees in these forests and the downward movement of water through cracks in the bedrocks. Partial clear cutting of open spruce forest in the Latte catchment located in Mt. Lozère, S. France, where there is a Mediterranean climate, increased the annual water yield from this catchment by 150 mm, in which the annual rainfall was 1,500 mm (Cosandey 1993). This increase in the water yield was mainly because of the decreased water use of the trees after the partial clear cutting.

Runoff Water Quality

Interaction of rainwater with the soil surface before runoff formation, and interaction of flowing runoff water with the soil surface could change the quality of the runoff water. This runoff quality is strongly affected by meteorological conditions, properties and cover of the soil surface, and micro-organism activities in the soil. The ranges of concentrations and the average values of some elements and compounds in runoff water samples that were collected from the runoff plots in Kaddori site during the winters of 2007–2008 and 2008–2009 are presented in Table 15.1. No cultivation or chemical fertilization were applied in these runoff plots. The average pH values in the runoff water in the three cover treatments were neutral (~7.2), and no significant differences were found among these values in the various cover treatments (Table 15.1). The relative high (21.1%) $CaCO_3$ content in the studied soil maintained high pH values in the soil solution and in the runoff waters in all the studied treatments (Table 15.1). The Cl^- concentrations in the runoff water in the three treatments ranged from 12.5 to 53.6 mg/L (Table 15.1). In contrast, Eriksson (1952) reported that the average Cl^- concentration in the rainwater in the

Table 15.1 Ranges of pH values and concentrations of total organic C (TOC) and Cl, K, NO_3, NH_4, and PO_4 in the surface runoff from plots with different cover treatments. Numbers in parenthesis are average values (Ben-Hur, unpublished data)

Treatment	pH	TOC (mg/L)	Cl (mg/L)	K (mg/L)	NO_3 (mg/L)	NH_4 (mg/L)	PO_4 (mg/L)
Bare	7.0–7.6 (7.3)	9.7–22.5 (15.1)	13–39.3 (22.6)	2.7–10.2 (5.5)	1.8–14.3 (5.7)	0.29–2.61 (1.0)	0.28–1.84 (0.97)
Mulch	6.9–7.6 (7.2)	11.2–38.8 (23)	12.5–53.6 (30.4)	2.0–13.8 (6.0)	1.7–16.7 (9.9)	0.1–1.73 (0.93)	0.72–2.0 (1.3)
Olive tree	7.0–7.5 (7.1)	27.8–95.4 (59.5)	23.4–39.1 (29.1)	1.6–16.4 (6.6)	3.8–23.6 (12.7)	0.32–1.43 (1.12)	0.25–1.0 (0.55)

interiors of most continents ranged from 2 to 6 mg/L. It is most likely that these high Cl^- concentrations in the runoff water (Table 15.1) were a result of the high Cl^- concentrations in the solution of semiarid soils that are characterized by their relatively high salinity; the Cl^- concentration in saturated paste of soil from the 0–10-cm layer in the runoff plots was 140 mg/L.

In spite of the low organic matter content in the soil at the Kaddori site – typical of semiarid soils – the total organic carbon (TOC) concentrations in the runoff water in the three cover treatments were relatively high (Table 15.1). Probably the relatively high winter temperatures that characterize the Mediterranean climate increased the micro-organism activity in the soil, which, in turn, enhanced the soil organic matter decomposition and increased the dissolved organic carbon concentration in the runoff water. The total organic carbon concentrations in the runoff water in the mulch and tree treatments were higher than that in the bare soil treatment (Table 15.1), probably because mulch decomposition and biological activity of the tree roots increased the concentration of dissolved organic carbon in the upper soil layer and, consequently, in the runoff water (Table 15.1). The concentrations of the nutrients, K, $NO_{3,}$ and PO_4 in the runoff water in all three treatments were relatively high (Table 15.1) compared with those in wetter regions, such as the semi-humid northern and mild humid regions discussed below. The relatively high contents of K and P in semiarid soils (Singer 2007) and the high decomposition rates of organic matter in this region caused the high concentration of these nutrients in the runoff water (Table 15.1).

Semi-humid Northern Climate

One of the widely occurring climates in the boreal zone of northern Europe is the semi-humid northern climate. The average annual precipitation in this climate is ranged from 550 to 700 mm, of which ~50% falls as snow (Tikkanen 2006). From records collected from 1979 to 2006 in Finland it was found that the mean annual temperature was ~1°C, the maximum temperature in the summer ranged from 18°C to 25°C, and the minimum temperature in the winter ranged from −3°C to −18°C (Tikkanen 2006). Because of the low evapotranspiration in the watersheds in regions with this climate, these watersheds are under non-water-limited conditions. Soil erosion is a considerable problem in forestry throughout Europe. However, in the boreal zone of Northern Europe, apart from the mountainous areas, the risk of large-scale soil erosion is low, because most of the slopes in this region are gentle (<3%) and the soils have low erodibility factor. Forest management practices, such as clear cutting, site preparation and land drainage, could decrease the forest cover and the biomass of tree stands, and change soil structure. All these can reduce the total evapotranspiration, enhance the exposure of the soil to the rainfall-driven erosion, and decrease the inherent sedimentation within the catchments, thereby possibly increasing overland flow and soil erosion. Detrimental effects of forestry treatments on water quality can, however, be controlled by practicable water-protection methods, such as: avoiding intensive forestry in areas with steep slopes or on soils that are sensitive to degradation processes; leaving unmanaged buffer zones around water bodies (Nieminen et al. 2005); and creating sedimentation ponds along drainage networks in order to collect suspended solids and organic and mineral nutrients (Joensuu et al. 1999).

Overland Flow Yield

This section and the following one discuss the effects of clear cutting and land drainage on overland flow and its quality in watersheds occupied by forest and under semi-humid northern climate conditions. For this study, four small upland catchments in eastern Finland were selected for application of a classical paired-watersheds approach; they included no agricultural or inhabited areas. Selected features of the studied catchments are presented in Table 15.2. The forests in the catchments contained mainly Scots pine (*Pinus sylvestris* L.) and Norway spruce (*Picea abies* L. Karst.), with average stand volumes ranging from 62 to 187 m^3 ha^{-1}. The bedrocks in the studied catchments were mainly gneiss, granite and granodiorite, covered with a layer of mineral soil and peat. The thickness of this layer in most area of the catchments was <2 m. The texture of the mineral soil was stony, sandy till containing <2% of fine earth clay. The slope of the terrain in the catchments varied between 0.3% and 1.5%.

The treatments that were studied in these catchments were (Table 15.2): (1) in Murtopuro catchment, 58% and 40% of the catchment areas were used for clearcutting and drainage (digging ditch networks mainly in the peat land), respectively, in 1983, and 4% of the area was again clear cut in 2002; (2) in Kivipuro catchment, 56% of the catchment area were clear cut in 1983, when an uncut buffer zone 30–50 m in width was left between the clear-cut area and the stream (Ahtiainen and Huttunen 1999); and (3) the Liuhapuro and Välipuro catchments were left unmanaged except for small-scale clearcutting that involved 9% of the catchment area, conducted in 2001 in the Välipuro catchment. The clearcut areas in all the cutting treatments were ploughed, mounded, and/or harrowed during 3 years after the forest cutting. In all the studied catchments, the daily amount of stream flow at the outlet from the

Table 15.2 Selected features of the studied catchments and the applied forest managements (Sarkkola, unpublished data)

Catchment	Coordinates of Location	Land Area	Elevation, a.s.l	Peat Land	Forest Managements	Application Year
		ha	m	%[a]		
Murtopuro	63°46′;28°28′	494	225	50	58%[a] Clear cutting	1983
					40% Land drainage	1983
					4% Clear cutting	2002
Liuhapuro	63°47′;28°29′	165	225	48	No management	
Kivipuro	63°52′;28°65′	54	200	32	56% Clear cutting	1983
Valipuro	63°52′;28°65′	86	200	56	9% Clear cutting	2001

a.s.l Above sea level

[a] Percent of the total catchment area

catchment was continuously measured from 1979 to 2006, and water samples from the stream were taken once or twice each month and were analyzed by standard methods for concentrations of total organic carbon, nitrogen, and suspended solids (Ahtiainen and Huttunen 1999).

Cumulative amounts of overland flow at the outlets of the four studied catchments for 3 meteorological years – 1980–1981, 1986–1987, and 2005–2006 – are presented in Fig. 15.5, as functions of cumulative rainfall. The meteorological year was defined, on the basis of precipitation and overland flow, as extending from 1st October until 30th September of the next year. The annual amounts of overland flow from the studied catchments in the 3 meteorological years were relatively high, ranging from 200 mm in the Kivipuro catchment in 2005–2006 to 580 mm in the Murtopuro and Liuhapuro catchments in 1980–1981 (Fig. 15.5), and the average, annual precipitations in these catchments was ~600 mm (Fig. 15.5). These large amounts of overland flow were mainly a result of the low water use (transpiration and interception) of the trees in the catchment because of the low potential evapotranspiration under semi-humid northern climatic conditions, the limited water-storage capacity of the soils, and the low permeability of the bedrock in these catchments. In the 3 meteorological years in the four studied catchments, the overland flow coefficient (the overland flow in millimeter per 1 mm of cumulative precipitation), as determined from the slopes of the graphs in Fig. 15.5 were lowest (lowest slopes) in the winter, most likely because of the frozen conditions in this season. In contrast, the highest overland flow coefficients (steepest slopes) occurred in the spring as a result of fast snow melting. In the summer and autumn, the overland

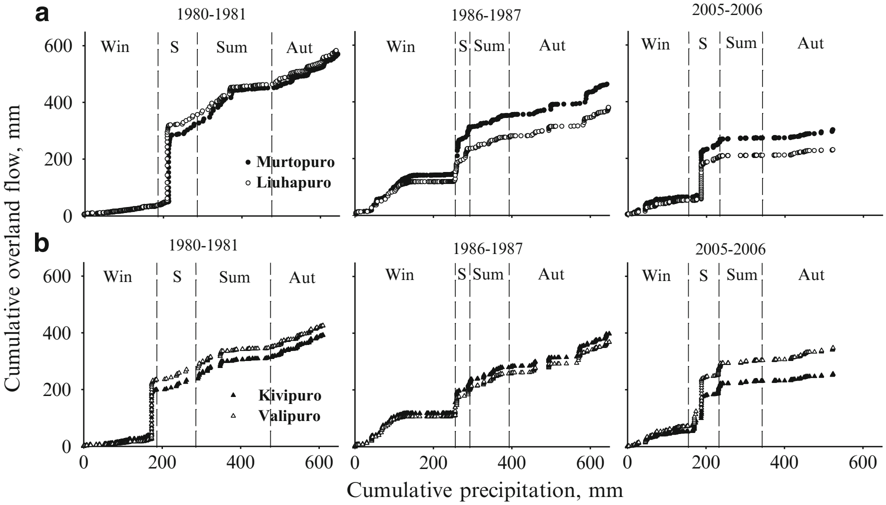


Fig. 15.5 Cumulative overland flow at the outlets of four catchments in eastern Finland subjected to different perturbations, as functions of cumulative rainfall during winter (Win), spring (S), summer (Sum), and autumn (Aut) in 3 meteorological years, 1980–1981, 1986–1987, and 2005–2006 (Sarkkola, unpublished data)

flow coefficients were moderate (Fig. 15.5), and were controlled mainly by the intensity and the amount of the rainstorms in these seasons.

The overland flow patterns in the paired catchments (Murtopuro and Liuhapuro; Kivipuro and Välipuro) before the forestry treatments (1980–1981) were similar (Table 15.2 and Fig. 15.5). In contrast, after the forest perturbations in the Murtopuro catchment, the overland flow amounts in this catchment during the meteorological years 1986–1987 and 2005–2006 were higher than those in the untreated Liuhapuro catchment (Fig. 15.5a). This higher overland flow in Murtopuro catchment could be mainly attributed to the following mechanisms: (1) decreased water use by the trees in the catchment, following the clearcutting; (2) formation of a denser network of water channels at the surface of the clearcut area; and (3) destruction of the soil structure in the clearcut area as a result of land drainage and site preparation. A very small repeat clearcutting, involving 4% of the catchment area, was conducted in 2002 in the Murtopuro catchment (Table 15.2), but no significant effect of this clearcutting on the overland flow in this catchment was observed in 2002–2003 (results not presented). Therefore, the significantly higher overland flow in the Murtopuro catchment in 2005–2006 than that in the untreated Liuhapuro catchment (Fig. 15.5a) suggests that the intensive perturbations that occurred in the Murtopuro catchment in 1983 had long-term (23 years) effects on the overland flow. This result indicates also that the main factor that increased the overland flow in the Murtopuro catchment after its perturbations was a decrease of water use by the trees as a result of their cutting, because the other two mechanisms had only short-term effects on the overland flow. In contrast, the recovery of water use by the trees seedlings depended on their regeneration rates, which are slow.

The intensive clear cutting, involving 56% of the catchment area, in the Kivipuro catchment in 1983 (Table 15.2) did not affect significantly the amount of overland flow in this catchment during 1986–1987 (Fig. 15.5b). This lack of response to the intensive clear cutting in the Kivipuro catchment was probably due to the limited change in the interception capacity of the forest stand as a result of the clear cutting, because the large amounts of tree brash that covered the felled area in this catchment could also have high interception capacity (Robinson et al. 2003). Moreover, the presence of debris dams in streams may have acted to attenuate surface runoff flow. In contrast, a small clear cutting, involving 9% of the catchment area, which was conducted in the Välipuro catchment in 2001, significantly increased the cumulative amount of overland flow compared with that in the Kivipuro catchment during 2005–2006 (Fig. 15.5b).

Overland Flow Quality

In the boreal regions of Northern Europe, such as Finland, leaching of chemical compounds by the overland flow in forests is particularly important, because millions of hectares of peatlands in this region have been drained to enable wood production (Paavilainen and Päivänen 1995). Leaching of these compounds and soil loss may have significant effects, resulting in enhanced eutrophication and deterioration of the quality of aquatic habitats (Grip 1982, Ahtiainen and Huttunen 1999). Concern for this effect on aquatic habitats is particularly relevant in boreal regions, because of the large numbers of lakes and watercourses that are sensitive to changes in nutrients concentrations (Schindler 1998).

The average annual concentrations of suspended solids and total N and organic C in the overland flow in the four studied catchments (Table 15.2) at various dates are presented in Fig. 15.6, in which the clear cutting and drainage that were conducted in

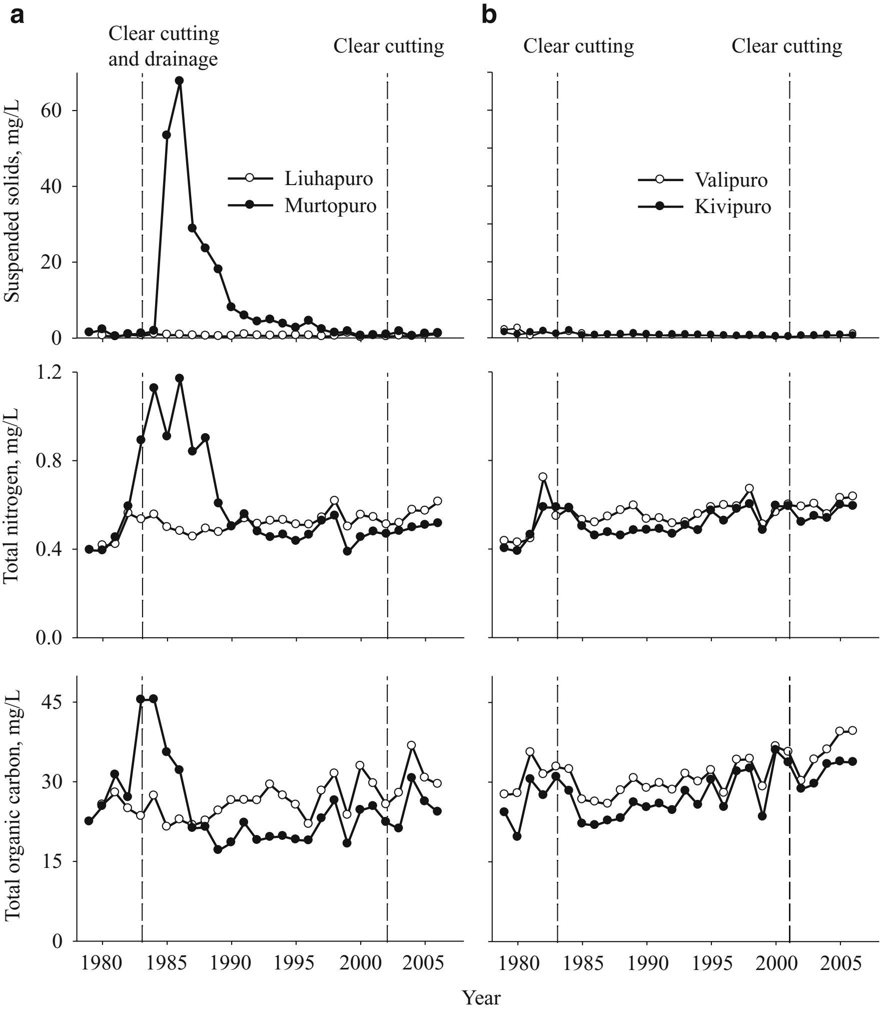


Fig. 15.6 Average annual concentrations of suspended solids, total N and organic C in overland flow at four catchments in eastern Finland on various dates. Implementation of clear cutting and land drainage in these catchments according to Table 15.2 indicated by *vertical dashed lines* (after Ahtiainen and Huttunen 1999; Sarkkola et al. 2009)

these catchments (Table 15.2) are indicated by vertical dashed lines. The intensive clearcutting and land drainage that were conducted in the Murtopuro catchment in 1983 significantly increased the concentrations of suspended solids and total N and carbon C in the overland flow, over periods of 14 years for the suspended solids, 6 years for total N, and 4 years for total organic C (Fig. 15.6a). Concentrations of these compounds in the overland flow after the clear cutting in the Murtopuro catchment were comparable with those measured in other small forested catchments after perturbations (e.g., Finer et al. 1997; Kortelainen and Saukkonen 1998). The increases in the concentrations of these compounds in the overland flow in the Murtopuro catchment after the intensive perturbations (Fig. 15.6a) were probably due to land surface disturbance caused by the clearcutting and soil drainage activities, and the high decomposition of the large amounts of the tree brash that covered the felled area.

The intensive clearcutting that was conducted in the Kivipuro catchment in 1983 (Table 15.2) had no significant effect on the concentrations of compounds in the overland flow (Fig. 15.6b). This lack of any effect of the intensive clearcutting in the Kivipuro catchment on the compounds concentrations was probably because: (1) no increase in the overland flow occurred after this clearcutting (Fig. 15.5b), and (2) the uncut buffer zone that was left in this catchment prevented intensive depletion of the concentrations of these compounds from the clearcut area into the stream water. The limited clear cutting that was conducted in the Murtopuro catchment in 2002 and in the Välipuro catchment in 2001 (Table 15.2) also caused insignificant effects on the concentrations of the compounds in the overland flow (Fig. 15.6). In these cases, the disturbance of the forest floor caused by the limited clear cutting was probably mild and not sufficient to lead to increases in these concentrations in the overland flow.

The average seasonal concentrations of suspended solids and total N and organic C in the overland flow in the unperturbed Liuhapuro and the slightly perturbed Välipuro catchments, which were measured over 28 years from 1979 until 2006 (Fig. 15.6), indicated that the average seasonal concentrations of total N and organic C were higher in summer and autumn than in winter and spring, and that the average concentration of suspended solids were higher in the spring than in the other seasons (results not presented). These high concentrations of total N and organic C in the summer and autumn were, probably, a result of the high temperatures in these seasons that increased the micro-organism activities in the soil, which, in turn, increased the soluble N and organic C levels in the soil surface. In contrast, the occurrence of the highest suspended solids concentration in the spring was probably due to the high overland flow rate and its erosivity in this season (Fig. 15.5), caused by the fast melting of snow in this season.

Mild, Humid Climate

The mild, humid climate is characterized by a mean annual temperature of ~14°C, and mean temperatures in the coldest and hottest months of ~9°C and ~20°C, respectively. The average annual rainfall in this climate is high, about 1,900 mm, with the autumn – from October to December – being the wettest season, and the summer – from July to September – being the driest. An example in Europe of a region with a mild, humid climate is northwest Spain (Galicia), with an ocean-influenced climate. In this region, a eucalypt species commonly used in forestry is the *Eucalyptus globulus* Labill, which covers ~335,000 ha under monocultural and mixed stands. It is grown mostly on steep slopes, in shallow and acidic soils that developed on granitic bedrock (Fernandez et al. 2006). The wood of this eucalypt species is the most economical raw material for paper-pulp production in southwest Europe (Gras et al. 1993). These eucalypt stands are characterized by their high growth, and are managed on short rotations of 10–15 years using mechanized skidding and intensive logging by means of slash manipulation after clear cutting. The effects of eucalypt forest cover on catchment water yield, as affected by various perturbations and management activities such as wildfires (e.g., Langford 1976; Mackay and Cornish 1982; O'Loughlin et al. 1982; Brown and Mitchell 1986; Scott 1993), clear cutting (e.g., Langford et al. 1982; Cornish and Vertessy 1993; Cornish and Vertessy 2001; Watson et al. 2001), and pests (e.g., Cornish and Vertessy 2001; Watson et al. 2001), have been extensively studied in regions where these species are natural growth or extensively planted, such as Australia and South Africa.

Overland Flow Yield

A case study that determined the effects of forest cover types (*Eucalyptus globulus* Labill and *Pinus pinaster* Ait) and of successive perturbations on the yield and quality of overland flow from small upland catchments under mild, humid climatic conditions is discussed below. The experimental catchments were located near the Atlantic Ocean inlet of Pontevedra, northwest Spain; selected features of these catchments are presented in Table 15.3. The soils were Humic Umbrisols, Alumiunbric Regosols and Alumiumbric Leptosols, with sandy and sandy-loam textures that developed on granitic or granodioritic bedrock. The effects of successive perturbations of forest on overland flow yield and quality were studied in the *Eucalyptus globulus* catchment (Castrove catchment, Table 15.3), in which the forest perturbations were: a moderate surface wild fire in July 1989 that burned the understorey and scorched the tree crowns; clearcutting of trees in summer 1992; a rotation during 1995 and 1996 based on coppice and coppice sprout selection that

Table 15.3 Selected features of the studied catchments (Fernandez et al. 2006)

Catchment	Land Area	Slope	Elevation, a.s.l	Mean Soil Depth	Mean Precipitation	Tree Species	Density
	ha	%	m	m	mm		Trees ha^{-1}
Castrove	9.9	20	348–445	0.5	2,020	*E. globulus*	1,251
Caldas	6.7	10	210–251	0.9	1,700	*P. pinaster*	329

a.s.l Above sea level

reduced the basal area by 70%; and an attack by *G. scutellatus* Gill that started in spring 2000 and caused moderate tree defoliation in most parts of the catchment area. Stream flow out of the studied catchments was continuously measured with a 90° V-notch. In water samples that were collected from the stream, concentrations of NO_3, PO_4, K, Na, Ca, and Mg were measured by using standard methods (Ahtiainen and Huttunen 1999).

During 1987–1989 under no forest perturbations, the quarterly overland flow yields at the outlet of the *Pinus pinaster* Ait catchment (Caldas catchment, Table 15.3) ranged from 23 to 417 mm, and the annual average of these overland flow yields during these years was 696 mm, which was 42% of the average annual rainfall in this catchment. In contrast, in the *Eucalyptus globulus* catchment (Castrove catchment, Table 15.3) with no perturbations in the same years, the quarterly overland flow yields ranged from 0 to 1,052 mm, and the annual average of these yields was 1,596 mm, which was 73% of the average annual rainfall. The soils in these catchments were shallow (Table 15.3), therefore their water-storage capacities were limited. Likewise, no recharge of the ground water was expected in these catchments. Consequently, the higher annual overland flow yield from the *Eucalyptus globulus* than from the *Pinus pinaster* catchment was mainly a result of the greater water use (transpiration and interception) by the *Pinus pinaster*, which was 1,034 mm, than by the *Eucalyptus globulus* trees, which was 914 mm, under the local conditions of the studied catchments.

Quarterly rainfall amounts and the differences between the quarterly overland flow yields, as measured at the studied *Eucalyptus globulus* catchment and the values predicted on the basis of the measurements in the unperturbed catchment, over the period from 1987 to 2005 are presented in Fig. 15.7. The wildfire in 1989 significantly increased the overland flow yield during the autumn and winter of meteorological years 1989–1990 and during the autumns of meteorological years 1990–1991 and 1991–1992 (Fig. 15.7). The highest quarterly increase in the overland flow yield as a result of the wildfire was 512 mm, which is 52% of the corresponding quarterly rainfall (Fig. 15.7). After the clear cutting in summer 1992, the overland flow yields increased significantly during autumn 1992–1993, autumn and winter 1993–1994, and autumn 1994–1995 (Fig. 15.7). The maximum increase in the overland flow yield after the clear cutting was 667 mm during autumn 1993–1994, which is 63% of the corresponding quarterly rainfall. After the coppice sprout selection conducted in 1995 the overland flow yields increased significantly during autumn and winter 1995–1996, and the autumns of 1996–1997 and 1997–1998 (Fig. 15.7); no significant effect was observed during the fourth post-thinning year (Fig. 15.7). The quick spread of the *G. scutellatus* pest during 1999–2000 and 2000–2001significantly increased the water yields during all the autumns from 1999 to 2004, and during the winters of 2000–2001 and 2002–2003. Calculation of the average values of the significant quarterly increase of the overland flow yield during 3 years after each perturbation indicated that the increases in overland flow yield after wildfire and clear cutting were greater than those that followed the perturbations caused by coppice sprout selection and *G. scutellatus*

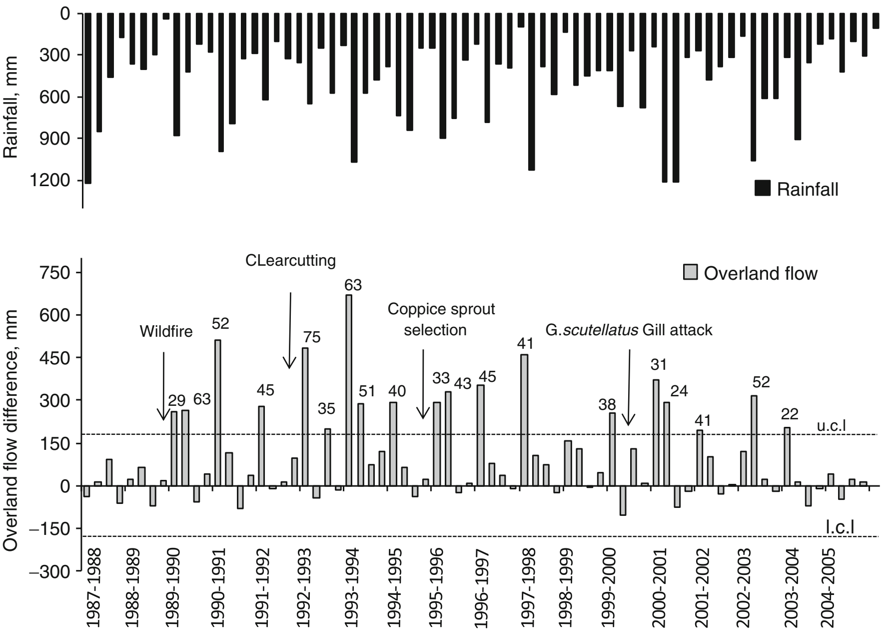


Fig. 15.7 Quarterly rainfall, and differences between the measured quarterly overland flow yields from perturbed *Eucalyptus globulus* catchment and values predicted from unperturbed catchment measurements, over the period from 1987 until 2005. *Lateral dashed lines* indicate the upper (u.c.l) and lower (l.c.l) confidence (95%) interval. The *numbers above the columns* indicate the quarterly increase in the overland flow as percentages of the corresponding quarterly rainfall (After Fernandez et al. 2006)

pest attack. This was probably because the wildfire and clear cutting affected the water use of the vegetation in the catchment more strongly than the perturbations caused by coppice sprout selection and *G. scutellatus* pest attack.

Overland Flow Quality

The average annual concentrations of some elements and compounds in water samples that were collected from the overland flow at the outlet of the Castrove catchment (Table 15.3) over the period from 1987 until 2002 are presented in Fig. 15.8; the forest perturbations are indicated by vertical dashed lines. The wildfire caused large increases in the concentrations of the majority of the studied elements and compounds in the overland flow, but this effect was short-lived – <2 years after the wildfire (Fig. 15.8). The immediate increase of the NO_3 concentration in the overland flow after the wildfire (Fig. 15.8) was most likely attributed to the oxidation of the organic materials on the forest floor caused by the heat of the fire, and the high nitrification rate that prevailed after the fire (Kutiel and Shaviv 1989).

Similar results were found by Kutiel and Inbar (1993), which indicated that burning of plant materials in the catchment caused increases in Ca, Mg, Na and K contents in the upper few centimeters of the soil. During the rainy season after the wildfire, the surface runoff leached the elements from the upper soil in the catchment and increased their concentrations in the overland flow (Fig. 15.8). After clearcutting of the forest, the concentrations of the nutrients, NO_3, K, Ca, and PO_4 in the overland flow in the catchment increased (Fig. 15.8). This nutrients increase resulted, most likely, from decomposition of the large amounts of tree brash that covered the felled area after the clearfelling that released these nutrients. However, the forest clearcutting had insignificant effects on Na and Mg concentrations in the overland flow (Fig. 15.8).

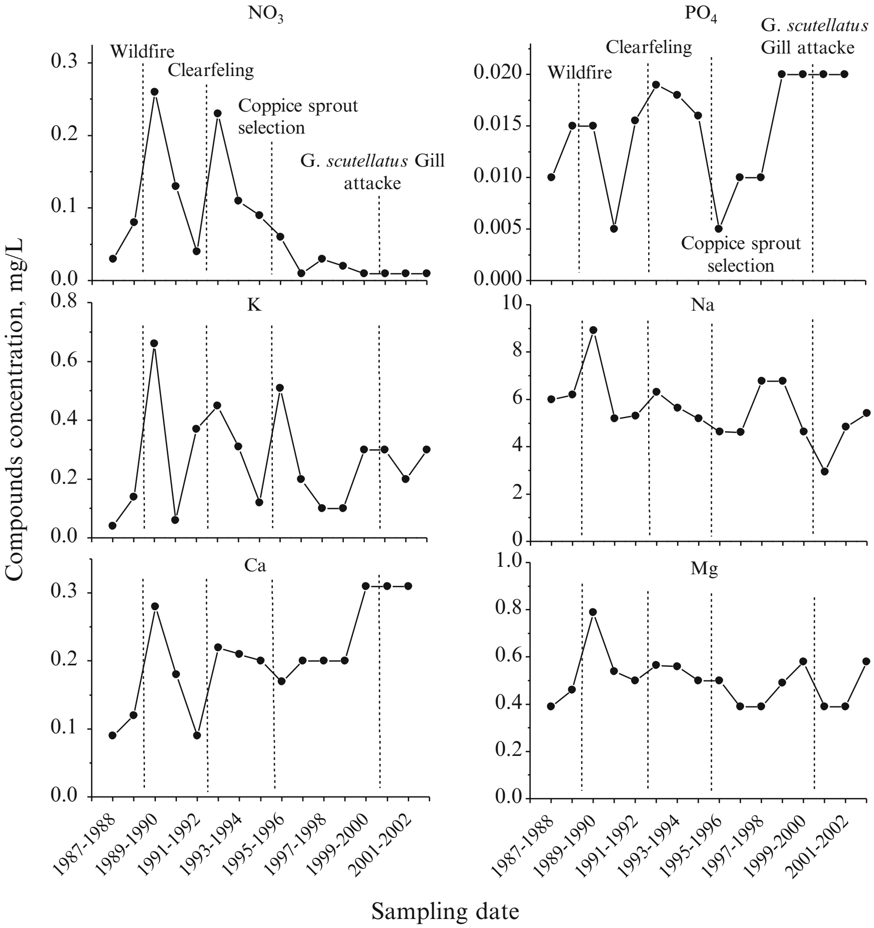


Fig. 15.8 Annual average concentrations of NO_3, PO_4, K, Na, Ca and Mg in water samples from overland flow at the *Eucalyptus globulus* catchment in various years. Forest perturbations marked by *vertical dashed lines* (After Fernandez et al. 2006)

Summary and Conclusions

It is well known that forest cover and its management and perturbations significantly affect the hydrology of watersheds, and especially the surface runoff, overland flow, and soil erosion. These effects, however, are strongly related to climatic conditions. With respect to water issue, watersheds were divided into two main types: (1) those under water-limited conditions; and (2) those under non-water-limited conditions. The overall objective of this chapter was to describe and discuss the relationships between forest and climate conditions, and their effects on surface runoff, over land flow, soil erosion and runoff water quality, when subjected to various managements and perturbations. Three different climate types were discussed in this chapter: (1) a Mediterranean, semi-arid climate with annual precipitation of ~500 mm; (2) a semi-humid northern climate with annual precipitation of ~600 mm and freezing temperatures and snow conditions in the winter; and (3) a mild, humid climate with average annual rainfall of ~1,800 mm and no freezing temperatures or snow conditions. The forested watersheds in the Mediterranean semiarid region are under water-limited conditions and in the semi-humid northern climate and mild, humid climate are under non-water-limited conditions.

In the Mediterranean semiarid climate region, high surface runoff and soil loss were obtained in bare soils and soils covered with annual because the unstable structure of the soils in this region and seal formation in the beginning of the rainy season. A litter layer on forest floor with deep soil and/or very cracked bedrock in Mediterranean semiarid climate region could be a very effective means in preventing surface runoff and soil erosion in deciduous forests. In this case, the litter acts as a protective layer against raindrop impact, and its decomposition increases the aggregate stability in the soil layer. The effects of the forest floor litter on water infiltration and its movement through the soil layer in forests in Europe are discussed in detail in Chapter 16 in the present book. Close cover forests in the Mediterranean semiarid region are very effective in preventing runoff and soil erosion mainly because of the high water use of the trees and the downward movement of water through cracks in the bedrocks. Partial clear cutting of forests in this region could increase the water yield from the catchment mainly because of the decreased water use of the trees, and exposing the soil surface to the raindrop impact and seal formation.

In the semi-humid northern climate and the mild, humid climate, the overland flow yield in small, upland forested catchments with no perturbations is relatively high. For example, annual cumulative amounts of overland flow at the outlets of four catchments contained mainly *Pinus sylvestris* L. and *Picea abies* L. Karst in the semi-humid northern climate ranged from 200 to 580 mm, when the average, annual precipitations in these catchments was ~600 mm. Likewise, in the mild, humid climate, the annual average of overland flow yields was 696 mm, which was 42% of the average annual rainfall, in *Pinus pinaster* Ait catchment and 1,596 mm, which was 73% of the average annual rainfall, in *Eucalyptus globulus* catchment. These large amounts of overland flow from catchments in the semi-humid northern and the mild, humid climates were mainly a result of low water use (transpiration and interception) of the trees because of the low potential evapotranspiration under these climates conditions, the limited water-storage capacity of the soils, and the low permeability of the bedrock in these catchments. In both climates, the risk of large-scale soil erosion is low, mainly because the soils in these regions have low erodibility factor.

In the semi-humid northern climate, the average seasonal of overland flow coefficient was lowest in the winter, most likely, because of the frozen conditions in this season, highest in the spring as a result of fast snow melting, and moderate in the summer and autumn, which was controlled mainly by the intensity and the amount of the rainstorms in these seasons. Significant perturbation of clearcutting and drainage in catchment in this climate increased significantly the overland flow amounts for a period of 23 years. This is because that the recovery of water use by the trees stumps depended on their regeneration rates, which are slow. Successive perturbations of a moderate surface wild fire that burned the understorey and scorched the tree crowns; clearcutting of trees, a rotation based on coppice and coppice sprout selection that reduced the basal area by 70%, and an attack by *G. scutellatus* Gill that caused moderate tree defoliation in most parts of the catchment in the mild, humid climate had significant effects on the yield of overland flow. The increases in overland flow yield after wildfire and clear cutting were greater than those that followed the perturbations caused by coppice sprout selection and *G. scutellatus* pest attack. This was probably because the wildfire and clear cutting affected the water use of the vegetation in the catchment more strongly than the perturbations caused by coppice sprout selection and *G. scutellatus* pest attack.

The interaction between forest cover and its management and perturbations and climate conditions have also effects on the quality of surface runoff and overland flow. In the Mediterranean semiarid climate region, the relative high content of $CaCO_3$, K and P in the mineral fraction of the semiarid soils and the high Cl^- concentration in their solutions maintained high pH values and Cl^-, K, and PO_4concentrations in runoff waters under various soil covers. In spite of the low organic matter content in semiarid soils, the total organic carbon concentration in the runoff water was relatively high. Probably, the relatively high temperatures, which characterize the Mediterranean climate, increased the activities of the soil fauna and trees roots, and consequently enhanced the dissolved organic carbon concentration in the runoff water. In contrast, in *Eucalyptus globulus* catchment in the mild, humid climate, the average annual concentrations of NO_3, PO_4, K, Ca, Mg, and Na in overland flow were very low. These low concentrations were mainly a result of the high overland flow yield in this catchment that diluted the elements and compounds concentrations in the runoff water.

In forested catchments with no perturbations in the semi-humid northern climate, the average seasonal concentrations in overland flow of total N and organic C were higher in summer and autumn than in winter and spring, and of suspended solids were highest in the spring. These high concentrations of total N and organic C in the summer and autumn were, probably, a result of the high temperatures in these seasons that increased the micro-organism activities in the soil, which, in turn, increased the soluble N and organic C levels in the soil surface. In contrast, the occurrence of the highest suspended solids concentration in the spring was probably due to the high overland flow rate and its erosivity in this season, caused by the fast melting of snow in this season. An intensive clearcutting and land drainage of catchment in this climate increased the concentrations of suspended solids and total N and carbon C in the overland flow over periods of >4 years. The concentrations increases of the of these components in the overland flow were probably due to soil surface disturbance caused by the clearcutting and soil drainage activities, and the high decomposition of the large amounts of the tree brash that covered the felled area.

References

Agassi M, Shainberg I, Morin J (1981) Effects of electrolyte concentration and soil sodicity on the infiltration rate and crust formation. Soil Sci Soc Am J 45:848–851

Ahtiainen M, Huttunen P (1999) Long-term effects of forestry managements on water quality and loading in brooks. Boreal Environ Res 4:101–114

Allen A, Chapman D (2001) Impact of afforestation on ground water resources and quality. Hydrogeol J 9:390–400

Andreassian V (2004) Water and forest: from historical controversy to scientific debate. J Hydrol 29:1–27

Assouline S, Mualem Y (1997) Modeling the dynamics of seal formation and its effect on infiltration as related to soil and rainfall characteristics. Water Resour Res 33:1527–1536

Baver LD, Gardner WH, Gardner WR (1972) Soil Physics. John Wiley, New York

Ben-Hur M, Shainberg I, Bakker D, Keren R (1985) Effects of soil texture and $CaCO_3$ content on water infiltration in crusted soils as related to water salinity. Irrig Sci 6:281–284

Ben-Hur M, Shainberg I, Morin J (1987) Variability of infiltration in a field with surface-sealed soil. Soil Sci Soc Am J 51:1299–1302

Ben-Hur M, Plaut Z, Shainberg I, Meiri A, Agassi M (1989) Cotton canopy and drying effects on runoff during irrigation with moving sprinkler systems. Agron J 81:751–757

Ben-Hur M, Agassi M (1997) Predicting interrill erodibility factor from measured infiltration rate. Wat Resour Res 33(10):2409–2415

Ben-Hur M (2008) Seal formation effects on soil infiltration and runoff in arid and semiarid regions under rainfall and sprinkler irrigation conditions. In: Zereini F, Jaeschke W (eds) Climatic changes and water resources in the Middle East and in North Africa. Springer-Verlag, New York, pp 429–452

Bosch JM, Hewlett JD (1982) A review of catchment experiments to determine the effect of vegetation changes on water yield and evapotranspiration. J Hydrol 55:3–23

Brown G, Mitchell DT (1986) Influence of fire on the soil phosphorus status in sand plain lowland fynbos, south-western Cape. S Afr J Bot 52:67–72

Bruijinzeel LA (2004) Hydrological functions of tropical forests: not seeing the soil for the tree? Agri Ecosyst Environ 104:185–228

Calder IR (1990) Evaporation in the Uplands. Wiley, Chichester, p 148

Calder IR (2007) Forests and water – ensuring forest benefits outweigh water costs. For Ecol Manage 251:110–120

Chen Y, Tarchitzky J, Brower J, Morin J, Banin A (1980) Scanning electron microscope observations on soil crusts and their formation. Soil Sci 130:49–55

Cornish PM, Vertessy RA (1993) The effects of logging and forest regeneration on water yields in moist eucalypt forest in New South Wales. J Hydrol 150:301–322

Cornish PM, Vertessy RA (2001) Forest age-induced changes in evapotranspiration and water yield in eucalypt forest. J Hydrol 242:43–63

Cosandey C (1993) Consequences hydrologiques d'une coupe forestiere. Le cas du bassin de la Latte (Mont-Lozere, France). In: Griselin M (ed) L'eau, la Terre et les Hommes: Hommage a Rene Frecaut. University Press, Nancy, pp 355–363

Cosandey C, Lavabre J, Martin C, Mathys N (2002) Consequences de la foret Mediterraneenne sur le ecoulements de crue. Synthese de recherches menees en France. Houille Blanche 3:38–42

Eriksson W (1952) Composition of atmospheric perecipitation. Tellus 4(215–232):280–303

Farley KA, Jobbagy EG, Jackson RB (2005) Effects of afforestation on water yield: a global synthesis with implications for policy. Glob Change Biol 11:1565–1576

FAO (2001) State of the world's forests. Food and Agriculture Organisation, Rome

Fernandez C, Vega JA, Gras JM, Fonturbel T (2006) Changes in water yield after a sequence of perturbations and forest management practices in *Eucalyptus globulus* Labill. watershed in Northern Spain. For Ecol Manage 234:275–281

Finér L, Ahtiainen M, Mannerkoski H, Möttönen V, Piirainen S, Seuna P, Starr M (1997) Effects of harvesting and scarification on water and nutrient fluxes. A description of catchments and methods, and results from the pretreatment calibration period. Research Papers 648. The Finish Forest Research Institute, 38 pp.

Foster SSD, Cripps AC, Smith Carington A (1982) Nitrate leaching to ground water. Phil Trans R Soc (London) 296:477–489

Gal M, Arcon L, Shainberg I, Keren R (1984) The effect of exchangeable Na and phosphogypsum on the structure of soil crust – SEM observation. Soil Sci Soc Am J 48:872–878

Gras JM, Vega JA, Bara S (1993) Six years of study on fast-growing forest plantations catchments in northwest Spain. Acta Geol Hisp 28:111–117

Grip H (1982) Water chemistry and runoff in forest streams in Kloten. UNGI Report 58. Division of Hydrology, Department of Physical Geography, Uppsala University

Hillel D (2004) Introduction to environmental soil physics. Elsevier Academic Press, Amsterdam

Hotzl H (2008) Water resources management in the Middle East under aspect of climatic change. In: Zereini F, Jaeschke W (eds) Climatic changes and water resources in the Middle East and in North Africa. Springer-Verlag, New York, pp 75–92

Hornbeck JW, Adams MB, Corbett ES, Verry ES, Lynch JA (1993) Long-term impacts of forest treatment on water yield: a summary for Northeastern USA. J Hydrol 150:323–343

Inbar M, Tamir M, Witternberg L (1998) Runoff and erosion process after a forest fire in Mount Carmel, a Mediterranean area. Geomorphology 24:17–33

Jackson RB, Jobbagy EG, Avissar R, Roy SB, Barrett DJ, Cook CW, Farely KA, Le Maitre DC, McCarl BA, Murray BC (2005) Trading water for carbon with biological carbo sequesration. Science 310:1944–1947

Johnes SM, Cummins T, Boyel GM, Aherne J, Farrell EP (1998) A pilot study into the effects of clearfelling on nutrient losses and sustainability. Monstac Project Final Report. Forest Ecosystem Research Group Rep 23, University College, Dublin, 62 pp

Joensuu S, Ahti E, Vuollekoski M (1999) The effects of peatland forest ditch maintenance on suspended solids in runoff. Boreal Environ Res 4:343–355

Johnson R (1998) The forest cycle and low river flows. A review of UK and international studies. For Ecol Manage 109:1–7

Kirby C, Newson MD, Gilman K (1991) Plynlimonresearch: the first two decades. Institute of Hydrology, Report No. 109, p. 188

Kortelainen P, Saukkonen S (1998) Leaching of nutrients, organic carbon and iron from Finnish forestry land. Water Air Soil Pollut 105:239–250

Kutiel P, Shaviv A (1989) Changes of soil N-P status in laboratory simulated forest fire. Plant Soil 120:57–63

Kutiel P, Inbar M (1993) Fire impact on soil nutrients and soil erosion in a Mediterranean pine forest plantation. Catena 20:129–139

Langford KJ (1976) Change in yield of water following a bushfire in a forest of *Eucalyptus regnans*. J Hydrol 29:87–114

Langford KJ, Moran RJ, O'Shaughnessy PJ (1982) The Coranderrk experiment – the effects of roading and timber harvesting in a mature mountain ash forest on streamflow yield and quality. In: O'Loughlin EM, Bren LJ (eds) Proceedings of the 1st national symposium on forest hydrology, Melbourne, pp 92–102

Mackay SM, Cornish PM (1982) Effects of wildfire and logging on the hydrology of small catchment near Eden, N.S.W. In: O'Loughlin EM, Bren LJ (eds) Proceedings of the 1st national symposium on forest hydrology, Melbourne, pp 111–117

McCulloch JSG, Robinson M (1993) History of forest hydrology. J Hydrol 150:189–216

McIntyre DS (1958) Permeability measurements of soil crust formed by raindrop impact. Soil Sci 85:158–189

Mendel HG (1996) Hochwasser-Gedanken uber Ursachen und Vorsorge aus hydrologischer Sicht. Bundesanstalt fur Gewasserkunde BfG1022, Koblenz

Morin J, Benyamini Y, Michaeli A (1981) The effect of raindrop impact on the dynamics of soil surface crusting and water movement in the profile. J Hydrol 52:321–335

Munich Re (2000) Natural catastrophes – the current position, special millennium issue. Munich Reinsurance Co, Munich, Germany

Nieminen M, Ahti E, Nousiainen H, Joensuu S, Vuollekoski M (2005) Does the use of riparian buffer zones in forest drainage areas to reduce the transport of solids simultaneously increase the export of solutes? Boreal Environ Res 10:191–201

O'Loughlin EM, Cheney NP, Burns J (1982) The Bushrangers experiment: hydrological response of an eucalypt catchment to fire. In: O'Loughlin EM, Bren LJ (Eds), Proceedings of the 1st national symposium on forest hydrology, Melbourne, pp 132–138

Onofiok O, Singer MJ (1984) Scanning electron microscope studies of surface crusts formed by simulated rainfall. Soil Sci Soc Am J 48:1137–1143

Paavilainen E, Päivänen J (1995) Peatland forestry – ecology and principles, vol 111, Ecological studies. Springer-Verlag, Berlin, Amsterdam, New York

Robinson M, Cognard-Plancq AL, Cosandey C et al (2003) Studies of the impact of forests on peak flows and baeflows: a European perspective. For Ecol Manage 186:85–97

Rothacher J (1970) Increases in water yield following clear-cut logging in the Pacific Northwest. Water Resour Res 6:653–658

Sarkkola S, Koivusalo H, Laurén A (2009) Trends in hydrometeorological conditions and stream water organic carbon in boreal forested catchments. Sci Tot Env 408:92–101

Schindler DW (1998) Sustaining aquatic ecosystems in boreal regions. Conserv Ecol 2:18

Scott DF (1993) The hydrological effects of fire in South Africa mountain catchment. J. Hydrol. 150 (2-4); 409-432

Sidle RC, Ziegler AD, Negishi JN et al (2006) Erosion processes in steep terrain – truths, myths, and uncertainties related to forest management in Southeast Asia. For Ecol Manage 224:199–225

Singer A (2007) The soils of Israel. Springer, Berlin, Heidelberg, New York

Swiss Re (various dates) Sigma. Swiss Reinsurance, Zurich

Thornthwaite CW (1948) An approach toward a rational classification of climate. Geograph Rev 38:55–94

Tikkanen M (2006) Unsettled weather and climate of Finland. In: Lindholm T, Heikkilä R (eds) Finland – land of mires. Finn Environ 23:7–11

Troendle CA (1983) The potential for water yield augmentation from forest management in the Rocky Mountain region. Water Resour Bull 9:359–373

Van Dijk AIJM, Keenan RJ (2007) Planted forests and water in perspective. For Ecol Manage 251:1–9

Wakindiki IIC, Ben-Hur M (2002) Soil mineralogy and texture effects on crust micromorphology, infiltration and erosion. Soil Sci Soc Am J 66:897–905

Waterloopkundig Laboratorium (1994) Onderzoek Watersnood Mass. Deerlaport No 4. Hydrologische Aspecten

Watson DA, Laflen JM (1986) Soil strength, slope rainfall intensity effect on interrill erosion. Trans Am Soc Agric Eng 29:98–102

Watson F, Vertessy R, McMahon T, Rhodes B, Watson I (2001) Improved methods to assess water yield changes from paired catchment studies: application to the Maroondah catchments. For Ecol Manage 143:189–204

West LT, Chiang SC, Norton LD (1992) The morphology of surface crusts. In: Sumner ME, Stewart BA (eds) Soil crusting: chemical and physical processes. Adv Soil Sci. Lewis Publishers, Boca Raton, FL, pp 73–92

Zhang I, Dawes WR, Walker GR (2001) Response of mean annual evapotranspiration to vegetation changes at catchment scale. Water Resour Res 37:710–708

Chapter 16
Forest Management Effects on Below-Ground Hydrological Processes

Viliam Novák, Viliam Pichler, Elisabeth Graf-Pannatier, Edward P. Farrell, and Marián Homolák

Intrinsic Variability of Soil Hydrologic Processes

Below ground movement of water can be viewed from various perspectives. It can be regarded within the soil–plant–atmosphere system framework, mainly when assessing the fluxes of water and energy in forests (Valentini 2003), or as a process determining ground water recharge, its fluctuations and the transport of solutes. Both perspectives are important in the realms of forest production and regulation functions. However, studies of the forest management impact on below-ground hydrologic processes have been sparse compared to the number of studies conducted in agro-ecosystems. It seemed unlikely in the 1980s that the future would see any heightened interest in the management of water relations in forests, but the present situation often requires active forestry measures, such as forest density reduction to improve extractable water availability and water yield from forests (Čermák et al. 1993). Although some findings can be transferred from agricultural to forest soils, some fundamental differences remain. The basic differences in soil water regimen between relatively homogeneous

V. Novák (✉)
Department of Soil Hydrology, Institute of Hydrology, Slovak Academy of Sciences,
Racianska 75, 831 02 Bratislava 3, Slovak Republic
e-mail: novak@uh.savba.sk

V. Pichler
Department of Natural Environment, Technical University in Zvolen, T.G. Masaryka 24,
960 53 Zvolen, Slovak Republic

E. Graf-Pannatier
Soil Sciences Unit, Eidg. Forschungsanstalt WSL, Zürcherstrasse 111, CH-8903
Birmensdorf, Switzerland

E.P. Farrell
Forest Ecosystem Research Group, Department of Environmental Resource Management,
University College Dublin, Belfield, Dublin, Ireland

M. Homolák
Department of Natural Environment, Technical University in Zvolen, T.G. Masaryka 24,
960 53 Zvolen, Slovak Republic

M. Bredemeier et al. (eds.), *Forest Management and the Water Cycle: An Ecosystem-Based Approach*, Ecological Studies 212, DOI 10.1007/978-90-481-9834-4_16,

arable land and stony, forest soils arise from heterogeneity and stoniness of forest soils, resulting in preferential paths for water. Another, but important difference is the significant interception of rain by forests in comparison to agricultural crops, leading to lower soil water content in forest stands.

Originally, soil water transport was thought to occur almost exclusively within the homogeneous soil matrix, and broad applications of the one-dimensional Richard's equation reflected this approach. Thus, Richards' equation describes water movement in homogeneous soil without preferential paths:

$$\frac{\partial \theta}{\partial t} = \frac{\partial}{\partial x}\left(K(h)\frac{\partial h}{\partial x} + 1 \right) - S(x,t) \tag{16.1}$$

where, θ is the soil water content (L^3L^{-3}), h is the soil matric water potential (L); K(h) is the hydraulic conductivity as a function of h (LT^{-1}); x is coordinate (L), and t, is time (T). S(x,t) is rate of soil water extraction by roots ($L^3L^{-3}T^{-1}$). At the same time, early preferential flow observations (Rode 1952) were considered an exception rather than a rule, partly due to the lack of experimental evidence and the use of instruments unsuited to the study of preferential flow, e.g. tensiometers. This status-quo began to change from the late 1970s and beginning of 1980s when several important works, e. g., Germann and Beven (1981), highlighted preferential flow through macrcopores as an important hydrologic phenomenon in soils, and in forest soils in particular. Recent findings clearly contradict predictions made by application of Richards' equation, which describes flow in soil without preferential paths. The reason for this failure can be also due to differences in contact angle of soil–water–air system, or due to organic matter coating (Novák et al. 2009). Extreme nonhomogeneity and presence of preferential paths in forest soil leads to nonhomogeneous and sometimes unstable flow (Fig. 16.1). That could explain why unstable flow is not predicted by widely used simulation model codes (Jury et al. 2003).

Both major flow types, i.e. the matrix flow and the preferential flow have a potential to either promote or compromise various ecosystem services provided by forests. They include immission load buffering (Pichler et al. 2006), waste water purification (Kermen and Janota-Bassalik 1987; Kim and Burger 1997; Barton et al. 1999; Livesley et al. 2007), the degradation of pesticides used in forestry, groundwater recharge (Aussenac 2000; Pichler 2007), runoff moderation (Badoux et al. 2006), carbon sequestration (Qualls et al. 2002) and many others.

In the light of these findings, numerous concepts have been developed with the aim of describing the movement of water including preferential flow, e.g. two-domain model that splits soil water into mobile and slowly mobile portions (Skopp et al. 1981), the dual – porosity approach (Gerke and van Genuchten 1993), the kinematic wave model (Germann and Beven 1985) and the stochastic-convective flow model with gravity being the main driving force (Jury and Scotter 1994), or combinations thereof (Steenhuis et al. 1990).

It follows that there is an intrinsic pattern of water and solute transport variability in soils and that to capture forest management effects against this backdrop poses a difficult task. Therefore, new dye tracer techniques and image analysis methods have been used in some forest soils studies (Stadler et al. 2000; Bogner et al. 2008). One

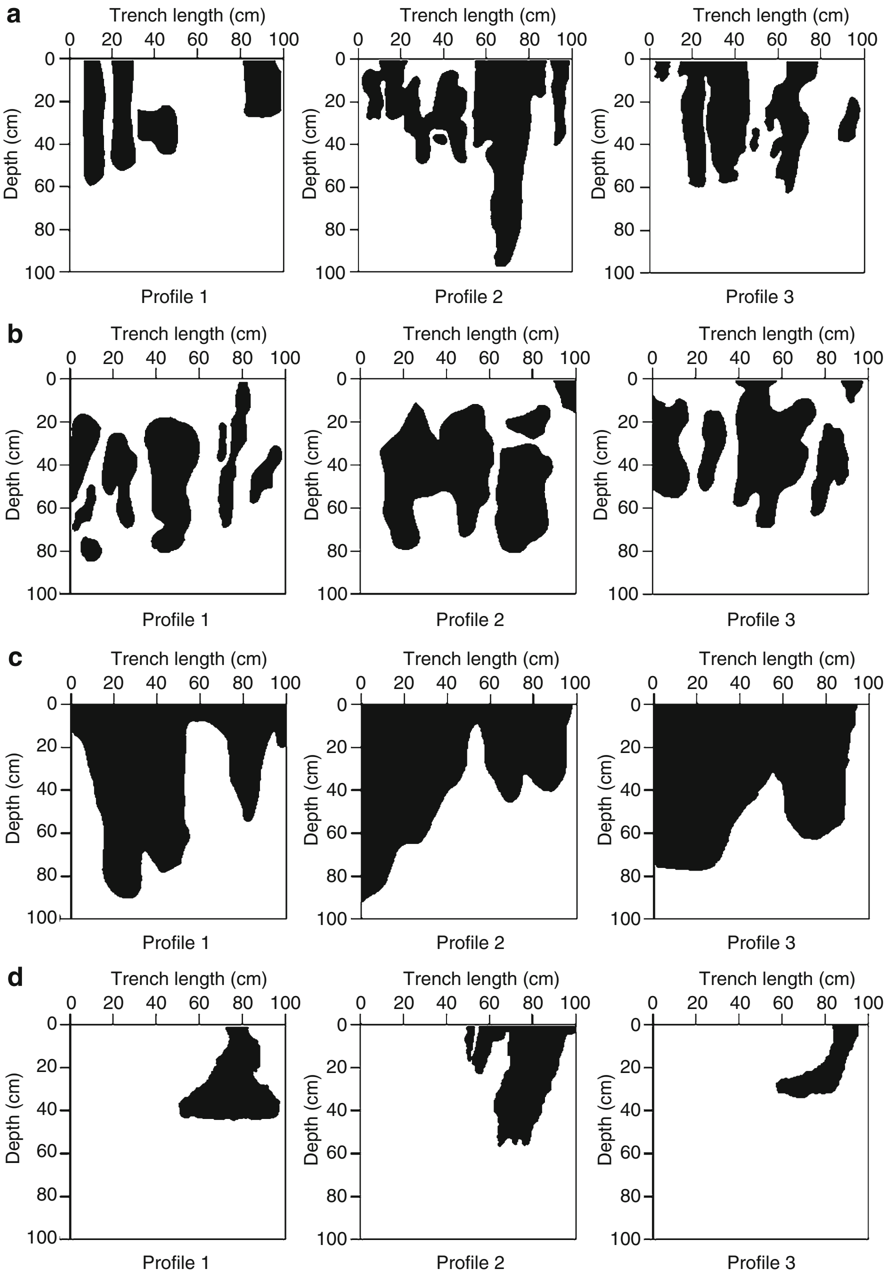


Fig. 16.1 Various types of water flow in soils depending on beech forest density as visualized by a dye tracer. (**a**) Prolonged infiltration plumes in a natural stand; (**b**) biomacropore flow, montane zone; (**c**) moderately heterogeneous matrix flow, clear-cut, motane zone; (**d**) highly heterogeneous matrix flow, stem-flow zone

Table 16.1 Common forest management practices and their objectives

Management practices	Objectives
Choice of tree species and structure	Improvement of water quality and quantity (hydraulic conductivity)
	Health of forests
Thinning	Biomass harvesting
	Recharge of aquifers
	Wood quality
	Increase of sturdiness of leaving trees
Clear-cutting, whole tree harvesting	Biomass harvesting for energy
Change in rotation	C storage
No management	Biodiversity, sustainability
	To preserve existing water quality and quantity
	Health of forests
	C storage
Fertilization	Increase of wood production
	Improvement of nutritional status
	Health of forests
	C storage
Site preparation	Improvement of regeneration and initial growth of seedlings
Ditching	Drainage
Prescribed burning	Forest fires prevention, fertilization, improvement of regeneration
Change in rotation	C storage

important question that can possibly be clarified through their application is the influence of forest management on the dominant type of soil water flow on the plot scale. The management practices with their objectives are summarized in Table 16.1 and their impact will be discussed through the following subchapters.

In the process of assessing the impact of forest management practices on below-ground hydrological processes, a comparatively higher soil skeleton content (Šály 1986), spatio-temporal variability of the vegetation cover (Kenderes et al. 2008; Nagel and Diaci 2006) and the presence of surface humus must be considered as important factors.

Above-Ground Biomass

Forest management and silvicultural practices, such as the choice of tree species composition, forest type, management system, thinning and biomass harvesting can significantly influence below-ground hydrologic processes. Our analysis starts with the controls on the three basic canopy characteristics used for hydrological processes modeling:

- Leaf area index (LAI)
- Albedo of the evaporating surface
- Roughness of the evaporating surface

Precipitation (in liquid and solid phase) is partly intercepted by canopies and evaporates back to the atmosphere, consuming latent heat of evaporation as a significant part of the net radiation in the Soil–Plant–Atmosphere System (SPAS). In the temperate zone, it amounts up to 80% of the net radiation during the vegetation period and about 40% annually (Novák 1995). Canopy interception of precipitation by forest stands is significant, particularly in coniferous forests. To comprehend the spatio-temporal dynamics and variability of below ground hydrological processes, it is necessary to appreciate forest interception, which not only influences the soil water regime, but can be relatively easily affected by forest management systems and interventions. In particular, they often result in changes of both canopy closure and forest density, such as in the shelterwood system.

High value LAI of forests in comparison to grass and crops and some particular properties of leaves (needles) results in high interception capacity of forests in comparison to crops. Interception losses vary due to various meteorological factors, such as air temperature, wind velocity, rain intensity, and canopy properties like species composition, stand density and canopy closure. On average, 37% of the total precipitation was intercepted by a temperate 80 year old Norway spruce forest canopy, compared to 18% intercepted by a deciduous forest at the same locality during ten vegetation periods (1981–1990) in Central Slovakia. Ten years of continuous measurements during snow-free season of spruce forest, indicated a maximum daily interception of about 5 mm under conditions of low wind velocity (Miklánek and Pekárová 2006; Novák and Kňava 2008). Kantor (1984) found considerable difference in annual interception between mountain spruce (212 mm) and beech (87 mm) forests. Therefore, the choice of tree species is of a paramount importance in influencing interception losses and infiltration rate, while considering site requirements and natural habitats of different species. Although similar evapotranspiration totals was estimated for both species (conifers, deciduous), the transpiration rates usually decrease with the reduction of forest density and LAI (Holst et al. 2004). On the other hand, even-sized forest stands with distinct LAIs may produce similar evapotranspiration rates if they are structurally rich. In this sense, forest evapotranspiration may be considered a conservative hydrological process (Roberts 1983, 2000).

In addition, albedo and the roughness of canopies, which are dependent on tree species composition and forest structure, affect both the energy balance and evapotranspiration rate (Hlavatá, et al. 2008). The less energy is needed for the evaporation of intercepted water, the more of it can be spent on transpiration. In spite of the fact that the water fluxes have been studied in several species (e. g., Aussenac 2000; Granier et al. 2003), information on the soil water content (SWC) differences resulting from various tree species compositions, forest densities and structures is limited. Aussenac and Granier (1988) reported a maximum 42 mm of SWC difference between a natural and a low density Douglas fir stand. Gregor (1999) identified significant soil moisture differences up to 5–10% between a shelterwood stand and a natural stand in a lower altitude beech forest.

Koshi (1959) and Breda et al. (1995) established that SWC values in low density oak stands averaged 25 or 40 mm higher than in natural stands under semi-arid or temperate conditions, respectively. Pichler et al. (2009) found a significant forest density effect on the SWC under a montane Western Carpathians beech forest (Fig. 16.2). In the natural forest, SWC was in average by 40 mm lower than SWC of shelterwood stand in the respective zone.

Such differences result from a comparatively lower interception and a reduced transpiration in low-density shelterwood stands featuring a reduced basal area and a simplified age–size structure. Two main factors usually contribute to large soil water content differences: (i) the dominance of a hormonally controlled seasonal regulation of growth under optimum soil moisture conditions; (ii) the absence of suppressed, overtopped trees in the shelterwood stand. From an intraspecific competition point of view, suppressed trees of some species show a high responsiveness to global irradiance. (Střelcová et al. 2002), and sudden increase in water extraction (Ježík and Voško 2002; Pichler et al. 2009) that manifests itself in the increase of sapflow and stem diameter, respectively. In other words, suppressed trees sustain transpiration under cloudy skies and may utilize instant SWC spikes, such as following intense precipitation, even if main canopy trees do not respond. Also, the fraction of stand evapotranspiration originating at the understorey increases as the water vapour pressure deficit increases and the soil dries (Black and Kelliher 1989). However, many studies have shown that

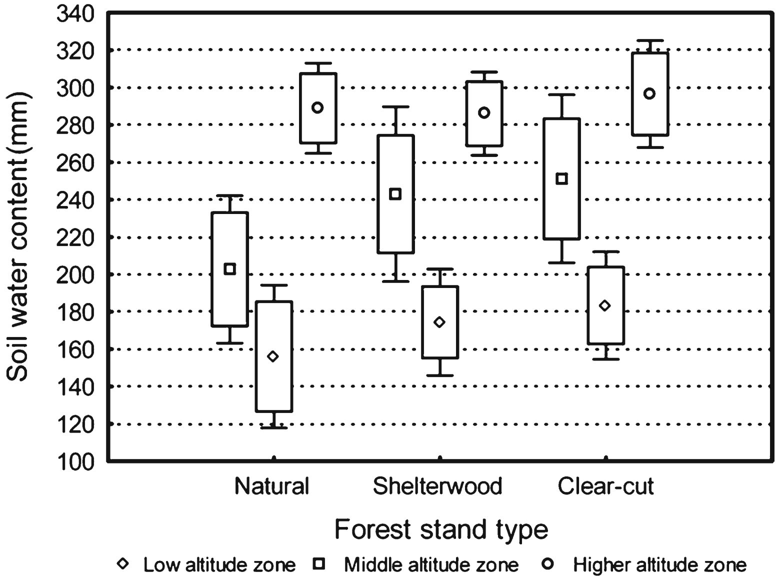


Fig. 16.2 Soil water content (SWC) under beech forest stands in the submontane, montane and supramontane zone. The mean values are represented by *markers*, 95% confidence intervals coincide with the *boxes* and *whiskers* indicate the standard deviation (Pichler et al. 2009)

water yields will return to pretreatment levels approximately 10–30 years after forest density reduction (Bosch and Hewlett 1982; Reiter and Beschta 1995; Stednick 1996; Scherer and Pike, 2003).

However, significant SWC differences between natural and shelterwood stands rarely occur in the arider submontane and moist supramontane zones due restricting climate patterns. It means that potential SWC differences due to distinct forest densities will likely be dampened by the precipitation deficit at lower altitudes or in arid conditions. Such conditions usually lead to the lack of extractable soil water as indicated by soil matrix potentials below approximately −700 hPa, at which the transpiration rate of some tree species adapted to the oceanic or suboceanic climates may decrease (Papritz et al. 1991; Pichler 2007). Consequently, timber harvesting disturbs the water balance in forests by causing the largest increases in water yield in areas where the moisture content of the soil is high during the growing season, or in the streamflow generation zones of a watershed. On the other hand, some evidence indicates that in tropical and subtropical areas logging a watershed may decrease water yield (after a brief period of increase) in contrast to what is expected in temperate regions (Reifsnyder 1988; Creedy and Wurzbacher 2001).

Moreover, the forest canopies cause deterministic processes that consistently redistribute precipitation to create patterns of throughfall that manifest a considerable temporal persistence (Keim et al. 2005). One of the most distinctive mechanisms that also produce spatial patterns of stand precipitation is stemflow. It occurs mainly in various beech species due to the specific habitat of the genus *Fagus* (Gersper and Holowaychuk, 1970; Johnson and Lehmann, 2006), but it can be observed in some other species to a lesser extent as well. Jochheim and Schäfer (1988) established that the beech stemflow zone received eight times more water enriched by particles and dissolved chemical compounds in comparison with areas only exposed to throughfall. Holko et al. (2009) found up to six times higher incoming water in the near stem area in comparison to the drip zone of spruce forest in High Tatras.

Raat et al. (2002) found a correlation between throughfall variability and surface humus layer wetting in a Douglas fir forest plot. Spatial patterns of throughfall water fluxes and forest floor water contents were consistent over time; but because of the spatial variability of forest floor thickness and drainage, it was not possible to relate patterns in throughfall water directly to patterns in water content. Even a more pronounced variability exists in forests subject to nature conservation or close-to-nature management regimes, in which the long-lived trees gain control over the water cycle by depositing organic debris from their photosynthesis. The presence of coarse woody debris in varying stages of decay is characteristic of unmanaged forests (Fig. 16.3). Studies have shown that the ability of coarse woody debris to store water increases with decay (Harmon and Sexton 1995; Brais et al. 2005). The issue of water redistribution on the forest soil surface is of a paramount importance from the viewpoint of the prevailing flow types in soils and as such it will be dealt with in the next chapter.

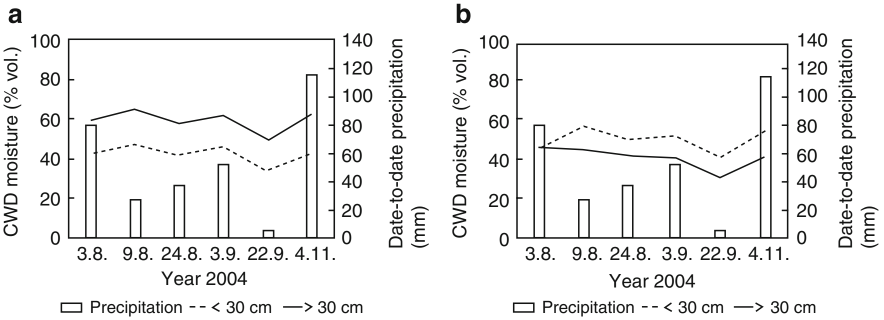


Fig. 16.3 Coarse woody debris moisture content (vol. %) in the second (**a**) and third (**b**) decomposition stage of fir logs (Poľana Biosphere Reserve, Western Carpahians, Slovakia). The temporal variability of coarse wood debris moisture increases with the advancing decomposition

Soil Surface Interface

Throughfall, canopy drip and stemflow infiltrate into soil. Part of the infiltrating water may directly recharge the groundwater, while another part accumulates in the soil (soil water retention) and becomes subject to subsequent redistribution, thus being a source of water for tree, brush and herb layer transpiration via root extraction; a small part of the water is evaporated from the soil surface. Richards' equation (16.1) used for solving water infiltration into soils can be rewritten in the form (Kutilek and Nielsen 1994):

$$\frac{\partial \theta}{\partial t} = \frac{\partial}{\partial z}\left(D(\theta)\frac{\partial \theta}{\partial z} \right) + \frac{\partial K}{\partial z} \tag{16.2}$$

$$D(\theta) = K(\theta)\frac{\partial h}{\partial \theta} \tag{16.3}$$

where term D is so called diffusivity, L $T^{-0.5}$ (it has nothing common with coefficient of diffusion, only its dimension is the same). Philip (1957a) solved Eq. (16.2) for a constant water content (or matric potential) and obtained a solution in the form of a depth below the soil surface z at which there is water content θ at a time t

$$z(\theta, t) = \lambda(\theta)t^{1/2} + x(\theta)t + \varphi(\theta)t^{3/2} + \ldots \tag{16.4}$$

where $\lambda(\theta)$ is termed sorptivity (S) and $x(\theta)$, $\varphi(\theta)$ account for gravity. Then, infiltration can be rapidly evaluated by an iterative procedure developed by Philip (1957b) and described in detail by other authors (e.g. Kirkham and Powers 1972; Warrick 2003). For practical purposes, $x(\theta)$ is sometimes replaced by K (saturated hydraulic conductivity in LT^{-1}). Solution (16.4) shows that the rate of infiltration is decreasing with time, finally reaching the rate equal to saturated soil hydraulic conductivity.

The maximum infiltration rate under given conditions, either ponded or non-ponded, is known as an infiltration capacity, which is typically high in forest soils, so that surface runoff can only rarely be observed. This is partly owing to the protection of soil surface from sealing, provided by the surface layer of undecomposed organic material, preventing soil erosion and allowing water entry into soil. The retention capacity of this layer is generally very small; but it serves as efficient highly conductive layer for water entry. The following layer of surface humus contributes to further spatial redistribution of stand precipitation, including throughfall, drip and stemflow (precipitation minus interception) and affects the very initial stage of infiltration process. Although the influence of vegetation on the distribution of water that reaches the surface of hillslope soils (as throughfall) has been studied in many ecosystems, the influence of the leaf litter layer on initiating infiltration and solute transport pathways in nonagricultural hillslope soils has not received much attention (Albers and Migge et al. 2004).

The role of surface humus forms on the flow types occurring in forest soils may well require a new classification of flow types defined not only by the dye stains horizontal width distribution (Weiler and Fluehler 2004), but also by the compartment or interface in/at which a particular flow type evolved. In this framework, the surface controlled flow type includes macropore flow and heterogeneous infiltration characterized by prolonged vertical dye tracer plumes protruding from the soil surface to deeper soil layers. In contrast, the matrix controlled flow type encompasses both homogeneous and heterogeneous matrix flow, as well as fingering. It follows that the distinct humus forms, as associated with different forest sites, tree species compositions, structures and development phases will influence the prevailing flow types, whose role has been explained before, in a predictive manner. While zero intervention management policy results in characteristics found in primeval forests, an application some kind of the selection system, continuous cover forestry methods and conservation management may approximate the low to intermediate disturbance regime with a high forest density. In such cases, moder humus forms, which contain a holorganic horizon, often prevail. This horizon is colonized by fine tree roots and their associated mycelia, mostly ectomycorrhizal basidiomycetes. The water capacity of moder humus layer as the interface between infiltrating water and the topsoil can be seen on Fig. 16.5c, where it directs the input of water into clearly visible funnels. For instance, there is a tendency of the beech litter to partially concentrate percolating water into streams (Walsh and Voigt 1977). They also observed that the formation of water rivulets is often accompanied by flow pulsation. These processes are promoted by the laminated mixing of leaf fragments and feces of litter-consuming enchytraeids, called "sandwiched material" by Ponge (1999). For the same reason, a considerable proportion of the deeper surface humus layers remain unwetted. Thus, under some conditions, concentrated water flows are dissipated into the grainy top (0.3–0.5 cm) organic horizon (O_h) and further down to the mineral soil after the wetting and passage through patches within the layer 1.5–2.5 cm (O_f) in the form of prolonged vertical plumes (Fig. 16.4).

As opposed to that, the removal of stands by the clear-cutting system has distinct effects on the infiltration and redistribution rate, which is generally increased, due

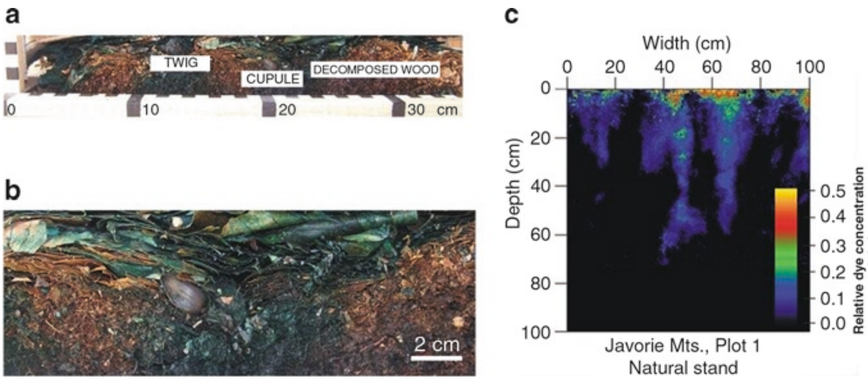


Fig. 16.4 (**a**) Dysmoder features sandwiched material that results from a laminated mixing of leaf fragments and feces of litter-consuming enchytraeids (natural stand, Javorie); (**b**) gradual wetting of the O_f layer forms a patchwork of permeable spots; (**c**) dye tracer plumes disperse as they protrude vertically to deeper soil layers

to the interception decrease. Firstly, the input of litter drops sharply and so the organic layer preventing drops impact is continuously diminishing. Then, the matrix flow is further promoted by a rapid forest floor mineralization that in turn reduces the water repellency of the soil surface. As observed by Ziegler and Zech (1989) and Dinel et al. (1990), the concentration of hydrophobic lipid compounds decreases when the decomposition becomes more rapid. Pronounced hydrophobicity effect may also be absent due to the humid and cooler climate, or where the moder humus effectively protects the amphiphilic compounds from drying. Specifically, drying is a precondition for pronounced hydrophobic behaviour (Novák et al. 2009). Hydrophobicity, as a phenomenon was not proved to play significant role in soil water movement, but it can appear in soils subject to drying and forest fires.

Finally, in stands, where stemflow is substantial, clear cutting shifts infiltration away from concentrated inputs into developed preferential flowpaths toward diffuse infiltration through the soil matrix. As a middle case, forest management systems and interventions that rest on an intermediate disturbance regime seem, such as the shelterwood system, sometimes accompanied by soil scarification, seem to promote bimacropore flow will play an important role in their soils, as explained in the next subchapter.

Vadose Zone and Groundwater

To fully understand hydrological processes in the forest soils in a comprehensive manner, it is necessary to mention some of their specific properties. However, these properties are rather stable and opportunities to affect them are limited when compared to agricultural soils. The rooting zone of forest soils is generally shallow; the cultivation of deep-rooting tree species increases the root density in the subsoil and therefore stimulates preferential infiltration to deeper, hydromorphic horizons and enlarges the water-storage space. However, the roots, either living or decaying, would likely play only a minor role in determining the flow if their horizontal or nearly horizontal growth prevails. This is not an unusual phenomenon, except for the zones proximal to the tree stems.

In the previous chapter, the role of surface humus in the process of infiltration was discussed. For instance, the preferential flow occurred with the intact oak forest litter in place, but it vanished when the litter was removed (Campbell et al. 2004). Generally, surface organic layers should minimize preferential flow during the infiltration, by the improving regularity of water distribution over the surface. But the surface humus effect will be either reinforced or dampened by the mineral soil properties, such as the presence of biomacropores created by anecic earthworms. Partial control on their populations may be introduced by forestry interventions such as shelterwood cuts that induce favorable conditions for the formation of mull humus forms. The reason consists in more favourable light, warmth and aeration conditions. Specifically, oligomull hosts earthworm communities composed of epigeic, anecic and endogeic species (Ponge and Delhaye 1995). Under such conditions, a certain portion of the organic top layer (O_f) discontinuities, typical of mull (Brêthes et al. 1995), overlap with the microrelief depressions and burrow openings. Therefore, patterns of infiltration under the frequently represented mull forms (e.g., oligomull and mesomull), as observed under the shelterwood stands, appear rather distinct from those exposed under the moder forms. As expected, fewer earthworm burrows were observed under natural stands with moder and in clear-cuts. In the former case, usually associated with the moder humus form, enchytraeids prevail over earthworms, while the endogeic biomass collapses within clearings (David et al. 1991; Nachtergale et al. 2002). In contrast, the characteristic occurrence of isolated dye stains under shelterwoods stands indicates the biomacropore flow (Fig. 16.5).

The activity of soil biota can strongly influence soil hydrophysical and hydraulic properties. The macropore flow due to subsurface initiation may suddenly occur under specific conditions, e. g. in layered soils with different characteristics (Torres 2002). Here we provide two illustrations of typical forest soils: a Dystric Cambisol on a moraine (Gőmőryová et al. 2008) contained an average of 60% of sand particles and 28% of silt (0.05–0.002 mm) (Bärwolf 2006). However, a significant stony fraction content (1–20 cm in diameter) was detected. In contrast, the textural composition of an Mollic Andosol was made up of only 35% of sand and 52% of silt. The soil's bulk density was only 0.80 g cm^{-3} and the matrix or uniform flow prevailed as the dominant process of the respective soil drainage. Other observations on andic soils (Magesan et al. 2003; Bartoli et al. 2007; Eguchi and Hasegawa 2008) produced similar conclusions.

Low soil bulk density values and high soil porosity are also typical features of forest soil matrix (fine earth). They may be expected to result in high hydraulic conductivity values (181–1,290 cm day^{-1}) and high infiltration capacity (Novák and Kňava 2008). In contrast, retention capacity of forest soils, containing a high ratio of rock fragments is relatively low. Because forest soils in mountainous regions are

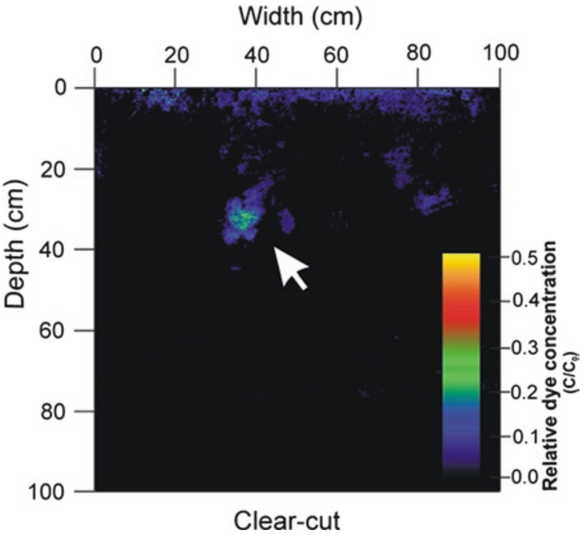


Fig. 16.5 Isolated earthworm borrow opening surrounded by stained matrix visualized by Brilliant Blue FCF dye tracer applied in a shelterwood stand (Homolák unpublished data)

often comparatively coarser and shallower due to their origin from slope deposits (Šály 1986), the stochastic-convective flow theory can often be applied to the respective transport of water and solutes. In this, a solute added uniformly to the soil surface is transported through stream tubes that form near the surface and remain isolated thereafter (Jury and Scotter 1994). Such conceptualizing can be justified for cases, in which the time required for mixing between stream tubes exceeds the time to reach depths near the surface.

If the water supply on the soil surface is sufficient, a portion of water will reach the groundwater table, mainly if it is located at shallow depths. Interestingly, quantitative information regarding the effects of mountaineous forest management versus no management on groundwater resources is sparse. This may be because the effects of forest management on groundwater are sometimes considered to be less important (Scherer and Pike, 2003) and because groundwater table depth is difficult to identify and measure on slopes. On the other hand, alluvial forests overlying shallow groundwater table (GWT) are contributing to the intense recharge rate and good quality of groundwater (Koivusalo et al. 2008). Also in clearfells, near-surface temporary water tables can be formed (Aussenac 2000). Specifically, tree density reduction in the shelterwood stands and seed felling strips allow the water table to be fed via preferential flow initiated at the soil surface interface. For example, it was observed that the SWC under a natural Carpathian beech forest continued to decrease by 1.5 mm day^{-1} during a dry period, while a moderate rise (ca. 1 mm day^{-1}) was observed under the shelterwood stand. This seemingly paradoxical phenomenon occurred due to a gradual capillary rise from water-logged slope deposits, supplied with water via preferential biopore flow during a preceding wet period.

Subsequently, a portion of the infiltrated water returned to the soil matrix by the way of rising capillary fringe.

Recently, revitalization of rivers has become an issue in many European countries. Strategy favouring river revitalization projects has an impact on forest management, since the forests that were generally rather dry hardwood forests under the canalized regime become alluvial forests with a new succession of vegetation. The increase in area of alluvial forests in Europe will be probably a trend in the near future with the river revitalization strategy, as opposed to formerly preferred canalization and drainage practicies (ditches) applied in the lowland forests.

Effects of Forest Management on the Quality of Seepage Waters

The water from forested catchments is usually of good quality compared to water from other land uses such as agricultural lands (De la Crétaz and Barten 2007; FOEN 2009). Therefore groundwater from forested areas is pumped for drinking water in several regions of Europe. However, air pollution and some forest management practices might affect the quality of water in forested areas. Forest management, including the choice of tree species, regeneration method, type of cutting, harvesting intensity, fertilization or liming may alter the quality of water resources (surface, seepage and ground water) in forested watersheds. The major issues related to water quality are the acidification of water resources, eutrophication of the aquatic system, nitrate contamination of groundwater, remobilization of heavy metals and organic pollutants, soil erosion and export in suspended particles into surface waters (Meesenburg et al. 2005).

In this chapter, we focus on management practices related to the forest composition and structure and on their effects on the chemistry of seepage waters through the forest soil and below the rooting zone. While the effects of forest management on water quality have been largely investigated and reviewed in the temperate regions of Europe (Hegg et al. 2004; Gundersen et al. 2006) and in Northern America (Brown and Binkley 1994; Stednick 2000), very little literature is available from the Mediterranean countries. The main issue related to the quality of seepage waters is nitrate, particularly in areas with chronic high nitrogen atmospheric deposition. Therefore, in this subchapter it will be analysed mainly nitrogen leaching.

Phosphorus leaching is rarely an issue except in peatland forests, where particularly following the application of fertilizer during the establishment of plantations, it may occasionally represent a significant source of eutrophication. While phosphorus leaching is rarely, if ever a problem in mineral forest soils, in many peat soils, the low clay, iron and aluminum contents, all of which reduce the solubility of phosphate (Nieminen and Jarva 1996) facilitate its mobilization. Surface runoff on peat soils of very low saturated hydraulic conductivity exacerbate this problem in the establishment phase. The leaching of phosphate to water bodies from forested peatlands had been widely investigated (Gibson 1976; Kenttamies, 1981; Ahti, 1983; Malcolm and Cuttle, 1983; Nieminen and Ahti, 1993; Joensuu et al. 1998; Nieminen 1999; Cummins and Farrell 2003).

Type of Cutting and Regeneration

The removal of the canopy during clear-cutting leads to abrupt changes in microclimate, throughfall input, evapotranspiration, nutrient uptake and litter input, which might enhance mineralization and nitrification, further acidification in the soil, leaching of nitrate, aluminium and nutrient cations (Robertson et al. 2000; Huber et al. 2004a). The strong decrease in nutrient uptake by the mature stand, the decomposition of the old litter, logging residues and dying fine roots release a substantial quantity of nitrogen.

Increased nitrate concentrations in seepage water after a clear-cut or comparable natural calamities like windthrow or insect outbreak have been recorded in many studies (Dahlgren and Driscoll 1994; Huber et al. 2004a, b; Weis et al. 2006; Legout et al. 2009). The registered high N output after clear-cuts result from the higher infiltration in the soil and elevated nitrate concentration due to enhanced nitrification. However, these high outputs of nitrate occur mostly during the first years after cut. If the high nitrate concentrations occur in fast draining soils, wells for the supply of drinking water may be affected at least temporarily depending on the size of the clear-cuts. At sites saturated with nitrogen high values have been reported reaching up to 200 mg N l^{-1} after clear-cut or windthrow (Huber et al. 2004a; Mellert et al. 1996). In contrast, several investigations of clear-cuts at nitrogen-limited sites reported no or only slightly increases of the nitrogen concentration (e.g Brown and Binkley 1994; Grace 2005). Stednick (2000) reported in his review that nutrient losses from silvicultural activities to be minimal and water quality not degraded throughout the United States.

Alternatives to clear-cut harvest include regeneration in gaps, selective cutting beneath the old trees to maintain a continuous forest cover. These practices might create forest of mixed ages with continuously higher N demand and N retention capacity on a regional scale (Gundersen et al. 2006). Stand age is an important parameter affecting N uptake rate in forest ecosystems (Horváth et al. 2009). Losses of nitrate usually increase as the growth rate of the forest declines. Rothe and Mellert (2004) observed a clear age-related pattern in spruce stands with negligible nitrate concentrations in stands younger than 40 years and higher concentrations in older stands (up to 100 years). In beech stands, nitrate concentrations were low and independent of age. The regeneration of Norway spruce with European beech in the shelterwood system as an example for continuous cover forestry might prevent high losses of N, Mg and Ca from the soil during the first years after initializing regeneration (Huber et al. 2004b; Rothe and Mellert 2004; Weiss et al. 2006). Whether or not the differences to clear-cut plots are significant depends on the N availability of the forest site (v. Wilpert et al. 2000), the abundance of ground vegetation and regeneration (Emmett et al. 1991a,b; Weis et al. 2006; Legout et al. 2009), its capability to store nutrients and the intensity of the shelterwood cut (Huber et al 2004b). The formation of gaps creating openings in the canopy might result in an increase in N deposition. In forest edges, throughfall deposition is increased up to four times compared to the forest interior (Wuyts et al. 2008).

Clearcutting of peatland forests can result in significant losses of mobile phosphorus to surface waters. The principal sources of phosphorus in this case are decomposing foliage and the decaying root system. It is likely that most of the loss of phosphate occurs in surface runoff.

Harvest Intensity

The rotation length and the degree of biomass removal during harvesting influence the magnitude of exports of nutrients from the system. Short rotations produce small amounts of biomass that is relatively high in nutrients and longer rotations produce more wood with lower nutrient concentrations (Fisher and Binkley 2000). The removal of logging residues during harvesting (whole-tree harvest) allows the reduction of nitrogen stocks in the forest ecosystem and the N export through the decomposition of the dead wood. However, Gundersen et al. (2006) did not find the general difference in nitrate leaching following stem-only harvest and whole-tree harvest but site specific differences are reported depending on pre-existing site conditions.

With the removal of logging residues, substantial amounts of nutrients are removed from the ecosystem, leading after several rotations to a depletion of base cations and phosphorus. The weathering, biological fixation and deposition cannot compensate for the removed quantities. Whole-tree harvest may result in net losses of P and in soil acidification (Gundersen et al. 2006). Stem-only or partial harvest logging reduces the impacts on nutrient release and exports in comparison to whole tree harvest (Kreutzweiser et al. 2008). Differences between tree species also exist. Merino et al. (2005) showed that nutrients exports due to harvesting of *Pinus pinaster* were considerably lower than losses caused by removal of *Eucalyptus globulus*.

Whole-tree harvesting is often associated with other disturbances. It includes mechanical disturbance with delayed re-growth of ground vegetation, damage to forest tree seedlings, compaction of the soil and less woody debris to support biodiversity and retain water and nutrients (Mou et al. 1993). Soil compaction and rutting caused by heavy forest machinery change physical soil properties by reducing soil infiltration capacity, soil aeration and by increasing soil resistance to root penetration (Kozlowski 1999; Grigal 2000). These changes might lead to anaerobic conditions and result in changes in the quality of seepage waters (Briggs et al. 2000). Road construction, skidder traffic and intensive site preparation may cause soil erosion, export of N and nutrients in suspended particles to forest streams and thus increase turbidity in streams (Grace and Clinton 2007; Stednick 2000). Increased N mineralization and nitrate losses might occur after mechanical operations such mounding, harrowing or scarification to improve soil conditions (Kubin 1998; Smolander et al. 2000; Piirainen et al. 2007).

Ditching

Artificial drainage or ditching in forested wetlands has been a common practice, especially in Fennoscandinavia, for gaining more forest areas, increasing forest productivity and improving regeneration. In boreal zone it has been found that nitrate leaching from the forested areas is generally low and most of the nitrogen load is consisting of organic nitrogen (Mattsson et al. 2003; Nieminen, 2003).The drainage of wetlands usually lead to the oxidation of organic compounds and the mobilization of large amounts of stored organic nitrogen (Gundersen et al. 2006) and to increase the load of suspended material in streams (Manninen 1998). Callesen et al. (1999) found high nitrate concentrations up to 140 mg NO_3-N l^{-1} in the soil solution in a drained and reafforested bog, which was probably caused by the mineralization and further nitrification of nutrient-rich organic matter accumulated in the bog. Prévost et al. (1999) also found that drainage favored aerobic mineralization and increased nutrient concentrations in the soil solution of forested peatland. In contrast Åström et al. (2002) found only a small impact of ditching on the leaching and export of nutrients to the streams in boreal forests in Finland. In particular they did not observe any change in nitrate leaching after ditching, which was explained by the high C:N ratio and the high acidity of the forest floor, leading to suppression of nitrification processes (Åström et al. 2005). Soil properties, especially its hydraulic conductivity, strongly influence drainage rate. The influence of timber harvesting on water quality differs also with the degree of drainage in the forest soil. Briggs et al. (2000) reported that nitrate concentrations increased after whole-tree clear-cutting in moderately well-drained soils, while they did not differ in poorly drained soils from those in the uncut watershed.

Outlook

The forest management effects on below-ground hydrological processes are difficult to generalize, due to their inherent temporal-spatial variability compared to a comparatively short time span of the studies. The extreme time and spatial variability of soil parameters, high soil permeability and low retention for water further complicate evaluation of the influence of management practices on subsurface hydrological processes. Mathematical models which could be employed as a decision support tools for acceptable management practices represent a prospective tool. Meanwhile, ecological disturbances occurring whether due to or irrespective of the climate change, e.g. forest fires, droughts and/or floods, melting permafrost in boreal forests, increased frequencies and scope of fungal diseases and insect attacks, will change the situation both in unmanaged and managed forests. They may even overshadow any effects of forest management on water quantity and quality. For that reason, comparative studies in managed forests and forests following natural dynamics are highly desirable. In spite of these challenges, important factors related to forest management have been identified and their influence described both qualitatively and quantitatively.

The choice of species shall consider their interactions with precipitation and soil water as determined by LAI and stomatal regulations. Such considerations should be sufficiently complex, because the regulations are controlled through species specific physiological responses to microclimate, plant hormones, soil water potential and both intraspecific and interspecific competition. Canopy closure, forest density, soil scarification and burning strongly affect properties of the soil surface which in turn determines the prevalent types of water flow in forest soils. Forest management, including the choice of tree species, regeneration method, type of cutting, harvesting intensity, fertilisation or liming may alter the quality of water resources (surface, seepage and ground water) in forested watersheds, but water from forested catchments is usually of good quality, in comparison with that from agricultural land.

Acknowledgments Parts of this work have been funded by the APVV project No. 0468/06, VEGA 1/0723/08 and MVTS related to FP0601.

References

Ahti E (1983) Fertiliser-induced leaching of phosphorus and potassium from peatlands drained for forestry. Comm Inst For Fenn 111:1–20

Albers D, Migge S, Schaefer M, Scheu S (2004) Decomposition of beech leaves (Fagus silvatica) and spruce needles (Picea abies) in pure and mixed stands of beech and spruce. Soil Biol Biochem 36:155–164

Åström M, Aaltonen EK, Koivusaari J (2002) Impact of forest ditching on nutrient loadings of a small stream – a paired catchment study in Kronoby, W. Finland. Sci Total Environ 297:127–140

Åström M, Aaltonen EK, Koivusaari J (2005) Changes in leaching patterns of nitrogen and phosphorus after artificial drainage of a boreal forest – a paired catchment study in Lappajärvi, western Finland. Boreal Environ Res 10:67–78

Aussenac G (2000) Interactions between forest stands and microclimate: ecophysiological aspects and consequences for silviculture. Ann Forest Sci 57:287–301

Aussenac G, Granier A (1988) Effects of thinning on water stress and growth in Douglas fir. Can Forest Res 1:100–105

Badoux A, Jeisy M, Kienholz H, Lüscher P, Weingartner R, Witzig J, Hegg C (2006) Influence of storm damage on the runoff generation in two sub-catchments of the Sperbelgraben, Swiss Emmental. Eur J For Res 125:27–41

Bartoli F, Regalado CM, Basile A, Buurman P, Coppola A (2007) Physical properties in European volcanic soils: a synthesis and recent developments, 2007. In: Arnalds O, Bartoli F, Buurman P, Oskarsson H, Stoops G, Garcia-Rodeja E (eds) Soils of volcanic regions in Europe. Springer-Verlag, Berlin/Heidelberg

Barton L, McLay CDA, Schipper LA, Smith CT (1999) Denitrification rates in a wastewater-irrigated forest soil in New Zealand. J Environ Qual 28:2008–2014

Bärwolf M (2006) Soil properties. In: Measurements of Max Planck Inst. for Biogeochemistry at the Tatra windthrow sites. Jena CD –ROM

Black TA, Kelliher FM (1989) Processes controlling understorey evapotranspiration. Phil Trans R Soc Lond B 324:207–231

Bogner C, Engelhardt S, Zeilinger J, Huwe B (2008) Visualisation and analysis of flow patterns and water flow simulations in disturbed and undisturbed tropical soils. In: Beck E, Bendix J, Kottke I, Makeschin F, Mosandl R (eds) Gradients in a tropical mountain ecosystem of ecuador, vol 198, Ecological studies. Springer Verlag, Berlin/Heidelberg

Bosch JM, Hewlett JD (1982) A review of catchment experiments to determine the effect of vegetation changes on water yield and evapotranspiration. J Hydrol 55:3–23

Brais S, Sadi F, Bergeron Y, Grenier Y (2005) Coarse woody debris dynamics in a post-fire jack pine chronosequence and its relation with site productivity. For Ecol Manag 220:216–226

Breda N, Granier A, Aussenac G (1995) Effects of thinning on soil water balance and tree water relations, transpiration and growth in oak forest (Quercus petraea (Matt) Liebl.). Tree Physiol 15:295–306

Brêthes A, Brun JJ, Jabiol B, Ponge JF, Toutain F (1995) Classification of forest humus forms: a French proposal. Ann For Sci 52:535–546

Briggs RD, Hornbeck JW, Smith CT, Lemin RC Jr, McCormack ML Jr (2000) Long-term effects of forest management on nutrient cycling in spruce-fir forests. Forest Ecol Manag 138:285–299

Brown TC, Binkley D (1994) Effect of management on water quality in North American Forests. USDA Forest Service General Technical Report RM-248

Callesen I, Raulund-Rasmussen K, Gundersen P, Stryhn H (1999) Nitrate concentrations in soil solutions below Danish forests. Forest Ecol Manage 114:71–82

Campbell CG, Garrido F, Ghodrati M (2004) Role of leaf litter in initiating tracer transport pathways in a woodland hillslope soil. Soil Sci 169:100–114

Creedy J, Wurzbacher A (2001) The economic value of a forested catchment with timber, water and carbon sequestration benefits. Ecol Econ 38:71–83

Čermák J, Matysek R, Kučera J (1993) Příčiny odumíraní vyspělých buků na těžkých půdach po náhlem proředení porostu. Lesnictví – Forestry 39:175–183

Cummins T, Farrell EP (2003) Biogeochemical impacts of clearfelling and reforestation on blanket peatland streams. I. Phosphorus. Forest Ecol Manag 180:545–555

Dahlgren RA, Driscoll CT (1994) The effects of whole-tree-clear-cutting at the Hubbard-Brook-Experimental-Forest, New-Hampshire, USA. Plant Soil 158:239–262

David JF, Ponge JF, Arpin P, Vannier G (1991) Reactions of the macrofauna of a forest mull to experimental perturbations of litter supply. Oikos 6:316–326

De la Crétaz AL, Barten PK (2007) Land use effects on streamflow and water quality in the northeastern United States. CRC Press/Taylor & Francis, Boca Raton, FL

Dinel H, Schnitzer M, Mehuys GR (1990) Soil lipids: origin, nature, content, decomposition, and effect on soil physical. In: Bollag JM, Stotzky G (eds) Soil biochemistry, vol 6. Marcel Dekker, New York

Eguchi S, Hasegawa S (2008) Determination and characterization of preferential water flow in unsaturated subsoil of andisol. Soil Sci Soc Am J 72:320–330

Emmett BA, Anderson JM, Hornung M (1991a) Nitrogen sinks following two intensities of harvesting in a Sitka spruce forest (N. Wales) and the effect on the establishment of the next crop. Forest Ecol Manag 41:81–93

Emmett BA, Anderson JM, Hornung M (1991b) The controls on dissolved nitrogen losses following two intensities of harvesting in a Sitka spruce forest (N. Wales). Forest Ecol Manag 41:65–80

Fisher RF, Binkley D (2000) Long-term Soil Productivity. In: Fisher RF, Binkley D (eds) Ecology and management of forest soils, 3rd edn. Wiley, New York

FOEN (2009) Ergebnisse der Grundwasserbeobachtung Schweiz (NAQUA). Zustand und Entwicklung 2004–2006. Umwelt-Zustand 0903. Federal Office for the Environment FOEN, Bern, Switzerland

Gerke HH, van Genuchten MTh (1993) A dual-porosity model for simulating the preferential movement of water and solutes in structured porous media. Water Resour Res 29:305–319

Gibson CE (1976) An investigation into the effects of forestry plantations on the water quality of upland reservoirs in Northern Ireland. Water Res 10:995–998

Germann P, Beven K (1981) Water flow in soil macropores I. An experimental approach. J Soil Sci 32:1–13

Germann P, Beven K (1985) Kinematic wave approximation to infiltration into soils with sorbing macropores. Water Resour Res 21:990–996

Gersper PL, Holowaychuk N (1970) Effect of stemflow water on a Miami soil under a Beech Tree. Morphological and physical properties. Soil Sci Soc Am Proc 34:779–786

Gömöryová E, Střelcová K, Škvarenina J, Bebej J, Gömöry D (2008) The impact of windthrow and fire disturbances on selected soil properties in the Tatra National Park. Soil Water Res 3(Special Issue 1):574–580

Grace JM III (2005) Forest operations and water quality in the South. Trans ASAE 48:871–880

Grace JM III, Clinton BD (2007) Protecting soil and water in forest road management. Trans ASABE 50:1579–1584

Granier A, Aubinet M, Epron D, Falge E, Gudmundsson J, Jensen NO, Köstner B, Matteucci G, Pilegaard K, Schmidt M, Tenhunen J (2003) Deciduous forests: carbon and water fluxes, balances and ecophysiological determinants. In: Valentini R (ed) Fluxes of carbon, water and energy of European forests, vol 163, Ecological Studies. Springer Verlag, Berlin

Gregor J (1999) Influence of beech stand density and relief on soil moisture. (In Slovak, with English abstract). Vedecké štúdie 7/1999/A. Technical University in Zvolen

Grigal DF (2000) Effects of extensive forest management on soil productivity. Forest Ecol Manag 138:167–185

Gundersen P, Schmidt IK, Raulund-Rasmussen K (2006) Leaching of nitrate from temperateforests – effects of air pollution and forest management. Environ Rev 14:1–57

Harmon ME, Sexton J (1995) Water balance of conifer logs in early stages of decomposition. Plant Soil 172:141–152

Hegg Ch, Jeisy M, Waldner P (2004) Wald und Trinkwasser. Eine Literaturstudie. Eidg. Forschungsanstalt WSL, Birmensdorf, Switzerland

Hlavatá H, Holko L, Kostka Z, Novák J (2008) Analysis of precipitation – runoff relations at small watershed of High Tatras (In Czech, with English abstract). In: Šír M, Tesař M, Lichner L (eds) Proceedings of the international conference on hydrology of small catchments. Institute of Hydrology, Czech Academy of Sciences, Praha

Holko L, Škvarenina J, Kostka Z, FričM, StaročJ (2009) Impact of spruce forest on rainfall interception and seasonal snow cover in the Western Tatra Mountains, Slovakia. Biologia 64:594–599

Holst T, Hauser S, Kirchgässner A, Matzarakis A, Mayer H, Schindler D (2004) Measuring and modeling plant area index in beech stands. Int J Biometeorol 48:192–201

Horváth B, Meiwes KJ, Meesenburg H (2009) Die Bedeutung von Baumart und Bestandesalter für die Nitrat-versickerung unter Wald in der Region Weser-Ems. Forstarchiv 80:35–4

Huber C, Weis W, Baumgarten M, Göttlein A (2004a) Spatial and temporal variation of seepage water chemistry after femel and small scale clear-cutting in a N-saturated Norway spruce stand. Plant Soil 267:23–40

Huber C, Baumgarten M, Göttlein A, Rotter V (2004b) Nitrogen turnover and nitrate leaching after bark beetle attack in mountainous spruce stands of the Bavarian Forest National Park. Water Air Soil Pollut Focus 4:391–414

Ježík M, Voško M (2002) Diameter increment of beech trees and its trends on permanent research plots with various stand densities. Ekol – Bratislava 21(suppl2):80–90

Jochheim H, Schäfer H (1988) “Die Baumfuss–Methode” dargestellt anhand einer Untersuchung der Immissionsbelastung von Nordwest–Jugoslawischen Buchenwälder. Z Pflanz Bodenkunde 15:81–85

Joensuu S, Ahti W, Vuollekoski M (1998) Quality of runoff water from old ditch networks in Finnish peatland forests. The Spirit of Peatlands. Jyväskyla, Finland

Johnson S, Lehmann J (2006) Stemflow and root-induced preferential flow: the double-funneling of trees. Ecoscience 13:324–333

Jury WA, Scotter DR (1994) A unified approach to stochastic-convective transport problems. Soil Sci Soc Am J 58:1327–1336

Jury WA, Wang Z, Tuli A (2003) A conceptual model of unstable flow in unsaturated soil during redistribution. Vadose Zone J 2:61–67

Kantor P (1984) Water-regulation function of Mountainous spruce and beech stands. Lesnictví – Forestry 30:471–490

Keim RF, Skaugset AE, Weiler M (2005) Temporal persistence of spatial patterns in throughfall. J Hydrol 314:263–274

Kenderes K, Mihók B, Standovár T (2008) Thirty years of gap dynamics in a Central European beech forest reserve. Forestry 81:111–123

Kenttamies K (1981) The effects on water quality of forest drainage and fertilisation in peatlands. Water Research Institute Publication No. 43. National Board of Waters, Finland

Kermen J, Janota-Bassalik L (1997) A note on the forest soil as a biological filter in the sanitary purification of municipal waste water evaluated on the basis of Escherichia coli titre. Acta Microbiol Pol 36:109–18

Kim DJ, Burger JA (1997) Nitrogen transformations and soil processes in a wastewater-irrigated mature Appalachian hardwood forest. Forest Ecol Manag 90:1–11

Kirkham D, Powers WL (1972) Advanced soil physics. Wiley Interscience, New York

Koshi PT (1959) Soil-moisture trends under varying densities of oak over story. US Forest Serv South Forest Expt Sta Occas Paper 167

Kozlowski TT (1999) Soil compaction and growth of woody plants. Scand J For Res 14:596–619

Koivusalo H, Ahti E, Laurén A, Kokkonen T, Karvonen T, Nevalainen R, Finér L (2008) Impacts of ditch cleaning on hydrological processes in a drained peatland forest. Hydrol Earth Syst Sci 12:1211–1227

Kreutzweiser DP, Hazlett PW, Gunn JM (2008) Logging impacts on the biogochemistry of boreal forest soils and nutrient export to aquatic systems. Environ Rev 16:157–179

Kubin E (1998) Leaching of nitrate nitrogen into the groundwater after clear felling and site preparation. Boreal Environ Res 3:3–8

Kutílek M, Nielsen DR (1994) Soil hydrology. Catena, Cremlingen-Destedt

Legout A, Nys C, Picard JF, Turpault MP, Dambrine E (2009) Effects of storm Lothar (1999) on the chemical composition of soil solutions and on herbaceous cover, humus and soils (Fougères, France). Forest Ecol Manag 257:800–811

Livesley SJ, Adams MA, Grierson PF (2007) Soil water nitrate and ammonium dynamics under a sewage effluent irrigated Eucalypt plantation. J Environ Qual 36:1883–1894

Magesan GN, Vogeler I, Clothier BE, Green SR, Lee R (2003) Solute movement through an allophanic soil. J Environ Qual 32:2325–2333

Malcolm DC, Cuttle SP (1983) The application of fertilisers to drained peat 1. Nutrient losses in drainage. Forestry 56:155–173

Manninen P (1998) Effects of forestry ditch cleaning and supplementary ditching on water quality. Boreal Environ Res 3:23–32

Meesenburg H, Jansen M, Döring C, Beese F, Rüping U, Möhring B, Hentschel S, Meiwes KJ, Spellmann H (2005) Konzept zur Beurteilung der Auswirkungen forstlicher Maßnahmen auf den Gewässerzustand nach den Anforderungen der EG-Wasserrahmenrichtlinie. Ber Freiburger Forstl Forschung 62:171–180

Mattsson T, Finér L, Kortelainen P, Sallantaus T (2003) Brook water quality and background leaching from unmanaged forested catchments in Finland. Water Air Soil Poll 147:275–297

Mellert KH, Kölling C, Rehfuess KE (1996) Stoffauswaschung aus Fichtenwaldökosystemen Bayerns nach Sturmwurf. Forstwiss Centralbl 115:363–377

Merino A, Balboa MA, Rodríguez Soalleiro R, González Á (2005) Nutrient exports under different harvesting regimes in fast-growing forest plantations in southern Europe. Forest Ecol Manag 207:325–339

Miklánek P, Pekárová P (2006) Interception assessment in experimental microbasins of IH SAS with spruce and hornbeam vegetation (In Slovak, with English abstract). J Hydrol Hydromech 54:123–136

Mou P, Fahey TJ, Hughes JW (1993) Effects of soil disturbance on vegetation recovery and nutrient accumulation following whole-tree harvest of a northern hardwood ecosystem. J Appl Ecol 30:661–675

Nachtergale L, Ghekiere K, De Schrijver A, Muys B, Luyssaert S, Lust N (2002) Earthworm biomass and species diversity in windthrow sites of a temperate lowland forest. Pedobiologia 46:440–451

Nagel TA, Diaci J (2006) Intermediate wind disturbance in an old-growth beech-fir forest in Southeastern Slovenia. Can J For Res 36:629–63

Nieminen M (1999) Phosphorus fertilizer leaching from drained ombrotrophic peatland forests: empirical studies and modeling. Finnish Forest Research Institute Research Paper 756

Nieminen M (2003) Effects of clear-cutting and site preparation on water quality from a drained Scots pine mire in southern Finland. Boreal Environ Res 8:53–59

Nieminen M, Ahti E (1993) Leaching of nutrients from an ombrotrophic peatland area after fertilizer application on snow. Folia Forestalia 814:3–22

Nieminen M, Jarva M (1996) Phosphorus adsorption by peat from drained mires in southern Finland. Scand J Forest Res 11:321–326

Novák V (1995) Evaporation of water and methods of its estimation. (In Slovak). VEDA, Bratislava

Novák V, Kňava K (2008) Forest soil water content annual courses as influenced by canopy properties. In: Rožnovský J, Litschmann T (eds) Bioklimatologické aspekty hodnocení procesu v krajině. Institute Creation and Protection of Landscape, Mendel University, Brno, Czech Republic (CD-ROM)

Novák V, Lichner L, Zhang B, Kňava K (2009) The impact of heating on the hydraulic properties of soils sampled under different plant cover. Biologia 64:483–486

Papritz A, Diserens E, Schneebeli M (1991) Einfluss des Bodenwasserhaushalts auf die Transpiration von Waldbäumen – eine Literaturübersicht In: Pankow W (ed) Schlussbericht des Nationalen Forschungsprogrammes Nr. 14: Luftverschmutzung und Waldschäden in der Schweiz. Verlag der Fachvereine, Zürich

Pichler V (2007) Beech stands density as a aool for the regulation of soils hydric and environmental functions. (In Slovak, with English abstract). Technical University in Zvolen, Zvolen

Pichler V, Bublinec E, Gregor J (2006) Acidification of forest soils in Slovakia – causes and consequences. J Forest Sci 52:23–27

Pichler V, Ďurkovič J, Capuliak J, Ďurkovič J, Pichlerová M (2009) Altitudinal variability of the soil water content in natural and managed beech (Fagus sylvatica L.) forests. Pol J Ecol 58:313–319

Piirainen S, Finér L, Mannerkoski H, Starr M (2007) Carbon, nitrogen and phosphorous leaching after site preparation at a boreal forest clear-cut area. Forest Ecol Manag 243:10–18

Prévost M, Plamondon AP, Belleau P (1999) Effects of drainage of a forested peatland on water quality and quantity. J Hydrol 214:130–143

Philip JR (1957a) The theory of infiltration: 1. The infiltration equation and its solution. Soil Sci 83:345–357

Philip JR (1957b) Numerical solutions of equations of the diffusion type with diffusivity concentration dependent II. Aust J Phys 10:29–42

Ponge JF (1999) Horizons and humus forms in beech forests of the Belgian Ardennes. Soil Sci Soc Am J 63:1888–1901

Ponge JF, Delhaye L (1995) The heterogeneity of humus profiles and earthworm communities in a virgin beech forest. Biol Fert Soils 20:24–32

Qualls RG, Haines BL, Swank WT, Tyler SW (2002) Retention of dissolved organic nutrients by a forested ecosystem. Biogeochemistry 61:135–171

Raat KJ, Draaijers GPJ, Schaap MG, Tietema A, Verstraten JM (2002) Spatial variability of throughfall water and chemistry and forest floor water content in a Douglas fir forest stand. Hydrol Earth Syst Sci 6:363–374

Reifsnyder WE (1988) Hydrologic process models. In: Reynolds ERC, Thompson FB (eds) Forests, climate, and hydrology: regional impacts. United Nations University, Tokyo

Reiter ML, Beschta RL (1995) Effects of forest practices on water. In: Beschta RL, Boyle JR, Chambers CC, Gibson WP, Gregory SV, Grizzel J, Haga JC, Li JL, McComb WC, Parzybok TW, Reiter ML, Taylor GH, Warila JE (eds) Cumulative effects of forest practices in Oregon: literature and synthesis. Oregon Department of Forestry, Salem, OR

Roberts J (1983) Forest transpiration: a conservative hydrological process? J Hydrol 66:133–141

Roberts J (2000) The influence of physical and physiological characteristics of vegetation on their hydrological response. Hydrol Process 14:2885–2901

Robertson SMC, Hornung M, Kennedy VH (2000) Water chemistry of throughfall and soil water under four tree species at Gisburn, northwest England, before and after felling. Forest Ecol Manag 129:101–117

Rode AA (1952) Soil Moisture. (In Russian) Izd AN SSSR, Moscow

Rothe A, Mellert KH (2004) Effects of forest management on nitrate concentrations in seepage water of forests in southern Bavaria, Germany. Water Air Soil Poll 156:337–355

Scherer R, Pike R (2003) Effects of forest management activities on streamflow in the Okanagan Basin, British Columbia: outcomes of a literature review and a workshop, vol 9, Forrex Series. Forrex, Kamloops, B.C

Skopp J, Gardner WR, Tyler EJ (1981) Solute movement in structured soils: two-region model with small interaction. Soil Sci Soc Am J 45:837–842

Smolander A, Paavolainen L, Mälkönen E (2000) C and N transformations in forest soil after mounding for regeneration. Forest Ecol Manag 134:17–28

Stadler D, Stähli M, Aeby P, Flühler H (2000) Dye tracing and image analysis for quantifying water infiltration into frozen soils. Soil Sci Soc Am J 64:505–516

Stednick JD (1996) Monitoring the effects of timber harvest on annual water yield. J Hydrol 176:79–95

Stednick JD (2000) Timber management. In: Dissmeyer GE (ed) Drinking water from forests and grasslands. A Synthesis of the scientific literature. Gen. Tech. Rep. SRS-39. US Department of Agriculture, Forest Service, Southern Research Station, Asheville, NC

Steenhuis TS, Parlange J-Y, Andreini MS (1990) A numerical model of preferential solute movement in structured soils. Geoderma 46:199–208

Střelcová K, Matejka F, Minďáš J (2002) Estimation of beech tree transpiration in relation to their social status in forest stand. J Forest Sci 48:130–140

Šály R (1986) Slope deposits and soil of the Western Carpathians. (In Slovak, with English abstract). Veda, Bratislava

Torres R (2002) A treshold condition for soil-water transport. Hydrol Process 16:2703–2706

Valentini R (ed) (2003) Fluxes of carbon, water and energy of European forests, vol 163, Ecological studies. Springer, Berlin

v. Wilpert K, Zirlewagen D, Kohler M (2000) To what extent can silviculture enhance sustainability of forest sites under the immission regime in Central Europe? Water Air Soil Poll 122:105–120

Walsh RD, Voigt PJ (1977) Vegetation litter: an underestimated variable in hydrology and geomorphology. J Biogeogr 4:253–274

Warrick AW (2003) Soil water dynamics. Oxford University Press, New York

Weiler M, Flühler F (2004) Inferring flow types from dye patterns in macroporous soils. Geoderma 120:137–153

Weis W, Rotter V, Göttlein A (2006) Water and element fluxes during the regeneration of Norway spruce with European beech: effects of shelterwood-cut and clear-cut. Forest Ecol Manag 224:304–317

Wuyts K, De Schrijver A, Staelens J, Gielis L, Vandenbruwane J, Verheyen K (2008) Comparison of forest edge effects on throughfall deposition in different forest types. Environ Pollut 156:854–861

Ziegler F, Zech W (1989) Distribution pattern of total lipid fractions in forest humus. Z Pflanzenernaehr Bodenkd 152:287–290

Chapter 17

Adjustment of Forest Management Strategies to Changing Climate

Špela Planinšek, Leena Finér, Antonio del Campo, Jorge Alcazar, Cristina Vega-García, Dimiter Dimitrov, and Jozef Capuliak

Introduction

As shown in Chapter 12, the composition and structure of forests and their functions vary significantly between the Boreal, Temperate and Mediterranean regions of Europe. There is also a high variation in hydro-meteorological conditions along with latitude, altitude, continentality and oceanicity, slope of the terrain and soil type. Demands on forest products and functions in these regions influence their management and impact on the quantity and quality of surface and ground water and related element fluxes (Chapters 15 and 16).

There are mounting concerns that global climate is changing and leading to warmer and often drier conditions in many forests ecosystems around the world (Parks and Bernier 2010). Global climate models predict marked changes in seasonal snowfall, rainfall and evaporation in many parts of the world (UN-ECE/FAO 2000). The influence of these changes on water quantity and quality may be negative or positive. The impacts of climate change on forests and trees, and the role of forests in mitigating impacts of climate change are high on the international agenda (GFRA 2005; Chang 2006; IUFRO 2007).

Š. Planinšek (✉)
Slovenian Forestry Institute, Vecna pot 2, 1000, Ljubljana, Slovenia
e-mail: spela.planinsek@gozdis.si

L. Finér
Finnish Forest Research Institute, Box 68, 80101 Joensuu Finland

A. del Campo
Polytechnic University of Valencia, Cami de Vera s/n, 46022, Valencia, Spain

J. Alcazar and C. Vega-García
University of Lleida Avda, Alcalde Rovira Roure 191, 25198, Lleida, Spain

D. Dimitrov
Forest Research Institute, Bulgarian Academy of Sciences, 132, St. Kliment Ochridski Blvd, 1756 Sofia, Bulgaria

J. Capuliak
National Forest Centre, T.G. Masaryka 22, 96053, Zvolen, Slovakia

M. Bredemeier et al. (eds.), *Forest Management and the Water Cycle: An Ecosystem-Based Approach*, Ecological Studies 212, DOI 10.1007/978-90-481-9834-4_17,

In order to enhance the capability of forests to adapt to climate change and to store carbon, forest managers shall gain insights into related processes between forests and water and find effective management methods to meet the – still uncertain – challenges of climate change. Some important processes linking forest management and environment have been addressed in the preceding chapters, constituting a big challenge and an opportunity for forest management at the regional and operational level.

This chapter focuses on the different views and knowledge about the regulation and protective functions of forests in Boreal, Temperate and Mediterranean regions of Europe. We review the impacts of current forest management and future climate change on the interaction between forests and surface or ground water. Then, we propose measures aimed at mitigating possible negative impacts. Approaches are different as the forest management strategies between regions differ. Utilizing specific forest dynamics for the purpose of mitigating climatic change requires an understanding of forest development in different regions and a thorough appraisal of the consequences of interventions during the long rotation periods of forests (IUFRO 2007).

Major Water-Related Forest Management Concerns Under Climate Change Uncertainties

Boreal Forests of Fennoscandia

The climate change scenarios (IPCC 1992, 2007; Räisänen et al. 2004; Bater et al. 2008) for the boreal regions of Europe forecast a 4–6°C rise in average annual temperature by the 2080s. The temperature will increase in all seasons, but especially in wintertime. The annual precipitation will increase and mostly in winter. Total precipitation in summer changes very little or might even decrease according to some models. In northern parts of the region the snow cover period is estimated to shorten. In the south, a significant part of the winter rainfall will be water and the snow cover periods will become shorter. Warming and changes in snow cover will affect soil frost. Climate change is expected to increase the occurrence of extreme weather phenomena. An increased number of storms and heavy rainfalls are forecasted. These changes can have multiple effects on the interactions between forests and water.

Many possible negative effects are currently being discussed. Forest pests and plant diseases may benefit from a warmer climate and can affect forest cover and species distribution. Trees will be more vulnerable to storm damage in winter. Areal distribution of tree species may change and deciduous trees will become more abundant and hardening of tree species before the winter season may be disturbed. The risk of forest fires is likely to increase.

Currently, forest fires represent a natural disturbing agent in the boreal environment. However, wildfires are currently effectively controlled and therefore forest management practices are the main factors affecting water and nutrient fluxes in boreal forests, even if annual forest operations are done on less than 2% of the forest area in the Nordic countries. While the general expectation is that runoff, erosion and nutrient leaching would increase due to more rapid climate changes, this would also unfold from their synergies with forest management operations, which mostly affect water and associated element fluxes even under present conditions. They are harvesting, soil scarification for regeneration, fertilization and forest drainage. The increase in the leaching of suspended solids, nitrogen and phosphorus after these forestry operations is causing the biggest problems in the associated watercourses (Kenttämies 2006; Ring et al. 2008). The leaching of suspended solids and subsequent change in the colour of water and sedimentation has negative effects on the aquatic organisms and the use of water and watercourses. The Baltic Sea and many of the fresh water ecosystems are nitrogen and phosphorus limited and the excess flux of these elements cause eutrophication. Paired catchment studies have been carried out to detect the changes in runoff, suspended solid and nutrient fluxes induced by these management practices (Table 17.1).

The effects of clear-cuttings and the subsequent soil scarification on increase runoff and leaching of suspended solids and nutrients have been variable and affected by e.g. the amount of removed timber, the location and area of the catchment, the distance to the watercourse, the type of the buffer zones, the timing of the cutting, and the type of the soil scarification method. The export of nutrients increase soon after the treatment, peaks within 3–5 years and lasts over period 5–15 years (Ahtiainen and Huttunen 1999; Finér et al. 2008; Ring et al. 2008). Increase in runoff and the leaching of nutrients is mainly caused by the ceasing of water and nutrient uptake by trees (Laurén et al. 2005; Löfgren 2007) but also by the decrease in rainfall interception. The development of the new generation of trees and the subsequent recovery of nutrient uptake is slow in boreal conditions. The understory vegetation, which is typically abundant in boreal forests is also disturbed by clear-cutting and soil scarification and cannot immediately compensate the role of trees in water and nutrient uptake (Palviainen 2005).

Forest fertilization has not been reported to affect runoff, but the effects on nutrient leaching have been clear over period of 2 years in the case of nitrogen fertilization

Table 17.1 The amount of background leaching kg ha^{-1} in 10 years (Mattsson et al. 2003; Kortelainen et al. 2006) and the excess nutrient load caused by different forestry practices (kg ha^{-1} sum of 10 years) (Finér et al. 2008; Mattsson et al. 2003; Kenttämies and Haapanen 2006)

Operation	Nitrogen	Phosphorus
Background (no operations)	8–23	0.20–1.8
Clear-cutting and soil scarification		
Upland	5–35	0.25–0.6
Peatland	18	0.42
Fertilization		
Nitrogen	15	
Phosphorus		2.3
Ditch maintenance	0	0.7

(150 N kg/ha) on upland soils and for 5 years after phosphorus fertilization on peatlands (Saura et al. 1995; Kenttämies 2006).

Forest drainage is carried out in the boreal region to improve forest growth on peatlands and paludified soils. That is done by digging open ditches to lower water table and to increase aeration in the root zone. In Finland half of the peatlands (5.9 Mha) in Sweden 15% (1.4 Mha) and in Norway 19% (0.42 Mha) have been drained for forestry purposes (Paavilainen and Päivänen 1995) mainly between 1960 and 1980. Peatlands are extremely vulnerable to climate change and thus no new pristine peatlands are drained. However, the once established ditch networks are maintained by ditch cleaning at 20–30 years intervals. Forest drainage has a big impact on the water and nutrient fluxes. The runoff and nutrient leaching usually increase for more than 10 years after the first drainage (Mattsson et al. 2003), whereas the effects of ditch maintenance are much smaller (Joensuu 2002).

The projected global climate change may also diminish ground water resources, whose quality may too be compromised, despite their current abundance and use as a source of drinking water in Finland and Sweden. The increase in nitrate concentration is the major concern related to forests management and it has been shown that nitrate concentration has increased after cuttings (Rusanen et al. 2004). However, increase of nitrate concentration in ground water is so small that it does not prevent its use as drinking water.

Flood risks will increase, especially in winter, and if sea level of the Baltic Sea rises. In contrast, if the growing season is longer, forest growth would increase. Natural regeneration will be favoured by less harsh conditions. Spring flooding risks may decrease and seasonal variation in water level of lakes and rivers may decrease. In fact, in the Fennoscandian boreal region the annual runoff is almost equal to the annual evapotranspiration, and typical for the region is that up to 50–60% of the annual runoff takes place in spring during the snowmelt period.

Temperate Mountain and Lowland Zones

According to research (IPCC 1992; ANL 1994; Flato et al. 1999; IPCC-DDC 1999; Knight et al. 2004) in the zone of natural coniferous and beech forests in southeast temperate mountains (elevations between 800–1,700 m a.s.l), projected warming varies between 1.0°C and 1.4°C and precipitation reduction oscillates between 12% and 15.9%. Under such hydrothermic conditions, natural *Picea abies* (L.) Karst. and *Fagus sylvatica* (L.) forests will not yet be seriously limited by soil water supply. On the other hand, model studies conducted in the Alps and the Carpathians predict altitudinal shifts of tree species and/or forest types. This could mean an increase in areas suitable for them in the short term, but an effective loss of such areas in the long run (e.g. Casalegno et al. 2010), followed by an area-wise loss of the respective water-related forest functions.

In lowlands (under 800 m a.s.l.) dominated by oaks, predicted warming reaches 2.0°C in mean annual temperature. Precipitation is predicted to be lower on average by 24.9–28.7%. These conditions would be extremely unfavourable for the natural oak vegetation. That could be one of the factors already causing drying of some *Quercus* spp. forests in Bulgaria, Croatia and Slovenia. Their traditional coppice management does not facilitate the economy of water either. With warmer temperatures, all forests will face more threats from diseases and insects. According to simulation results (Vilhar and Fajon 2007; Vilhar and Simoncic 2007) the increased air temperatures and decreased precipitation could cause a decrease in drainage fluxes and eco-physiological drought could become a frequent event, possibly affecting regeneration. Due to increasing temperatures, change of precipitation regime, windiness, cloudiness and increased climatic extremes, the most affected forest stands in middle Europe will be those on extreme sites, stands not adapted to the site, stands with exotic species, and old, mature stands (Diaci 2007). In the temperate lowland zone, the frequency of years with no or little groundwater recharge and continuous decline of groundwater levels will increase.

This is inasmuch important that ground water provides drinking water supply both in lowlands and in mountain areas. In the mountain regions, the main sources of drinking water are springs or ground water reservoirs. They are fed by water largely purified by forests and forests soils that retain or transform hazardous substances from wet or dry deposition. Here, the influence of forest management on water regime and water quality has been found to extend further than forestry operations to depend also on forest cover, tree species composition, vertical as well as horizontal forest structure, preservation of forests (Table 17.2) and land use on a larger area (Vilhar and Fajon 2007).

Effects of the aforementioned change in precipitation regime and more frequent weather extremes would be probably further aggravated by improperly planned forest management operations, which mostly affect water and associated element fluxes

Table 17.2 Positive (+) or negative (–) impact of forest management measures on drinking water quality indicators (Vilhar and Fajon 2007)

Indicators of forest management measures	Concentration of pollutants in the water	Nitrogen content in the water	Erosion	Runoff volume	Water temperature
Clearing area	––	–––	–––	+++	–––
Freq., intensity, technique of felling	–––	–	––	++	–
Combination of tree species	++	+++	++	–––	+
Crown density, screen	+	+++	+++	–––	++
Growth phases and their distribution	+	++	++	––	+++
Vertical and horizontal forest structure	+	++	++	––	+
Reforestation and forest regeneration, ground vegetation	–	––	++	––	–
Deadwood	+	++	–	–	–

Fig. 17.1 Forest skid trails are prone to severe erosion (Photo: Špela Planinšek)

through the construction of forest roads, their poor maintenance and subsequent massive road drainage (Fig. 17.1), as well as final cutting and soil scarification.

In the lowlands, however, ground water supply is usually limited because of its low rate of renewal. The main bodies of groundwater are in gravel plains, which are also attractive for settlement and other economic activities (Kladnik et al. 2004). In the last 2 or 3 decades, nitrate and toxic element concentrations have increased in ground water due to the intensive agriculture in the lowlands. The changes in water regimes caused by anthropogenic activities such as damming and the changes of the ground water level by drainage have greatly influenced the hydrology of lowland forests in Central Europe.

The Mediterranean Region

The Mediterranean region is expected to suffer considerable impacts from climatic change under different emissions change scenarios (Alcamo et al. 2007). Climate change models predict a temperature increase between 2°C and 4°C over the next century. In Southern and Western Europe the warming is projected to be greater during the summer months and precipitation could decrease up to 70% in this period, although the reduction in precipitation extends year round. Nevertheless, under specific scenarios, winter precipitation may increase in the northern Mediterranean. The pattern of rainfall distribution is expected to change in frequency, intensity and duration. Extreme weather events will affect Mediterranean ecosystems: more frequent drought periods, heavy storms, more hot days and heat waves, strong winds, large wildfires, violent floods, and fewer cold-days spells. A decrease in annual runoff by up to 36% is expected. The drier and warmer regions of the Mediterranean South zone (Metzger et al. 2005) are expected to grow about 28% by 2080 at the expense of the Mediterranean north and Mediterranean mountains zones (Metzger et al. 2008). These changes will lengthen water scarcity and drought, identified as one of the critical ecological factor affecting Mediterranean vegetation growth. So, droughts would not only be more frequent but longer and more intense, although their gravity may differ across various regions. It is because the mean annual runoff is highly variable around the Mediterranean basin. It is lowest in south eastern Spain and highest in western Greece (>1,000 mm). In some of these areas there is overuse of ground water, leading to overexploitation of aquifers, marine water intrusion, nitrate pollution and salinization (MMAMRM 2006). Water resources are scarce; agricultural and urban demands are high and desertification risk a very real threat.

Overall, Mediterranean ecosystems and forests are expected to suffer important alterations in their structure and function due to significant disruptions in forest disturbances. Extended periods of high temperatures or heat waves, low relative humidity and strong winds, will in all likelihood alter the frequency, intensity and extent of wildfires (the fire regimes). Insect outbreaks, pathogens, and invasion of exotic species are all expected to increase and strain debilitated forest stands (Brasier 1996; Resco de Dios et al. 2007). In this context, the main forestry-related activities potentially affecting water and associated element fluxes include forest road construction, lack of maintenance of forest roads and final cutting soil disturbance. The low-intensity silviculture practiced, though, limits these effects to a negligible level compared to other disturbances.

On the other hand, the elimination of the human seasonal disturbance pattern at the landscape level caused by development and policies in the UE has promoted forest and scrub expansion (Puigdefábregas and Mendizabal 1998), fuel built-up and a change in the Mediterranean fine-grained ecosystems to coarse grained. This fact may have desirable effects on soil conservation, contributing to the reduction in runoff and -suspended solids in forested watersheds, but may also sustain the spread of large wild land fires (Vega-García and Chuvieco 2006). The absence of silvicultural practices, due to the marginal benefits, predisposes the stands to fire but also to other important disturbance agents like pests, diseases or dieback associated to abiotic factors. Paradoxically, unmanaged forest expansion may lead to forest cover elimination and associated effects of soil erosion. Also, forest expansion has lead to an increase in evapotranspiration and decreases in runoff in downstream areas were water scarcity is already a problem (e.g. semiarid areas of Spain). In France, Rambal (1987) reported a runoff decrement of 80 mm equivalent to 15% of precipitation, due to vegetation changes after agricultural land abandonment. Similarly Gallart and Llorens (2002, 2003) reported from the Ebro river headwaters significant stream flow decreases in the last decades due to forest densification. From these and other studies in Mediterranean conditions

Puigdefábregas and Mendizabal (1998) have concluded that there is no runoff from forest watersheds where annual precipitation is <400 mm.

It is especially relevant in Mediterranean mountain forests the fact that future weather conditions may elevate the tree line for most species. Model simulations suggest that deciduous trees, such as *Fagus sylvatica* or *Quercus petraea* L., could occupy the subalpine belt. Conifers would invade the current alpine zone, which would become afforested if soils capable of supporting high forest are available or exist (Resco de Dios et al. 2007). However, in despite all these changes, when rainfall has been predicted to increase in some areas under climatic change, the modeling of growth for Mediterranean forests has been projected to increase (Sabaté et al. 2002). Recent field studies provide evidence of these predicted changes in Mediterranean forests health, composition and distribution as a consequence of observed changes in climatic trends (Resco de Dios et al. 2007).

Adaptation of Forest Management Strategies Aimed at Protecting Forest Hydrological Functions

Boreal Forests

Boreal forests cover a significant portion of the land area of the Fennoscandian countries, from 39% in Norway to 79% in Finland. Two native coniferous species, Scots pine (*Pinus sylvestris* (L.)) and Norway spruce (*Picea abies* (L.) Karst.), dominate the forests and are mixed with deciduous species, mainly birch (*Betula pendula* Roth. and *B. pubescens* L.). The tolerance of these trees species varies in relation to drought and water logging (Niinemets and Valladares 2006). Scots pine is more tolerant both to drought and waterlogging than Norway spruce, and birch has an intermediate behavior. Soils on upland forests are mainly well-drained podzolized glacial tills. Peatlands are abundant, covering from one third to one tenth of land area in Finland, Sweden and Norway. The tree species which grow on upland soils are characteristic species also for forested peatlands.

Within Fennoscandia there is a high variation in hydrometeorological conditions. Evaporation is low and prolonged periods of drought are rare. Considerable periods with the ground under snow cover and frozen topsoil layers during the winter season allow for forestry operations to be conducted. Highly mechanized harvesting produces industrial roundwood in an intensive forest management based on a sustainable yield principle. Artificial regeneration is a common practice, and forest tending is based on thinning

While forests are mostly sustainably managed for timber purposes, they also serve other functions like biodiversity, carbon sequestration, recreation and hunting. Less than 10% of forests are protected only to maintain biodiversity. Consequently, forests are for the most part monospecific, young (<80 years old) and even-aged due to management practices.

At present, several measures are taken to decrease nutrient leaching and soil erosion after the forestry operations (Tapio 2007; Skogforsk 2008). Water protection plans are done before starting the operations and they include buffer zones between clear-cut and fertilized areas and watercourses (Fig. 17.2). In forest drainage, sedimentation ponds and pits and infiltration areas are used to catch soil erosion (Fig. 17.3). Sedimentation ponds can catch on an average 50% of the erosion and infiltration areas up to 80% (Joensuu 2002; Nieminen et al. 2005). Only the infiltration areas have been shown to trap some of the dissolved nutrients (Nieminen et al. 2005). In forest drainage, the water protection plans are done for the whole catchment area affected by the operation. Special care is taken to manage the forests on ground water areas. Fertilizations and heavy soil scarification methods are avoided.

Fig. 17.2 A buffer zone left between clear-cut area and a watercourse (Photo: Erkki Oksanen, Metla)

Fig. 17.3 A sedimentation pond constructed to stop soil erosion from a drainage area (Photo: Leena Finér)

In Finland there are no measures being currently applied to manage the forests only for water. The other functions of forests and mainly the productive function are prevalent. However, the situation may change by the changing climate. Forested areas could be used as flood protection areas to cut the flood peaks in large rivers. Increase runoff and leaching of nutrients scenarios demand to find ways to manage the forests to minimize these impacts. These management tools could involve continuous cover forestry, planting of deciduous trees with high transpiration demand and good adaptation capacity and moving more towards a multi-scale management of forests, from the catchment to the stand planning basis (Kellomäki and Leinonen 2005; Peltola and Kellomäki 2005).

Temperate Mountain and Lowland Forests

Temperate forests occur in a wide range of situations in Western, Central and some parts of Eastern Europe. While the lowland forests grow along big rivers, mountain forests cover important areas in the Pyrenees in Spain, in the Alps, and reach through Central to Eastern Europe. Well-defined seasons with a distinct winter characterize this forest biome. A moderate climate and a growing season lasting 4–6 frost-free months distinguish the temperate forests. Variations in geomorphology, climate, as well as in silviculture practices explain the large diversity of forest types, which in turn require the implementation of different approaches, e.g. shelterwood, selection and clear-cutting forest management systems. Some of these approaches tend to rely on the ecological plasticity of tree species, such as in the case of the shelterwood system in relation to European beech. The preference for natural regeneration is growing for both ecological and economic reasons. Forest tending is based on the principle of selecting single trees or small to medium sized bio-groups according to the criteria of vitality, adaptability to the site, their ecosystem role, health condition and quality. The tree species composition of lowland forests depends on the water regime of soil and the level of ground water. The soils are fertile and the lowland forests are some of the most productive terrestrial ecosystems maintaining an exceptional richness of both fauna and flora.

Currently, the recognition of the importance of the hydrological role of forests, especially near infiltration basin areas, is rising. As an expression of this, general guidelines for forest development in Central Europe emphasizing the role of changing hydrological factors have been drafted (Fajon et al. 2008; Fajon 2007; Diaci 2007). They underscore several major points:

- Structural diversity of forests should be promoted (uneven-aged stands, even distribution of growth-phases in watershed areas).
- A dynamic balance of trees in various development phases should be maintained.
- Sites which are suitable for selective structures with natural species present should be identified and restored.

- Species with wide ecological amplitude (*Quercus robur*, *Pseudotsuga menziesii*) and thermophilous tree species (*Tilia* sp., *Robinia* sp., *Pinus nigra*, *Pinus sylvestris*) should be promoted and preserved.
- Promotion of *Picea abies* (Norway spruce) is no longer a silviculture goal (because of its vulnerability to drought and natural disturbances), but *Pinus sylvestris* and *Larix europaea* are still a choice.
- Spruce stands on sites more suitable for deciduous species should be converted to the natural vegetation suitable for that area.
- Shift of tree species can be made also with planting of seedlings of natural (mostly deciduous) vegetation.
- Stands where spruce trees prevail, especially those showing little stability and quality of wood, can be regenerated sooner than planned. As pointed out in Chapter 1, spruce stands were promoted in the past for productive reasons, but are now inadequate for enhancing ecological goals, because they are often affected by insects, wind and snow damages.

Among the aims of lowland forest management in response to projected climate changes, restoring the ecosystem integrity of flood plains separated by original flood protection dikes, represents one of the challenges, along with the revitalisation of the banks of human-channelled rivers and degraded riparian zones. Optimization should lead to a return to naturalness and auto-regulation of the entire ecosystem as a complex. This includes not only provision of the optimal life conditions for as large number of species as possible, but also preservation of the natural biodiversity, maintaining amplitude and frequency of changes (dynamic balance) and preservation of the reparation and regeneration processes (Varga et al. 1997; Neštický et al. 1999).

Given the degree of uncertainty in the predictions about climate change in Central Europe, a shift from currently successful sustainable and close-to-nature forest management may not be advisable. On the other hand, there is a pressure to produce very specific solutions and recipes, whose application might narrow our options for future approximations. Therefore, maintaining options for forests adaptation, based on their high structural and species diversity, ranks as one of the main forest management objectives. In addition, measures for reducing forest vulnerability from extreme weather events and adaptive forest management strategies will have to be further developed to protect the water cycle. For instance, where traditional coppice management of oak forests is not favourable for the economy of water, they will have to be converted to high forest.

Mediterranean Forests

The composition and configuration of the Mediterranean forests cannot be understood without considering the millennia of human intervention. Human impact and high environmental variability (cold winters, hot and dry summers, topographic chains oriented E–W) in the Mediterranean region have made the forest ecosystems very heterogeneous in their structure and composition (Metzger et al. 2005; EEA 2006). The spatial structure of this fine-grained mosaic of forests retains the highest levels of biodiversity in Europe (Puigdefábregas and Mendizabal 1998), with a very large number of endemic species. The main forest types represented in the Mediterranean region include (EEA 2006) thermophilous deciduous or semideciduous forest in the supra-Mediterranean climatic region (Quézel and Médail 2003), dominated by oaks (*Quercus* spp.), but frequently associated to maples and in the eastern Mediterranean region to hop-hornbeam, ash (*Fraxinus* spp.), hornbeam (*Carpinus* spp.), or lime (*Tilia* spp.), which may be the dominant species in some sites. The natural composition of these forests has been simplified to just one or two dominant tree species of commercial value and good resprouting capacity, accompanied by secondary species and/or an understory. Simple structures predominate, shaped by silvicultural systems traditionally established for grazing and firewood production: coppice, mixed coppice and coppice with standards.

Another type is represented by broadleaved evergreen forests, woodlands dominated by the evergreen sclerophyllous oaks, carob tree (*Ceratonia siliqua*) or wild olive tree (*Olea europaea* var. *sylvestris*). Most holm-oak forests, in the meso-Mediterranean belt, are coppices (*Q. ilex*) or *dehesas* (*Q. rotundifolia*), crops and pasture land with a very open, park-like canopy. *Dehesas* in Spain and *montados* in Portugal are important agro-forestry systems where cork oak (*Q. suber*) often dominates. The thermo-Mediterranean belt of kermes and alder-leaved oak forest (*Quercus coccifera*, *Q. alnifolia*) stretches in woodlands in the Iberian Peninsula, Greece, Cyprus and Anatolia. These arborescent, open or steppe formations are rich in endemic species.

Coniferous xerophytic forests of pines and other species consist of firs (*Abies* spp.), junipers (*Juniperus* spp.), cedar (*Cedrus* spp.), cypress (*Cupressus sempervirens*), and yew (*Taxus baccata*). The pinewoods include thermophilous *Pinus* species (*P. pinaster*, *P. pinea*, *P. halepensis*, *P. brutia*, *P. canariensis*) on lowlands and coastal areas occur on very poor soils and extremely harsh sites, where they withstand severe drought conditions. Management of these shade-intolerant species, when commercially feasible, favours even-aged structures through shelterwood or group selection treatments.

According to the management objectives, a gross classification of Mediterranean forests could divide them as exclusive protection forests, where only marginal returns could be expected, and protection forests with some returns from wood (conifer forests), cork (cork oak forests), hunting, pastures and domestic livestock (*dehesa* and *montado* forests) or other non-timber forest products. In any case, biological conservation and physical protection objectives are strong concerns when managing Mediterranean forests and in most cases preponderate over productive objectives.

Mediterranean mountain forests provide significant ecosystem services among which are prominent those related to soil conservation, improved catchment of surface waters and supply to underground water reservoirs. Forest management goals in exclusive protection forests, such as those of Aleppo pine (*Pinus halepensis* Mill.), typically found on steep slopes, shallow soils and on torrential storm prone areas, are mainly oriented to preserve the forest cover. The silvicultural treatments applied shift from carefully planned, multi-stage shelterwood systems to selection systems, removing single trees or small groups of trees throughout the forest for sanitary reasons.

Based on the facts previously exposed, Mediterranean forests will require active management to cope with global change. Due to the characteristics of the Mediterranean climate (low P/ETP values), some forest ecosystems are already limited for tree development, so projected changes and disturbances could turn many types of woodland in shrublands in a few decades, having little chance to recover the forest condition. In addition, water scarcity downstream may be related to the presence of dense and closed forests in headwaters (due to interception and evapotranspiration losses). In this sense, water-yield or water-saving silviculture must play an important role in forest management in the future, which is in good agreement with wildfire prevention silviculture (Velez 1990) and other silvicultural techniques that promote stand and tree resistance to biotic attacks. Economic implications thereof are unfathomable. Allowing the forestry sector to take share in payments for water supply may motivate foresters and forest owners to continue or better enhance operations contributing to the quality water yield from forests.

Water saving silviculture pursues spatial structures that balance the advantages related to higher stand densities for soil conservation or flood control, and those related to lower densities, as the lower evapotranspiration component, the increase in runoff and underground drainage. This can be achieved by defining forest structures and cover in accordance with site characteristics (Belmonte Serrato et al. 1999). Interception losses can reach values up to 49% of precipitation under very dry atmospheric conditions (Llorens et al. 1997). Stem flow is also important as it is the responsible to drive water from canopy to soil and hence it influences soil infiltration. Although stem flow values are often very low (below 10% of gross precipitation, but usually lower (Llorens et al. 1997; Levia and Frost 2003)) it can be stimulated by several silvicultural treatments. Water yield silviculture has been early accomplished in several parts of the world. In Mediterranean ecosystems, Hibbert et al. (1982) demonstrated that silvicultural techniques as thinning can increase runoff when annual precipitation is above 450 mm. The magnitude of the increment can range between 25–165 mm year^{-1} depending on thinning intensity and annual precipitation. Bosch and Hewlett (1982) established that an augment of 40 mm of runoff could be expected by each 10% of forest area (from gum or coniferous forests) transformed to grassland. This augment of runoff is reduced to 10 mm in shrublands. An alternative to a whole-stand thinning is to clear small patches (about 1 ha in size), which can be more cost-effective, can create more ecotones and hence promote biodiversity and generates a fine-grained mosaic landscape. Troendle et al. (2001) presented some results in this line reporting increments in runoff (17% increment for a 24% of cleared area), both in experimental and larger watersheds and no effect in peak flows or sediment load in the streams was detected.

Traditionally, unproductive Mediterranean forests have had a low intensity management according to the narrow or null margin of benefits obtained from conventional forest products. In the future, it will be necessary to consider water as a direct product from upstream forests and hence define and establish explicitly silvicultural treatments that optimize the water cycle (infiltration, run-off, stream flow). In this sense, the question is to manage Mediterranean forests with hydrological criteria and keeping the highest possible range of desired benefits (groundwater and reservoirs recharge, flush floods protection, prevent reservoirs silting up, protection from forest fires, non-wood forest products, hunting, and many more forest services). At the stand level, more efficient forest structures could be achieved by maintaining moderate densities and managing species composition according to water use efficiency criteria. Increasing forest resilience and biodiversity will need a better integration of the knowledge on species ecophysiology and auto-ecology with restoration techniques in order to better define microhabitats for secondary or accessory species and enrich stand composition according to the limiting factors. At the catchment level, non-forest patches should be considered for runoff production and groundwater recharge.

Conclusions

Forest management performs an essential and highly valuable influence on the forest cover which should be planned and monitored in two substantially different ways. The first aspect includes the protection of existing ground and surface waters as well as waterbeds and the prevention of erosion and flooding which could, in combination with high waters, directly endanger values-at-risk in their way. The second aspect, still underestimated and scientifically as well as operationally less known, includes the maintenance and improvement of forest structures and stands of trees, which indirectly increase water-retention capacity of the soil. Current trends toward the abandonment of thinning and tending (and even felling) could be problematic, since it has negative impacts on the water regime and protective functions of forests.

Water supply for the vegetation is one of the most limiting factors in many forest ecosystems, affecting health, productivity and stability of forests. Natural disturbances are an important component of forest regeneration cycles. The impact of projected changes in climatic conditions, and uncertainties thereof, on both water availability and disturbance regimes will force to technically support the stability of forests to protect forest values. We should therefore review and enlarge the precautionary silvicultural measures to mitigate climate change in Europe. Further in-depth knowledge needs to be gained in all its regions. In view of the differences in site and forest conditions, modelling the complex forest–water system can provide some tools to simulate and predict the scenarios of climate change and their interactions with alternative silvicultural measures. To further develop and apply close-to-nature, sustainability and multiple-purpose principles, while adjusting to changing conditions in designing forest plans poses one of the main challenges for forestry in the years to come.

References

Ahtiainen M, Huttunen P (1999) Metsätaloustoimenpiteidne pitkäaikaisvaikutukset purovesien laatuun. (in Finnish). Boreal Environ Res 4:101–114

Alcamo J, Moreno JM, Nováky B, Bindi M, Corobov R, Devoy RJN, Giannakopoulos C, Martin E, Olesen JE, Shvidenko A (2007) Europe. Climate change 2007: impacts, adaptation and vulnerability. In: Parry ML, Canziani OF, Palutikof JP, van der Linden PJ, Hanson CE (eds) Contribution of working group II to the fourth assessment report of the intergovernmental panel on climate change. Cambridge University Press, Cambridge, UK, pp 541–580

Argonne National Laboratory (ANL) (1994) Guide for Vulnerability and Adaptation Assessments. U.S. Country Studies Program, Washington D.C.

Bater B, Kundzewicz ZW, Wu S, Pallitikof JP (2008) Technical Paper on Climate Change and Water Climate Change and Water. IPCC Secretaria Geneva 210

Belmonte SF, Romero DA, López BF, Hernández Laguna E (1999) Óptimo de cobertura vegetal en relación a las pérdidas de suelo por erosión hídrica y las pérdidas de lluvia por interceptación. Papeles Geografía 30:5–15

Bosch JM, Hewlett JD (1982) A review of watershed experiments to determine the effects of vegetation changes on water yield and evapotranspiration. J Hydrol 55:3–23

Brasier CM (1996) Phytophthora cinnamomi and oak decline in southern Europe. Environmental constraints including climate change. Ann Forest Sci 53:347–358

Casalegno S, Amatulli G, Camia A, Nelson A, Pekkarinen A (2010) Vulnerability of Pinus cembra L. in the Alps and the Carpathian mountains under present and future climates. Forest Ecol Manag 259:750–761

Chang M (2006) Forest hydrology: an introduction to water and forests. CRC Press LLC, Boca Raton, FL

Diaci J (2007) Adapting silviculture to climate change. In: Jurc M (ed) Podnebne spremembe – vpliv na gozd in gozdarstvo (in Slovenian). Department of Forestry and Renewable Forest Resources, Slovenia, Ljubljana, pp 117–133

European Environment Agency (2006). European forest types: categories and types for sustainable forest management reporting and policy. EEA Technical Report 9. 111 p. http://reports.eea.europa.eu/technical_report_2006_9/en/eea_technical_report_9_2006.pdf. Accessed 27 August 2010

Fajon Š et al (2008) Model za vrednotenje hidrološke vloge gozda (Model for evaluation of the forest's hydrological role) (in Slovenian). In: GIS v Sloveniji 2007–2008. Znanstvenoraziskovalni center SAZU, Ljubljana, pp 207–218

Fajon Š (ed) (2007) Forest and water: summary of project Interreg IIIA. Slovenian Forestry Institute, Slovenian Forest Service, Ljubljana, 16

Finér L, Ahti E, Joensuu S, Koivusalo H, Laurén A, Makkonen T, Mattsson T, Nieminen M, Tattari S (2008) Metsätalouden vesistökuormituslaskelmat Kansallisen metsäohjelman 2015 valmistelua varten. (In Finnish) In: Uusivuori J, Kallio M, Salminen H (eds) Vaihtoehtolaskelmat kansallisen metsäohjelman valmistelua varten. Metlan työraportteja/Working Papers of the Finnish Forest Research Institute 75: 68–86

Flato GM et al (1999) The Canadian Centre for Climate Modeling and Analysis, global coupled model and its climate. Climate Dyn 16:451–457

Gallart F, Llorens P (2002) La cubierta forestal de la cuenca del Ebro: caracterización espacio-temporal y la afección en la disminución de recursos hídricos. Confederación Hidrográfica del Ebro, Zaragoza

Gallart F, Llorens P (2003) Observations on land cover changes and water resources in the headwaters of the Ebro catchment Iberian Peninsula. Phys Chem Earth 29:769–773

Global Forest Resources Assessment (GFRA) (2005) Progress towards sustainable forest management. 2006. FAO forestry paper 147. Rome 2006. FAO, Rome, 320 pp

Hibbert AR, Davis EA, Knipe OD (1982) Water yield changes resulting from treatment of Arizona chaparral. In: Conrad CE, Oechel WC (eds) Proceedings of the symposium on dynamics and management of Mediterranean-type ecosystems. USDA Forest Service, Gen. Tech. Rep., PSW-58

Intergovernmental Panel on Climate Change (IPCC) (1992) Climate change: the IPCC 1990 and 1992 assessments. WMO/UNEP, Cambridge

Intergovernmental Panel on Climate Change Data Distribution Centre (IPCC-DDC) (1999) Data information supplied by the IPPC data distribution centre of climate change and related scenarios for impact assessments, Version 1.0, April 1999

Intergovernmental Panel on Climate Change (IPCC) (2007) The Fourth Assessment Report (AR4)

IUFRO strategy 2006–2010 (2007) Global Science Cooperation for the Benefit of Forests and People, 52

Joensuu S (2002) Effects of ditch network maintenance and sedimentation ponds on export loads of suspended solids and nutrients from peatland forests. Metsäntutkimuslaitoksen tiedonantoja 868, 83

Kellomäki S, Leinonen S (2005) Management of European forests under changing climatic conditions. Univeristy of Joensuu, Faculty of Forestry, Research Notes 163, 427

Kenttämies K (2006) Metsätalouden fosfori- ja typpikuormituksen määritys. Julkaisussa. (in Finnish). In: Kenttämies K, Mattsson T (eds) Metsätalouden vesistökuormitus. MESUVE-projektin loppuraportti, vol 816. Suomen ympäristö, Helsinki, pp 9–25

Kenttämies K, Haapanen M (2006) Metsätalouden toimenpiteiden aiheuttama kasviravinteiden huuhtoutuminen ja kansallisen metsäohjelman suositustason vaikutus siihen (in Finnish). Julkaisussa. In: Kenttämies K, Mattsson T (eds) Metsätalouden vesistökuormitus. MESUVE – projektin loppuraportti, vol 816. Suomen ympäristö, Helsinki, pp 29–40

Kladnik D, Rejec Brancelj I, Smrekar A (2004) Integral burdening in gravel plains in Slovenia – case study of Ljubljansko polje. In: Prospects of applied geography. Filozofska fakulteta, Oddelek za geografijo, Ljubljana, 113

Knight C, Raev I, Staneva M (eds) (2004) Drought in Bulgaria a contemporary analog for climate change. Ashgate, Hampshire

Kortelainen P, Mattsson T, Finér L, Ahtiainen M, Saukkonen S, Sallantaus T (2006) Controls on the export of C, N, P and Fe from undisturbed boreal catchments, Finland. Aquat Sci 68:453–468

Laurén A, Finér L, Koivusalo H, Kokkonen T, Karvonen T, Kellomäki S, Mannerkoski H, Ahtiainen M (2005) Water and nitrogen processes along a typical water flowpath and streamwater exports from a forested catchment and changes after clear-cutting: a modelling study. Hydrol Earth Syst Sc 9:657–674

Levia DF, Frost EE (2003) A review and evaluation of stemflow literature in the hydrologic and biogeochemical cycles of forested and agricultural ecosystems. J Hydrol 274:1–29

Llorens P, Poch R, Latron J, Gallart F (1997) Rainfall interception by a Pinus sylvestris forest patch overground in a Mediterranean mountainous abandoned area I. Monitoring design and results down to the event scale. J Hydrol 199:331–345

Löfgren S (eds) (2007) How to estimate N and P losses from forestry in northern Sweden (http://www.ksla.se/sv/retrieve_file.asp?n=1122). Kungl. Skogs- och Lantbruksakademiens Tidskrift 2/2007: 14–17. Accessed 2 September 2010

Mattsson T, Finér L, Kortelainen P, Sallantaus T (2003) Brook water quality and background leaching from unmanaged forested catchments in Finland. Water Air Soil Poll 147:275–297

Metzger MJ, Bunce RGH, Jongman RHG, Mücher CA, Watkins JW (2005) A climatic stratification of the environment of Europe. Global Ecol Biogeogr 14:549–563

Metzger MJ, Bunce RGH, Leemans R, Viner D (2008) Projected environmental shifts under climate change: European trends and regional impacts. Environ Conserv 35:64–75

MMAMRM. (2006) Anuario de estadísticas forestales. Ministerio de Medio Ambiente y Medio Rural y Marino. http://www.mma.es/portal/secciones/biodiversidad/montes_politica_forestal/estadisticas_forestal/indice_estadisticas_forestales_2006.htm. Accessed 20 August 2010

Neštický Š, Varga L., Bucha T et al (1999) Monitorovanie lesných ekosystémov v roku 1999 pre potreby plnenia medivládnej dohody z 19. apríla 1995, Lesnícky výskumný ústav Zvolen, Výskumná stanica Gabčíkovo (in Slovak)

Nieminen M, Ahti E, Nousiainen H, Joensuu S, Vuollekoski M (2005) Capacity of riparian buffer zones to reduce sediment concentrations in discharge from peatlands drained for forestry. Silva Fenn 39:311–339

Niinemets Ű, Valladares F (2006) Tolerance to shade, drought, and waterlogging of temperate northern hemisphere trees and shrubs. Ecol Monogr 76:521–547

Paavialinen E, Päivänen J (1995) Peatland Forestry – Ecology and Principles, Ecological Studies vol. 111, Springer, Berlin

Parks CG, Bernier P (2010) Adaptation of forests and forest management to changing climate with emphasis on forest health: a review of science, policies and practices. Forest Ecol Manag 259:657–659

Palviainen M (2005) Logging residues and ground vegetation in nutrient dynamics of a clear-cut boreal forest. Dissertationes Forestales 12. 38

Peltola H, Kellomäki S (2005) Ilmastonmuutoksen vaikutukset metsäekosysteemin toimintaan ja rakenteeseen sekä metsien hoitoon ja ainespuun tuotantoon. (in Finnish) Metsäntutkimuslaitoksen tiedonantoja 944, pp 99–113

Puigdefábregas J, Mendizabal T (1998) Perspectives on desertification: Western Mediterranean. J Arid Environ 39:209–224

Quézel P, Médail F (2003) Ècologie et biogéographie des forêsts du bassin Méditerranéen. Elsevier, Paris

Rambal S (1987) Evolution de l'occupation des terres et ressources en eau en región Méditerranéenne karstique. J Hydrol 93:339–357

Räisänen J, Hansson U, Ullerstig A, Döscher R, Graham LP, Jones C, Meier M, Samuelsson P, Willén U (2004) European climate in the late 21st century: regional simulations with two driving global models and two forcing scenarios. Clim Dyn 22:13–31

Resco de Dios V, Fischer C, Colinas C (2007) Climate change effects on Mediterranean forests and preventive measures. New For 33:29–40

Ring E, Löfgren S, Sandin L, Högbom L, Goedkoop W (2008) Skogbruk och Vatten. En kunskapsöversikt. Forestry and Water. (In Swedish with English summary) A review. Redogörelse från Skogforsk. Nr 2. 64

Rusanen K, Finér L, Antikainen M, Korkka-Niemi K, Backman B, Britschgi R (2004) The effect of forest cutting on the quality of groundwater in large aquifers in Finland. Boreal Environ Res 9:253–261

Sabate S, Gracia C, Sanchez A (2002) Likely effects of climate change on growth of Quercus ilex, Pinus halepensis, Pinus pinaster, Pinus sylvestris and Fagus sylvatica forests in the Mediterranean region. Forest Ecol Manag 162:23–37

Saura M, Sallantaus T, Bilaletdin Ä, Frisk T (1995) Metsälannoitteiden huuhtoutuminen Kalliojärven valuma-alueelta. (in Finnish). In: Saukkonen S, Kenttämies K (eds) Metsätalouden vesistövaikutukset ja niiden torjunta. METVE-projektin loppuraportti, vol 2. Suomen ympäristökeskus, Helsinki, pp 87–104

Skogforsk (2008) Skogsburk med hänsyn till vatten – en handledning från Skogforsk. (In Swedish) 64

Tapio (2007) Hyvän metsänhoidon suositukset. (In Finnish) 100 http://www.tapio.fi/tapio_100_vuotta_1. Accessed 2 September 2010

Troendle CA, Wilcox MS, Bevenger GS, Porth LS (2001) The Coon creek water yield augmentation project: implementation of timber harvest technology to increase streamflow. Forest Ecol Manag 143:179–187

UN-ECE/FAO, 2000. Temperate and Boreal Forest Resource Assessment. Main Report

Varga L et al (1997) Monitorovanie lesných spoločenstiev v záujmovom území vodného diela Gabčíkovo (in Slovak) (Súhrnná záverečná správa za rok 1996). Lesnícky výskumný ústav, Zvolen

Vega-García C, Chuvieco E (2006) Applying local measures of spatial heterogeneity to Landsat-TM images for predicting wildfire occurrence in Mediterranean landscapes. Landscape Ecol 21:595–605

Velez R (1990) Preventing forest fires through silviculture. Unasylva 162. http://www.fao.org/docrep/t9500E/t9500e03.htm#preventing%20forest%20fires%20through%20silviculture. Accessed 20 May 2010

Vilhar U, Fajon S (2007) Forest hydrology. In: Forest and water: summary of project Interreg IIIA. Slovenian Forestry Institute, Slovenian Forest Service. Ljubljana, pp 15–21

Vilhar U, Simoncic P (2007) Climate change and drought in dinaric silver fir-beech forest in Kocevski Rog. In: Jurc M (ed) Podnebne spremembe. Biotechnical Faculty, Department of Forestry and Renewable Forest Resources Slovenia, Ljubljana, pp 381–399

Section III

Scales and Approaches for Forest – Water Interactions

Chapter 18
Introduction

Elve Lode

Scale is the main concept of the following chapters of this section of the book. Although the *scale* issue has been highly attractive in landscape ecology and fluvial systems research since the early 1980s, its use is increasingly challenging, especially in the fields of integrated research or management objectives. Together with improved availability for high quality technical use, the importance of the *scale* issue has arisen in relation to the limitations of the equipment used or the systems it selves.

Scaling as a specific procedure involving the aggregation or disaggregation of existing information or data, e.g. *upscaling* or *downscaling,* is an important tool both for serving purely research ambitions, or assessing management options and societal needs.

The huge variety of existing meanings of *scale* and *scaling* procedures is reflected in the following chapters of this book section, – however, the section is strongly related to the topic of forest management and the water cycle.

Alongside the fundamentals of forest scaling ecology, more applied background *upscaling* and *downscaling* approaches are demonstrated in the field of water factors in forest management. Content ranges from the plot to the landscape level, and in the framework of the small catchment of a mosaic structured landscape. Several limitations of meso- and macroscale forest and water interactions are emphasised, via water budget and rainfall-runoff modelling results, as well as existing results gained from geographic information modelling systems. GIS based upscaling limitations of aerial photographs at the plot and macroscale levels are demonstrated in the field of forested peatland images. Difficulties of forest and water management in urban areas have been highlighted via social and institutional issues. The meaning of *scale* has expanded via qualitative and quantitative characteristics of stable and extreme forest and forestry conditions.

E. Lode (✉)
Tallinn University, Institute of Ecology, Uus Sadama 5 10120 Tallinn, Estonia
and
Department of Soil and Environment, Swedish University of Agricultural Sciences,
SE-75007, Uppsala, Sweden
e-mail: elve.lode@tlu.ee

M. Bredemeier et al. (eds.), *Forest Management and the Water Cycle: An Ecosystem-Based Approach*, Ecological Studies 212, DOI 10.1007/978-90-481-9834-4_18,

The variety of the scale and approach examples presented in this section has been bound together with an integrated introduction chapter providing contemporarily used keywords, together with corresponding references related to forest management and water cycle *scale* and *scaling* topics.

Chapter 19
Forest Management and the Water Cycle: An Integrated Introduction to Scaling

Elve Lode, Matthias Langensiepen, Jüri Roosaare, Gebhard Schueler, and Harri Koivusalo

Introduction

Human, ecological and hydrological systems are multi-component non-linear systems with their own complex dynamics, but their internal processes cannot be de-coupled from changes that are induced by interactions between systems and system components (Bithell and Brasington 2009). Modelling approaches must consider relevant interactions and characterize appropriate time scales at which the various processes operate (Sturtevant et al. 2007; FORSYS 2008). S*ystem level* understanding about combined effects of physical, ecological and social processes is still limited and has led to the development of a large number of reductionist models which operate at *small levels* of system organization where explicit

---

E. Lode (✉)
Institute of Ecology, Tallinn University, Uus Sadama 5, 10120 Tallinn, Estonia
and
Department of Soil and Environment, Swedish University of Agricultural Sciences,
SE-75007 Uppsala, Sweden
e-mail: elve.lode@tlu.ee

M. Langensiepen
Institute of Crop Science and Resource Protection, Crop Science Group, University of Bonn,
Katzenburgweg 5, 53115 Bonn Germany
e-mail: matthias@langensiepen.net

J. Roosaare
Institute of Ecology and Earth Sciences, University of Tartu, Vanemuise St. 46,
51014, Tartu, Estonia
e-mail: juri.roosaare@ut.ee

G. Schueler
Research Institute for Forest Ecology and Forestry Rheinland-Pfalz, Hauptstrasse 16,
D-67705, Trippstadt, Germany
e-mail: schueler@forestclim.eu

H. Koivusalo
Department of Civil and Environmental Engineering,
Aalto University School of Science and Technology, FI-00076 Aalto, Finland
e-mail: harri.koivusalo@tkk.fi

M. Bredemeier et al. (eds.), *Forest Management and the Water Cycle: An Ecosystem-Based Approach*, Ecological Studies 212, DOI 10.1007/978-90-481-9834-4_19,

parameterization is possible. S*caling* model representations of forest systems components to larger scales is a challenging task, because properties emerge at higher levels which cannot be characterized by interactions between system components at lower levels (i.e. the whole is not the sum of its parts – see Pattee 1973; Isachenko 1973; Turner et al. 2001). A thorough understanding of forest ecological processes at different scales is an important prerequisite for reducing errors in scaling exercises (Chapter 21, this volume). Limited understanding or simplistic quantifications of small scale processes are significant error sources which easily propagate through a model framework and can thereby render scaling exercises useless.

Pattee (1973) proposed a *hierarchy theory* which provides solutions for characterizing the complex interactions in landscape-level human-dominated ecological systems (Wu and David 2002, but see also criticism of Wilby 1994). Depending on *scale invariance* and *multiscalarity* options, different scaling approaches can be applied such as *isometric* and/or *allometric* scaling (Peterson and Parker 1998), *self-organized criticality* (Bak 1996), or *generalized scale invariance* (Schertzer and Lovejoy 1985). *Generalized scale invariance* (Pecknold et al. 1997) can be applied, for example, to characterize *forest covers* as *fields* which represent *tree densities*. Categorizing and choosing temporal scales, appropriate for quantifying dynamic changes of forest ecological processes at given spatial scales, is another challenge.

The word *scale* is used in different contexts of *space* and *time* (e.g. Chapter 20, this volume) and can be seen as a method of *trading* which uses certain assumed sets of scales as references (e.g. Goodchild and Quattrochi 1997).

Scale as a term appeared in cartography and mapping during the eighteenth century (Sheppard and McMaster 2004). A *spatial scale* is defined as a geographic *extent* or *resolution* of an area. In cartography, *up-scaling* has been seen as a question of map *generalization*, which in turn is an aspect of human cognition, learning and knowledge representation (Buttenfield and McMaster 1991). In that sense, *scaling* is an interdisciplinary concept ranging from *naturalia* to *humaniora*.

Most of the pattern based processes in landscapes are scale dependent (Tate and Atkinson 2001). Extrapolation across scales is thus bound to certain limits (Wu 2004; Uuemaa et al. 2005).

In case of known *hierarchical* structures, *up-scaling* is applicable if the same system characteristics are reflected in model representations of small and large scale systems. Scaling may require changes in the model structure, its parameters or introduction of *multiscale* parameters (*allometric equations*, *Bayes theorem* for example – Chapter 21, this volume).

The concept of *down-scaling* originates from geophysics and applies to field variables such as air temperature, pressure, precipitation, ground roughness which are important variables in meteorological modelling studies (Benestad et al. 2008). At the landscape level, where many discrete changes with high local variances exist, remotely sensed data and cartographic methods are useful tools (e.g. Riitters 2005).

In ecology, the effects of scales on statistical data are termed as *the ecological fallacy problem* (Sankey 2008). *Individualistic fallacy* is characterized as the erroneous inference of *coarse-scale* properties from *fine-scale* properties. *Cross-level fallacy* occurs when inferences are made from single population observations and then erroneously generalized although the repeatability of processes, patterns and scales do not actually allow this. *Ecological fallacy* arises when erroneous inferences from *coarser* scales of observations are translated to *finer* scales.

Changes in forest structure, water flow patterns and translocation of chemicals are important research fields in forest hydrology. Changes in landscapes induced by forestry need to be evaluated with respect to forest internal and downstream water dynamics, stream geometry and their characteristics (National Research Council 2008). *Cumulative watershed effects* (CWEs) include hydrologic effects resulting from multiple land use activities over time and cause-effect relationships among forests, and watersheds over *large spatial* and *temporal* scales (Ibid).

Models become increasingly important in integrating forest hydrological processes at different *temporal* and *spatial* scales. Each chosen model representation of a scale has its unique limitation and complication in terms of parameterization, validation, input data requirement and application. Assessing model strengths and weaknesses is thus an important and integral part of scaling studies (McNulty et al. 1997).

Changing the structure of a model at a certain level of scale is one solution for translating inherent hypotheses to higher levels of system organization (Bithell and Brasington 2009), whereas another would be to change the time evolution of coupled components (*ergodic hypothesis*). Applying the ergodic hypothesis in the context of Earth sciences implies that parameters of temporal statistics can be substituted by those of spatial statistics. This is one of the substantial arguments of integrating geographic information systems (GIS) into modelling of complex systems.

Any kind of forest cover change in landscapes could be seen also as a part of the water management resting on the spatial information of the land-use and land-use changes due to human impact on land-cover, state of vegetation and stream network morphology on the catchment (de Fraipont and Tholey 2007). Quantifying such information for *large* river basins has traditionally been achieved by compiling traditional land maps. During the last decades, satellite imagery and *repeatable map products* facilitated *temporal* analyses of landscape pattern changes (Kasischke et al. 2004). Already in 1972, the first satellite images were produced by Landsat-1 for such purposes. Spatial resolutions of satellite images have been consistently improved since then. Multi-temporal image data allow the discrimination of land cover changes through digital change detection techniques (Lillesand and Kiefer 2000). Since landscape ecosystems are subjected to continuous *subtle* or *abrupt* changes, appropriate selections of satellite image acquisitions allow *periodic* tracking of land cover changes (Schlerf et al. 2007).

Criticism raised by the Consortium of Universities for the Advancement of Hydrologic Science (CUAHSI) showed that although remote sensing products provide adequate information about land cover, land use changes, states of vegetation, and stream network morphology, they nevertheless provide highly uncertain estimates of hydro-meteorological variables such as precipitation distribution, soil moisture variation, transpiration intensity, and aquifer recharge. Remote sensing techniques generally provide little information about geological heterogeneities in below-ground aquifers, their flow and nutrient loads in streams (CUAHSI 2007). A recently developed method allows remote sensing data determinations for spatial soil moisture patterns in top soil layers (Schüler et al. 2007). Identification of spatial patterns of low soil moisture content distribution at the ground level became feasible through the introduction of high resolution multi-temporal optical satellite imagery and validation with thermal remote sensing data (Seeling et al. 2007).

When human geographers began using *cartographic analysing methods*, they realized that their results depended on data aggregation methods and zoning schemes. *Grain size*, *zoning* and *areal extent* were three important parameters in Openshaw and Taylor's (1981) *modifiable areal unit problem* (MAUP) and its optimisation for concrete study cases (Uuemaa et al. 2005).

Developments in remote sensing, geographical mapping and environmental monitoring promoted the use of spatial data in forest vegetation management and decision-making practices (e.g. Pitt et al. 2000; FORSYS 2008).

Forest Ecohydrological Processes and Scaling

Estimating water balances of forested catchments and quantifying the consequences for stream- and groundwater flow intensities at different scales requires an understanding of the underlying regulation processes which is still fragmentary. After decades of hydrological modeling, available prediction tools are still incomplete and sometimes not noticeably better than simple rules of thumb (Klemes 1982; Kokkonen and Jakeman 2003; Passioura 1996). Recent advances in geostatistics, sensor technology and remote sensing provide new opportunities, however, for monitoring hydrological processes at increasingly *finer spatial* and *temporal scales*. The resulting data can be used for categorizing *spatial* and *temporal* scales of forest hydrological processes, model parameterization and development of decision support systems.

In forest hydrology, forest managers are primarily concerned about two questions (Landsberg et al. 1997): (1) How do forest management practices such as thinning and clear cutting affect water yield and quality from the catchment? (2) What is the relation between forest water balance and tree growth?

A solution to the first question can be found by scaling hydrological processes from *lower* to *higher levels* of forest hydrological system organization. The relevance of processes at different levels must be evaluated during such exercises. An answer to the second question requires reductionist scaling of hydrological information from the *forest stand* to *individuums* (i.e. trees).

Internal water fluxes in forest stands are commonly quantified with a mass balance equation

$$P - I - R - SW - D - ET - \Delta S = 0, \tag{19.1}$$

where P is precipitation, I intercepted water, R surface runoff, SW soil water flux, D drainage into deeper soil layers, ET the actual forest evapotranspiration and ΔS the change in the total water storage of the system. Water can be stored in tree canopies, snowpacks, soil surface ponds, soils and ground water reserves.

The regional distribution of precipitation is determined by the evolution of frontal or convective weather systems which interact with the topography of landsurfaces. The spatial and temporal extents of resulting precipitation events are difficult to predict (Kottegoda et al. 2003) and might thus better be monitored by regional weather station networks. Statistical analyses of measured precipitation patterns can be used for predictive purposes at various scales.

Rainfall interception is commonly determined by comparing rain amounts measured above a forest canopy or in an open site against throughfall measured below a forest canopy (Ward and Trimble 2003). The intercepted amounts of rainfall vary considerably between 10% and 30% (Landsberg et al. 1997) and depend on forest composition, tree geometrical characteristics and seasonal changes in foliage cover. In snow-affected zones, canopy interception of snowfall can become as high as 30% of precipitation in dense forests and as important as rainfall interception during frost-free periods (e.g. Lundberg and Koivusalo 2003). Forest managers can influence the amount of rain- and snowfall interception through species selection, planting arrangement and thinning.

Recent advances in geostatistics provide a large range of methods (Webster and Oliver 2007) for characterizing heterogeneities of forest soils and their consequences for spatial and temporal distribution of water in the vadose zone. Soil moisture content and hydraulic pressure head are commonly related by empirical water retention curves (Hillel 2003), which do not account for the impact of physical scaling factors on water retention characteristics and their spatial variability. Tuli et al. (2001) recently provided a new concept for simultaneous scaling of soil water retention and unsaturated hydraulic conductivity functions, which improves the determination of spatial variability of soil hydraulic functions. However, variability in soil physical properties and highly nonlinear behavior of transport processes prevent the application of general rules for scaling water transport from *pore* to *watershed*. An overview of different aggregation methodologies is given in Hopmans et al. (2002).

The amount of water available for tree stand transpiration is largely determined by the soil water volume accessed by tree roots. Root elongation cannot be characterized in isolation of other tree growth processes because they compete for growth resources (Lambers et al. 2009). A large number of feed-forward and feedback mechanisms interact in plant systems and determine the rates at which tree organs grow. They respond to changes in root zone soil moisture. Transpiration and adaptation mechanisms to changes in soil moisture supply and atmospheric evaporative demand vary considerably among tree species. The soil moisture regime is additionally altered by water uptake from the understory vegetation, particularly in young or sparse tree stands where it acts as a competitor for water resources. Forest managers have a range of options available for adjusting spatial and temporal patterns of soil water uptake at different spatial and temporal scale levels through planting and vegetation management strategies.

Groundwater systems can be broadly divided into zones of recharge and discharge. Their spatial magnitudes and locations depend on the topographic setting and stratification of sedimentary layers. Discharge areas are the lower boundaries of the forest vadose zone. Estimation of water discharge rates must consider the anisotropy and isotropy of the underlying sediment or rocks (Smith and Wheatcraft 2003).

In boreal and mountainous regions, snow hydrology is strongly influenced by the architecture and spatial arrangement of forest canopies. Winter precipitation is accumulated to the snowpack which can melt over relatively short periods of time in spring and then generate the main flood event of the year. The spatial pattern of snow accumulation is greatly influenced by topography and tree canopies, which decrease snow depths in dense forests through snow interception losses (Jost et al. 2007). Shading effects cause spatial differences in snowmelt, which is often driven by incident solar radiation on snow surface during periods of spring melt (Koivusalo and Kokkonen 2002).

Stream flow from densely forested areas is often delayed, because organic material on the soil surfaces, biological activity in soils, and complex root systems strongly promote infiltration. Forest cutting and thinning thus have large effects on water discharge dynamics from forested areas. Bosch and Hewlett (1982) reviewed 94 catchments experiments from all over the world and showed that water yield from catchment areas increases after vegetation removal. Landsberg et al. (1997) distinguish between three types of time courses after tree removal:

1. Initial increase in stream flow with a return to original levels when the vegetation cover is re-established.
2. Increase in stream flow followed by an extended period below the original level.
3. Sustained increase in stream flow particularly when land-use is changed to pasture or agricultural crops.

The characteristic roles of forests in preventing flood damages are strongly site and event specific and can thus not easily be generalized. The effectiveness of flood damage protection is also strongly dependent on vegetation management (Calder 2005). Some field studies indicate that forest management activities such as ditch drainage, road construction and soil compaction during logging, also influence outflow and flood responses rather than the presence or absence of forests themselves (Jones and Grant 1996). The highest floods are often generated after major large-scale storm events which lead to soil saturation of large watershed areas. The role of forests in reducing peak flows with long return periods easily become less important in such cases than their role in affecting flood peaks with intermediate return periods.

Scales and Scaling in Fluvial Systems

Hydrological processes take place at many terrestrial *scales* ranging from pores to continents. *Horizontally*, they occur in hill slopes and headwater streams (dimensions of 1–1,000 m), river basins and regional aquifers (dimensions of 1–1,000 km) or entire continents. *Vertically,* they appear at the micro scale of single soil pores, leaf stomata, or microbial communities, as well as throughout the entire atmosphere–plant–soil–bedrock continuum. The associated fluvial processes and properties cannot be explained by simple *up-scaling* from *finer* to *coarser* scales through aggregation. New processes and properties might emerge at each level of complexity instead (CUAHSI 2007).

The fluvial system is a *drainage basin*, i.e. the spatial geomorphic area occupied by a river network. It contains process-response components including the morphologic component of channels, floodplains, hill slopes, and cascading components of water and sediment (Schumm 1988). There are many spatial and temporal scales which are commonly dealt with in land level research and managing fluvial systems which confirm a great *scale dependency* in the nature of water related systems. Since there are no universal relationships underlying hydrologic processes at all scales, data *aggregation/disaggregation* with some *degree of confidence* has been seen as a principle approach in scaling hydrological processes (Blöschl 2001). For example, infiltration is a variable with complicated spatial patterns. It is typically measured at the point scale at which characteristics are not representative for the *watershed* scale. Instead, the *unit area* of impermeable surface (i.e. 0% of infiltration) and the *unit area* where 100% of precipitation infiltrates may be relevant for rainfall-runoff model applications.

Spatial information of infiltration characteristics may be compiled for an expert system integrated into a GIS system (Schüler 2006) in which the spatial distribution of runoff generating areas are deduced from the infiltration patterns. The accuracy of discriminating spatial runoff units depends on the quality of site classification conducted in a GIS system. A map with *runoff-generating site-units* at a *high resolution* is the common outcome of such studies. Interaction between discharge processes with hydrological processes in neighbouring areas can be captured by rainfall-runoff models (Casper 2004).

Disaggregation of landscape surfaces into sub-areas of quasi-homogeneous behaviour or hydrologically uniform response units, e.g. hydrotopes, can be chosen as a *meso-* and *macroscale* hydrological modelling approach (Becker and Braun 1999). Vertical processes (water fluxes and storages) within disaggregated units need to be quantified for this purpose and lateral flow processes characterized by providing linkages between scale units in landscapes and river basins. *Model scaling* does hence not need to be equalised with parameterisation for land surface discontinuities (e.g. surface water bodies, wetlands etc.). The calculated hydrotope or hydrotope-class-related fluxes are considered instead. The real landscapes can then take into account separate possible non-contiguous modelling units, and the area of the modelling interest can be extended up to the size, where the similarity of meteorological conditions is still valid. The model input databases of landscape surface characteristics can be extracted directly from GIS and hydrotope related characteristics (areal mean, storage capacity, porosity, root depth etc.). They could be applied to both, large and small reference areas (i.e. scaling issues becoming less relevant (Ibid)). The key issue in the application of the hydrotope approach for large areas is that the role of fluvial processes increases with the size of the area. Terrestrial processes described in landscape units dominate the discharge dynamics in small headwater catchments, whereas the river network processes govern the flow dynamics in large watersheds while the terrestrial processes primarily describe evapotranspiration losses.

GIS Based Scaling, Modelling and Monitoring

In cartography, a *map scale* is defined as the *ratio* between unit distance on the map and its equivalent distance in reality (e.g. *large scale* map of 1:10,000 scale equals *ratio* of $1/10{,}000 = 10^{-4}$ equals 1 mm on the map and equals 10 m on the land, and *small scale* map of 1:1,000,000 equals *ratio* of 10^{-6} and equals 1 mm on the map and equals 100 m on the land). Hence the *map* is a kind of *model* representation of reality. The formal *ratio* is only one scale aspect in spite of others inherent to modelling processes (Goodchild and Quattrochi 1997; Atkinson and Tate 2000).

In computer cartography with its electronic maps and possibilities to *zoom in* and *out,* the consistency between map scale and presented details is not preserved automatically, but needs special efforts. The widespread use of GIS with its easy change of *tessellation* (pixel size, vector-raster-TIN modes of spatial data presentation) has promoted a generic value of *geographic scale*, where the *small* linear dimensions are defined by the limited *spatial resolution* or *pixel size* (Goodchild and Quattrochi 1997). The *large* linear dimensions could thus be defined by the geographic extent of the study project or data collection effort. While the linear dimensions were well defined for some types of digital data it is not at all well-defined for landscape ecological characteristics.

To quantify the functioning of existing ecological systems, identification of landscape composition, structure and pattern from remotely sensed images is essential (Urban et al. 1987; Turner 1989). The perforation, dissection, fragmentation, shrinkage, attrition and other scale-dependent parameters of *landscape metrics* (Wu et al. 2002) are important targets for detecting image patterns and monitoring forest ecological systems for management over the time (Franklin et al. 2000). Since the nature of remotely sensed data is *grid-based*, combinations of grid data representation and grid based hydrologic functions for instance, allow for the conversion of forest management impact on downstream processes (Krogstad and Schiess 2000).

In comparison to drainage basins of main fluvial water systems, forested landscapes are comparably small and often fragmented by management and other disturbances. Traditional forest and related hydrological research is therefore targeted at understanding *small scale* watershed processes to better understand and improve management tools. Remote sensing and GIS are widely used tools for expanding visualisation and improving the prediction of forest hydrological processes within defined parts of larger watersheds (Anon 2008).

GIS plays a classical role in those modelling approaches where pre-processing input data and post-processing results are necessary (Fig. 19.1). Advances in the development and application of GIS also required an increasing integration of environmental modelling (Fedra 1994) and remote sensing techniques (Mesev 2007).

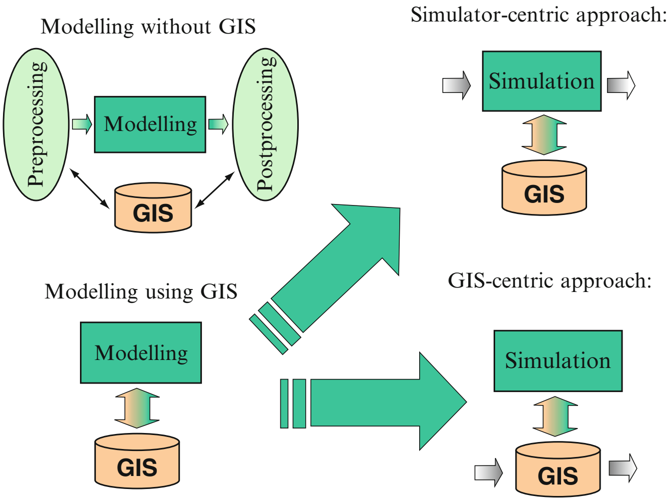


Fig. 19.1 A principle scheme for using of GIS in environmental modelling. *GIS-centric* modelling uses e.g. ArcGIS environment and models as extensions; *simulator-centric* modelling uses e.g. Stella and server-side GIS software

Spatial components are used in two different model approaches: (a) *Local dynamic models* where the same functional model is applied to all grid cells and no interactions between neighbouring grid cells are taken into account. (b) Models with *dynamically changing structures*, where multiple models for different grid cells have changing properties for representing interactions between neighbouring grid cells. One local model may be replaced with a new one (Miller et al. 2005).

Contemporary component models of geographical information systems include models of balances (MBs), thematic maps (TMs) and geostatistical features (GsFs) (e.g. Lenz et al. 2001; Mauser et al. 2001; Costanza and Voinov 2004). They are powerful tools for characterizing the spatial complexities of collected site data, simulating their dynamic changes in time and supporting management decisions in practice (Lovett and Appleton 2008). The availability of such tools provides opportunities for integrating ecological models into resource management applications, i.e. for (i) establishing inventories of resources, (ii) implementing spatial forecasting tools, and (iii) evaluating the effectiveness of management strategies in retro-perspective (Fournier et al. 2000; Matthies et al. 2007). Such computer-based *decision support systems* (DSS) are increasingly used to improve environmental management and planning. They were originally introduced by Matthies et al. (2007). DSS integration with GIS (Lovett and Appleton 2008) resulted in spatial DSS (SDSS), which have many applications such as forest harvest planning (Lubello 2008) or protecting forests to achieve sustained yields from water resources (Vacik and Lexer 2001).

According to Fournier et al. (2000), the application of ecological models at different spatial scales provides options for similarity analyses. Networks of forest inventory plots could be established for this purpose and used for testing whether observations at the plot level can be scaled to regional levels.

Zhang et al. (2006) illustrate an up-scaling approach for studying the spatiotemporal patterns of ecosystem processes which integrates simulation modelling, GIS, remote sensing, and field-based observations. Using a direct extrapolation scheme, a *patch* level ecosystem model was scaled up to quantify *landscape* level patterns of primary productivities and carbon source-sink relationships. *Hierarchical patch dynamic* scaling was then applied for scaling these processes from patch to landscape levels. Patches were defined according to plant species, age class, and richness criteria. The local ecosystem model was then executed for all patches with spatially varying parameter and input values. The landscape level variables of interest were subsequently computed by summing up model outputs generated for each single patch. Zhang et al. (2006) stress that meteorological conditions are critically important determinants of primary ecosystem productivity due to their direct and indirect effects on carbon and water cycling processes. Vegetation classification thus directly affects a range of physiological and ecosystem parameters.

Digital remote sensing uses spectral discrimination between land patterns for extracting information from images (Fournier et al. 2000). Optical sensors, which operate in the near and middle infrared wavebands, are most commonly used in forest research and management. Their spatial resolution varies widely between *centimetres* and *kilometres* (King 1995). Spatial resolution and revisit capability also highly correlate. Satellites can revisit specific locations more frequently, but have a relatively coarse resolution. The opposite is true for sensors with pointing capabilities. High spectral discrimination of digital outputs can be achieved with digital sensors. Hyperspectral sensors, which use a greater number of narrower spectral bands, are further improving this advantage and are particularly well suited for obtaining spectral indices of vegetation states (Chen 1996). In summary, the selection of a proper digital sensor for a given application is based on four main considerations: (i) The resolution must be appropriate for characterizing the area under study, (ii) the spectral discrimination should be sufficient, (iii) the revisiting interval must be frequent enough, and (iv) the costs of image acquisition and processing should be reasonable.

Direct pixel-based image analysis has several known limitations. *Spectral-mixture-models* are pointed out as useful alternatives for mapping degraded forest environments since the technique separates between soil, vegetation and shade abundances at the sub-pixel scale (Souza et al. 2003). *Smart joining* of pixels or segmentation for identifying spatial objects are important alternatives (Blaschke et al. 2008).

Gaps between digital remote sensing and aerial photography are gradually disappearing. Digital photography benefited from methodological developments in different fields of image analysis, such as geometric correction, image enhancement, and feature discrimination. Texture discrimination algorithms and spatial operators, which use spatial information contained in digital images, increase the potential for information retrieval.

In very high-resolution imagery, traditional per-pixel accuracy assessments are not suitable for evaluating the accuracies of tree counts. With many pixels falling on branches and background (soil, likely vegetation, logging slash or dead herbaceous), their inclusion in the accuracy assessment results into a negative bias. An object-based accuracy assessment based on classification of the area in relation to the central portion of a tree species and survival counts can be employed to avoid such bias (Haddow et al. 2000). Best results can be obtained by applying automated tree detection and crown delineation algorithms before GIS classification so that objects being counted could vary in size (Ibid). Most studies recognise that the lower limit of tree detection and classification is determined by pixel size limitations of narrow band imagers. Pixels of 30 cm cannot be used to accurately detect and count trees that are 10–30 cm in diameter (the core of the crown). Pixel sizes of 5–10 cm may be more appropriate for canopy-based sampling, or if trees more sparsely, but it would reduce capabilities for sampling within tree individuals (Ibid).

Similar to the limitation of scales and levels of research and mapping (Chapter 26, this volume) there are also scale limitations to *remote sensing, aerial photo monitoring* and *airborne laser scanning* which strongly depend on the extension of study areas, forest structure, and type as well as age of tree coverage (Sedykh 1995; Hershey and Befort 1995; Yu 2007).

Conclusions

Whatever decisions drive human activities, actual management is always paralleled by variations in natural resources and externalities which in turn result from natural and human disturbances, ecological succession, and recovery from previous disturbances. Relations between forest patterns, structure, functions and landscape as well as socio-economic environments are thus highly complex.

Since knowledge is never perfect, uncertainties are inevitably associated with modelling exercises and must be dealt with. Quantitative information about model uncertainties is a crucial requirement for a successful establishment of management procedures and decision support systems. A forester wants to know, for example: How accurate are the predictions of a forest model and what economical or ecological risks might evolve from its uncertainties? Such questions plainly bridge the gap between practitioners and scientists. Each scenario prediction is afflicted with a certain degrees of uncertainty. The larger the scale – temporal, spatial or in terms of complexity – the higher the uncertainties that might arise which potentially have severe consequences for practitioners. One way of avoiding this dilemma is reductionist scaling which is a good measure for our understanding of forest systems.

S*caling* facilitates the application of a range of relatively new measurement and modelling techniques which provide new and exciting insights into the complexities of forest ecological functioning in the frame of fluvial systems. Various decision-support systems will evolve from such scaling exercises in the future. They are useful tools for improving forest management efficiencies and our understanding of forest and water cycling responses to climate variability. The following chapters of Part III provide introductions to the principles and fundamentals of scaling using information from a large number of practical case studies which have been recently conducted in Europe.

Acknowledgments Work was prepared in the frame of COST Action FP0601 “Forman” project in collaboration of University of Tartu (Department of Geography), Estonia, the Swedish University of Agricultural Sciences (Soil and Environment Department), Institute of Ecology of Tallinn University, Estonia, and the Plant Production Systems Group of Wageningen University, The Netherlands. Used financial sources were the Estonian basic research foundation of SF0280009s07 project of “Impact of disturbances on wetland ecosystems in Estonia” and SF0180127s08 “Material cycling of landscapes in changing climate and land use conditions and ecotechnological control thereof” integrated with FP0601.

References

Anon (2008) Hydrological effects of a changing forest landscape. In: Committee on Hydraulic Impacts of Forest Management, Water Science and Technology Board, Division on Earth and Life Studies, National Research Council of the National Academies. The National Academies Press, Washington, DC, www.nap.edu 2009-06-10

Atkinson PM, Tate NJ (2000) Spatial scale problems and geostatistical solutions: a review. Prof Geographer 5(4):607–623

Bak P (1996) How nature works: the science of self-organized criticality. Springer, New York, 212 pp

Becker A, Braun P (1999) Disaggregation, aggregation and spatial scaling in hydrological modelling. J Hydrol 217:239–252

Benestad RE, Hanssen-Bauer I, Chen D (2008) Empirical-statistical downscaling. World Scientific Publishing Company, Singapore, 228 pp

Bithell M, Brasington J (2009) Coupling agent-based models of subsistence farming with individual-based forest models and dynamic models of water distribution. Environ Modell Softw 24(2):173–190

Blaschke T, Lang S, Hay GJ (eds) (2008) Object-based image analysis. Spatial concepts of knowledge-driven remote sensing applications. Springer, Berlin, Heidelberg. NY, 817pp+CD

Blöschl G (2001) Scaling in hydrology. Hydrol Process 15:709–711

Bosch JM, Hewlett JD (1982) Reference review of catchment experiments to determine the effects of vegetation changes on water yield and evapotranspiration. J Hydrol 55:3–23

Buttenfield BP, McMaster RB (1991) Map generalization: making rules for knowledge representation. Longman, London, 245 pp

Calder I (2005) Blue revolution: Integrated land and water resources management, 2nd edn. Bath Press, London, 353 pp

Casper M (2004) Zur Problematik der Prozessabbildung in kleinen Einzugsgebieten: Prozessstudien und Modellvergleich. http://ubt.opus.hbz-nrw.de/volltexte/2004/284

Chen JM (1996) Evaluation of vegetation indices and a modified simple ratio for boreal applications. Can J Remote Sens 22:229–242

Costanza R, Voinov A (eds) (2004) Landscape simulation modeling: a spatially explicit, dynamic approach. – Series: Modeling dynamic systems, XIII. Springer, New York, 330 pp

CUAHSI (2007) Hydrology of a Dynamic Earth. A decadal research plan for hydrologic. Consortium of Univ-ties for the Advancement of Hydrologic Science, Inc., Draft version 7.0. April 2, 2007, Released for community review and comment

Fedra K (1994) GIS and environmental modeling. In: Goodchild MF, Parks BO, Steyaert LT (eds) Environmental modeling with GIS. Oxford University Press, New York, pp 35–50

FORSYS (2008) COST Action FP0804 – Forest Management Decision Support Systems. http://fp0804.emu.ee/?id=stsm de Fraipont P, Tholey N (2007) Map anthropic and climate changes within the Alsatian Forests. In: Schüler G, Gellweiler I, Seeling S (eds) Dezentraler Wasserrückhalt in der Landschaft durch vorbeugende Maßnahmen der Waldwirtschaft, der Landwirtschaft und im Siedlungswesen. Mitteilungen aus der Forchungsanstalt für Waldökologie und Forstwirtschaft Rheinland-Pfalz, Nr 64/07. Zentralstelle der Forstverwaltung, Trippstadt, pp 267–275

Franklin SE, Dickson EE, Farr DR, Hansen MJ, Moskal LM (Nov/Dec 2000) Quantification of landscape change from satellite remote sensing. Forestry Chronicle 76(6):877–886

Fournier RA, Guindon L, Bernier PY, Ung C-H, Raulier F (2000 Nov/Dec) Spatial implementation of models in forestry. Forestry Chronicle 76(6):929–940

Goodchild MF, Quattrochi DA (1997) Scale, multiscaling, remote sensing, and GIS. In: Goodchild MF, Quattrochi DA (eds) Scale in remote sensing and GIS. Lewis Publishers, Boca Raton, FL, pp 1–26

Haddow KA, King DJ, Pouliot DA, Pitt DG, Bell FW (2000 Nov/Dec) Early regeneration conifer identification and competition cover assessment using airborne digital camera imagery. Forestry Chronicle 76(6):915–928

Hillel D (2003) Introduction to environmental soil physics. Academic, Amsterdam, 494 pp

Hershey RR, Befort WA (1995) Aerial Photo Guide to New England Forest Cover Types. USDA Forest Service, Northeastern Forest Experiment Station, General Technical Report NE-195

Hopmans JW, Nielsen DR, Byrant RB (2002) How useful are small-scale soil hydraulic property measurements for large-scale vadoze zone modelling? In: Raats PA, Smiles PAC, Warrik AW (eds) Environmental mechanics: Water, mass, and energy transfer in the biosphere, vol 129, Geophys Monogr. AGU, Washington, 247 pp

Isachenko AG (1973) Principles of landscape science and physical-geographic regionalization. University of Melbourne Press, Melbourne, Australia, 311 pp

Jones JA, Grant GE (1996) Peak flow responses to clear-cutting and roads in small and large basins, western Cascades, Oregon. Water Resour Res 32:959–974

Jost G, Weiler M, Gluns DR, Alila Y (2007) The influence of forest and topography on snow accumulation and melt at the watershed-scale. J Hydrol 347:101–115

Kasischke ES, Goetz S, Hansen MO, Ozdogan M, Rogan J, Ustin SL, Woodcock CE (2004) Temperate and boreal forests. In: Ustin SL (ed) Remote sensing for natural resource management and environmental monitoring. Wiley, Hoboken, NJ, pp 147–238

King DJ (1995) Airborne multispectral digital camera and video sensors: a critical review of systems designs and applications. Can J Remote Sens 21:245–273

Klemes V (1982) Stochastic models of rainfall-runoff relationship. In: Singh VP (ed) Statistical analysis of rainfall and runoff. Water Resources Publication, Littleton, CO

Koivusalo H, Kokkonen T (2002) Snow processes in a forest clearing and in a coniferous forest. J Hydrol 262:145–164

Kokkonen TS, Jakeman AJ (2003) Structural effects of landscape and land use on streamflow response. In: Beck MB (ed) Environmental foresight and models: a manifesto. Elsevier, Oxford

Kottegoda NT, Natale L, Raiteri E (2003) Reference parsimonious approach to stochastic multisite modeling and disaggregation of daily rainfall. J Hydrol 274:47–61

Krogstad F, Schiess P (2000) Cumulative impact estimation for landscape scale forest planning. In: 4th international conference on integrating GIS and environmental modeling (GIS/EM): problems, prospects and research needs. Banff, Alberta, Canada, 2–8 Sept 2000: GIS/EM4 no 59

Lambers H, Stuart Chapin F III, Pons TL (2009) Plant physiological ecology. Springer, Berlin, 610 pp

Landsberg JJ, Gower ST, Roy J (1997) Applications of physiological ecology to forest management. Academic, San Diego, CA, 354 pp

Lenz R, Haber W, Tenhunen JD (2001) A historical perspective on the development of ecosystem and landscape research in Germany. Ecol Stud 147:17–35

Lillesand TM, Kiefer RW (2000) Remote sensing and image interpretation. Wiley, New York, 763 pp

Lovett A, Appleton K (eds) (2008) GIS for environmental decision-making. CRC Press, Boca Raton, FL/Taylor & Francis Group, London, 259 pp

Lubello D (2008) A rule-based SDSS for integrated forest harvesting planning. Ph.D. Thesis, Università degli. Studi di Padova, Padova. www.isocarp.net/Data/case_studies/1605.pdf 2009-06-10

Lundberg A, Koivusalo H (2003) Estimating winter evaporation in boreal forests with operational snow course data. Hydrol Process 17(8):1479–1493

Matthies M, Giupponi C, Ostendorf NB (2007) Environmental decision support systems, current issues, methods and tools. Environ Modell Softw 22:123–127

Mauser W, Tenhunen JD, Schneider K, Ludwig R, Stolz R, Geyer R, Falge E (2001) Remote sensing, GIS and modelling: assessing spatially distributed water, carbon, and nutrient balances in the Ammer river catchment in Southern Bavaria. Ecol Stud 147:583–619

McNulty SG, Vose JM, Swank WT (1997) Scaling predicted pine forest hydrology and productivity across the Southern United States. In: Quattrochi DA, Goodchild MF (eds) Scale in remote sensing and GIS. Lewis Publisher, Boca Raton, FL, pp 187–209

Mesev V (ed) (2007) Integration of GIS and remote sensing. Wiley, New York, 296 pp

Miller I, Knopf S, Kossik R (2005) Linking general-purpose dynamic simulation models with GIS. In: Maguire DJ, Batty M, Goodchild MF (eds) GIS, spatial analysis and modeling. ESRI Press, Redlands, CA, pp 113–129

National Research Council (2008) Hydrologic Effects of a Changing Forest landscape (Free Executive Summary). Committee on Hydrologic Impacts of Forest management, http://www.nap.edu/catalog/12223.html 2009-06-10

Openshaw S, Taylor PJ (1981) The modifiable areal unit problem. In: Wrigley N, Bennett RJ (eds) Quantitative geography: a British view. Routledge & Keegan, London, pp 60–70

Passioura JB (1996) Simulation models: Science, snake oil, education, or engineering. Agron J 88:690–694

Pattee HH (ed) (1973) Hierarchy theory: the challenge of complex systems. G. Braziller, New York, 156 pp

Pecknold S, Lovejoy S, Schertzer D, Hooge C (1997) Multifractals and resolution dependence of remotely sensed data: GSI to GIS. In: Goodchild MF, Quattrochi DA (eds) Scale in remote sensing and GIS. Lewis Publishers, Boca Raton, FL, pp 361–394

Peterson DL, Parker VT (eds) (1998) Ecological scale: theory and applications. Columbia University Press, New York, 608 pp

Pitt DG, Runesson U, Bell FW (2000) Application of large- and medium –scale aerial photographs to forest vegetation management: a case study. Forestry Chronicle 76(6):903–928

Riitters KH (2005) Downscaling indicators of forest habitat structure from national assessments. Ecol Indicators 5(4):273–279

Sankey TT (2008) Scale, effects. In: Shekar S, Xiong H (eds) Encyclopedia of GIS. Springer, Berlin, pp 1021–1026

Schertzer D, Lovejoy S (1985) Generalized scale invariance in turbulent phenomena. Phys-Chem Hydrodyn J 6:623–635

Schlerf M, Borens S, Seeling S, Hill J (2007) Land use and land cover change in Rhineland-Palatinate based on multi-temporal image classification. In: Schüler G, Gellweiler I, Seeling S (eds) Dezentraler Wasserrückhalt in der Landschaft durch vorbeugende Maßnahmen der Waldwirtschaft, der Landwirtschaft und im Siedlungswesen. Mitteilungen aus der Forchungsanstalt für Waldökologie und Forstwirtschaft Rheinland-Pfalz, Nr. 64/07. Zentralstelle der Forstverwaltung, Trippstadt, pp 237–245

Schüler G (2006) Identification of flood-generating forest areas and forestry measures for water retention. For Snow Landsc Res 80(1):99–114

Schüler G, Gellweiler I, Seeling S (2007) Dezentraler Wasserrückhalt in der Landschaft durch vorbeugende Maßnahmen der Waldwirtschaft, der Landwirtschaft und im Siedlungswesen. Mitteilungen aus der Forchungsanstalt für Waldökologie und Forstwirtschaft Rheinland-Pfalz, Nr. 64/07. Zentralstelle der Forstverwaltung, Trippstadt, 337 pp

Schumm SA (1988) Variablity of the fluvial system in space and time. In: Rosswall T, Woodmansee RG, Risser PG (eds) Scales and global change. Spatial and temporal variability in biospheric and geospheric processes. SCOPE 35. Wiley, New York, pp 225–250

Sedykh VN (1995) Using aerial photography and satellite imagery to monitor forest cover in western Siberia. Water Air Soil Pollut 82:499–507

Seeling S, Nink S, Vohland M (2007) Methodenentwicklung zur Ableitung von Raummustern der Oberbodenfeuchte aus Fernerkundungsdaten. In: Schüler G, Gellweiler I, Seeling S (eds) Dezentraler Wasserrückhalt in der Landschaft durch vorbeugende Maßnahmen der Waldwirtschaft, der Landwirtschaft und im Siedlungswesen. Mitteilungen aus der Forchungsanstalt für Waldökologie und Forstwirtschaft Rheinland-Pfalz, Nr. 64/07. Zentralstelle der Forstverwaltung, Trippstadt, pp 255–266

Sheppard E, McMaster RB (eds) (2004) Scale and geographic inquiry: nature, society, and method. Wiley–Blackwell, Oxford, 288 pp

Smith L, Wheatcraft SJ (2003) Groundwater flow. In: Maidment DR (ed) Handbook of hydrology. McGraw-Hill, New York, pp 6.1–6.58

Souza C Jr, Firestone L, Silva LM, Roberts D (2003) Mapping forest degradation in the Eastern Amazon from SPOT 4 through spectral mixture models. Rem Sens Environ 87:494–506

Sturtevant BR, Fall A, Kneeshaw DD, Simon NPP, Papaik MJ, Berninger K, Doyon F, Morgan DG, Messier C (2007) A toolkit modeling approach for sustainable forest management planning: achieving balance between science and local needs. Ecol Soc 12(2):7, URL: http://www.ecologyandsociety.org/vol12/iss2/art7/2009-06-10

Tate NJ, Atkinson PM (2001) Modelling scale in geographical information science. Wiley, New York, 292 pp

Tuli A, Kosugi K, Hopmans JW (2001) Simultaneous scaling of soil water retention and unsaturated hydraulic conductivity functions assuming lognormal pore-size distribution. Adv Water Resour 24:677–688

Turner MG (1989) Landscape ecology: the effects of pattern on process. Ann Rev Ecol Syst 20:171–197

Turner MG, Gardner RH, O'Neill RV (2001) The critical concept of scale. In: Turner MG, Gardner RH, O'Neill RV (eds) Landscape ecology in theory and practice: pattern and process. Springer, New York, pp 25–45

Urban DL, O'Neill RV, JrHH S (1987) Landscape ecology: a hierarchical perspective can help scientists understand spatial patterns. BioScience 37:119–127

Uuemaa E, Roosaare J, Mander Ü (2005) Scale dependence of landscape metrics and their indicatory value for nutrient and organic matter losses from catchments. Ecol Indic 5:350–369

Vacik H, Lexer MJ (2001) Application of a spatial decision support system in managing the protection forests of Vienna for sustained yield of water resources. For Ecol Manage 143:65-76

Ward AD, Trimble SW (2003) Environmental hydrology, 2nd edn. CRC Press, Boca Raton, FL, 504 pp

Webster R, Oliver MA (2007) Geostatistics for environmental scientists, 2nd edn. Wiley, New York, 330 pp

Wilby J (1994) A critique of hierarchy theory. Syst Pract Action Res 7(6/Dec):653–670

Wu J (2004) Effects of changing scale on landscape pattern analysis: scaling relations. Landscape Ecol 19(2):125–138

Wu J, David JL (2002) A spatially explicit hierarchical approach to modeling complex ecological systems: theory and applications. Ecol Modell 153:7–26

Wu J, Shen W, Sun W, Tueller PT (2002) Empirical patterns of the effects of changing scale on landscape metrics. Landscape Ecol 17(8):761–782

Yu X (2007) Methods and techniques for forest change detection and growth estimation using airborne laser scanning data. Dissertation of the degree of Doctor of Science, Publications of the Finnish Geodetic Inst, N:0 137, Kirkkonummi

Zhang N, Yu Z, Yu G, Wu J (2006) Scaling up ecosystem productivity from patch to landscape: a case study of Changbai Mountain Nature Reserve, China. Landscape Ecol 22:303–315. doi:10.1007/s10980-006-9027-9

Chapter 20
Multidimensionality of Scales and Approaches for Forest–Water Interactions

Ivan Pilaš, Karl-Heinz Feger, Urša Vilhar, and Andreas Wahren

Introduction

The development and implementation of scaling methods up to the present have contributed to a better understanding of the effect of different land-use and forest managerial strategies on water quantity and quality. The main topics that are particularly relevant for policy-making in forest hydrology are: the comparative advantages and disadvantages of forest cover in maximizing downstream water yields; the role of upstream forests in season; water-quality preservation (Calder et al. 2007a). Contrary to the conservative approaches in which forests are mainly assessed for wood production, in the new emerging circumstances traditional forestry should be balanced against other forest goods and services and the role of European forests can be expected to develop further towards multi-functionality. Also, the EU is committed to biodiversity preservation to halt biodiversity loss within the EU (European Environmental Agency 2008). In order to determine the most efficient strategies in respect to the aforementioned goals, the scaling, i.e. linkage of scales from policy level to smaller scale ecosystem level, proves unavoidable. Scaling methods aim to

---

I. Pilaš (✉)
Division of Ecology, Croatian Forestry Institute, Cvjetno Naselje 41, 1450
Jastrebarsko, Croatia
e-mail: ivanp@sumins.hr

K-H. Feger
Institute of Soil Science and Site Ecology, Technische Universität Dresden Pienner Street,
19 D-01735 Tharandt, Germany
e-mail: feger@mail.zih.tu-dresden.de

U. Vilhar
Department of Forest Ecology, Slovenian Forestry Institute, Večna pot 2, 1000
Ljubljana, Slovenija
e-mail: ursa.vilhar@gozdis.si

A. Wahren
Dr. Dittrich and Partner Hydro-Consult GmbH, Gerlinger Straße 4, D-01718
Bannewitz, Germany
e-mail: kunde.wahren@gmx.de

M. Bredemeier et al. (eds.), *Forest Management and the Water Cycle: An Ecosystem-Based Approach*, Ecological Studies 212, DOI 10.1007/978-90-481-9834-4_20,

bridge the gap between policy makers operational on a broad scale and various landscape management services functional on different operational scales: forest managers, operational on forest stand scale, mostly concerned with the sustainability of forest production, environmental protection which emphasises the biodiversity issue in the range of single species to community and hydrologists concerned mostly with water supply, flood protection and water conservation on the watershed scale.

One of today's main concerns in perspective to future forest and water relations present awareness about irreversible changes in the earth's climate due to mostly human induced global warming. The scenarios of future development of the main climate variables, temperature and precipitation put strong pressure on landscape managers, policy makers and researchers to adopt commonly established traditional forest managerial practices towards mitigation and the diminishing of the very possible intensification of future extreme events (droughts, floods, forest fires, soil erosion). An increasing possibility of occurrence, together with an enlarged intensity of extreme events, undergo the commonly established knowledge, extending the inputs towards evaluating some outlier scenarios as an obviously very possible expected outcome in the nearer future.

Climate change is mostly hydrological change due to the shifting of temperature and precipitation as the basic inputs of the hydrologic cycle. The reconstruction of monthly and seasonal surface temperature fields for Europe show that in the late twentieth and early twenty-first century European climate is very likely (>95% confidence level) warmer than that of any time during the past 500 years (Luterbacher et al. 2004). However, the future rise of global temperature in the next century from 2°C to 6°C based on scenarios by General Circulation Models is very certain (IPCC). Observations prove the existence of a global warming trend: the average global temperature has increased by 0.8°C since 1900 and the 12 hottest years observed globally since 1880 have all occurred after 1990. Even greater hazards present extremes at regional scales, such as the dry spell in 2003 which affected many European areas with larger amplitudes than extremes at a global scale (Luterbacher et al. 2004). Apart from a global rise in temperature, the long-term precipitation records across Europe are regionally shifted; the long term decrease of precipitation is evident in Southern Europe and the Mediterranean basin and an increasing trend of precipitation is common in Northern Europe (Trenberth et al. 2007). These trends broadly determine the intensification of hydrological extremes which determine future activities of forest and water management at the regional gradient in Europe; the mitigation of droughts to retain long-term sustainability of forest production in the south and mitigating the floods in the north.

Stable Conditions: Water, Quantity, Quality and Management – Impact of Forest Management on Water Quantity and Quality

Forests and Water Retention Capacity

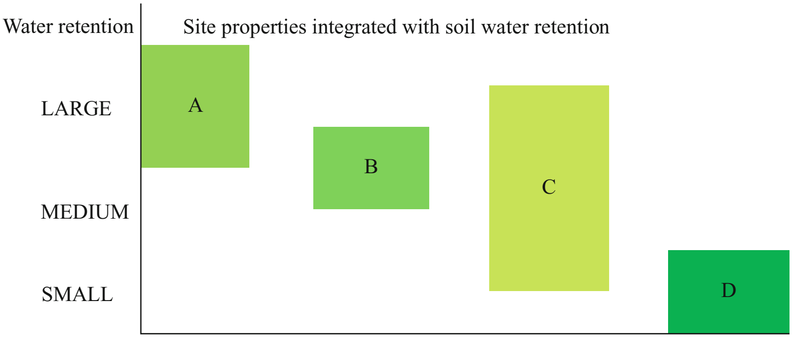
Stable forests in good condition growing on appropriate sites are most suitable land use category for retaining water in case of extremely abundant precipitation events (Frehner et al. 2005). Influence of forest cover on runoff quantity depends on medium- and long-term improvement of soil conditions, especially on water retention capacity of soils. Forest cover can increase water retention capacity of soils in case of deep but slowly drained soils and in case of very wet soils. Improvement of water retention capacity of well drained soils by forest cover will be much smaller. The soil compaction by forest mechanization can decrease the water retention capacity of soils significantly (Croke et al. 2001). Infiltration of water decreases on compacted soils which can result in surface soil erosion. However, we must emphasize that forests can only retain water in periods when the water saturation of soils is not reached. Forests can reduce peak flows for storm of short duration and low intensity, but cannot prevent the occurrence of floods that are of high intensity and long duration over a large area (Chang 2003). As the forest soils absorb large quantities of water, a forest gradually loses its capacity to retain high runoff and flooding, and eventually, when critical amount of precipitation is reached, fails to control the flood (Frehner et al. 2005). This can be the case by:

1. Short and intense precipitation events in small areas
2. Long lasting intense precipitation in large areas
3. Intense precipitation in large areas, when soil water saturation was already reached (mainly in spring and autumn)

Figure 20.1 shows the influence of forest condition on water retention capacity of soils in case of large precipitation amount, where: (A). The largest impact of forest condition on water retention capacity of soils is on deep, slowly drained soils (C). For these areas forest management practices will have largest impact on water quantity. In case of low water retention capacity of soils, independently of forest condition (D), the soils are extremely shallow, extremely permeable or extremely wet. The water retention capacity of soils depends on both soil properties and forest condition. On deep, well drained soils the water retention capacity of soils is maximal, irrespective of forest conditions. In such areas the continuous forest cover should be priority (Vilhar 2009). Different forest management practices in such areas will not improve soil water retention capacity but can aim towards decreased surface runoff and soil erosion.

Five Most Important Indicators for Water Quality and Quantity Influenced by Different Forest Management Practices

Influence of forest cover on total runoff from a drainage basin and water quality in rivers is increasing according to increasing share of forest cover in the basin. Clearing of forest areas in the drainage basin can lead to sediment loss because of erosion processes (Binkley and Macdonald 1994; Prybolotna 2006), increase of high waters after storms and snow melting (Von Burger 1954a, b) or even increase of floods and their frequency (Veny 1986). Forest removal can result in increased exposure of water bodies to solar radiation and consequently increased water temperatures at clear skies (Binkley and Macdonald 1994). Concentration of oxygen can be altered which could change the productivity and composition of



A – large water retention capacity of soils, irrespective of forest condition. Sites on deep, well drained soils.

B – medium water retention capacity of soils, irrespective of forest condition. Sites on medium deep, well drained soils.

C – large water retention capacity of soils in case of good forest condition / low water retention capacity of soils in case of bad forest condition. Sites on deep, slowly drained soils.

D – small water retention capacity of soils, irrespective of forest condition. Sites on extremely wet, shallow or permeable soils.

Fig. 20.1 Influence of forest condition on soil water retention capacity in case of large precipitation amount (Frehner et al. 2005)

water ecosystems with accelerated algae growth and eutrofication (Lee 1980). Nitrogen and phosphorous concentrations are lower in water bodies from forested drainage basins compared to drainage basins with prevailing agriculture land (Binkley and Macdonald 1994; Chang 2003). Main source of nitrogen and pollutants in water are use of pesticides and fertilizers, pollution induced by housing, forest roads, forest mechanization and air deposition by local pollutants. Most frequent source of pesticides in freshwaters is neighboring agricultural land. Strips of riverside vegetation can reduce or even prevent the input of pesticides into freshwaters significantly (Binkley and Macdonald 1994).

According to above mentioned influences of forest management on drinking water quality and quantity five most important indicators have been selected (Vilhar and Fajon 2007):

- Concentration of pollutants in the water
- Nitrogen concentration in the water
- Sediment loss/erosion
- Runoff
- Water temperature

Other forest management practices apart from clear cut system are rarely discussed in hydrological studies. Thinning by 20% reduction in wood stock is reported to be the threshold for influencing the runoff quantity and sedimentation loss (Vilhar and Fajon 2007). Nevertheless a study by Serengil et al. (2007) showed that even a 11% reduction of wood stock of a deciduous forest can alter stream nutrient discharge with increased sedimentation due to thinning. Frequency and intensity of thinning does influence hydrology of a forested drainage basing, but the rate of influence depends on local site conditions: climate, soil and vegetation. In alpine and subalpine forests the impact of forest management measures can be observed 80 years after they were executed (Chang 2003). In mixed deciduous forests the impact of forest management measures ceases in 4 years (Ibid). Also the presence of ground vegetation is very important for hydrological regime. Creation of a gap in a spruce stand with lush ground vegetation will not alter the nitrogen concentration in the groundwater significantly (Weis et al. 2001). On the contrary, removal of trees in a stand with sparce ground vegetation layer can result in increased nitrogen concentration in the groundwater (Ibid). In a Dinaric silver fir – beech forest a creation of a gap with bare soil resulted in increased drainage compared to untouched stand. Advanced growth of ground vegetation layer and natural regeneration diminished the drainage differences between gaps and stands (Vilhar 2006). In such forest ecosystems small-scaled forest management measures should be taken to avoid soil degradation and erosion (Vilhar 2009). Mixed tree species composition is important for hydrological regime mostly because of rooting which influences the soil and stand stability. Natural tree species composition and adequate share of each species strengthen self-preservation capability of forest sites (Frehner et al. 2005).

Harvesting and forest mechanization can chronically pollute drinking water sources in forests. Most evident reasons are use of principally forbidden chemicals in forests, inappropriate forest roads construction, use of mineral oils for chainsaws and hydraulic systems of forest mechanization (Košir 2006). One drop of mineral oil can pollute 25 l of water to become undrinkable (Barlow and Clarke 2002). One liter of mineral oil, poured into sewage system can pollute 1 ha of water surface (Arnšek 2001).

Different forest management practices have different impact rate on selected drinking water quality and quantity indicators. Eight forest management measures were selected that can impact the drinking water quality and quantity and are usually part of forest management planning:

- Share of clear cut area in the drainage basin
- Frequency, intensity and technique of harvesting
- Tree species composition
- Crown density, cover percentage
- Distribution of growth classes
- Vertical and horizontal stand structure
- Forest regeneration, ground vegetation
- Coarse woody debris

Table 20.1 Impact of forest management measures on drinking water quality and quantity indicators

Forest management measure/Indicator	Concentration of pollutants in the water	Nitrogen content in the water	Sediment loss (erosion)	Runoff	Water temperature
Clear cut area	*	***	***	***	***
Frequency, intensity, technique of harvesting	***	*	**	**	*
Tree species composition	**	***	**	***	*
Crown density, cover percentage	*	***	***	***	**
Distribution of growth classes	*	**	**	**	***
Vertical and horizontal stand structure	*	**	**	**	*
Forest regeneration, ground vegetation	*	**	**	**	*
Coarse woody debris	*	**	*	*	*

* – Low impact ** – Medium impact, *** – High impact

Frequency, intensity and technique of harvesting have much higher impact on concentration of pollutants in the drinking water than vertical and horizontal stand structure, for example. Share of clear cut area in the drainage basin has much larger impact on sediment loss than on concentration of pollutants in the drinking water. These relations between forest management measures and selected indicators for drinking water quality and quantity are presented in the Table 20.1.

Best Management Practice Guidelines for Water Quality and Quantity in the Frame of Close-to-Nature Forest Management

Close-to-nature forest management assures that with the human interference into forest ecosystem sustainability and multifunctionality are preserved. One of basic principles of close-to-natural management is to imitate natural structures and natural regeneration patterns, for example natural disturbances (Bončina and Diaci 1998). They cause gap formation of different size and influence stand dynamics and horizontal forest structure. The following recommendations for enhancing water quality and quantity aim to improve practices in forestry and other activities in forested areas where drinking water resources are of great importance. Recommended actions are classified on the basis of selected indicators for drinking water quality and quantity, presented in Table 20.2.

Table 20.2 Recommendations for forest management related to drinking water quality and quantity indicators

Factor	Recommendation for forest management
Concentrations of pollutants and nitrogen in the water	Compulsory use of biodegradable chainsaw lubricants and oils in hydraulic machines in forest (Košir 2006).
	Prohibition of washing, maintenance and repair of forest mechanization in a forest (Trontelj 2006).
	Selective thinning should be performed in narrow riparian zones of forest to remove old and unstable trees.
	Strip of riparian vegetation (including trees and shrubs) should be established and maintained to prevent or mitigate pesticide and other pollutants in the freshwater (Binkley and Macdonald 1994).
	Storage facilities for fuel and oil should be kept away from water courses and other water bodies (Mulkey 1980).
	High concentrations of ash and dust must be avoided in close distance of water courses (Mulkey 1980).
	Closure of forest roads when there is no harvesting. This enables to stabilize the roadway and recovery of the vegetation along the roads (Mulkey 1980).
	Use of heavy machinery on erodible soils should be minimized or avoided, in the riverbed prohibited (Mulkey 1980).
Sediment loss (erosion)	Surface of the whole catchment area should be covered with vegetation. In areas whit bare soils pioneer vegetation should be established (Frehner et al. 2005).
	Continuous presence of natural regeneration, developing under shelter of adult trees (Frehner et al. 2005)
	Old, unstable trees, fallen trunks, uprooted stumps should be removed from the riverbed and its immediate vicinity to prevent blocking the channel and flooding in the case of high waters (Frehner et al. 2005).
	Landslides can be prevented with planting of tree species having deep root system to stabilize the soils as well as draining excessive water (ash, oak, maple, black alder and fir tree, dwarf pine, Austrian pine) (Frehner et al. 2005).
	Construction of skidding tracks and transportation roads on less steep slopes to avoid excessive ground stripping and excessive use of trenches and the length of forest routes (Mulkey 1980)
	Avoidance of any activities which could damage upper layer of forest roads (i.e. log skidding) (Mulkey 1980).
	During the period of heavy and persistent rainfall, closure of forest roads can prevent formation of ruts and their flooding (Mulkey 1980).
	Sections of already constructed forest roads with steep inclination should be preventively cemented or covered with asphalt (Mulkey 1980)..
	Frequently used roads and entry points on erodible soils should be hardened with rocks, lumbers or branches (Mulkey 1980).
	In the event of erosion urgent preventive measures should be taken and no cutting, burning or damaging of upper soil layer at least 100 years after the event (Twery and Hornbeck 2001)
	Working in dry weather (in spring or summer) can significantly decrease the risk of erosion near watercourses and drinking water collectors (Nisbet 2001).
	With consent of local communities, water supply can be temporarily suspended if large-scale activities are performed in forest, thus preventing pollution of drinking water.

(continued)

Table 20.2 (continued)

Runoff	The most appropriate forest structure for preventing high flows is an uneven-aged stand with dense canopy cover, diverse vertical structure and even distribution of growth-phases (Frehner et al. 2005). To decrease high flows the total share of non-forested areas, gaps and regeneration younger than 10 years should be less than 25% of the drainage basin (Twery and Hornbeck 2001). To decrease high flows the canopy cover of all forest stands should be more than 70% in the whole drainage basin as well as in the riparian cone (Twery and Hornbeck 2001).
Water temperature	To maintain water temperature the total share of gaps should be less than 10% of the drainage basin (Twery and Hornbeck 2001). The canopy cover of forest stands in the riparian cone should be more than 70% (Twery and Hornbeck 2001). Natural regeneration should take place in gaps, smaller than 0.1 ha (Twery and Hornbeck 2001) Coarse woody debris should be present in the forest stands and water bodies, single old trees should be left untouched as "habitat trees" (Twery and Hornbeck 2001).

Drought and Decay of Woodlands

Exposure of European Forests to Drought

Drought is a natural hazard, i.e., a long period of abnormally low rainfall which adversely affects growing or living conditions. Drought differs from aridity, which characterises low rainfall regions, and should be considered relative to some long-term average condition of balance between precipitation and evapotranspiration. Meteorological drought on the whole encompasses the number of days with precipitation less than a specified threshold (in regions characterized by a year-round precipitation regime) or actual precipitation departures to average amounts on monthly, seasonal or annual time scales. Hydrological drought is emphasised on a watershed or river basin scale i.e. how the deficiency in precipitation reflects on the hydrologic system. Drought can be quantified by the application of various less or more complex hydrological indices based solely on precipitation data or included water storage and evapotranspiration. Drought indices are appropriate for analysing the series of weather data with the aim of obtaining the fundamental characteristics of drought events for a given location or region; the drought strength, the number, the mean duration, the maximum duration of droughts of a given intensity, and the trend in drought incidence. During the last century (1901–1999) extreme droughts, as classified by the PDSI (The Palmer Drought Severity Index), occurred with greater frequency over continental Eastern Europe. On the other hand, droughts were rarer along the North West European seaboard, the Mediterranean seaboard, and the Alps (Lloyd-Hughes and Saunders 2002). The same results confirm that drying tendencies exist in central Eastern Europe and western Russia which is in accordance with the IPCC findings. According to Vogt and Somma (2000) drought, especially in the Mediterranean area, is a reccurring phenomenon, i.e. its behaviour comprises a cyclical periodic pattern. The events and locations of droughts in Europe in recent times occurred as follows: 1973 (Austria, Germany, Czech Republic and Slovakia), 1976 (Northern Europe – Scandinavia to France), 1984 (North and west UK), 1988–1992 (most of Europe), 1992 (Germany, Hungary, Bulgaria and much of western Russia), 1990–1995 (Spain, Portugal), 1995 (Ireland, UK, Norway and Sweden), 1999 (Finland), 2003 (much of continental Europe) (Source: Ben Lloyd-Hughes, Benfield Hazard Research Centre, UCL).

Projections from catchment areas across Europe (according to IPCC SRES scenario A2) (Dankers et al 2007), including the catchment area above 1,000 km^2 predict a reduction of discharge of up to 40% for the period 2071–2100 in respect to the base period 1961–1990 most severely in Mediterranean basin and South-eastern Europe but also in the continental part (Fig. 20.2). An increase of discharge is predicted overall for the Northern part of Europe. The simulations of ECHAM4 and HadCM3 climate models and WaterGAP predict a very strong increase of frequency of recurrence of today's 100-year drought events to less than 10 years at 2070 in the Mediterranean area and South-eastern Europe (DMP Report 2007). Drought in forestry considers precipitation shortages and its impact on particular forest tree species in relation to its water requirements and hydraulic properties. Drought causes a decrease in the soil water potential and large increases in the hydraulic resistance at the soil-root interface. The main effect of drought is the occurrence of cavitation and xylem embolism in trees when the water potential in the xylem falls below specific vulnerability thresholds (Cruiziat et al. 2002). The limited water uptake by trees also has consequences on their nutrition, vitality and health status (defoliation and discoloration of the canopy) (Potočić et al. 2005). Recent studies which worked with various aspects of drought and heat waves in forest ecosystems (Granier et al. 2007; Pichler and Oberhuber 2007; van der Werf et al. 2006; Desprez-Loustau et al. 2006; Archaux and Wolters 2006; Breda et al. 2006; Rebetez et al. 2006) took into consideration drought and heat waves in 2003 which were the most severe in last 500 years and spread over large parts of the European continent.

The predicted increase of temperature, carbon enrichment and nitrogen deposition as a consequence of global warming can enhance the current limits of forest growth. On the contrary, the future availability of soil water presents a strong growth limitation. Soil water depletion in new environmental circumstances can be induced by regional precipitation deficits but also as a result of enhanced physiological processes, lead to higher soil water uptake of trees in the regions where more favourable conditions for forests could be expected. According to the climate scenarios based on scenarios which assume a doubling of CO_2 emissions in the twenty-first century (Echam and Hadley climate), the future development of water scarcity across Europe have a strong regional gradient (Mannerkoski 2005, Silvistrat project). Future forest and water relations in the main bioclimates of Europe according to the above models are:

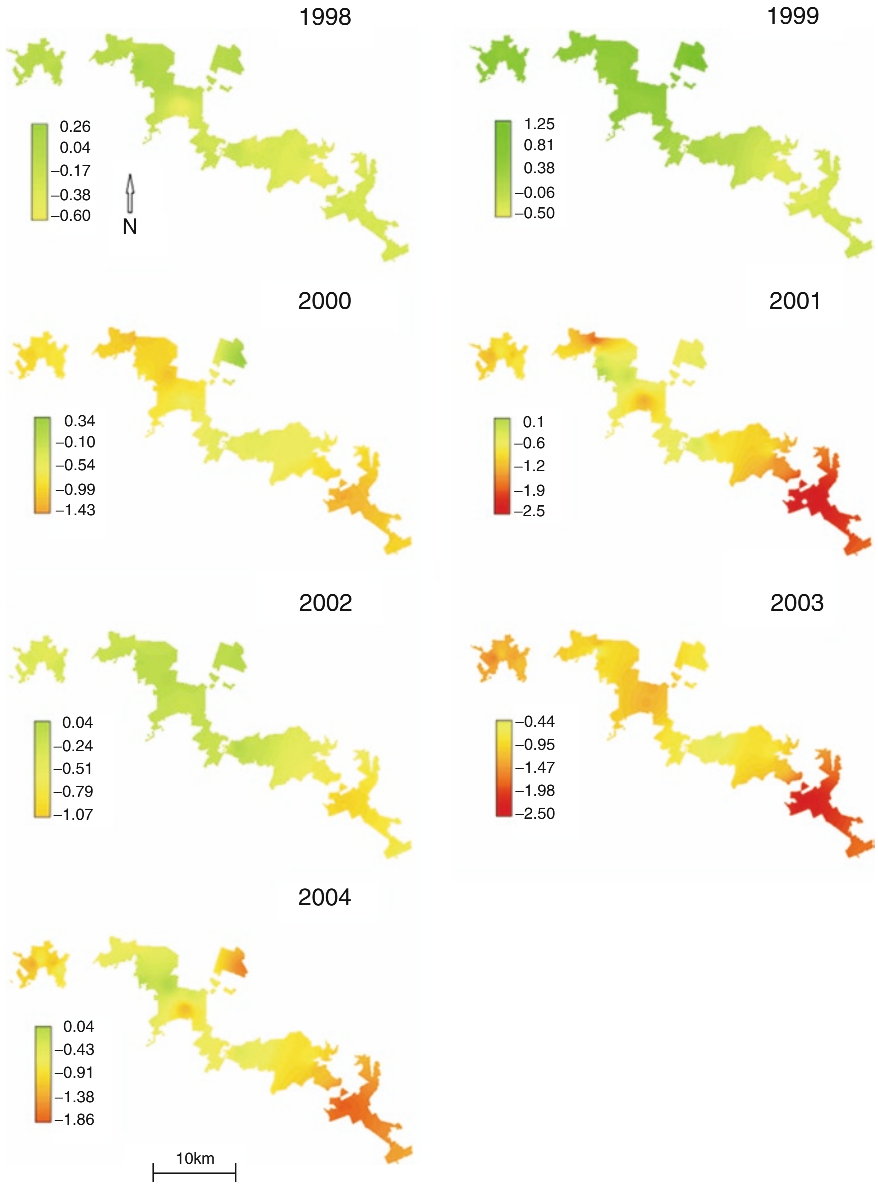


Fig. 20.2 *The onset of climate change*: decline of groundwater discharge during the drought episode 2000–2004 with yearly groundwater levels up to −250 cm in relation to stable state conditions (1997). The interpolated spatial trend was calculated from 22 piesometric locations across the Našice lowland forest area in eastern Croatia

Boreal Forests

Overall higher temperatures and precipitation, with a shorter duration of snow cover and soil frost. On the western edge of the boreal forests, reduced snow accumulation may decrease on shallow soils the soil moisture in early spring and early summer and this may cause problems for *Picea abies* and other shallow-rooted tree species.

Atlantic Forests

Increases in temperature and precipitation during winter and a decreased amount of precipitation during the growing season in summer. Due to higher atmospheric nitrogen depositions and CO_2 feeding, vegetation is expected earlier in spring which may lead to premature depletion of soil water in summer.

Continental Forests

Changes in precipitation are uncertain but in general, an increase of precipitation during winter and a decreased amount of precipitation during the growing season in summer is expected. Together with increasing temperatures, CO_2 and nitrogen potential, earlier depletion of water in summer can also be assumed. Decreased precipitation during summer along with increased evapotranspiration due to the higher temperatures will result in increased drought stress in all but those areas that have access to groundwater supplies from elsewhere.

Mediterranean Forests

Lower precipitation in summer or equal or slightly higher precipitation in winter. Very restrictive conditions regarding water availability, precipitation accounts for about one half of the potential evapotranspiration, and actual evapotranspiration is very often limited.

Forest-Tree Response at Various Spatio-Temporal Scales

Drought presents one of the strongest stress factors of forest mortality, but its effect on tree health status (defoliation) or growth indicators such as tree ring increment usually become obvious with a time lag of 1–2 years. The effect of the dry spell in 2003 is reflected in increased tree mortality in consecutive years, in 2004 in particular. In most cases, drought does not result in acute visible damage to the trees, but induces physiological disorder by decreasing carbon and nutrient assimilation and, possibly, by induction of xylem embolism that blocks the water transport to leaves (Breda et al. 2006). During this time, the trees are probably particularly susceptible to additional stress factors such as defoliation by insect larvae (which also reduces their carbon reserves) or severe winter frost (whose impact is aggravated by the preceding reduction of the carbon reserves). Drought with other abiotic and biotic stress factors comprise an interrelated factorial complex (Thomas 2008) which makes it difficult to clearly distinguish its significance in the cause effects assessments of forest mortality (Klap et al. 1999). Therefore, new approaches aimed towards a mechanistic understanding of drought stress on plant metabolism should present one of the key issues of improvement of resilience of ecosystems to climate change.

The existing broad scale assessments of forest cover, forest health conditions and overall performance of vegetation across different regions of Europe together with predicted discharge from watersheds can serve as a guide towards critical spots where particular future activities of forest and water management should be emphasised towards drought mitigation. The most recent results of broad scale monitoring of forest conditions in Europe on 4,834 plots, $16 \times 16\ \text{km}^2$ grid (Institute for World Forestry 2008), established that 21.9% of all trees assessed had a needle or leaf loss of more than 25% and were thus classified as either damaged or dead. The most threatened forest tree species in Europe by far are European and sessile oak with the share of damaged and dead trees being 35.2% (2007) and with a high positive trend. The progression of climatic trends over the regional gradient across Europe (Trenberth et al. 2007) at this stage can not be strictly confirmed in the crown damage status but there is evidence that the highest increasing rates of defoliation occur in Mediterranean species such as Holm oak and Maritime pine. Damage to the Holm oak in 2007 even exceeds damage to the European beech which was recently considered the second most threatened tree species but with a more stable trend over time. On the contrary the “northern” species such as Norway spruce and Scots pine comprise a decreasing trend of defoliation, i.e. an improvement in health conditions.

On the broad European scale the intraseasonal response of drought events on forests can also be assessed by remote sensing through anomalies in phenology using indices such as FAPAR (Fraction of Absorbed Photosynthetically Active Radiation) (Verstraete et al. 2008). FAPAR represents the fraction of solar energy which is absorbed by vegetation and presents an indicator of the presence and state of the vegetation cover, and can also serve as a response in cause effect assessment of spatial and temporal soil water depletion. The spatial extent of drought and forest regions which are mostly affected can easily be distinguished by comparing monthly anomalies in respect to the base period (as, for example, the severe drought in 2003) (Gobron et al. 2006). The areas in Europe with the lowest vegetation feedback of FAPAR index are Mediterranean areas of Spain, France, Italy, Croatia and Greece.

The response of trees and forest cover to drought is mostly ecosystem dependent. It is the result of the present state of climate, coupled with hydraulic properties of trees, soils and boundary interactions. Drought mitigating practices should therefore be downscaled and orientated to a particular forest type and environmental conditions. Timely recognition of drought, either as a periodical recurring event (dry spells) or long term water scarcity (i.e. the effect of a decrease of groundwater supply), presents a first step towards operational performance of short term drought mitigation measures at a forest stand level. Drought can influence temporal systematic patterns of tree water fluxes as diurnal variations, intra-seasonal yearly variations, decadal and multidecadal variations. These temporal systematic patterns occur mainly as a result of variations in physiological processes that govern forest growth, i.e. physical factors, radiation, air temperature, air humidity and wind, and ecological factors such as nutrient availability, canopy architecture and leaf longevity (Landsberg and Gower 1997).

Diurnal variations of tree water relations can be assessed at leaf level (stomatal conductance and water potential) and whole tree level (sap flow measurements). Through the use of very sensitive dendrometers, the upward flow of water in the xylem of mostly dissolved nutrients from soils and downward flow of photosynthetic products (Sevanto 2003) can be distinguished. The daily transpiration pattern, which can roughly be approximated with bell shaped distribution, resulted from diurnal regulation of stomatal aperture which depends on signals from light, the current stomatal aperture, crown water potential, and tree water deficit (Zweifel et al. 2007). The stomata typically opens at twilight but their aperture during the day is non linear and is very dependent of water availability. Extreme drought in general causes only partial opening of stomata which remain closed for the rest of the day and as such, causes a significant change in the daily transpiration distribution, i.e. the asymmetry to the left towards the dawn. In drought conditions (Infante et al. 2003) daily amplitude of water potential decreases if predawn water potential become more negative.

Intra-seasonal variations in tree transpiration throughout the year or vegetation season occur with much stronger amplitude than diurnal variations, and also comprise a decline-dependence pattern in relation to a drought. Due to the usually increased soil water content, the strongest amplitude of transpiration and tree diameter swelling (Fig. 20.3) occurred during the beginning of the growth season (budding and leaf formation). Later during the season, the amplitude of variations decrease as the soil water content is reduced until the late summer when tree stem growth seizes. The empirical allometric ratio between maximal amplitude of tree diameter growth at the beginning of season and the cumulative tree stem growth at the end can be established (Pilas et al. 2007) according to which the effect of drought can be forecasted on a seasonal scale. The time of manifestation of drought during the year implies the extent of its impact on the forest stand condition; at seasonal level trees are much more sensitive to water depletion during the early stage of growth, i.e. spring rather than in later summer and autumn.

Drought and Adaptive Forest Management Strategies

In newly emerging environmental conditions, forest management should aim to establish adaptive forest managerial measures which aim to reduce as many

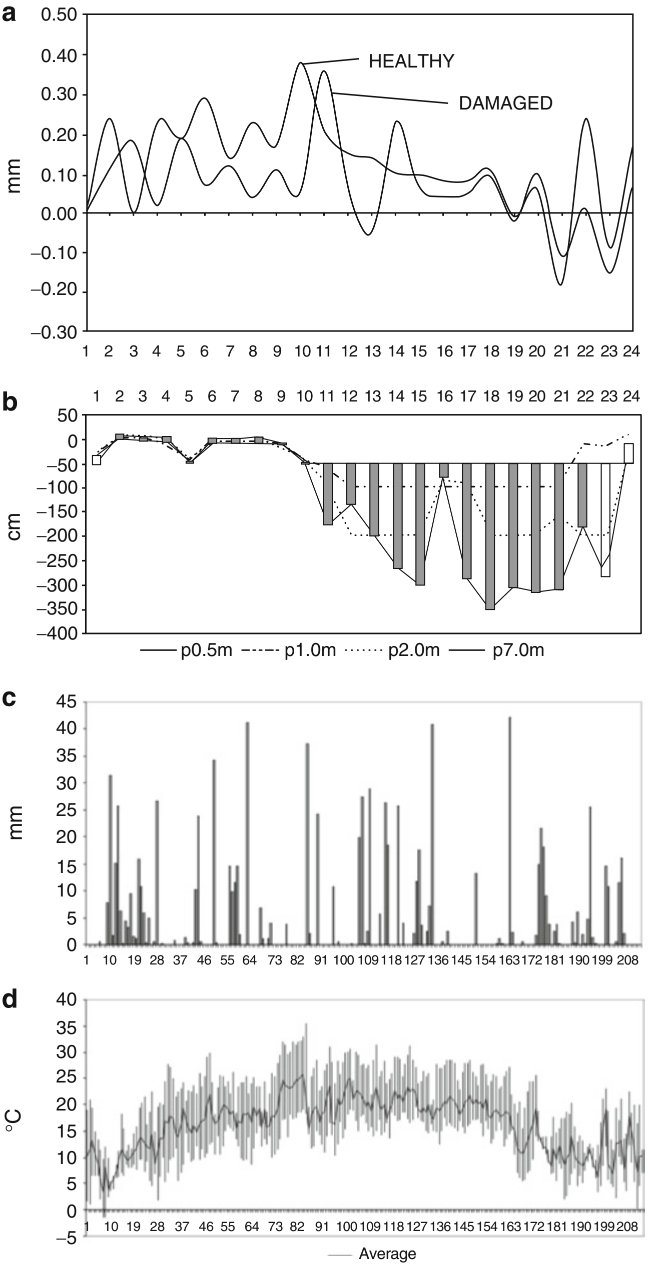


Fig. 20.3 (**a**) Differences of seasonal *Quercus robur* tree stem diameter variations of trees with different health status (healthy with <10% crown damage and damaged >50%), where (**b**) with respect to seasonal groundwater status (The phase delay between healthy and damage trees is evident in spring period of water availability), (**c**) daily precipitation and (**d**) daily temperature 2.4 Scaling Forest Effects on Flood Formation and Water Retention

ancillary stresses on forest resources as possible (Nabuurs et al. 2007). The adaptation of forestry to climate change needs to: establish objectives for future forestry as affected by climate change; determine the vulnerability of ecosystems, forest communities and societies; develop present and future cost-effective adaptation activities; manage the forests to reduce vulnerability; monitor the state of the forests and identify when critical thresholds are reached; manage to reduce the impact when it occurs, speed recovery and reduce vulnerability to further climate change (Spittlehouse and Steward 2003). The concept of adaptation includes assessment of the three main components of vulnerability: sensitivity – the degree to which systems will respond including beneficial and harmful effects; adaptive capacity – the degree to which adjustments in practices, processes or structures can moderate or offset the potential for damage or take advantage of opportunities created; exposure – the degree to which the system is exposed to climate hazards. The adaptation strategies comprise various options for forest management such as: selection of tree species/provenances, thinning regimes, stand regeneration techniques, forest protection routines, nurseries, tree breeding and genetic engineering (Lexer 2009).

Drought mitigation measures should also consider impact management, i.e. to evaluate the possibilities of the various practices of water cycle manipulation in the range from upper (stand canopy) to lower (soil) forest ecosystem boundaries in a variety of forest types and environmental conditions. Timely thinning in drought exposed forest ecosystems can provide additional water availability for the rest of the trees in the reduced forest stand due to the reduction of forest canopy interception and transpiration demands. Thinning practices can also contribute to increased resilience of forest stands by selection and removal of trees with low health conditions, based usually on indices of tree crown status (defoliation, discoloration). Although the danger of increased water scarcity in forests is unavoidable in future perspective of some European regions, the occurrence of more extreme storm events, even in areas affected by drought, is very certain. The only possible seasonal increase of precipitation, particularly in the continental part of Europe, could be expected in the winter season during the latency stage of vegetation. Therefore, drought impact management at a landscape scale should also examine the possibilities of increase of water storage in forests and temporal retention of accumulated winter precipitation further into the growing season. Potential landscape management practices (Schüler et al. 2007) should therefore examine various suitable measures for that purpose: halting the runoff inside of orographic waterways, closing drainage ditches, avoiding linear structures, water drainage from forest roads into adjacent forest areas, the promotion of a stable soil pore structure by liming acidified forest soils, water table management in lowland and riparian forests.

While the future simulations of annual precipitation changes in Europe show a clear north–south divide, the increase of extreme events is an outcome of the future projections all over Europe (European Environmental Agency 2008; Bates et al. 2008). The increase of flood occurrences is a logical consequence of a higher probability of heavy rain events. The more frequent and more severe floods in the recent years, e.g. the extreme summer flood 2002 in the Elbe River watershed, brought on a vital discussion and a high number of research activities dealing with flood assessment and prevention (research initiatives such as WaReLa, FLOODsite, CRUE ERA-Net etc.). A result is the EU-Flood Directive (EC 2007). It aims at reducing and managing the risks that floods pose to human health, the environment, cultural heritage and economic activity. First, the Member States have to identify the river basins and associated coastal areas at risk of flooding. For such zones they would then need to draw up flood risk maps and establish flood risk management plans focused on prevention, protection, and preparedness. The management plans should contain a combination of technical flood protection measures, non-structural measures, and extended protection measures to reduce the probability of flooding and its potential consequences. The non-structural measures of such basin-oriented integrated flood protection strategies consider amongst others the increase of the water retention in the landscape. Therefore, the debate on the role of afforestation and forest conversion as measures for preventive flood protection has received new impulses (e.g. Calder et al. 2007c; Laurance 2007). It is quite clear that the quantification of effects of forest conversion and afforestation on flood runoff at the catchment scale can only be achieved by reliable model calculations combining the various hydrological scales (soil – plot – catchment).

Forests produce low levels of stormflow and greater soil stability than any other vegetation type. This is due to their high infiltration rates, protective ground cover, high consumption of soil water via evapotranspiration, particularly interception, and high tensile strength of roots (FAO 2003). For that reason afforestation of grassland or arable land can be seen as a non-structural measure to increase the water retention in the landscape as announced in the EU-Flood Directive (EC 2007). The ‘forest effect’ in comparison to other land-uses on floods can be split into two general parts: retention by providing additional storage (higher water consumption and higher interception) and decelerating runoff by shifting water into slower pathways (improved infiltration and vertical percolation, cf. Wahren et al. 2007b). However, it has to be considered that beneficial flood-related effects of forests in catchments interfere with several other hydrological functions. On a larger regional scale, there are many, sometimes conflicting demands (domestic, agricultural and industrial water use, ecological needs) in both upstream and downstream areas. A major issue in this context is the conflict between higher retention (and simultaneously lower erosion) from forest land and the lower water yield which results from higher evapotranspiration losses. Especially in areas with scarce water supply due to low and unevenly distributed rainfall (e.g. in Mediterranean countries or in the semi-arid NW of China), a lack of water is often a limiting factor not only for the forest vegetation itself but also for water demands of other sectors, notably agricultural irrigation (Feger 2007). A key challenge for land-use practitioners and planers is to harmonize and/or to optimize the wide range of multisectoral forest benefits without detriment to water resources and ecosystem function (Calder 2006; Calder et al. 2007b). There has been promising efforts to strengthen research activities of ‘eco-hydrology’ as new discipline within traditional forest sciences (cf. Angelstam et al. 2005).

Forest Measures for Flood Protection

A river catchment is a complex living ecosystem, which means that it is a large, interconnected web of land, water, vegetation, structural habitats, biota and the many physical, chemical and biological processes which link these among themselves and with the embedded settlements and their infrastructure. The duality of use and protection of the landscape is irresolvable as long as people live from and on these areas.

As long linear ecosystems, rivers and streams are particularly vulnerable to fragmentation. The structural diversity and flow continuity of river courses are major points of focus in mountainous regions and respective floodplains. Likewise, of great importance is the linkage between the amphibious, aquatic habitat and the adjacent land area (SMUL 2000; Tockner 2007). It is logical that the structural diversity declines, when the river courses are tangent to or cross settlements and transport infrastructure in narrow valleys. There, the technical hydraulic precepts are generally valid, which require an unhindered discharge with minimal water levels (Sönnichsen 2005).

The following shows how the flood mitigating potential of forests can be purposefully developed. There are two complexes of measures that should be distinguished.

Measures to reduce runoff formation – Forestry practices minimize runoff formation or at least reveal a delayed peak occurrence. This complex of measures covers the entire spectrum from the forest maintenance through the forest conversion (site plasticity and functional diversity of the forests) up to afforestation.

Measures to mitigate the adverse impacts of running waters – The damaging impacts of floods and running water are mitigated punctually by individual trees at critical sites, and through forest structures (e.g. groves) along the water courses. Individual trees can stabilize margin areas and forest structures increase the site roughness, which is able to reduce the flow velocity and retain sediments as well as flotsam (Initiative Weißeritz-Regio 2007).

Landscape

A stable structured successional mosaic of trees ensures meaningful flood generation prevention. This is due to the fact that close-to-nature forests are adapted to the natural site conditions. Hence, such forests represent the best flood prevention that the nature can provide). Through its biodiversity (tree species composition – high number of species and low number of individuals, shrubs, ground vegetation and organisms in the forest floor and mineral soil layers) forest ecosystems possess a high capacity for self-organization and are therefore less destructible (wind, snow, pests/calamities). They provide as well a protection belt for water bodies.

Therefore, related forestry measures must not only to be seen under an economic point of view, but primarily in the light of site adaptation and ecological interactions which in synergism contribute significantly to flood mitigation. Due to the long development of trees, the achievements of afforestation or forest conversion can only be evaluated with a distinct lag of time. Thus, an overall monitoring plan covering measures in forestry and nature conservation is needed to observe the efficiency of such measures in space and time.

In general an increasing elevation implies a temperature decrease and a precipitation increase. So, to be sustainable, forest conversion and afforestation measures have to take into consideration the height dependency of the potential natural vegetation, along with the site specific soil and landscape features, these lead to a highly diversified pattern.

Forest Conversion

Forest conversion is the gradual conversion of monocultural even-aged (mostly coniferous) plantations into site-adopted mixed forests, which are quasi natural with respect to ecosystem stability and elasticity. Such silvicultural measures serve primarily not the flood prevention but the structure quality, the stability, and other beneficial effects of forest land-use.

Afforestation

In many agriculturally dominated landscapes a general increase of forest coverage has to be a major goal. Afforestation in catchments is aimed to have long-lasting positive effects with respect to flood and erosion control. To be more specific and economically efficient priority areas for afforestation in the catchment should be defined. The methodology should be based on an integrated model-based analysis of runoff formation taking into account relief, geology, and soil properties (e.g. Naef et al. 2000; Schüler 2006; Wahren et al. 2007a).

Forest Management

The rotational maintenance of the existing stocks should be in accordance with forestry legislation. The multifunctional character of forests can only be controlled and sustainably achieved by following a balanced management plan. Thus, flood prevention has to be integrated in such forest management plans which may contribute to preserve and even improve the water retention functions of forests within structured landscapes. Many synergies to other ecosystem functions and services and related protection aims (e.g. soil conservation, water quality protection etc.) can be met when establishing flood protection measures. Such include close-to-nature silviculture with a maximum of horizontal and vertical structure and provision of a continuous vegetation cover; afforestation of extended agricultural areas; promoting a stable mixed forest; timber harvesting and logging technologies depending on soil moisture regime and structure; forest road density as low as necessary; water drainage from forest roads into adjacent forest areas; avoiding linear structures and closing of drainage ditches; implementation and preservation of close-to-nature retention basins in forest parts, which are sensitive to generation of quick runoff as well as renaturation of forest creeks, flood plains and wetlands (Schüler et al. 2007).

River Course

The river course contains the riverbed and the riparian zone. The maintenance lies often in governmental responsibility and has to assure amongst others the unhindered discharge of the water. The Water Framework Directive (EC 2000) requires a nearly continuous permeability of the river.

During a major flood event the strain of the river banks and the riparian vegetation can be extremely high. The above-ground vegetation removes energy from the water and retards the flood wave. Thereby, river bank erosion declines. Additionally, the riparian soil is stabilized by the vegetation roots. The conservation measures should, therefore, assure a high quality. As a consequence, only site-adopted riparian vegetation which resists the high water levels next to the river should be used for new plantations along river courses. Furthermore, a site- adapted forest in inundation zones contributes to a retardation of running water and efficient retention of sediments and water.

Limitation and Scales

However, the protection that forest cover provides is naturally limited (FAO 2003). This limitation depends on the nature of the catchment as well as on the intensity and duration of the rainstorm event. Thus, a maximal retention potential in every basin can be defined. To estimate such potentials modeling techniques, that quantify land-use effects on flood formation and retention, have to be refined (Wahren et al. 2007a, 2008).

A lack of comprehensive data and the limitations of the rainfall-runoff-model systems lead to considerable uncertainties in the quantification of land-use effects on floods. In many cases modeling approaches neglect important aspects when these models are parameterized. Hence, most models just consider vegetation parameters (root depth, leaf area index (LAI), canopy height, etc.). Some more advanced spatially distributed models (e.g. AKWA-M® : Münch 2004; Münch et al. 2007, WASIM-ETH: Schulla and Jasper 2001) also include the pre-event soil water storage by calculating land-use specific evapotranspiration and related soil water dynamics. In addition, one should also be aware that changes in the vegetation cover (e.g. conversion from arable land into grassland or forest) in the mid- to long-term will also result in distinct changes in soil hydraulic properties (infiltration, percolation, retention). While the descriptions of the vertical water fluxes at the point often use approaches with a (measurable) physical background the lateral fluxes are calculated with more conceptual algorithms particularly in models for larger scales. A reasonable prognosis of the change of lateral pathways especially on a larger scale is not feasible although a distinct modification by newly established root systems or changed soil hydraulic properties realistically can be expected (Wahren et al. 2009). It can be stated that the land-use impact on runoff generation at the plot-scale can be modeled much better than the impact on runoff concentration in the landscape (Wahren et al. 2008).

It is commonly believed that forests are necessary to regulate stream flow and reduce runoff, and to some extent this is true (FAO 2005). However, the value of the flood reducing effect of forests compared to other land-uses decreases with the increase of the return period of the flood, originating rainstorm event and with increasing catchment size (Bronstert et al. 2003). The above-mentioned measures can be very effective in small or mesoscale catchments where short convective rainstorms with large intensities cause floods. However, in large European river basins, floods are caused by huge adjective rainfall events, which have a smaller intensity, but affect a very large area and have a high cumulative precipitation sum. Thus, the positive ‘forest-effect’ disappears under such circumstances becoming the geomorphologic factors predominant.

Finally, it has to be considered that floods are integral parts of the natural regime of a river. The ‘good ecological’ status of rivers as pronounced in the EU WFD (EC 2000) also needs the occasional occurrence of flooding events in order to assure intact well-structured aquatic ecosystems. To minimize flood damages the inhabitants should be aware of the occurrence of such events and take the flood risk into consideration when settling in such areas. This means that there should be allowed sufficient space for rivers, notably in close-to-nature forested floodplains.

With respect to flood protection nature-orientated forestry includes timber harvesting techniques following soil conservation standards, adapted forest road building, high stand stability and biodiversity, revitalization of wetlands and afforestation of suitable (‘efficient’) sites. Altogether this might contribute to reestablish the original water retention potential in anthropogenically disturbed river basins and decreasing the human-made artificial portion of floods. Model tools to quantify the impact of such measures on floods should consider land cover changes (vegetation) just as changes in soil hydraulic properties and have to be further investigated.

Water Quality

Forests and woodlands are known to produce groundwater and surface water of high quality. The complex interactions between rainwater, atmospheric conditions, and plant–soil systems in forests have strong controls on water quality.

All processes controlling water quality are closely linked to water fluxes and their pathways in the catchment. As a consequence, the modification of water fluxes (e.g. by a changed climate and/or vegetation cover) will also be reflected in a changed water chemistry. All land-use activities will be reflected in specific water chemistry. On the other land-management and notably forestry measure can be targeted to optimize water quality. Thus, a multi-functional and sustainable forest management should adequately consider the protection, maintenance and – where necessary – the restoration of water resources. However, conservation of water quality represents a highly integrative effort encompassing various temporal and spatial scales of forests as components of landscapes.

In many regions of Europe water resources in forests have increasingly become necessary to guarantee drinking water standards. In many cases, public drinking water supply relies on water from forests. Most of the world‘s 105 biggest cities and metropolitan areas rely fully or partly on drinking water from forested catchments (cf. Dudley and Stolton 2003). In general there are two strategies to provide urban areas with high quality water from forests: (1) long-distance transport of water from remote forest areas (e.g. New York, Vienna); (2) conservation and extension of forests in co urban areas (e.g. Mannheim, Hannover, Germany). According to a joint World Bank/WWF study (Dudley and Stolton 2003) it might economically be better and more sustainable to invest money in the establishment and in the target-oriented management of water protection forests rather than in water treatment plants. In many regions water from forests is increasingly needed in order to dilute water originating from other land-use systems (agriculture, urban-industrialised areas) affected by specific symptoms of contamination. In addition, forest land-use in many ways facilitates aquatic wildlife including recreational and commercial fisheries.

With respect to water quality the EU Water Framework Directive (EU-WFD) (EU 2000) represents an integrated legal instrument fostering the protection and sustainable use of water resources on the scale of river catchments and/or large hydrogeological aquifer units. With the establishment of integrated river basin management plans a new water pricing policy will be introduced. This may lead to the formation of ‘strategic alliances’ between forestry and water resource management which is based on the principle of recovery of costs of water services including environmental and resource costs. In the future this may therefore offer some opportunities for forest owners to increase their income by managing their forests with respect to an optimal water quality rather than to focus exclusively on timber sale. Such a water-related forest management (*Water management through forest management*) is a challenge not only for practical forestry but also for forest sciences to provide the natural data for the necessary socio-economical evaluation processes.

In modern perceptions sustainable forestry also integrates the protection of water and soil resources. A crucial role in that context plays the historical use of forests by the society. In many regions of Europe the excessive removal of plant biomass had led to a severe degradation of forest soils which from their natural conditions had been poor before (acidification, decrease in pools of soil organic matter) (e.g. Glatzel 1991; Feger 1993). Accordingly the filter, buffer, and transformation functions of soils have considerably been decreased.

Since the beginning of industrialization the emission of acidifying compounds has accelerated soil acidification with distinct effects on water quality (loss of ANC, increased concentrations of Al and heavy metals). However, soils have mediated negative effects on the hydrosphere as they retained considerable amounts of the atmospheric input (notably S, N, and heavy metals). According to the external loads (e.g. acid deposition) soils may have accumulated considerable amounts of elements which be released under modified environmental and management conditions. Armbruster et al. (2003) gave an example for such a memory effect (S accumulation) for forested catchments in the Eastern Ore Mountains (Saxony, Eastern Germany) which received extremely high S deposition during the period 1960 until 1990. Hence, forest management must consider the element pools in the soil and their potential mobilization. Since atmospheric N inputs into forests are still on a high level the potential ‘N saturation’ of forest ecosystems may also negatively affect water quality. Potential risks are elevated concentrations of nitrate resulting from increased nitrification even at low pH levels, but also a stimulation of humus mineralization (Feger 1997/1998). Notably in sandy soils, the buffer and filter functions of soils are mainly linked to the soil humus. With respect to an increasing N availability soil liming (which has become a common practice to counteract soil acidification) might become critical since liming stimulates microbial transformations in the topsoil. An alternative to liming could be the application of base silicate rock materials (Kreutzer 1995, Feger 1997/1998).

The conversion of monospecific stands of even-aged conifers into unevenly-structured mixed stands is expected to create long-term positive effects also on water quality. However, in the transition phase strong thinning in order to favor planted deciduous trees in the understorey may bear some risks with respect to excess mineralization (e.g. Huber et al. 2004). Therefore, smooth silvicultural techniques should be applied. On the other hand, in most cases the growth of abundant ground vegetation has appeared to compensate for excess leaching of nutrients.

The most severe threaten of water quality from forestland may arise from major disturbances in the soil-vegetation system (Swank 1988). In most cases such a disturbance results from a sudden breakdown or removal of the forest vegetation (clear-cut: Likens et al. 1970; Mellert et al. 1996; wind-cast, insect infestations: Huber 2005). Therefore, a crucial role to ensure mostly tight nutrient cycles in forested watersheds is to assure permanent stand vegetation with a high degree of stability. Thinning and timber harvest should be achieved avoiding strong disturbances by regular low-intensity thinning and small-scale harvesting notably in a femel-type of timber removal.

According to the differing requirements of principal uses of water from forests (biomass production, utilization for drinking purposes, aquatic wildlife and recreation) obliging guidelines for a ‘proper’ forest management with respect to water resources have to be worked out. Even though forest management is conducted at the stand scale water-related questions in forests inherently need to be addressed on the catchment scale. Very often there is a gap in scales (Fig. 20.4). In this context, it can be helpful to think of water moving through three contiguous zones within a catchment: the adjacent land, the riparian zone and the aquatic zone. The riparian

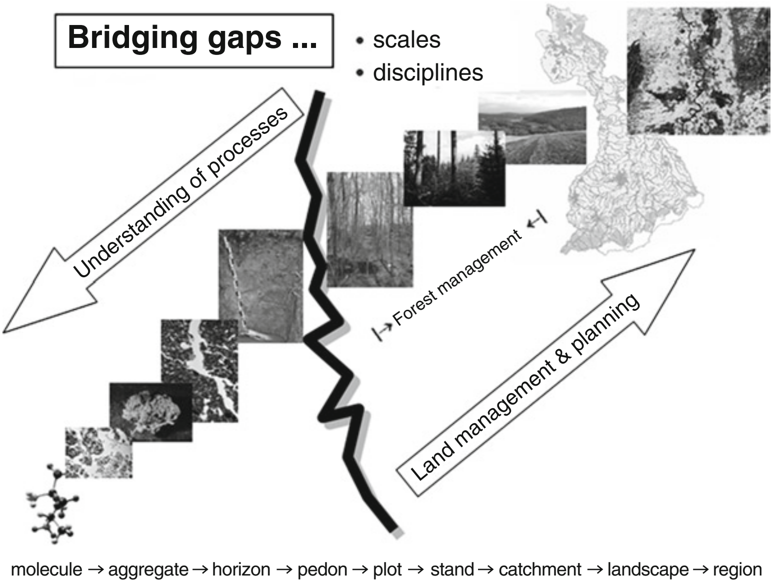


Fig. 20.4 The *scaling* diagram; linkage of various spatial dimensions

zone with it's highly hydromorphic soils is a crucial interface controlling element fluxes from the terrestrial to the aquatic systems (e.g. Armbruster et al. 2000).

Nevertheless, the catchment approach has to be combined with the traditional stand-based plot approach in order to obtain a reinforced understanding of underlying processes in water and nutrient cycling. Beside the study of interactions of water on its way through various types of forested catchments an experimental approach is essential. Short-term forestry operations (harvesting, thinning, road construction, fertilisation/liming) have to be studied in model catchments as well as effects of changed silvicultural concepts (e.g. conversion of conifer plantations into age-structured mixed stands) on various time scales. Long-term silvicultural manipulation experiments on the catchment scale appear to be essential in order to generate controlled side-conditions for a successful interdisciplinary research.

Modelling of water and element fluxes has to be developed further as a valuable tool to link processes on various temporal and spatial scales in catchments. Up-to-now most models are restricted to the plot scale and need a considerable amount of 'point information'. The application of such models on the landscape scale using GIS and regionalisation techniques will be a major research challenge.

The following research topics should be integrated in a landscape based research strategy (cf. Andersson et al. 2000, 2005):

- To define the relationships between natural site conditions, forest management and keeping water quality standards (e.g. for drinking water)
- To improve understanding of element cycling (notably on 'gaps' such as gaseous losses, lateral flows)

- Preferential flow and seepage water quality
- To predict potential effects on sink-source conditions at forest sites (climate change and forest management: thinning, forest transformation, afforestation)
- To detect ecosystem indicators for water quality
- To establish concepts of nutrition management which integrate forest growth and vitality with soil functioning and water quality
- To quantify the contribution of forest management to water quality including aquatic biodiversity as a basis for the implementation of EU-WFD
- To estimate the effects of a changing climate on nutrient cycling and associated drainage losses

In conclusion, a multi-functional and sustainable forest management should adequately consider the protection, maintenance and – where necessary – the restoration of soil and water resources. However, conservation of water quality represents a highly integrative effort encompassing various temporal and spatial scales of forests as components of landscapes. The objective to optimize water quality on a landscape scale necessitates a superordinated planning approach which harmonizes other forest functions in the landscape.

Conclusions

- Forests in good condition on appropriate sites present land use category most suitable for retaining water and reducing discharge. The function of forest to reduce runoff is decreasing with top soil compaction, the current saturation status and the character of precipitation. Forest gradually loses its capacity to retain high runoff when critical amount of precipitation is reached in cases of short and intense precipitation events in small areas; long lasting intense precipitation in large areas and intense precipitation in large areas, when soil water saturation was already reached (mainly in spring and autumn).

- Forest management activities which can influence the water status (quality and quantity) of forests are: Frequency, intensity and technique of harvesting; tree species composition; crown density, cover percentage; distribution of growth classes; vertical and horizontal stand structure; forest regeneration, ground vegetation; coarse woody debris. The efficiency of forest management on forest hydrology can be determined through five different indicators: Concentration of pollutants in the water; nitrogen concentration in the water; sediment loss/erosion; runoff; water temperature. The best forest managerial practice as close to nature forestry is most suited in stable environmental conditions for optimizing forest hydrological functions.

- By gradual shifting of climate parameters across the European gradient a new demands from forests emerges. Predicted scenarios of global warming required determination of two basic limits of forests: what are the limits of forests to reduce runoff and flooding under conditions of more extreme precipitation events and what are the thresholds for forests and related biodiversity to survive more intensive droughts. The current accelerating changes of hydrologic cycle across EU gradient and related uncertainties present new challenge for policy makers and scientists to determine most efficient forest adaptation measures in cases of droughts and landscape orientated measures to reduce flooding. These questions put additional pressure on scaling towards explaining and transposing various tree orientated physiologic processes on landscape level to secure timely feedback by forest management to reduce ancillary stress.

- A multi-functional and sustainable forest management should adequately consider the protection, maintenance and – where necessary – the restoration of soil and water resources. However, conservation of water quality represents a highly integrative effort encompassing various temporal and spatial scales of forests as components of landscapes. The objective to optimize water quality on a landscape scale necessitates a superordinated planning approach which harmonizes other forest functions in the landscape.

References

Andersson FO, Feger KH, Hüttl RF, Kräuchi N, Mattsson L, Sallnäs O, Sjöberg K (2000) Forest ecosystem research – priorities for Europe. For Ecol Manage 132:111–119

Andersson, FO, Angelstam P, Feger KH, Hasenauer H, Kräuchi N, Mårell A, Matteucci G, Schneider U, Tabbush P (eds) (2005) A research strategy for sustainable forest management in Europe. – COST Action E25 ENFORS Technical Report 5, GIP ECOFOR, Paris, 149 p. ISBN 2-914770-08-1

Angelstam P, Feger KH, Hasenauer H, Schneider U, Mårell A (2005) Chapter 4: A holistic research strategy. In: Andersson F, Angelstam P, Feger KH, Hasenauer H. Kräuchi N, Mårell A, Matteucci G, Schneider U, Tabbush P (eds) A research strategy for sustainable forest management in Europe. – COST Action E25 ENFORS Technical Report 5, GIP ECOFOR, Paris: 39–62 ISBN 2-914770-08-1

Archaux F, Wolters V (2006) Impact of summer drought on forest biodiversity: what do we know? Ann For Sci 63:645–652

Armbruster M, Abiy M, Feger KH (2003) The biogeochemistry of two forested catchments in the Black Forest and the eastern Ore Mountains (Germany) – effects of changing atmospheric inputs on soil solution and streamwater chemistry. Biogeochemistry 65:341–368

Armbruster M, Köhler H, Feger KH (2000) Chemische Zusammensetzung zweier quellnaher Waldbäche im Hochschwarzwald – Abflussabhängige Variabilität und Einfluss einer Bodenkalkung. Forstw Cbl 119:249–262

Arnšek A (2001) Aktualni trendi razvoja traktorskih olj. Trendi v razvoju kmetijske tehnike: zbornik simpozija. Društvo kmetijske tehnike, Radenci, pp 167–174

Barlow M, Clarke T (2002) Blue gold: the fight to stop the corporate theft of the World's water. New York Press, New York, p 278

Bates BC, Kundzewicz ZW, Wu S Palutikof JP, (eds) (2008) Climate change and water; Technical paper of the intergovernmental panel on climate change; IPCC Secretariat; Geneva; 210 pp; ISBN: 978-92-9169-123-4

Binkley D, Macdonald LH (1994) Forests as non-point sources of pollution, and effectiveness of Best Management Practices. National Council for Air and Stream Improvement, New York, 57 pp

Boczon A (ed) (2006) Assesing of soil and water conditions in forests. Forest Research Institute, Center for Exellence PROFOREST for Protection of Forest Resources in Central Europe, Warsaw, pp 141–145

Bončina A, Diaci J (1998) Contemporary research on regeneration patterns of Central European virgin forests with recomendation for future research. Zbornik Gozdarstva Lesarstva 56:33–53

Breda N, Huc R, Granier A, Dreyer E (2006) Temperate forest trees and stands under severe drought: a review of ecophysiological responses, adaptation processes and long-term consequences. Ann For Sci 63:625–644

Bronstert A, Niehoff D, Fritsch U (2003) Auswirkungen von Landnutzungsänderungen auf die Hochwasserentstehung. Petermanns Geogr Mitt 6:24–33

Calder I, Hofer T, Vermont S, Warren P (2007a) Towards a new understanding of forests and water. Unasylva 58(4):3–11

Calder IR (2006) Forest and floods: moving to an evidence-based approach to watershed and integrated flood management. Water Int 31:87–99

Calder IR, Smyle J, Aylward B (2007a) Debate over flood-proofing effects of planting forests. Nature 450:945 (published online 12 Dec 2007) Accessed 7 June 2008

Calder IR, Hofer T, Vermont S, Warren P (2007b) Towards a new understanding of forests and water – In: Perlis A (ed) Forests and water. Unasylva 229; Vol. 58, 2007/4; Food and Agriculture Organization of the United Nations; Rome; Web-document. http://www.fao.org/docrep/010/a1598e/a1598e02.htm Accessed 17 Oct 2008

Chang M (2003) Forest hydrology: an introduction to water and forests. CRC Press LLC, Boca Raton, FL, 392 pp

Croke J, Hairsine P, Fogarty P (2001) Soil recovery from track construction and harvesting changes in surface infiltration, erosion and delivery rates with time. For Ecol Manag 143:3–12

Cruiziat P, Cochard H, Ameglio T (2002) Hydraulic architecture of trees: main concepts and results. Ann For Sci 59:723–752

Dankers R, Christensen OB, Feyen L, Kalas M, de Roo A (2007) Evaluation of very high-resolution climate model data for simulating flood hazards in the Upper Danube Basin. J Hydrol 347(3–4):319–331

Desprez-Loustau ML, Marçais B, Nageleisen LM, Piou D, Vannini A (2006) Interactive effects of drought and pathogens in forest trees. Ann For Sci 63:597–612

DMP Report (2007) Drought management plan report: Including agricultural, drought indicators and climate change aspects. Water scarcity and droughts expert network. European Commission. http://ec.europa.eu/environment/water/quantity/pdf/dmp_report.pdf. Accessed 21 Dec 2008

Dudley N, Stolton S (2003) Running pure: the importance of forest protected areas to drinking water. A research report for the World Bank / WWF Alliance for Forest Conservation and Sustainable Use

EC (2000) Directive 2000/60/EC of the European Parliament and of the Council; published in the Official Journal of the European Union; L 288/27; 22 Dec 2000

EC (2007) Directive 2007/60/EC of the European Parliament and of the Council; published in the Official Journal of the European Union; OJ L 327; 6 Nov 2007

European Environmental Agency (2008a) About climate change; European Environment Agency; Web-document. http://www.eea.europa.eu/themes/climate/about-climate-change

EU (2000) DIRECTIVE 2000/60/EC OF THE EUROPEAN PARLIAMENT AND OF THE COUNCIL of 23 October 2000 establishing a framework for Community action in the field of water policy. L 327/1 Official Journal of the European Communities

European Environmental Agency (2008b) European forests – ecosystem conditions and sustainable use. http://www.eea.europa.eu/publications/eea_report_2008_3. Accessed 16 Jan 2009

European Environmental Agency (2008c) European forests – ecosystem conditions and sustainable use. http://www.eea.europa.eu/publications/eea_report_2008_3

FAO (2003) Sustainable use and management of freshwater resources: the role of forests – In: Perlis A, Wearne L, Moore B (Eds.) State of World's Forests (2003) Published by Center for International Forestry Research Food and Agriculture Organization of the United Nations; Rome; http://www.fao.org/DOCREP/005/Y7581E/y7581e09.htmP0-27

FAO (2005): Forests and floods - Drowning in fiction or thriving on facts?; RAP Publication 2005/03 Forest Perspectives 2; Published by Center for International Forestry Research Food and Agriculture Organization of the United Nations; 40 pp.; ISBN 979-3361-64-6; http://www.fao.org/docrep/008/ae929e/ae929e00.htmContents Accessed 12 July 2008

Feger KH (1993) Influence of soil development and management practices on freshwater acidification in Central European forest ecosystems. - In: C.E.W. Steinberg and R.W. Wright (Eds.): Acidification of Freshwater Ecosystems: Implications for the Future. John Wiley & Sons Ltd.:67–82

Feger KH (1997/98) Boden- und Wasserschutz in mitteleuropäischen Wäldern - I. Rahmenbedingungen. - Bodenschutz 2: 18-23; II. Gefährdungspotentiale und Bewertung. - Bodenschutz 2: 134-138; III. Waldbauliche Möglichkeiten und Maßnahmen des technischen Bodenschutzes. - Bodenschutz 3: 103–108

Feger KH (2007) Forests and water cycle – thematic introduction and symposium overview. – In: Feger K.H., Wang Yanhui, C. Bernhofer, Jörg Seegert (Eds.) (2007): Progress in Hydro Science and Engineering; Dresden Water Center Volume 3/2007:13–20; Dresden; ISBN: 978-3-86780-074-7

Frehner M, Wasser B, Schwitter R (2005) Nachhaltigkeit und Erfolgskontrolle im Schutzwald. Wegleitung für Pflegemassnahmen in Wäldern mit Schutzfunktion. Bundesamt für Umwelt, Wald und Landschaft (BUWAL), Bern, Switzerland p.20: 879–881

Glatzel G (1991) The impact of historic land use and modern forestry on nutrient relations of Central European forest ecosystems. Fertilizer Research 27: 1–8

Gobron N, Pinty B, Aussedat O, Chen J, Cohen WB, Fensholt R, Gond V, Hummerich KF, Lavergne T, Melin F, Privette JL, Sandholt I, Taberner M, Turner DP, Verstraete MM, Widlowski J-L (2006) Evaluation FAPAR products for different canopy radiation transfer regimes: Methodology and results using JRC products derived from SeaWiFS against ground-based estimations. J Geophysical Res 111:D13110. doi:10.1029/2005JD006511

Granier A, Reichstein M, Breda N, Janssens IA, Falge E, Ciais P, Grünwald T, Aubinet M, Berbigier P, Bernhofer C, Buchmann N, Facini O, Grassi G, Heinesch B, Ilvesniemi H, Keronen P, Knohl A, Köstner B, Langergen F, Lindroth A, Longdoz B, Loustau D, Mateus J, Montagnani L, Nys C, Moors E, Papale D, Peiffer M, Pilegaard K, Pita G, Pumpanen J, Rambal S, Rebmann C, Rodrigues A, Seufert G, Tenhunen J, Vesala T, Wang Q (2007) Evidence for soil water control on carbon and water dynamics in European forests during the extremely dry year: 2003. Agric Meteorol 143:123–145

Huber C., Weis W., Baumgarten M., Göttlein A. (2004) Spatial and temporal variation of seepage water chemistry after femel and small scale clear-cutting in a N-saturated Norway spruce stand. Plant and Soil, 267: 23-40

Huber C (2005) Long lasting nitrate leaching after bark beetle attack in the highlands of the Bavarian Forest National Park. J. Environ. Qual. 34: 1772–1779

Infante JM, Domingo F, Fernandez Ales R, Joffre R, Rambal S (2003) Quercus ilex transpiration as affected by prolonged drought period. Biol Plant 46(1):49–55

Initiative Weißeritz-Regio (2007) Waldwirkung und Hochwasser; Ein Leitfaden für Landnutzer und Entscheidungsträger (in German); Arbeitsgruppe Wald und Hochwasser in der Initiative Weißeritz-Regio;Bärenfels; 29pp; http://www.tudresden.de/ioer/statisch/weisseritz/html/frame.htm. Accessed 5 September 2010

Institute for World Forestry (2008) The condition of forests in Europe. International Co-operative Programme on Assessment and Monitoring of Air Pollution Effects on Forests. http://www.icp-forests.org/pdf/ER2008.pdf . Accessed 29. November 2008

Klap JM, Oude Voshaar JH, de Vries W, Erisman JW (1999) Effects of environmental stress on forest crown condition in Europe. Part IV: statistical analysis of relationships. Water Air Soil Pollut 119:387–420, 2000

Kreutzer K (1995): Effects of forest liming on soil processes. Plant and Soil 168/69: 447–470

Košir P (2006) Gozdnogospodarski vidiki ohranjanja voda visokega krasa na primeru GE Draga. Magistrsko delo. Univerza v Ljubljani, Biotehniška fakulteta, Oddelek za gozdarstvo. Univrza v Ljubljani, Ljubljana

Landsberg JJ, Gower ST (1997) Applications of physiological ecology to forest management. Academic, San Diego, CA

Laurance WF (2007) Environmental science: Forests and floods, Nature 449: 409–410

Lee R (1980) Forest hydrology. Columbia university press, New York, p 349

Lexer MJ (2009) Expected Climate Change & Options for European Silviculture: WG2 Adaptation. COST Action FP703 – Echoes, 1st scientific conference, 22–25 January, Florence, Italy

Likens GE, Bormann FH, Johnson NM, Fisher DW, Pierce RS (1970) Effects of Forest Cutting and Herbicide Treatment on Nutrient Budgets in the Hubbard Brook Watershed-Ecosystem. Ecol. Monographs 40: 23–47

Lloyd-Hughes B, Saunders MA (2002) A drought climatology for Europe. Int J Climatol 22:1571–1592

Luterbacher J, Dietrich D, Xoplaki E, Grosjean M, Wanner H (2004) European seasonal and annual temperature variability, trends and extremes since 1500. Science 303(5663):1499–1503

Mannerkoski H (2005) Management of European forests under changing climatic conditions. Research Notes 163. University of Joensuu, Faculty of Forestry. http://www.efi.int/portal/project/silvistrat/ Accessed 19 Jan 2009

Mellert K-H, Kölling C, and Rehfuess KE (1996) Bioelement leaching from Norway spruce ecosystems in Bavaria after windthrow. Forstw. Cbl. 115:363–377

Mulkey LA (ed) (1980) An Approach to Water Resources Evaluation on Non-Point Silviculture Sources (A Procedural Handbook). Forest Service. United States Department of Agriculture, Washington DC

Münch A, Dittrich I, Wahren A (2007) The effects of changes in tree species composition and of afforestation on water budget and flood dynamics calculated wit AKWA-M®, Ore mountains. In: Feger KH, Wang Yanhui C Bernhofer, Jörg Seegert (eds) Progress in hydro science and engineering; Dresden Water Center Volume 3/2007:331–337; Dresden; ISBN: 978-3-86780-074-7

Münch A (2004) AKWA-M® – Teilflächen basiertes Wasserhaushalts- und Hochwassermodell (in German); Dr. Dittrich & Partner Hydro-Consult GmbH; Bannewitz

Nabuurs GJ, Masera O, Andrasko K, Benitez-Ponce P, Boer R, Dutschke M, Elsiddig E, Ford-Robertson J, Frumhoff P, Karjalainen T, Krankina O, Kurz WA, Matsumoto M, Oyhantcabal W, Ravindranath NH, Sanz Sanchez MJ, Zhang X (2007) Forestry. In: Metz B, Davidson OR, Bosch PR, Dave R, Meyer LA (eds) Climate change 2007: mitigation. Contribution of working group III to the fourth assessment report of the intergovernmental panel on climate change. http://www.ipcc.ch/ipccreports/assessments-reports.htm. Accessed 9 Sept 2008

Naef F, Scherrer S, Thoma C, Weiler M, Fackel P (2000) Die Beurteilung von Einzugsgebieten und ihrer Teilflächen nach der Abflussbereitschaft unter Berücksichtigung der landwirtschaftlichen Nutzung – aufgezeigt an drei Einzugsgebieten in Rheinland-Pfalz (in German); Landesamt für Wasserwirtschaft Rheinland-Pfalz, Mainz/Germany; Bericht Nr. B003. 186 pp

Nisbet TR (2001) The role of forest management in controlling diffuse pollution in UK forestry. For Ecol and Manag 143:215–226

Pichler P, Oberhuber W (2007) Radial growth response of coniferous forest trees in an inner Alpine environment to heat-wave in 2003. For Ecol and Manag 242:688–699

Pilaš I, Lukić N, Vrbek B, Dubravac T, Roth V (2007) The effect of groundwater decrease on short and longterm variations of radial growth and dieback of mature pedunculate oak (Quercus robur L.) stand. Ekologia (Bratislava) 26(2):122–131

Potočić N, Ćosić T, Pilaš I (2005) The influence of climate and soil properties on calcium nutrition and vitality of silver fir (Abies alba Mill.). Environ Pollut 137:569–602

Prybolotna N (2006) Content of the sediment in runoff in the small Beskid's watershed. In: Boczoń A (ed) Assessing of soil and water conditions in forests. Forest Research Institute, Warsaw

Rebetez M, Mayer H, Dupont O, Schindler D, Gartner K, Kropp JP, Menzel A (2006) Heat and drought 2003 in Europe: a climate synthesis. Ann For Sci 63:569–577

Schüler G, Gellweiler I, Seeling S (Eds.) (2007) Dezentraler Wasserrückhalt in der Landschaft durch vorbeugende Maßnahmen der Waldwirtschaft, der Landwirtschaft und im Siedlungswesen; Das INTERREG IIIB NWE Projekt WaReLa – Ergebnisse aus vier Jahren Umsetzung und Forschung für einen nachhaltigen Hochwasserschutz in der Fläche (in German); Mitteilungen aus der Forschungsanstalt für Waldökologie und Forstwirtschaft Rheinland-Pfalz; Nr. 64/07

Schüler G (2006) Identification of flood-generating forest areas and forestry measures for water retention. Forest Snow Landscape Res 80:99–114

Schulla J, Jasper K (2001) Model DescriptionWaSiM-ETH; Internal report; IAC; ETH Zürich; 166 pp

Serengil Y, Gökbulak F, Özhan S, Hızal A, Şengönül K (2007) Alteration of stream nutrient discharge with increased sedimentation due to thinning of a deciduous forest in Istanbul. For Ecol and Manag 246:264–272

Sevanto S (2003) Tree stem diameter change measurements and sap flow in Scots Pine. Academic dissertation. Report Series in physics, University of Helsinki. http://ethesis.helsinki.fi/julkaisut/mat/fysik/vk/sevanto/treestem.pdf. Accessed 12 March 2006

SMUL (2000) Ökologische Studie – Beiträge zur Entwicklung eines ökologischen Leitbildes für die Flusslandschaften am Beispiel der Jahne, einem Nebenfluss der Elbe in Sachsen; Publ.: Saxon State Ministry of the Environment and Agriculture; Dresden

Sönnichsen D (2005) Q oder W – worauf konzentriert sich Hochwasservorsorge? (in German); WasserWirtschaft; Zeitschrift für Wasser und Umwelt; Verlag Vieweg & Sohn Braunschweig/Wiesbaden; No. 6:20–26

Spittlehouse DL, Steward RB (2003) Adaptation to climate change in forest management. BC J Ecosyst Manag 4(1):1–11

Swank WT (1988) Stream chemistry responses to disturbance. in W.T. Swank and D.A. Crossley (eds) Forest Hydrology and Ecology at Coweeta. Ecological Series 66, Springer Verlag. New York, 339–358

Thomas FM (2008) Recent advances in cause-effect research on oak decline in Europe. CAB Rev: Persp Agric Vet Sci Nutr Nat Resour 37(3):1–12

Tockner K (2007) River restoration: linking science with application. Ecol Civil Eng 10(1):15–25

Trenberth KE, Jones PD, Ambenje P, Bojariu R, Easterling D, Klein Tank A, Parker D, Rahimzadeh F, Renwick JA, Rusticucci M, Soden B, Zhai P (2007) Observations: surface and atmospheric climate change. In: Solomon S, Qin D, Manning M, Chen Z, Marquis M, Averyt KB, Tignor M, Miller HL (eds) Climate change 2007: the physical science basis. Contribution of working group I to the fourth assessment report of the intergovernmental panel on climate change. http://www.ipcc.ch/ipccreports/ar4-wg1.htm. Accessed 15 Nov 2008

Trontelj M (2006) Vzdrževanje in obratovanje lovilcev olj in mašńob. V Xlll.R. Slovenska vojska, SOP št. 1407, 6 pp

Twery MJ, Hornbeck JW (2001) Incorporating water goals into forest management decisions at a local level. For Ecol Manag 143:87–93

Van der Werf GW, Saas-Klaassen GW, Mohren GMJ (2006) The impact of the 2003 summer drought on the intra-annual growth pattern of beech (Fagus sylvatica L.) and oak (Quercus robur L.) on dry site in the Netherlands. Dendrochronologia 25:103–112

Veny ES (1986) Forest harvesting and water: the lake states experience. Water Resour Bull 22:1039–1047

Verstraete MM, Gobron N, Aussedat O, Robustelli M, Pinty B, Windlowski JL, Taberner M (2008) An automatic procedure to identify key vegetation phenology events using the JRC-FAPAR products. Adv Space Res 41:1773–1783

Vilhar U (2006) Vodna bilanca dinarskega jelovo-bukovega gozda na Końevskem rogu. WATER BALANCE OF A DINARIC SILVER FIR - BEECH FOREST IN KOńEVSKI ROG. Doktorska disertacija. Ph.D.Thesis. Biotehniška fakulteta, Odd. za gozdarstvo in obnovljive gozdne vire. Univerza v Ljubljani, Ljubljana

Vilhar U (ed) (2009) Vpliv gospodarjenja na vodno bilanco jelovo-bukovih gozdov Dinarskega krasa / Influence of management on water balance of the silver fir-beech forests in the dinaric karst. Studia forestalia Slovenica 133. Gozdarski inštitut Slovenije, Ljubljana, 122pp

Vilhar U, Fajon Š (2007) Vpliv gozda in gozdnogojitvenih ukrepov na hidrološki režim vodozbirnega območja. In: Kovač M (ed) Gozd in voda: rezultati projekta [Interreg III A]. Gozdarski inštitut Slovenije, Zavod za gozdove Slovenije, Ljubljana, pp 16–21

Vogt JV, Somma F (2000) Drought and drought mitigation in Europe, Advances in Natural and Technological Hazards Research. Kluwer, Dordrecht, The Netherlands

Von Burger H (1954a) Einfluss des Waldes auf den Stand der Gewaesser. II. Mitteilung. Der Wasserhaushalt im Sperbel- und Rappengraben von 1915/16 bis 1926/27. Mitteilungen der Schweizerischen Anstalt fuer forstliche Versuchswesen XVIII:311–416

Von Burger H (1954b) Einfluss des Waldes auf den Stand der Gewaesser. V. Mitteilung. Der Wasserhaushalt im Sperbel- und Rappengraben von 1942/1943 bis 1951/52. Mitteilungen Schweizerischen Anstalt Forstliche Versuchswesen XXXI:9–58

Wahren A, Feger KH, Schwärzel K, Münch A (2009) Land-use effects on flood generation – considering soil hydraulic measurements in modeling. Adv Geosci 7:1–9

Wahren A, Münch A, Dittrich I (2007a) Identification and model based assessment of the potential water retention caused by land use changes. Adv Geosci 11:49–56, http://www.adv-geosci.net/11/index.html1

Wahren A, Schwärzel K, Feger KH (2007b) Flood formation based on contrasting socio-economically founded land-use scenarios. In: Proceedings of the scientific conference on "Integrated catchment management for hazard mitigation", Trier, pp 60–68. http://ubt.opus.hbz-nrw.de/volltexte/2007/438/

Wahren A, Schwärzel K, Feger KH (2008) Uncertainties in the parameterisation of rainfall-runoff-models to quantify land-use effects in flood risk assessment. In: Samuels P, Huntington S, Allsop W, Harrop J (eds) Flood risk management: research and practice. CRC Press/Balkema proceedings and monographs in engineering; Water and Earth Science; pp. 1479–1483; Taylor & Francis Group, London, ISBN: 978-0-415-48507-4

Weis W, Huber C, Goetlein A (2001) Regeneration of mature Norway spruce stands: early effects of selective cutting and clear cutting on seepage water quality and soil fertility. Optimizing nitrogen management in food and energy production and environmental protection: proceedings of the 2nd international nitrogen conference on science and policy. Sci World 1(S2):493–499

Zweifel R, Steppe K, Sterck F (2007) Stomatal regulation by microclimate and tree water relations: interpreting ecophysiological field data with a hydraulic plant model. J Exp Bot 58(8):2113–2131

Chapter 21
Fundamentals of Model Scaling in Forest Ecology

Matthias Langensiepen

Introduction

Forests occupy about one-third of land area and can be classified according to numerous characteristics including geographical location, seasonality and floristic composition. They affect the hydrology of watersheds by soil water uptake, transpiration, runoff interception, and through various effects on regional microclimate. Water, geomorphology and climate, in turn, strongly influence the structure and function of forest ecosystems. Mutual relations between forests and the water cycle are thus highly complex and not easy to understand.

Quantitative modelling is an effective approach for characterizing the consequences of forest functions at different spatial and temporal scales. Functional relations between independent and dependant variables are often established through power laws which are scale invariant. One of the most widely applied scaling relations is the allometric equation, originally introduced by Huxley (1932):

$$Y = Y_0 M^b, \tag{21.1}$$

where Y_0 is a constant which varies with the type of structural or functional variable, M a size measure such as mass, diameter, circumference or length and b is an allometric scaling coefficient.

Scaling is applied in many contexts and commonly refers to quantifications of object or process similarities across spatial and temporal dimensions. A scale is defined by both grain, the finest level of spatial or temporal resolution, and extent. The establishment of scaling rules requires quantitative characterizations of spatial, temporal, organizational levels and system boundary conditions. Reciprocal relations between system components and their dynamics are quantified by experimental observations and mathematical modelling.

M. Langensiepen (✉)
University of Bonn, Faculty of Agriculture, Institute of Crop Science and Resource Protection, Crop Science Group, Katzenburgweg 5, 53115 Bonn, Germany
e-mail: matthias@langensiepen.net

M. Bredemeier et al. (eds.), *Forest Management and the Water Cycle: An Ecosystem-Based Approach*, Ecological Studies 212, DOI 10.1007/978-90-481-9834-4_21,

Scaling across different scales involves at least five procedures which will be explained in the following sections using different examples: First, the process of interest must be defined by quantitative modelling approaches. Second, the spatial and temporal dynamics and ranges of the process must be specified. Third, the relative importance of model parameters and variations of driving variables across different scales must be understood. Fourth, methods for translating models from one scale to another must be explored and declared if this is possible. Fifth, the approach must be validated across the defined scaling range and possibly in different environmental contexts to assert its universal applicability.

Application of Scaling in Forest Ecohydrology

Allometric relations between tree organ characteristics have been established and tested in many ecological studies to determine of tree biometric features and consequences for forest–water relations. Stem diameter at breast height is one of the most widely applied tree measures in practical forest management. Cermak et al. (1998), for example, used this information to calculate tree and crown base heights with

$$h = a[1 - \exp^{-x/b}]^{c}, \tag{21.2}$$

where x is the tree diameter at breast height and a, b and c are empirical fitting parameters. The Raleigh equation was applied in this study for determining vertical needle distributions in Scot pine crowns:

$$y = \frac{P_1 * (h_{top} - h_i)}{P_2} * \exp\{[-(h_{top} - h_i)^{P_4}] * P_3\}, \tag{21.3}$$

where h_{top} is the tree top height, h the crown layer height at level i and P_1, P_2, P_3 and P_4 are parameters which must be determined in field experiments. Total needle area and density distribution were finally determined using either information of tree diameter at breast height or basal area, and by applying a second order polynomial function. Karlik and McKay (2002) followed a similar procedure for determining allometric relationships in an oak stand in the Sierra Nevada. Using data from 45 tree species of the tropics, Enquist et al. (1999) established a mass-production law as a function of basal stem diameter D that is explicitly independent of wood density:

$$\frac{dM}{dt} = \left(\frac{C_G C_B}{C_D^2}\right) D^2, \tag{21.4}$$

where M is the tree mass and C_G, C_B and C_G are proportionality constants. In spite of a wide variation in growth rates they concluded that production within species scales with $M^{3/4}$ and relative growth rate $(1/M)(dM/dt)$ with $M^{-1/4}$. In a strongly discussed paper about a general model for the origin of allometric scaling laws in biology, West et al. (1997) postulate that for biological rates, times and dimensions the exponent b in Huxley's equation takes on a limited set of values which are multiples of 1/4; they include (West and Brown 2005) metabolic rate ($b \approx 3/4$), lifespan ($b \approx 1/4$), growth rate ($b \approx -1/4$) and tree height ($b \approx 1/4$). They further proposed a model based on fractal geometry and their quarter power law to quantify the structure and allometry of plant vessel-bundle vascular systems and their effects on whole plant hydraulic conductance (West et al. 1999). It is based on the assumption that scaling of morphological and anatomical features follows an allometric law that is universal for all branches and plant species (Mencuccini 2002). Kozlowski and Weiner (1997) cautioned that interspecific allometries *per se* do not reflect any functional relationships and are a result of optimization. Based on extensive studies Becker et al. (2000), Meinzer et al. (2005), Mencuccini (2002) and others showed that water transport does not necessarily scale universally with tree size, because partial buffering of hydraulic resistance from transport path-length effects can occur even for moderate tapering. Based on theoretical considerations, Mäkelä and Valentine (2006) have even demonstrated that hydraulic resistance is not constant in a fractal arrangement of tapering tubes with equal transport properties.

This brief account of a controversy draws the attention to an important question which is frequently raised in different contexts: Is scaling useful overall? West et al. (1999) view the complexity of plant ecophysiology from a high level of integration which facilitates the establishment of simple allometric relations that are universally applicable under practical forest management conditions. Kozlowski and Weiner (1997) demonstrated that interspecific allometries arise from intraspecific patterns which are not necessarily universal applicable. However, both theories assume that energy allocation is optimized (Cates and Gittlemen 1997) which is important from a methodological point of view since logical connections between top-down and bottom-up approaches are established. This example demonstrates that scaling is useful in at least four different ways: (1) It supports search for universal relations and patters which has practical implications. (2) Theories must be expressed in the language of mathematics which eliminates the ambiguities associated with descriptive approaches. (3) Required validations of underlying theories reduce uncertainties in model predictions. (4) New research is motivated when theories fail.

Huxley's law has not only been applied in quantifying allometric relations of single trees, but also for characterizing processes and patterns at larger scales. Yoda's law is another widely applied example (Yoda et al. 1963). It relates plant density d with average plant biomass w:

$$w = k\,d^{-3/2} \; or \; B = k\,d^{-1/2}, \tag{21.5}$$

where k is an intrinsic constant whose value depends on growth form and B the shoot biomass per area. The law rests on the assumption that the canopy maintains a 100% cover (Yoda et al 1963) or has reached its ceiling LAI (Sackville Hamilton et al. 1995). The average interaction between individuals is believed to become constant at this stage and hence should result into regular self-thinning (Li et al. 2000). A controversial discussion similar to the one above revolved around Yoda's law which has been extensively covered in the literature (Lonsdale 1990; Weller 1991; White 1981; Roderick and Barnes 2004; Sackville Hamilton et al. 1995). In spite of this controversy, relations between sizes, masses and densities can be established with reasonable accuracies (Enquist et al. 1998; Pretzsch and Mette 2008) and provide the basis for practical model applications in forest water management.

Canopy architecture influences forest microclimate and water transport into the atmosphere in many ways. Under near neutral conditions, vertical distribution and variation in wind speed are often quantified with the logarithmic scaling law (Kaimal and Finnigan 1994):

$$u(z) = \frac{u^*}{k} \ln \frac{z-d}{z_m}, \tag{21.6}$$

where $u(z)$ is the wind speed at height z, u^* the friction velocity, d the displacement height, z_m the roughness length, and k the von Karman constant. The terminology, practical application and theory are extensively covered in the literature (Brutasert 2005, Barenblatt and Chorin 1998, Campbell and Norman 1998). Vertical distributions of mass, heat and momentum have been inferred from the logarithm law, but the underlying theory (e.g. "K"-Theory) has been criticized, particularly in the context of forest studies. Heat transfer becomes strongly ejection dominated under forest conditions, while momentum transfer is still related to sweeps or gusts. Such countergradient conditions cannot be characterized with the K-theory (see Kaimal and Finnigan 1994; Finnigan and Raupach 1987 for a detailed discussion). Unfortunately, alternative higher-order closure or Langrangian solutions require complicated instrumentation and extensive parameterization which prevent the practical implementation of these methods (Langensiepen 2008).

Evapotranspiration is the connecting link between the forest's water and energy balances. A scaling relation must be employed to account for dynamic changes in energy and water supply to allow for extrapolations of tree water uptake in space and time. Hatton and Wu (1995) suggest that leaf area L is the most appropriate covariate of tree water flux T:

$$T = a\,I\,L + b\Psi_s L^f, \tag{21.7}$$

where I is the radiation intercepted by the foliage, ψ_s the soil water potential, f a scaling exponent, and a and b are model constants. The equation can be summed over all individual trees to obtain stand water use. Another scaling equation has been proposed for estimating the effect of forest cover F on water discharge Q from a drainage area A (Mosley and McKerchar 1993; Thomas and Benson 1975):

$$Q = a\,A^b\,P^c F^d I^e, \tag{21.8}$$

where P is precipitation, I soil infiltration and a, b, c, d, and e are fitting parameters. It is interesting to note that the outputs of both equations depend on the extent of forest cover (e.g. L and F) while stomatal control of tree water flux is totally neglected. McNaughton and Jarvis (1991) provide evidence that this approach is justified for larger scales where ratios of leaf to boundary-layer and canopy to surface air layer conductances are large. They showed that at each level of scale a control system can be treated as a component of a larger scale system. Changes in stomatal conductance cause proportional changes in vapour transport at the extremely small leaf patch scale. Negative feedback loops at the much greater leaf and canopy scales reduce the sensitivity of the system to stomatal control. It becomes completely dominated by feedbacks through the planetary boundary layer at the regional scale when water availability does not limit transpiration. Stomatal control becomes important, however, when canopy conductance is small. This case demonstrates that the relative importance of small scale processes must be known to allow for simplifications at larger scales. Such certainty can only arise from modelling which is the initial and probably most difficult part of scaling analyses (Barenblatt 2003).

Many forest models were developed based on long-tem observations, but increasing fluctuations of environmental and societal conditions prevent the future application of such models. Kimmis (2008) quotes German growth and yield tables in this context which were developed in the late 1700s and based on 50 years of observations. They proved to be eventually inadequate as landless peasants had reduced soil-fertility.

Reductionist scaling is a measure of our ability to understand the complexity of systems. Search for model similarities across different scales enhances our understanding of complex forest ecosystems. Models are thus indispensible prerequisites for scaling studies. Different models and scaling examples have been outlined in this section. The following section extends the discussion to the organismal and molecular levels.

Functional–Structural Plant Models and Systems Biology

There are two concepts, which form a common basis of many models and are suited for interfacing adjoining scales, structure and function. Quantifications of structures provide information about how parts of a system are geometrically related to each other. Functions are employed for mapping one or several system input variables to an output. Interrelations between functions give information about the sequence of logical events in a system.

Based on empirical work of the Danish botanist Peter Boysen Jensen (1932), Masami Monsi and Toshiro Saeki were the first to formulate a model which accounted for the mutual relation between plant canopy structure and one of its most important functions, the production of biomass (Monsi and Saeki 1952). It provided the theoretical basis for calculating photosynthetic production of plant communities based on light–response curves of single leaves. Although the paper was written more than 50 years ago it is still a source of inspiration for plant ecologists (Hirose 2005).

Today, the characterization of plant architecture is carried out with three alternative approaches (Godin 2000): (1) tree graphs, which encode the topography of connected plant organs (Prusinkiewicz and Lindenmayer 1990; Kurth 1994); (2) graphical information about the spatial orientation of organs, which must not necessarily be connected with each other; and (3) three-dimensional cells, called voxels, which carry statistical information about the plant organs they enclose (leaf area density, orientation distribution, etc.). The three methods are applied in quantifying the exposures of single plants against their aerial and soil environments, as well as for characterizing the geometrical composition and extent of plant canopies.

Using structural information, it is possible to localize and quantify functional relations between plant organs and their physical and biological environments at different spatial and temporal scales.

At the canopy scale, dynamic changes in boundary layer flow and exchanges of mass, energy, and momentum strongly depend on the geometrical exposition of leaves against air-movement. Accurately defining flux-profile relations above and within vegetation canopies still poses a major challenge (Raupach 2001; Kaimal and Finnigan 1994). Since the relations between biological processes at canopy or plant scales and external state variables are often non-linear and difficult to quantify, Chelle (2005) recently proposed that microclimate research should be focussed more on the physical environments perceived by individual plant organs instead of whole plant communities. The novelty of this approach is the implicit use of 3D structure for integrating local interactions between organs and their aerial environments over whole plant communities.

Detailed knowledge about spatial patterns of environmental conditions is an important prerequisite for facilitating simulations of fluxes and information in plants. A good understanding of the processes that govern water transport is of particular importance, because they have a major influence on gas exchange properties and, ultimately, on the physiological ecology of plants (Mencuccini 2003; Schulze 1991). The hydraulic architecture approach unites three often differently treated quantitative concepts of characterizing plant–water relations, the Van den Honert approach, the cohesion–tension theory, and quantifications of vessel and branching anatomies (Cruiziat et al. 2002).

Hydraulic (Meinzer 2002) and chemical (Srivastava 2002) signals propagate through the hydraulic system and affect stomatal control of vapour diffusion into the atmosphere. The generation, propagation, and perception of these signals are interrelated with other plant processes such as carbon and nitrogen metabolisms (Stitt 1994). Fluctuations in root zone water availabilities, for example, alter the relative rate of nutrient uptake and circulation, which affect the xylem sap pH and, in consequence, the partitioning of abscisic acid within the stem circulation system (De Boer and Volkov 2003). Changing water availability also affects the pressure within the phloem, thereby altering carbon relations and sugar metabolism (Schulze 1991; Stitt 1994). Metabolic signalling networks related to nitrogen and carbon dynamics (Stitt and Fernie 2003; Stitt et al. 2002) thus interact with hormone signalling pathways to control photosynthetic gene expression and primary production (Paul and Foyer 2001).

Systems biology aims at examining and understanding the entire structure of cellular and organismal function (Kitano 2002). Three types of information are required in such analyses (Minorsky 2003): First, the expression levels of a large number of mRNAs, proteins, structural polymers, and metabolites must be simultaneously measured. Second, molecular changes must be determined at a high temporal resolution. Third, the spatial location of molecular processes in plants must be determined at the level of the cell type. The question is, whether this reductionist approach does not impede progress in understanding plant functioning at larger scales as there are billions of possibilities how molecular components can be fitted together. In light of the extraordinary complexity of plant biological systems it seems unlikely that unique solutions to the puzzle can be found (Sweetlove and Fernie 2005). There are exciting studies, however, which show that even the currently available noisy protein maps can be used for exploring the organization of biological networks, and to reveal interconnected modules, which control specific biological processes (Uetz and Finley 2005). Highly connected molecules, termed as 'party hubs', function as central components of large complexes, which only become active when the rest of the complex components are present. Specific proteins called 'date hubs' only become active at particular times with different partners (Aloy and Russell 2006). Based on this information it is possible to establish the node architectures of cellular networks, which seem to be scale-free. The probability that a node has k links can be calculated with (Barabasi and Oltvai 2004):

$$P(k) \sim k^{-\gamma}, \tag{21.9}$$

where γ is a degree exponent which determines the role of hubs and takes on values between 2 and 3 in most biological networks. It is remarkable that such networks share common principles with engineered networks, namely modularity, robustness to component tolerances, and use of recurring circuit elements. A detailed introduction to the exiting new area of systems biology can be found in Alon (2007).

The application of the highly formalized syntaxes of mathematics, software languages and database standards promote exchange of knowledge across the disciplines which have been mentioned so far. It must be cautioned, however, that model-based exchange in multidisciplinary research is only effective when model components are based on explicit knowledge structures (Boulton et al. 2005; Holyoak and Morrison 2005). A model always evolves in certain contexts, such as molecular plant biology, soil physics or meteorology that may be inaccessible to someone working in a different discipline. It is the joint use of standardized languages, mathematics, protocols, formalisms, and data structures which facilitates multidisciplinary dialogue and evolution of complex knowledge structures. This evolution would be impossible without maintaining disciplinary boundaries which ensure that each part of a model is based on firm conceptual knowledge.

The following section provides a brief overview of the model-based reasoning process which serves as a starting point of any scaling activity. The subsequent section focuses on the process of dimensional model analysis from which scaling laws eventually emerge.

Model-Based Reasoning: The Connecting Link Between Observation and Quantification

Model construction is initialized by the desire to solve problems which are perceived through cognitive skills (Holland et al. 1986; Johnson-Laird 2001; Rodgers 2000) and then mapped into a hypothetical solution structure which can be expressed in linguistic, formulaic, and imagistic informational formats (Nersessian 2002).

Writing an executable model requires the formulation of an abstract logical framework, which does not leave room for the ambiguities of everyday language. This goal can only be attained by using the symbolic language of mathematics (Steiner 2001; Spelke 2005; Dehaene 1997; Thagard and Zhu, 2003). Unlike human language, mathematical syntax consists of a very restricted set of symbols and rules. In order to transform linguistic statements into a mathematical form, a modeller must analyze the inner structure of a problem and use logical reasoning for creating alternative theorems that can be proven or falsified (Nersessian 1999; Ferrari 2003; Varley and Siegal 2000; Rodgers 2000).

After the logical structure of a mathematical model has been specified, the constants of its equations must be determined for matching simulations with reality. This procedure is termed “parameterization” and is accomplished by statistical analyses. When parameters cannot be determined precisely, sensitivity and uncertainty analyses (Cacuci 2003) are carried out for quantifying the effects of parameter variations on simulation precision. It is also possible that the complexity of a biological system prevents the complete identification of a model. Gadkar et al. (2005) proposed an iterative solution for this problem based on a parametric sensitivity analysis through the use of the Fisher Information Matrix and application of the State Regulator Problem algorithm for estimating the unmeasured quantities of complex biological system models. The results are used for defining experimental procedures that generate the most informative measurements for the subsequent iteration.

Model validation is a statistical process where systematic and random patterns in the experimental and modelled results are quantified and compared against each other (Refsgaard and Henriksen 2004). Since it is almost impossible to define the boundary conditions of biological systems it is hardly possible, however, to ascertain that plant simulation models provide accurate predictions. The usefulness of model validation is thus frequently debated amongst modellers of environmental systems (Oreskes 2003). It is a question whether such debates could ever lead to fundamental recommendations for the model validation process as there are serious doubts about whether reasoning can lead to finding absolute truths at all (Steiner 2001). Regardless of this philosophical issue there is no doubt, however, that modelling is a crucial search instrument for testing alternative research hypothesises and optimizing engineering procedures (Pearl 2000; Passioura 1996).

Search is based on goal-satisfaction constraints which lead to the formulation of a particular model whose simulation is tested against a target domain. When initial expectations are not met, hypothetical solutions for the unexpected problem sources are added to the model, which is then tested against the target domain and possible additional constraints. This procedure is iterated until a converging solution is found (Nersessian 2002). As close-formed solutions can seldom be found, numerical techniques are often used for approximating the dynamic behaviour of biological systems (Press et al. 2007). Various techniques of mathematical optimization can be employed for minimizing errors of model predictions (Snyman 2005). Search can also be supported by the application of computational intelligence, such as artificial neural networks, fuzzy systems, belief networks, and evolutionary computation (Konar 2005; Pearl 2000).

Scale Identification and Scaling

Scaling and modelling are coupled processes which require an identification of relevant spatial, temporal and organizational scales. Scales are observer-dependant criteria and must be appropriate for the phenomenon of interest. An experienced observer, well informed about the dynamic behaviour of system components, is able to make proper choices of scales (Haefner 2005). The application of statistical techniques can further support the identification of scales.

Autocorrelation analysis is employed to test whether there is any systematic pattern in the spatial distribution of an entity (Dale 1999) or if a variable is significantly dependant on a variable at another location (Turner et al. 1989). Spatial patterns of two species can be identified with different covariance techniques. The method of multiscale ordination is applied in detecting vegetation patterns at different scales (Noy-Meir and Anderson 1971; Ver Hoef and Glenn-Lewin 1989) and to relate them to underlying gradients such as water availability (Turner et al. 1989). The purpose of spectral analysis is to identify characteristic length scales which explain variability in data sets. Patterns in these sets are compared against the courses of the known forms of sine or cosine waves. The method is related to the Fourier transform technique which is applied to simplify complicated periodic functions in the same manner. Discrete Fourier Transform (DFT) decomposes a set of values into components of different frequencies. The Fast Fourier Transform (FFT) technique produces the same results but is much faster. Fractal models can be applied in analysing landscape structures and functions at multiple scales and for providing information about their grain and extent.

Extrapolating ecosystem processes across different scales requires a good understanding of the role of spatial heterogeneity in ecosystem processes. Wiens (2000) distinguishes between four forms of heterogeneity: spatial variance, patterned variance, compositional variance and locational variance. The degree of heterogeneity and responses of organisms are scale-dependant. Inferring mutual relations between heterogeneities and responses using spatial data alone would violate nearly every requirement for parametric statistical analysis (Meentemeyer and Box 1987). The relative importance of model parameters at different scales must thus be well understood. Scaling is facilitated when parameters values are determined from self-similar structures. Barenblatt (1996, 2003) provides a general guideline how to achieve this goal:

The basic requirement is a model with independent variables, parameters determined from experimental analyses, as well as specifications of initial and boundary conditions. The development of the scaling law is then based on dimensional analysis which establishes relationships between quantities which characterize a phenomenon of interest:

$$a = f(a_1,\ldots,a_k,b_1,\ldots,b_m), \tag{21.10}$$

where a is the quantity to be determined and $b_1,\ldots,b_m$ are dependant parameters which are expressed as products of powers of the dimensions (e.g. time, length etc.) of the independent parameters $a_1,\ldots, a_k$. They can be transformed into dimensionless parameters

$$[\mathrm{a}] = [a_1]^p \ldots [a_k]^r ; [\mathrm{b}_1] = [a_1]^{p_1} \ldots [a_k]^{r_1}, \ldots, [\mathrm{b}_\mathrm{m}] = [a_1]^{p_m} \ldots [a_k]^{r_m} . \tag{21.11}$$

The function under study is then expressed in a dimensionless form *Φ* (e.g. the similarity law)

$$\Pi = \Phi(\Pi_1,\ldots,\Pi_2). \tag{21.12}$$

which has fewer arguments than the original expression:

$$\Pi = \frac{a}{a_1^p \cdots a_k^r}, \Pi_1 = \frac{b_1}{a_1^{p_1} \cdots a_k^{r_1}}, \ldots, \Pi_m = \frac{b_m}{a_1^{p_m} \cdots a_k^{r_m}}, \tag{21.13}$$

The scaling law becomes valid when the function Φ can be replaced by its finite non-zero limit C

$$a = f = C\, a_1^p \ldots a_k^r, \tag{21.14}$$

in which the dependant arguments b_1, b_2,…b_m of the original parameter set disappear and the powers $p,\ldots,r$ are determined through dimensional analysis.

Spatial heterogeneity constraints the application of scaling laws to certain dimensions. Model grain and extent can be manipulated to translate models across specified range of scales. King (1991) proposes three different methods for this purpose: (1) Lumping where parameter values are varied depending on scale, a procedure which is also called calibration. It assumes that the mathematical structure of the model does not change with scale. The method produces large scaling errors when ecological processes have a non-linear behaviour. (2) Extrapolation by increasing model extent either through direct extrapolation or expected value. Grain remains constant during both processes. Small-scale models applied to characteristic landscape elements are simply summed up with the first approach. The method can become impractical however when local models consist of large sets of differential equations which must be solved numerically. The second approach assumes that larger-scale representations of small-scale processes are the product of the landscape area and the outputs from small-scale models. The frequency distribution of the variables describing landscape heterogeneity must be known for this purpose. (3) Upscaling through explicit integration is based on the assumption that spatial heterogeneity can be quantified by a set of functions of space. Complex, nonlinear models cannot be integrated analytically, however, and often prevent the practical application of this method (King 1991). Lumping and direct extrapolation are the most common methods of spatial scaling (Haefner 2005).

Conclusion

Forestry is the art, practice and science of managing forested landscapes in sustainable manners. Disturbances of forest systems by global change and increasing environmental fluctuations require a better understanding of the driving mechanisms to achieve this goal. Expressing knowledge by the language of mathematics provides the basis for better predictions of environmental and human effects on forest ecology. The ability to link models across different scales of forest system organizations is a measure of our understanding of forest complexity. The concept of scale is relatively old and can be traced back to the early twentieth century. However, the complex, interdisciplinary and quantitative nature of scaling and its importance for an improved ecological understanding has been recognized only recently (Schneider 2001). Laws evolving from scaling analyses have a huge practical relevance, because they provide practical access to the complexity of interactions between forests and their environments. It is amazing that the same mathematical relations can be applied in quantifying interactions in protein networks (Barabasi and Oltvai 2004; Alon 2007) and self-organization in ecosystems (Sole and Bascompte 2006). However, this positive outlook should not deceive us to disregard the division between science and management. Humans are the major drivers of forest change (Kimmis 2008). A thorough understanding of resulting effects requires fusions of ecological and socio-economic concepts which are still rare.

Acknowledgments I gratefully acknowledge the support of this work through the European COST action “Forest Management and the Water Cycle”.

References

Alon U (2007) An introduction to systems biology. Chapman & Hall, London

Aloy P, Russell RB (2006) Structural systems biology: modelling protein interactions. Nat Rev 7:188–197

Barabasi AL, Oltvai ZN (2004) Network biology: understanding the cell’s functional organization. Nat Rev 5:101–114

Barenblatt G (2003) Scaling. Cambridge University Press, Cambridge

Barenblatt GI, Chorin AJ (1998) New perspectives in turbulence: scaling laws, asymptotics, and intermittency. SIAM Rev 40:265–291

Becker P, Gribben RJ, Lim CM (2000) Tapered conduits can buffer hydraulic conductance from path-length effects. Tree Physiol 20:965–967

Boulton A, Panizoon D, Prior J (2005) Explicit knowledge structures as a tool for overcoming obstacles to interdisciplinary research. Conserv Biol 19:2026–2029

Boysen Jensen P (1932) Die Stoffproduktion der Pflanzen. Gustav Fischer, Jena

Brutasert W (2005) Hydrology. Cambridge University Press, Cambridge

Cacuci DG (2003) Sensitivity and uncertainty analysis. Chapman & Hall, London

Campbell GS, Norman JM (1998) *An Introduction to Environmental Biophysics* (2nd ed.). New York: Springer-Verlag. 286 pp

Cates S, Gittlemen J (1997) Reading between the lines – is allometric scaling useful? Tree 12:338–339

Cermak J, Riguzzi F, Ceulemans R (1998) Scaling up from the individual tree to the stand level in Scots pine: I. Needle distribution, overall crown and root geometry. Ann Sci For 55:63–88

Chelle M (2005) Phylloclimate or the climate perceived by individual plant organs: what is it? How to model it? What for? New Phytol 166:781–790

Cruiziat P, Cochard H, Ameglio T (2002) Hydraulic architecture of trees: main concepts and results. Ann For Sci 59:723–752

Dale M (1999) Spatial pattern analysis in plant ecology. Cambridge University Press, Cambridge

De Boer A, Volkov V (2003) Logistics of water and salt transport through the plant: structure and functioning of the xylem. Plant Cell Environ 26:87–101

Dehaene S (1997) The number sense: how the mind creates mathematics. Oxford University Press, Oxford

Enquist B, Brown J, West G (1998) Allometric scaling of plant energetics and population density. Nature 395:163–165

Enquist B, West G, Charnov E, Brown J (1999) Allometric scaling of production and life-history variation in vascular plants. Nature 401:907–911

Ferrari P (2003) Abstraction in mathematics. Phil Trans R Soc Lond B 358:1225–1230

Finnigan JJ, Raupach M (1987) Transfer processes in plant canopies in relation to stomatal characteristics. In: Zeiger E, Farqhuar G, Cowan I (eds) Stomatal function. Stanford University Press, Stanford, CA, pp 385–444

Gadkar K, Gunawan R, Doyle F (2005) Iterative approach to model identification of biological networks. BMC Bioinformatics 6:155. doi:10.1186/1471-2105/6/155

Godin C (2000) Representing and encoding plant architecture: A review. Ann. For. Sci. 57: 413–438

Haefner JW (2005) Modeling biological systems. Springer, New York

Hatton T, Wu H (1995) Scaling theory to extrapolate individual tree water use to stand water useful? Hydrol Process 9:527–540

Hirose T (2005) Development of the Monsi-Saeki theory on the canopy structure and function. Ann Bot 95:483–494

Holland J, Holyoak K, Nisbett R, Thagard P (1986) Induction – processes of inference, learning, and discovery. MIT Press, Cambridge

Holyoak K, Morrison R (2005) Cambridge handbook of thinking and reasoning. Cambridge University Press, Cambridge

Huxley J (1932) Problems of relative growth. Methuea, Methuea

Johnson-Laird P (2001) Mental models and deduction. Trends Cogn Sci 5:434–442

Kaimal J, Finnigan J (1994) Atmospheric boundary layer flows. Oxford University Press, New York

Karlik J, McKay A (2002) Leaf area index, leaf mass density, and allometric relationships derived from harvest of blue oaks in a California Oak Savanna. USDA Forest Service Gen. Tech. rep. PSW-GTR-184

Kimmis J (2008) From science to stewardship: harnessing forest ecology in the service of society. For Ecol Manage 256:1625–1635

King A (1991) Translating models across scales in the landscapes. In: Turner M, Gardner R (eds) Quantitative methods in landscape ecology. Springer, Berlin

Kitano H (2002) Systems biology: a brief overview. Science 295:1662–1664

Konar K (2005) Computational intelligence: principles. techniques and applications. Springer, New York

Kozlowski J, Weiner J (1997) Interspecific allometries are by-products of body size optimization. Am Nat 149:352–380

Kurth W (1994) Growth grammar interpreter GROGRA 2.4: A Software for the 3-Dimensional Interpretation of Stochastic, Sensitive Growth Grammar in the Context of Plant Modeling, Introduction and Reference Manual. Technical Manual

Langensiepen M (2008) Scaling transpiration from leaves and canopies. In: Trimble S, Stewart B Howell T (eds) Encyclopedia of water science. Marcel Dekker, New York

Li B, Wu H, Zou G (2000) Self-thinning rule: a causal interpretation from ecological field theory. Ecol Model 132:167–173

Lonsdale W (1990) The self-thinning rule: dead or alive? Ecology 71:1373–1388

McNaughton K, Jarvis P (1991) Effects of spatial scale on stomatal control of transpiration. Agric For Meteorol 54:279–301

Meentemeyer V, Box E (1987) Scale effects in landscape studies. In: Turner M (ed) Landscape heterogeneity and disturbance. Springer, New York

Meinzer F (2002) Co-ordination of vapour and liquid phase water transport properties in plants. Plant Cell Environ 25:265–274

Meinzer F, Bond BJ, Warren J, Woodruff D (2005) Does water transport scale universally with tree size? Funct Ecol 19:558–565

Mencuccini M (2003) The ecological significance of long-distance water transport: short-term regulation, long-term acclimation and the hydraulic costs of stature across plant life forms. Plant Cell Environ 26:163–182

Mencuccini M (2002) Hydraulic constraints in the functional scaling of trees. Tree Physiol 22:553–565

Minorsky P (2003) Achieving the in Silicio Plant. Systems biology and the future of plant biological research. Plant Physiol 132:404–409

Monsi M, Saeki T (1952) Über den Lichtfaktor in den Pflanzengesellschaften und seine Bedeutung für die Stoffproduktion. Jpn J Bot 14:22–52

Mosley M, McKerchar A (1993) Streamflow. In: Maidment D (ed) Handbook of hydrology. McGraw-Hill, Columbus

Mäkelä A, Valentine H (2006) The quarter-power law scaling model does not imply size-invariant hydraulic resistance in plants. J Theor Biol 243:283–285

Nersessian N (2002) The cognitive basis of model-based reasoning in science. In: Carruthers P, Stich S, Siegal M (eds) The cognitive basis of science. Cambridge University Press, Cambridge

Nersessian NJ (1999) Model-based reasoning in conceptual change. In: Magnani L, Nersessian NJ, Thagard P (eds) Model-based reasoning in conceptual change. Model-based reasoning in scientific discovery. Kluwer, New York

Noy-Meir I, Anderson D (1971) Multiple pattern analysis, or multiscale ordination: towards a vegetation hologram? In: Patil G, Pielou E, Water W (eds) Many species populations, ecosystems and systems analysis. Pennsylvania State University Press, University Park, PA

Oreskes N (2003) The role of quantitative models in science. In: Canham CD, Cole JJ, Lauenroth WK (eds) Models in ecosystem science. Princeton University Press, Princeton, NJ

Passioura J (1996) Simulation models: science, snake oil, education, or engineering? Agron J 88:690–694

Paul M, Foyer C (2001) Sink regulation of photosynthesis. J Exp Bot 52:1383–1400

Pearl J (2000) Causality: models, reasoning, and inference. Cambridge University Press, Cambridge

Press W, Teukolsky S, Vetterling W, Flannery B (2007) Numerical recipes: the art of scientific computing. Cambridge University Press, Cambridge

Pretzsch H, Mette T (2008) Linking stand-level self-thinning allometry to the tree-level leaf biomass alloetry. Trees 22:611–622

Prusinkiewicz P, Lindenmayer A (1990) The Algorithmic Beauty of Plants. Springer, New York

Raupach M (2001) Inferring biogeochemical sources and sinks from atmospheric concentrations: general considerations and application in vegetation canopies. In: In Schulze ED, Heiman M, Harrison S, Holland E, Lloyd J, Prentice IC, Schimel D (eds) Global biogeochemical cycles in the climate system. Academic, San Diego, CA

Refsgaard J, Henriksen H (2004) Modeling guidelines – terminology and guiding principles. Adv Water Resour 27:71–82

Roderick M, Barnes B (2004) Self-thinning of plant populations from a dynamic viewpoint. Funct Ecol 18:197–203

Rodgers N (2000) Learning to reason: an introduction to logic, sets, and relations. Wiley, New York

Sackville Hamilton NR, Matthew C, Lemaire G (1995) In defence of the –3/2 boundary rule: a re-evaluation of self-thinning concepts and status. Annals of Botany 76: 569–577

Schneider D (2001) The rise of the concept of scale in ecology. BioScience 51:545–554

Schulze ED (1991) Water and nutrient interactions with plant water stress. In: Mooney H, Winner W, Pell E (eds) Response of plants to multiple stresses. Academic, New York

Snyman J (2005) Practical mathematical optimization: an introduction to basic optimization theory and classical and new gradient-based algorithms. Springer, Berlin

Sole R, Bascompte J (2006) Self-organization in complex ecosystems. Princeton University Press, Princeton, NJ

Spelke ES (2005) Sex differences in intrinsic aptitude for mathematics and science? Am Psychol 60:950–958

Srivastava L (2002) Plant growth and development. Academic, San Diego, CA

Steiner G (2001) Grammars of creation. Yale University Press, New Haven, CT and London

Stitt M (1994) Flux control at the level of the pathway: studies with mutants and transgenic plants having a decreased activity of enzymes involved in photosynthesis partitioning. In: Schulze ED (ed) Flux control in biological systems. Academic, San Diego, CA

Stitt M, Fernie A (2003) From measurements of metabolites to metabolomics: an ‘on the fly’ perspective illustrated by recent studies of carbon-nitrogen interactions. Curr Opin Biotechnol 14:136–145

Stitt M, Müller C, Matt P, Gibon Y, Carillo P, Morcuende R, Scheible W, Krapp A (2002) Steps towards an integrated view of nitrogen metabolism. J Exp Bot 53:959–970

Sweetlove L, Fernie A (2005) Regulation of metabolic networks: understanding metabolic complexity in the systems biology era. New Phytol 168:9–24

Thagard P, Zhu R (2003) Acupuncture, incommensurability, and conceptual change. In: Sinatra G, Pintrich P (eds) Intentional conceptual change. Lawrence Erlbaum Associates, Mahwah, NJ

Thomas A, Benson M (1975) Generalization of streamflow characteristics from drainage basin characteristics. U.S. Geological Survey. Water-Supply Paper

Turner M, Dale V, Gardner R (1989) Predicting across scales: theory development and testing. Landscape Ecol 3:245–252

Turner S, O’Neill R, Conley W, Conley M, Humphries H (1991) Pattern and scale: statistics for landscape ecology. In: Turner M, Gardner R (eds) Quantiative methods in landscape ecology. Springer, Berlin

Uetz P, Finley R (2005) From protein networks to biological systems. FEBS Lett 579: 1821–1827

Varley R, Siegal M (2000) Evidence for cognition without grammar from causal reasoning and theory of mind in an agrammatic aphasic patient. Curr Biol 10:723–726

Ver Hoef J, Glenn-Lewin D (1989) Multiscale ordination: a method for detecting pattern at several scales. Vegetatio 82:59–67

Weller D (1991) The self-thinning rule: dead or unsupportable? – a reply to Donsdale. Ecology 72:747–750

West G, Brown J (2005) The origin of allometric scaling laws in biology from genomes to ecosystems: towards a quantitative unifying theory of biological structure and organization. J Exp Biol 208:1575–1592

West G, Brown J, Enquist B (1999) A general model for the structure and allometry of plant vascular system. Nature 400:664–667

West G, Brown J, Enquist B (1997) A general model for the origin of allometric scaling laws in biology. Science 276:122–126

White J (1981) The allometric interpretation of the self-thinning rule. J Theor Biol 89:475–500

Wiens J (2000) Ecological heterogeneity: an ontogeny of concepts and approaches. In: Hutchings M, John E, Stewart A (eds) The ecological consequences of environmental heterogeneity. Blackwell, Oxford

Yoda K, Ogawa KH, Hozumi K (1963) Self-thinning in overcrowded pure stands under cultivated and natural conditions (Intraspecific competition among higher plants XI). Journal of Biology Osaka City University 14:107–129

Chapter 22
Up- and Downscaling Model Approaches for Water Relations in Forest Management from Plot to Landscape Level

Chris S. Eastaugh, Stephan A. Pietsch, Richard Petritsch, Elisabeth Pötzelsberger, and Hubert Hasenauer

Introduction

Water cycles and forest growth cycles are co-dependant, in that water movement through an ecosystem drives forest growth, and forest growth has fundamental influences on water cycling. Neither process can be successfully modeled without proper consideration of the other. Water is one of the key factors in determining land cover, and in many environments limited precipitation is the determinant of whether a forest or non-forest biome will be found. Forest management practices may have a substantial impact on how water-related processes are affected by interactions with vegetation.

Water enters terrestrial ecosystems from (i) the atmosphere by rainfall, (ii) the snowpack by melting, (iii) lateral inflows like flooding, and (iv) the ground water table by soil water recharge. Rainfall is partially intercepted by the canopy. If the amount of rainfall exceeds the potential canopy interception, it is routed to the soil water pool. Water is released from a snow pack and adds to the soil water pool, if the energy input from temperature and radiation exceeds the energy of melting. Lateral water inflow occurs when water input from rainfall or snowmelt exceeds the capacity of standard water discharge routes, e.g. brooks and rivers.

The upper limit for soil water content is the soil porosity, which equals the total volume of soil pores. Ground water recharge, of the soil volume accessible by plants, occurs only if the soil water content falls below the field capacity. Field capacity is equal to the soil water content when the gravitational force is equal to the retention force of soil water potential and is lower than soil water saturation. Below the field capacity, the upward pressure of the soil water potential exceeds the downward pressure from gravitation. The amount of ground water ascent is defined

---

C.S. Eastaugh (✉), S.A. Pietsch, R. Petritsch, E. Pötzelsberger, and H. Hasenauer
Department of Forest and Soil Sciences, BOKU University of Natural Resources and Applied Life Sciences, Institute of Silviculture, Vienna, Peter-Jordan-Str. 82, A-1190, Vienna, Austria
e-mail: Chris.eastaugh@boku.ac.at; Stephan.pietsch@boku.ac.at; Richard.petritsch@boku.ac.at; Elisabeth.poetzelsberger@boku.ac.at; Hubert.hasenauer@boku.ac.at

M. Bredemeier et al. (eds.), *Forest Management and the Water Cycle: An Ecosystem-Based Approach*, Ecological Studies 212, DOI 10.1007/978-90-481-9834-4_22,

by the net upward force and the soil hydraulic conductivity, which depend on soil texture and soil water content (Pietsch et al. 2003).

Water leaves the ecosystem by soil evaporation, snow sublimation, soil water outflow and plant transpiration. Evaporation of soil water is governed by the atmospheric vapor pressure deficit and the temperature. Sublimation of snow water is determined by the radiation energy input. Soil water outflow includes all water, which is above soil water saturation and a proportion of soil water above field capacity (Chang 2003).

Plant transpiration depends on the atmospheric vapor pressure deficit, leaf temperature and water potential, and stomatal conductance to water vapor. Stomatal conductance is regulated by the leaf water potential, atmospheric vapor pressure deficit and temperature. Wind speed also has a direct effect on canopy transpiration and soil water evaporation. Forest management operations (harvesting, thinning, regeneration etc.) have a direct effect on overall transpiration, and are a major influence on catchment scale water relations. The influence of forest health and forest management on catchment scale hydrological processes is such that water planning at the catchment scale cannot be complete without considering the impacts of forests (Zhang et al. 2001). Clearly then there are links and co-dependencies between physical, physiological and forest management processes at all scales.

Combining processes that operate at the cellular level in short time frames (such as stomatal conductance) with others in up to the long-term landscape level (i.e. groundwater discharge) is often problematic, and so information must be adapted in some way so as to integrate and assess ecosystem processes at the appropriate spatial and temporal scale.

Information and understanding of processes is often not available at the temporal or spatial scale needed to provide meaningful direct outputs. Mechanistic process models ideally apply physically justified algorithms to observed data and relationships to produce validated outputs, thus developing understanding of the processes involved and providing a means of testing alternative scenarios. Empirical models extrapolate mathematical and statistical relationships found between observed data to new circumstances, in an effort to predict outcomes or discover inconsistencies. In either case, there is often a mismatch between the temporal or spatial scale of the known data, the level of understanding of the processes and relationships, the computational power realistically available, and the desired outputs.

Scaling may be needed in either (or both) spatial and temporal dimensions. Upscaling denotes a change to a coarser, more generalized representation of the data, either using larger spatial units or longer temporal periods. Downscaling conversely infers information at a finer resolution or shorter time period than the original. Caution should be taken with terminology, as there is some potential for confusion in that one upscales to a lower resolution, and downscales to higher levels of detail. Scaling may also describe a conceptual perspective, as either moving from a detailed consideration of elementary processes to a descriptive whole (bottom up) or vice versa (top down).

This chapter will examine the principles of model scaling, and provide a series of example studies illustrating some of the means by which climatic, forest and hydrological data at different scales may be adapted and combined to give results at a desired final scale. The interdependence of forest management and the water cycle are highlighted through the use of four examples drawn from current or recent research.

Scaling Options

Modeling in general may be divided into two broad groups; ‘bottom-up’ and ‘top-down’ approaches. Each has their uses and limitations, but issues of scale are common to both. Bottom-up modeling is essentially an attempt to assemble data and process understanding into a greater whole, while top-down modeling tries to disassemble observable results into their component features.

Bottom-up models begin from the point of understanding certain physical processes, and combining these with data to build a representation of a system. These models seek to integrate as many as possible of the known factors in determining the output of a system – in the case of water modeling, issues of different vegetation types, soils, management factors and detailed climatic information are required. Bottom-up models were criticized in the past for excessive error propagation and a resulting lack of predictive power (i.e. Mohren and Burkhart 1994), but recent advances have demonstrated the utility of bottom up modeling in forest water relations in many circumstances (i.e. Pietsch and Hasenauer 2002; Thornton et al. 2002; Pietsch et al. 2003; Merganičová et al. 2005; Bond-Lamberty et al. 2007).

Top-down models apply broad scale, often empirical relationships to generally easy measured data such as catchment outflow or tree diameter, and infer from this more detailed information regarding (for example) water usage or growth rates. Although error propagation is less of an issue with these models, they often lack the ability to account for impacts at scales finer than those used for the initial parameterization.

Approaches combining both bottom up and top down modeling have been suggested (i.e. Zeide 2003; Wang et al. 2006), and this chapter will give examples of how this has usefully been applied to water catchment modeling.

Upscaling

The context/situations in which upscaling is generally relevant are (i) predictions of some higher level ecosystem states or processes (e.g. runoff from a whole catchment, growth of a forest stand over the whole rotation period or life time), which typically include the attempt to generalise measurement results and (ii) scenario analysis, where the ecosystem behaviour in a changing environment is addressed. Considering only statistical relationships for upscaling is not enough, because measurements are only snap shots. The ecosystem features and behaviour result from multitudinous interactions/processes. Ecosystems are characterised not by simple (linear) relationships, but by heterogeneity in space and time.

For the description of a living ecosystem that experiences environmental changes, stress and human impact we need biogeochemical mechanistic ecosystem modeling. Such models are designed to reproduce, quantify, and describe ecosystem processes explicitly including the water cycle. According to (Pacala et al. 1996) the three conflicting interests in modeling – simplicity, observability and biological realism – have to be equally addressed. In addition, scaling in space and time is a crucial issue for designing a well balanced model. Mechanistic ecosystem modeling can be looked at as a step-wise upscaling procedure. Reality, as it can be measured, seen or experienced in any other way, undergoes an abstraction process within a model and the integration of results from field research and validation is an important aspect. Ecosystem processes are formalised and information on the state or the functioning of a system are integrated in a framework that allows predictions beyond the study site or on the system development. This can be viewed as an extrapolation in space and time.

In the course of upscaling for modeling ecosystem behaviour, on the one hand it is necessary to generalize (i) processes (complex stomata regulation and conductive resistance into transpiration, C and N allocation), (ii) states (soil) and (iii) functions (species to biome, tracheae and tracheids to sapwood) and knowledge can be more easily applied in a broader field. On the other hand spatial and temporal upscaling can prompt modellers to see a process (formally believed to be described in sufficient detail) in a broader context and eventually set it in a more complex system where more feedback loops are integrated. What was considered as a state parameter in the smaller scale system may have to be considered as a variable – (i) an independent model driver or (ii) a component of one or more new feedback loops – in a larger scale system (e.g. atmospheric CO_2 content, soil texture, soil depth – temporal upscaling).

Upscaling will be required or desirable in a wide range of modeling situations. In many cases the model drivers are only available at a higher temporal scale than the process theoretically is understood and therefore the processes must be simulated at the lower temporal resolution. If the model drivers are available at a sufficient temporal resolution, but information on the state parameters is rough (typical for simulations beyond the study site), then the processes must be simulated at the lower spatial resolution.

For processes being integrated from a system of sub-processes showing feedback loops upscaling will be necessary if some sub-processes work on higher temporal or spatial scales. For a well balanced model the scales of the interfering processes should be similar. In other cases a process operates so slowly that it makes no sense to deal with it in a very high temporal resolution, or there is no need to gain more detailed information/outputs and so computational power and time can be saved working at coarser scales. Variations or information created by the more detailed description of a process or system may be negligible when seeing it in a broader context or integrating it in a larger system, and working at high resolutions may introduce unnecessary noise into the system.

The major challenges and goals in forest ecosystem research are the holistic understanding and description of the functioning and behaviour of specific ecosystems. This requires analysis on many different scales and the assembly and structuring of the knowledge. Thus, there is a demand for a concept of a system representing the real forest ecosystem. This system will comprise a high number of interrelated subsystems, depending on where the interests lie (e.g. forest–water interactions). A model of the system can be built when mathematical descriptions of interactions between and the processes within the subsystems can be found. Knowing the model drivers, a description of the functioning of the ecosystem is possible.

The development of a forest ecosystem model represents an upscaling process. A good example is the mechanistic model Biome-BGC (Running and Hunt 1993) which considers all major essential ecosystem processes and so is able to simulate the development and behaviour of a forest stand under specific and changing environmental conditions. In the course of designing this model, the various plant parts (cell organelles, cells, small tissues and organs) and other constituents of the ecosystem were generalized into a smaller number of key ecosystem compartments, such as sun- and shade-leaves, live and dead-stem, fine and coarse roots, litter and coarse woody debris and soil (described by its texture and depth).

The innumerable very fine and coarse scaled natural processes were reduced to the amount absolutely necessary for representing the key ecosystem fluxes of energy and the most relevant substances (carbon, water and nitrogen). Carbon uptake through the stomata, photosynthetic fixation and allocation within the plant, litter fall and mineralization, precipitation interception by the canopy, water uptake through roots and loss through the leaves, and many other processes more are calculated as functions of several meteorological variables, physical site parameters and species specific eco-physiological parameters. A forest stand is regarded as homogenous, which means that only the dominating tree species is simulated and conditions do not vary within a stand. With all these simplifications fundamental upscaling steps have been made.

Downscaling

Downscaling is an effort to derive values at finer spatial scales or shorter temporal periods from information at coarser or longer levels. In many cases this is done simply by assuming that all values at the smaller scale will be equal to that at the large (soil types and depths for example are often represented as being spatially constant at fine scales, based on estimates of the broad-scale mean), although more sophisticated methods are sometimes used to ‘smooth’ the transition. An example of this is used in some GIS applications when downscaling elevation data from coarse to finer spatial resolutions. Temporal downscaling may assume that short scale values change in definable ways, and may thus be inferred from longer scale data. For example, if temperature over a 24-h period is assumed to follow a particular sinusoidal pattern, knowledge of maximum and minimum temperatures at a daily scale may be used to infer hourly values. The accuracy of these values will be dependant on the accuracy of the assumed pattern.

Interpolation of point data provides some semantic difficulties. In one respect it may be considered upscaling (from points to wider areas), but as the aim is to produce more detailed information than that which is initially available it should perhaps best be described as a form of downscaling. Several standard interpolation methods may be applied to spatial data but these methods fall into one of two categories: deterministic (such as inverse distance weighted methods or spline functions) or geostatistical (i.e. kriging, conditional simulation). Simpler methods are more computationally efficient and easily understood, and are appropriate for functions that change relatively smoothly between data points such as climatic data, i.e. the truncated Gaussian filter IDW interpolations used in DAYMET climate interpolations (Thornton et al. 1997). Functions with discrete changes are better interpolated with methods that consider a wider range of points, which may have relevant similarities less linked to spatial attributes. Soil interpolation is a good example of this (Petritsch and Hasenauer 2007).

The greatest risk in downscaling is the assumption that local conditions at the fine scale will not appreciably impact the values obtained from the coarse scale. In some circumstances this assumption is justified, where conditions are effectively homogenous or the main drivers of the system are factors that operate over the broad scale. In other cases, particularly in ecological or climate modeling, local conditions have strong effects on the system and simply downscaled outputs may have little relationship with observable realities. Where coarse scale values are highly dependant on Monte Carlo simulations requiring random inputs (such as, for example, ecological effects of forest fires or management operations at a landscape scale (McCarthy and Lindenmayer 1998)), it is dangerous to assume that these values will hold true at finer scales. These simulations inherently amalgamate a large number of partly random finer scale results, and it is not valid to then infer information at the original fine scale.

Statistical downscaling is increasingly being applied to develop regional climate scenarios from coarser scale global models, although the accuracy of these results is sometimes uncertain as different techniques can give significantly different results (reviewed in Wilby et al. 2004). These techniques use coarse scale data as ‘predictors’ of small scale results, based on the assumption that the statistical relationships between coarse and fine scale values are constant. Care should be taken that the errors margins inherent in these relationships are not excessive when considered in combination with the error margins of the original coarse scale values. Weather generators based on first-order Markov chains can be used to generate fine scale temporal data from coarser series (i.e. Wilks 1999; Jones and Thornton 1999).

Dynamic downscaling uses coarse results as inputs into models that operate at finer scales (nested models). Some downscaling techniques have been found to have too strong a dependence on boundary conditions supplied from the coarse scale data (i.e. Beltrán-Przekurat et al. 2008). Dynamic downscaling is computationally expensive. In some cases results from dynamic downscaling require statistical readjustments (Oh et al. 2004).

Mean Values of Fluctuating Processes

A special case of scaling is apparent in methods that infer fixed values for fluctuating processes. This is commonly more complex than simply taking the mean of the maximum and minimum, as the relevant desired value may reflect processes following non-linear patterns. Two examples presented here show methods for deriving a single value for daytime temperature, and for daily stomatal conductance.

The length of the time step of the recursion in process modeling defines the temporal resolution within the model. For large scale models it may range from one second (ORCHIDEE, Krinner et al. 2005) to 1 month (TEM, Mellilo et al. 1993), but typically it is 1 day (LPJ, Sitch et al. 2003; Biome-BGC, Thornton 1998; MC1, Bachelet et al. 2001). Regardless of the chosen time step the model drivers with fast dynamics have to be adapted to fit the time step. For meteorological drivers this adaptation is achieved by averaging the values over the time step that is implemented in the model. For a daily time step, e.g., daytime air temperature (t_{DAY}), which affects leaf temperature during photosynthesis, may be estimated from the daily minimum and maximum air temperature (t_{MIN}, t_{MAX}) as:

$$t_{DAY} = 0.45\left(t_{MIN} - \frac{t_{MIN} - t_{MAX}}{2}\right) + \frac{t_{MIN} - t_{MAX}}{2}. \tag{22.1}$$

Other meteorological model drivers may be fitted to a daily time step by averaging the values over the length of the daylight period. The length of the daylight period may be calculated from daily earth-sun geometry and the position on earth (Carbone 2007). The amount of daily photosynthetic carbon assimilation is then calculated using t_{day} for RubisCO limited photosynthesis or mean daytime radiation for electron transport limited photosynthesis. The minimum of these is then multiplied with the length of the daylight period in order to give an estimate for photosynthetic carbon assimilation over the modeled time step (e.g. 1 day). Note that the average output value of nonlinear functions (e.g. photosynthesis rate) may not match the output value achieved by using average values of variables (e.g. for radiation or temperature) in nonlinear equations. For a daily time step, the calculated value may be lower or higher versus the average output value of the non linear function. It is important to check that the averaging procedure does not lead to differences in the mean values of the nonlinear function if calculated over multiple simulation steps.

A similar averaging approach is applied to the ecophysiological parameters controlling stomatal conductance (g) to water vapor and CO_2. Stomatal conductance has an upper limit (g_{MAX}) which is down-regulated to actual stomatal conductance (g_{ACT}) by environmental factors, such as leaf water potential (Ψ), previous minimum air temperature, daytime vapor pressure deficit (VPD) and radiation (PHAR):

$$g_{ACT} = g_{MAX} \cdot m_{TMIN} \cdot m_{VPD} \cdot m_{\psi} \cdot m_{PHAR} \tag{22.2}$$

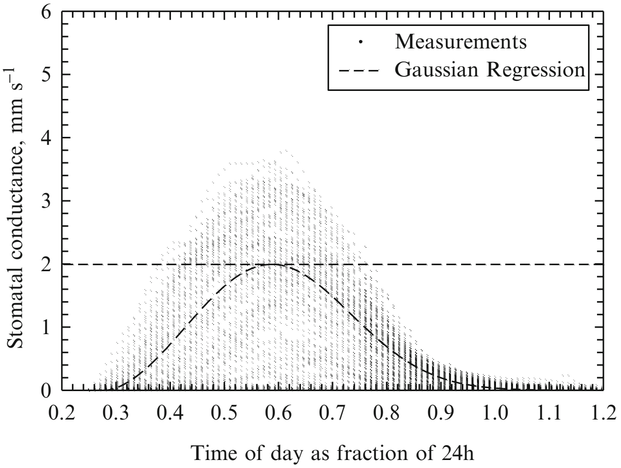


Fig. 22.1 Stomatal conductance of Norway spruce in Kreisbach, Austria. Measurements were performed every 15 min throughout the growing season 1999. The maximum of the daily average stomatal conductance used in a model with a daily time step results from the peak of a Gaussian regression as indicated by the *dashed line*. For Norway spruce the value of the maximum of daily average stomatal conductance equals 2 mm s^{-1}

The variables m_x are multipliers with values between 0 and 1, which are used to reduce g_{MAX} to g_{ACT}. Since radiation and vapor pressure deficit are mean values for a given time step the parameter g_{MAX} also needs to be averaged for the time step. Figure 22.1 illustrates this averaging step for daily g_{MAX} of Norway spruce.

Examples

In the following section four examples were, not was selected for demonstrating of scaling is an integral part of modeling water relations in forest management. Each example shows the relevance of forest management to water relations or vice versa, and how issues of scale may be addressed. The four examples examine a diverse range of issues regarding groundwater influences on forest growth, water runoff from forest roads, catchment water yield modeling and wildfire risk assessments.

Groundwater

The treeline at lower altitudes in many environments is often determined by precipitation limitations. Commonly, regions with precipitation <400 mm/year mark the forest/steppe boundary (Zhicheng 1995; Novenko et al. 2009). The presence of forest in areas with less than this level of rainfall indicates the likelihood of other sources of water which must be considered in modeling or management scenarios.

Forest productivity is modeled using precipitation as an input, but in some environments groundwater may also be an extremely important source of water for growth or a limitation due to flooding and soil saturation. Pietsch et al. (2003) studied the observed and modeled growth responses of forests in the floodplains of southern Moravia, and developed conclusions regarding growth responses due to historical land use change. A major recent change was the construction of a canal system in 1971/1972 that prevents spring flooding.

This implementation of the BIOME-BGC model added two daily water flux variables, one for flood water flux ($\mathrm{fw}_{\mathrm{in}}$) and one for ground water flux ($\mathrm{gw}_{\mathrm{in}}$) into the soil water pool, along with one nitrogen flux variable ($\mathrm{NDep}_{\mathrm{fw}}$) to account for flood induced deposition influx to the soil mineral nitrogen pool.

Infiltration of ground water was calculated based on the deviation of soil water potential from water potential at field capacity. Soil water potential consists of two different forces counteracting on soil ground water movement. The first – gravitation – is acting downwards, and the second – soil matric potential given by the deviation from the matric potential at field capacity – is directed upwards when soil water content is below field capacity or equals zero otherwise. The sum of both equals the amount of energy available for water ascent and drives the infiltration of water from the ground water table into the rooting zone (Eq. 22.3), whereby the amount of water ascent per day depends on soil hydraulic conductivity (Eq. 22.4)

$$\Delta\Psi = (-\Pi_{fc} + \Pi_{act}) + \rho_w g(h_1 - h_2), \tag{22.3}$$

where $\Delta\Psi$ =Water Potential, Π_{fc} = matric potential at field capacity, Π_{act} = matric potential at actual soil water content, ρ = soil water density, g = acceleration due to gravity, and h_1 and h_2 are potential soil water heights.

$$g_{w,in} = K(\Delta\Psi), \tag{22.4}$$

where $g_{w,in}$ = groundwater entry to the rootzone, and K is the soil hydraulic conductivity coefficient (Saxton et al. 1986).

Daily modeling in BIOME-BGC (using upscaled values of fluctuating processes described above) of stand transpiration in 6 years between 1972 and 1982 was tested against observations by Cermak et al. (1973, 1982). Figure 22.2 shows results for 1973.

Figure 22.3 demonstrates the differences in LAI dependent on access to groundwater in the previous season. Catchment-wide data at a broad (annual) temporal scale is used here to inform a daily stand-level model.

Validated daily results allow the construction of long-term (century scale) simulations in BIOME-BGC (Fig. 22.4), illustrating the effects of watershed management and different historical forest management strategies on carbon and nitrogen pools at a landscape scale.

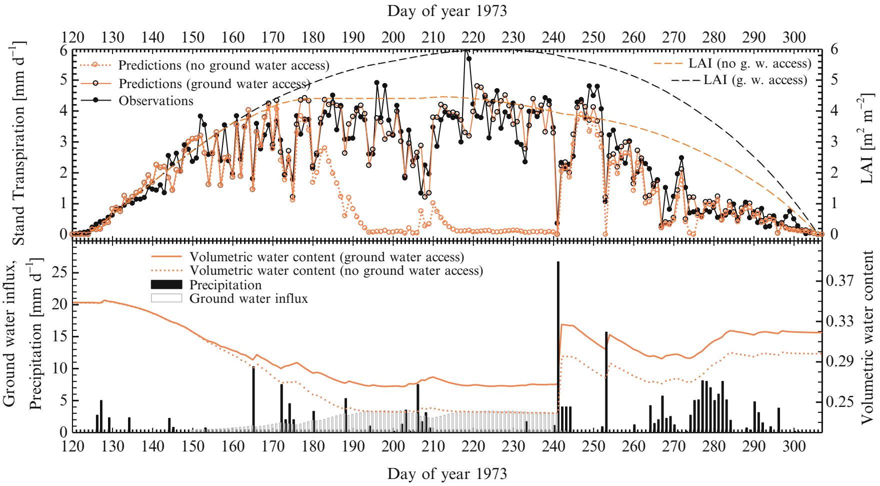


Fig. 22.2 Seasonal course of observed and predicted stand transpiration, predicted leaf area index (LAI), predicted soil volumetric water content, and observed precipitation for the 1973 growing season under the two scenarios with and without ground water access in 1973

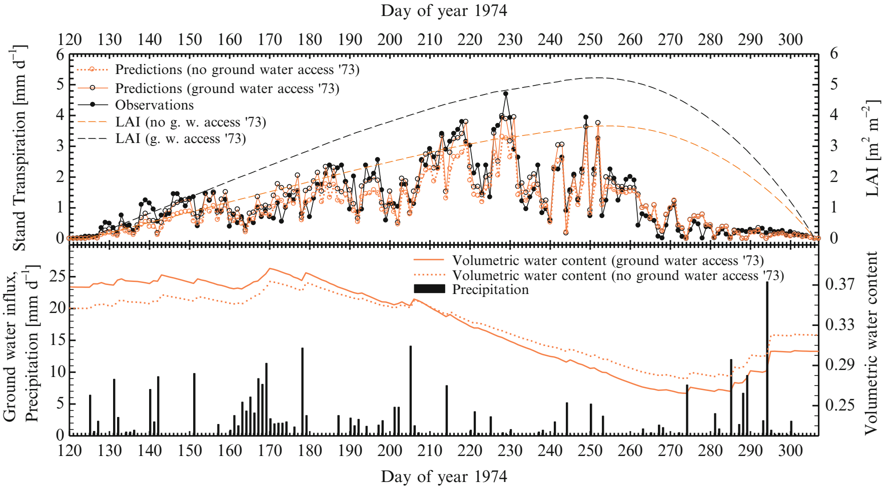


Fig. 22.3 Seasonal course of observed and predicted stand transpiration, predicted leaf area index (LAI), predicted soil volumetric water content, and observed precipitation for the 1974 growing season under the two scenarios with and without ground water access in 1973

Road Runoff

Forest management operations may impact on water relations through biophysical or purely physical means. Forest operations such as road construction can increase stream sedimentation, change water runoff pathways and reduce catchment response times to precipitation events (Wemple et al. 1996). In this example we

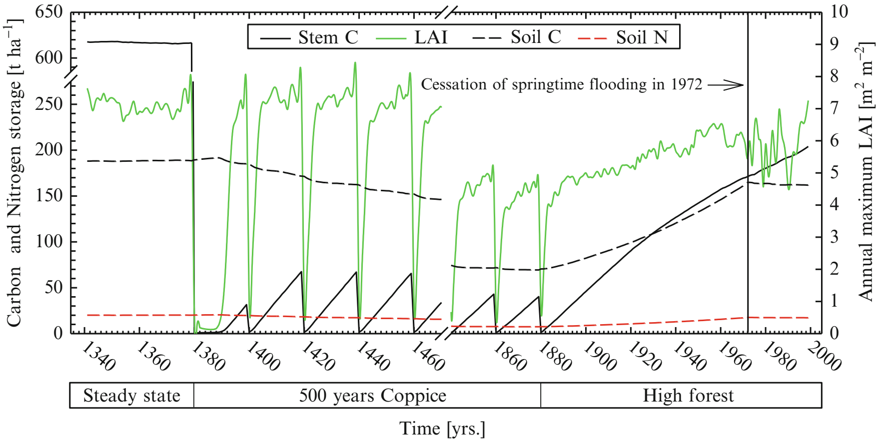


Fig. 22.4 Century-scale fluxes of carbon and nitrogen and LAI changes across a period of historical land use change and altered groundwater management

examine how scaling techniques may be used to reduce uncertainty in overland flow modeling and help to inform forest management decisions.

Water quality in forested catchments is largely dependant on the volume of sediment-bearing water that enters stream systems. Process models such as WEPP (Water Erosion Potential Prediction, Flanagan and Nearing 1995) may be highly scale-dependant, and give different results depending on model resolution (Rhee et al. 2004). An alternative approach useful in some situation is probabilistic modeling.

In some areas one of the main sources of sediment is contained in runoff water from forest roads. Eastaugh et al. (2008) studied the likelihood of sediment-bearing runoff water from forest roads reaching the stream network in part of Canberra's water supply area. Water that reaches the stream without infiltrating will compose of a 'direct' component (where water is discharged directly from road to stream) and a 'diffuse' component where some proportion of the water will infiltrate into the soil. Hairsine et al. (2002) had developed probabilistic expectation of the mean proportion of runoff water that could be expected to travel more than 5 m from a drainage outlet, and the variance around that mean. By measuring the flowpath length from each drainage outlet in the study area to the stream network individual mean proportions and variances of runoff water reaching the stream (rather than infiltrating) was able to be calculated for each drainage outlet. The proportion of non-infiltrating runoff water (P) was calculated as

$$P = \frac{V_{out} - 67X}{V_{out}}, \tag{22.5}$$

where V_{out} = the volume of water expected to run off the road segment (road surface area * rainfall depth), and X = distance from drainage outlet to stream.

The variance (σ^2) of a flow volume (V_X) reaching a distance (X) was measured by Hairsine et al. (2002) as

$$\sigma_{V_X}^2 = \frac{X^2}{25} * 35{,}600(l^2). \tag{22.6}$$

Statistical methods were then used to sum the variances of each drainage outlet i in cases where there was some predicted probability of runoff reaching the stream ($P > 0$) and where there was not ($P < 0$) (Fig. 22.5). The total proportional variance σ^2_{tot} is then calculated as

$$\sigma_{tot}^2 = \sum^{P>0} \left(\frac{A_{df_i}}{A_T} \right)^2 \frac{\sigma_i^2}{V_{out_i}^2} + \sum^{P=0} \left(\frac{A_{df_i}}{A_T} \right)^2 \left(\frac{4}{3} z - \frac{o_i^2}{V_{out_i}^2} \right), \tag{22.7}$$

where $\dfrac{A_{df}}{A_T}$ is the ratio of the diffusely connected road area to the total road area, and z is a factor chosen to reflect the desired degree of confidence in the results.

The total hydrological connectivity (C) of the road network with the stream network could then be expressed as the sum of the area of the directly connected road segments A_D, the summed expected proportions from diffusely connected segment areas and the associated uncertainties

$$C = \sum A_D + \sum A_{df_i} P_i \pm z \sqrt{\sigma_{tot}^2 \sum A_{df_i}}. \tag{22.8}$$

Although the variance of individual predictions was very high, upscaling from the individual drain level to encompass a whole road reach reduced the uncertainty to such a level that the results are useful from a road management perspective (Fig. 22.6). Upscaling from single sites to a catchment level can increase the signal to noise ratio and reduce the uncertainty of the summed predictions.

Catchment Yield Modeling

Catchment level water runoff is an important issue for human water supplies and flood control, among other concerns. Forest management has a significant influence on water yield, but the demonstration of this in a modeling environment requires a series of explicit scaling steps.

The influence of forest cover on water balance dynamics was demonstrated by Pötzelsberger (2008) through modeling the forest transpiration in the Schmittenbach catchment. Evapotranspiration in the Biome-BGC model is calculated as the sum of evaporation, transpiration and snow-sublimation and is the main factor that determines how much water is potentially left for outflow from the full amount of

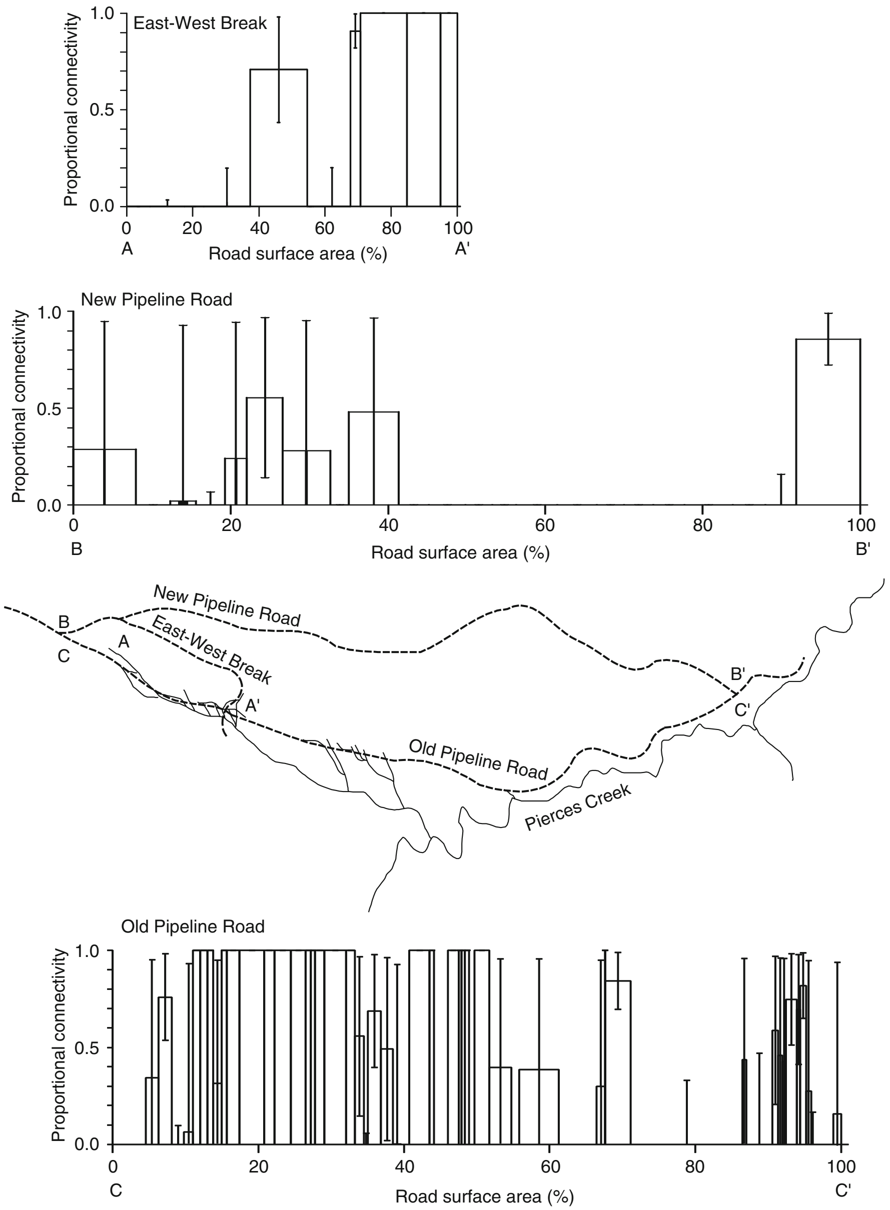


Fig. 22.5 Individual road segment connectivity, related to geographic position (Modified from Eastaugh et al. 2008)

available water. The free, available water comes primarily from precipitation, but also from snowmelt in spring or from soil water that is already present at the stand. In this model, precipitation is either intercepted by the canopy or travels into the soil. Rainfall interception by the canopy is a function of the leaf area index (LAI).

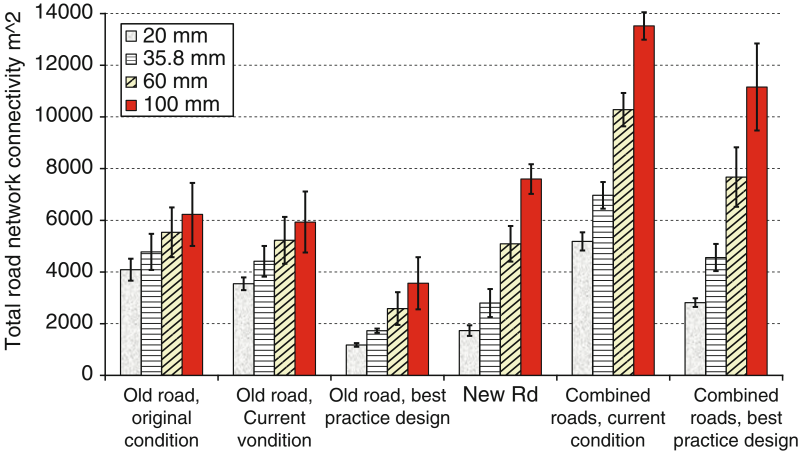


Fig. 22.6 Summary of connectivity variations in relation to rainfall amount and road management scenario (Modified from Eastaugh et al. 2008)

When temperatures are below zero precipitation comes down as snow stored in a snowpack over winter. Water can sublimate from the snow but mainly evaporates from the soil and from the canopy. Water that does not evaporate from the canopy until the evening is also added to the soil water pool. Outflow is calculated as the difference between the amount of water that enters the soil and the amount of water that can be kept when the soil is fully saturated. In addition, there is slow leakage that consists of the difference between the amount of soil water at full saturation and the water that can be held at field capacity. The water regime, therefore, is a very dynamic system influenced by several stand and site characteristics.

Water use efficiency (WUE) is a plant characteristic that describes how much water is transpired to achieve a particular amount of Net Primary Production (NPP). The WUE is given in g C m^{-2} $year^{-1}$ mm^{-1}. A lower NPP/water transpiration ratio can be seen as negative for the plant because it needs more water for achieving the same amount of NPP. This has the negative implication that the danger of water stress is higher and so water can become a limiting factor earlier. Figure 22.7 shows WUE in comparison to other stand and topographic variables.

Outflows are of course strongly dependant on precipitation, which is higly variable at an annual scale. To smooth some of the this variability Pötzelsberger (2008) divides annual runoff (summed from daily data) by precipitation. Figure 22.8 compares the LAI of two differently managed plots with annual runoff/precipitation over a 45-year period.

Figure 22.9 shows the correlation between the modeled and observed catchment outflows from the Schmittenbach runoff gauging station from 1981 to 2005. Correlation appears reasonable until 1997. Possible causes of error include: artificial snowmaking with lake water (areal spatial scope issue), deep drainage losses (vertical spatial scope issue), incorrect assumptions for outflow coefficients in non-forest areas (downscaling issue) or uncertainties in outflow modeling.

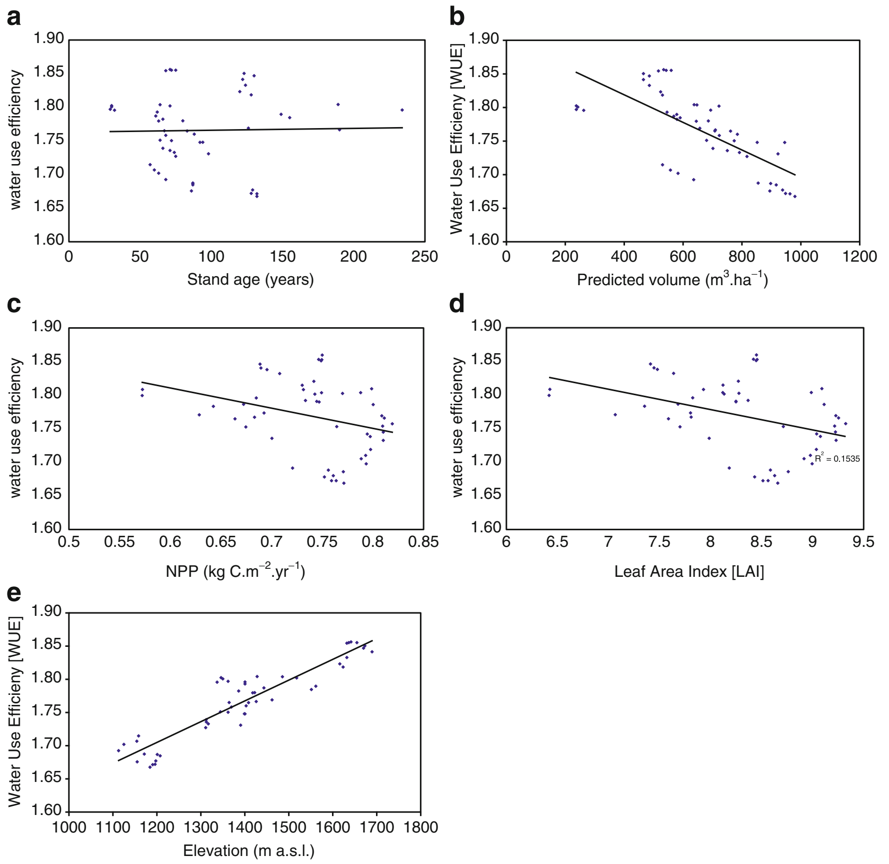


Fig. 22.7 Water use efficiency (g C m^{-2} year^{-1} mm^{-1}) versus Stand age (**a**), predicted volume (**b**), net primary production (**c**), LAI (**d**) and elevation (**e**) for all plots in 2005

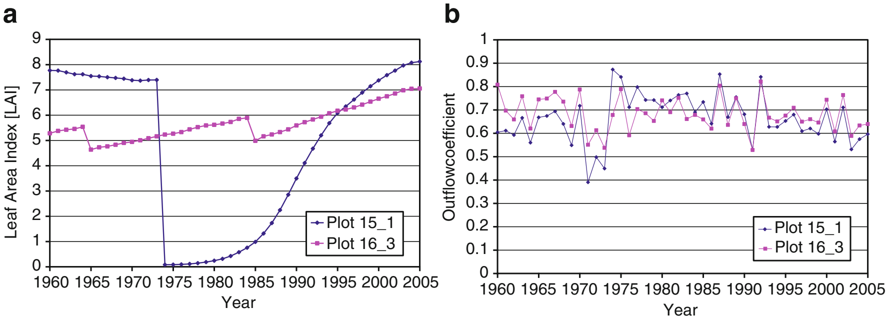


Fig. 22.8 Development of the leaf area index (**a**) and the annual outflow/precipitation ratio (**b**) in the years 1960–2005 for two different plots showing all data of the years 1960–2005. Plot 15_1 was clear-cut with 100 years in 1974 with subsequent planting. Plot 16_3 is an old growth stand with 145 years of age in 1960 (thinning of 30% was simulated twice during the observation time)

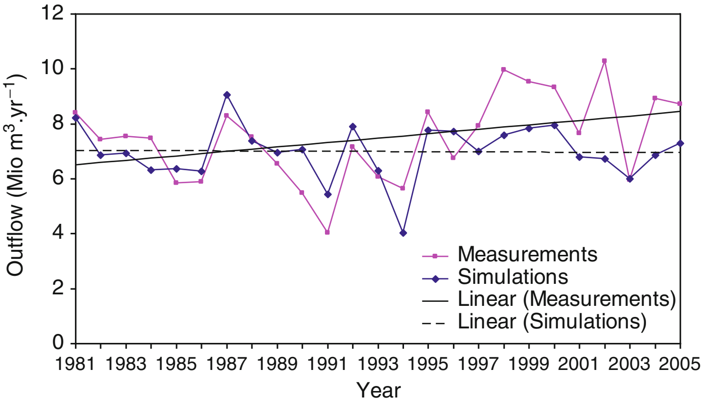


Fig. 22.9 Correlation between modeled and observed catchment outflows from 1981 to 2005

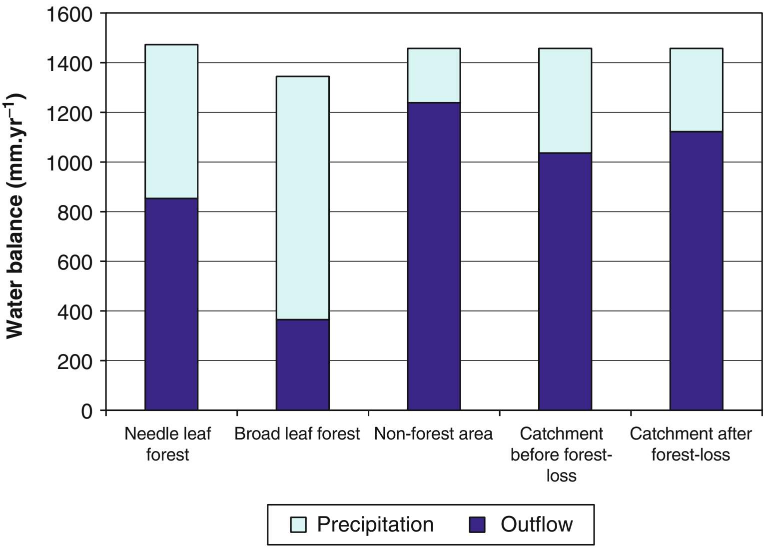


Fig. 22.10 Water balance of different forest types and conditions

Modeling from the Schmittenbach study was then applied to a management question regarding the protective functions of forest in the Kaprun watershed (Fig. 22.10, Pötzelsberger et al. 2009). An additional 3 million litres of outflow was modelled for the year 2005 (an extra 8%) due to forest loss from wind throw and beetle infestation. This has implications for flood control and slope stability.

In this example daily precipitation was modeled into daily forest growth. Results were summed and smoothed to an annual, catchment scale for model validation. The validated technique was then applied to a separate catchment for the modelling of one-time events (wind throw, beetle damage) on longer-term processes (catchment runoff).

The studies presented in this example demonstrate the impact of vegetation cover on the water cycle at the catchment scale, and show how appropriately scaled mechanistic modeling may be used to derive important management principles and inform management actions. An examination of the sensitivity of catchment vegetation cover to possible disturbance events will allow a better assessment of how the protective function of forests may be compromised, thus providing necessary inputs to risk analysis in these areas.

Drought Index

Apart from the direct impact of low precipitation on growth, forest management may also be impacted through an increase in fire risk resulting from long periods without rainfall. This example shows the use of appropriately scaled climate data in informing fire risk assessments.

The risk of an outbreak of wildfire is strongly influenced by fine fuel moisture, which may be modeled by the Keetch–Byram Drought Index (KBDI, Keetch and Byram 1968; Burgan 1988). Prolonged dryness (high KBDI) influences the intensity of a potential fire due to the missing moisture of the fuel. The value of the drought index ranges from 0 (no drought) to 800 (extreme drought) and is based on estimates of the soil water holding capacity. KBDI is actively used in the Wildland Fire Assessment System (WFAS; http://www.wfas.net) of the United States Forest Service (USFS).

In calculating of the index the value of the previous day is on the one side decreased by the amount of precipitation and on the other side increased by the potential evaporation from the soil. For precipitation a buffer of 0.508 cm (0.2 in.) is used. The factor for drying-out (dq_i) is affected by the index of the previous day (Q_{i-1}), the maximum temperature ($tmax_i$) and the long-time mean of the annual precipitation (R_{avg})

$$dq_i = (800 - Q_i)\frac{0.000968e^{0.08748t\max_i - 6.7448}}{1 + 10.88e^{-0.112014R_{avg}}}. \tag{22.9}$$

In the above formula temperature and precipitation are used in degree Celcius and centimetres as the parameters of the original formula (given in degree Fahrenheit and inches) were adjusted, accordingly.

A typically hot summer gives a high fire danger levels in the eastern part of Austria whereas during quite cool summers the fire danger (defined here as the integer value of KBDI/100) did not exceed level 3 for the whole of Austria. Likewise, the extreme drought in summer 2003 is well reproduced with very high fire danger levels for large parts of Austria (Fig. 22.11). Between 1960 and 2008, 915 (~5.1%) days were recorded with fire danger level 6 or higher and 53 (~0.3%) with fire danger level 8. Figure 22.12 plots the KBDI for a given point for each day of the year in 2003 and 2005, a year more representative of average conditions.

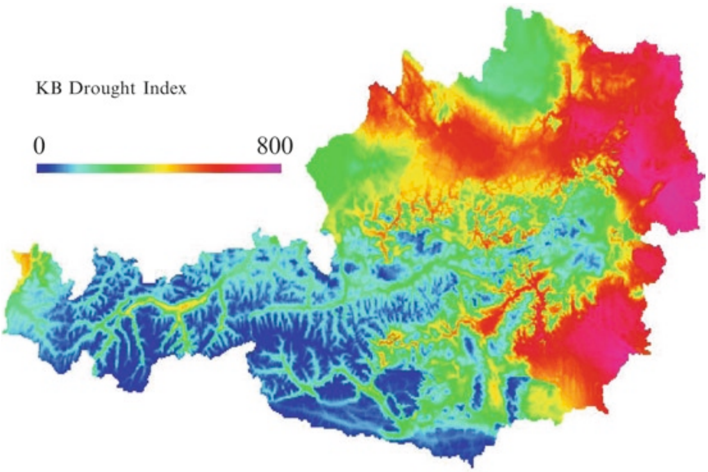


Fig. 22.11 KBDI at 1 km^2 resolution for an August day in 2003

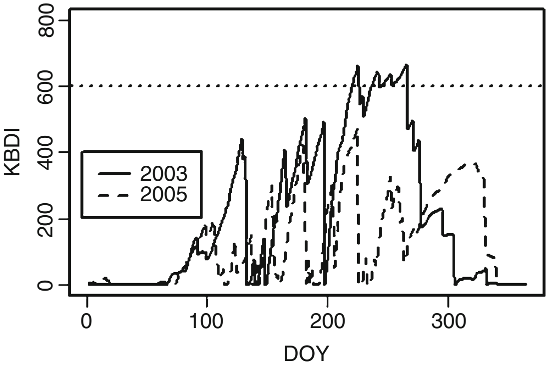


Fig. 22.12 Comparison of daily KBDI between the abnormally hot and dry 2003 and a more average 2005

Climatic data was sourced from the DAYMET model, which can scale down the nation-wide climate station network (roughly 500 km^2 resolution) to a 1 km^2 regular grid Petritsch and Hasenauer (2009). Validation of the interpolation process showed a mean absolute error nation wide of 1.17° for minimum temperature, 1.01° for maximum temperature and 3.0 mm for precipitation (Hasenauer et al. 2003). Respective mean errors however were only 0.00°, –0.01° and 0.1 mm, suggesting that subsequent upscaling of interpolated data can reduce errors to practically negligible levels.

Modeling was done with a daily time step. The available climate inputs from DAYMET match the required inputs for calculating the KBDI, which in turn match a useful reporting period. This procedure is not possible with coarse temporal scale data, and accuracy would be greatly reduced with data at a coarser spatial scale.

The spatial scale of these results is large compared to the scale of wildfire events in Austria, but as a means of informing a daily risk assessment of fire danger across regions it is useful. It is however important to realize that this (comparatively) large scale modeling must be used in combination with fine scale, site specific local details to give locally relevant information. Non-climatic factors such as topography, vegetation cover and type and ignition sources must also be considered.

Conclusions

Water relations are fundamental to forest growth, and will influence the development of forest structures at all levels from the cellular to the catchment scale, and at time scales from the micro to the inter-annual. Similarly, forests have a large bearing on water-related processes, and thus forest management issues must be considered in water modeling exercises.

This chapter has outlined the links between physiological processes in forest and catchment hydrological issues, and demonstrated how appropriate scaling is commonly required. Scaling may variously be needed to represent continuously fluctuating processes as discrete values, to include long-scale processes in short time-step models, or to develop fine scale, short step data into catchment wide annual results and simulate the impact of one-off events.

The examples presented in this chapter have demonstrated the application of scaling techniques to groundwater influences on forest growth and processes, water runoff from forest roads, catchment level water yield modeling and wildfire risk assessments. In each case, processes have been variously upscaled or downscaled in order to provide an appropriate and meaningful means of representing each process in the modeling environment. Judicious scaling can reduce the error of data derived through statistical procedures such as probabilistic modeling or geostatistical interpolations, increasing the overall confidence in the results.

Acknowledgments Work presented in this chapter was compiled specifically for COST Action FP 0601 ‘Forest Management and the Water Cycle’, with individual examples drawn from work at BOKU University of Natural Resources and Applied Life Sciences Institute of Silviculture (Vienna), the Austrian Forest Fire Research Initiative (AFFRI) and the Commonwealth Scientific and Industrial Research Organisation Land and Water Division (Canberra).

References

Bachelet D, Lenihan J, Daly C, Neilson R, Ojima D, Parton W (2001) MC1: a dynamic vegetation model for estimating the distribution of vegetation and associated ecosystem fluxes of carbon, nutrients and water. USDA Forest Service, Pacific Northwest Station, General Technical Report PNW-GTR-508

Beltrán-Przekurat A, Marshall CH, Pielke RA Sr (2008) Ensemble reforecasts of recent warm-season weather: impacts of a dynamic vegetation parameterization. J Geophys Res vol.113, 18 pp

Bond-Lamberty B, Peckham SD, Ahl DE, Gower ST (2007) Fire as the dominant driver of central Canadian boreal forest carbon balance. Nature vol.450:89–93

Burgan RE (1988) 1988 Revisions to the 1978 National Fire Danger Rating System. Res. Paper SE-273. U.S. Department of Agriculture, Forest Service, Southeastern Forest Experiment Station, Asheville, NC

Carbone G (2007) Exercises in weather and climate. Prentice Hall, Upper Saddle River, NJ

Cermak J, Deml M, Penka M (1973) A new method of sap flow rate determination in trees. Biol Plantarum vol.15:171–178

Cermak J, Ulehla J, Kucera J, Penka M (1982) Sap flow rate and transpiration dynamics in the full-grown oak (*Quercus* robur L.) in floodplain forest exposed to seasonal floods as related to potential evapotranspiration and tree dimensions. Biol Plantarum vol. 24:446–460

Chang M (2003) Forest hydrology: an introduction to water and forests. CRC Press LLC, Boca Raton, FL

Eastaugh CS, Rustomji PK, Hairsine PB (2008) Quantifying the altered hydrologic connectivity of forest roads resulting from decommissioning and relocation. Hydrol Process 22(14):2438–2448

Flanagan DC, Nearing MA (eds) (1995) USDA-water erosion prediction project: hillslope profile and watershed model documentation. NSERL Report #10, USDA-ARS National Soil Erosion Research Laboratory, West Lafayette, IN

Hasenauer H, Merganicova K, Petritsch R, Pietsch SA, Thornton PE (2003) Validation of daily climate interpolations over complex terrain in Austria. Agric For Meteorol vol.119:87–107

Hairsine PB, Croke JC, Mathews H, Fogarty P, Mockler SP (2002) Modelling plumes of overland flow from logging tracks. Hydrol Process vol.16:2311–2327

Jones PG, Thornton PK (1999) MarkSim: software to generate daily weather data for Latin America and Africa. Agron J vol.92:445–453

Keetch JJ, Byram G (1968) A drought index for forest fire control. Res. Paper SE-38. U.S. Department of Agriculture, Forest Service, Southeastern Forest Experiment Station, Asheville, NC

Krinner G, Viovy N, de Noblet-Ducoudré N, Ogée J, Polcher J, Friedlingstein P, Ciais P, Sitch S, Prentice C (2005) A dynamic global vegetation model for studies of the coupled atmosphere-biosphere system. Global Biogeochem Cycles vol.19, 33 pp

McCarthy MA, Lindenmayer DB (1998) Multi-aged mountain ash forest, wildlife conservation and timber harvesting. For Ecol Manage vol.104:43–56

Mellilo JM, McGuire AD, Kicklighter DW, Moore B III, Vorosmarty CJ, Schloss AL (1993) Global climate change and terrestrial net primary productivity. Nature vol.363:234–240

Merganičová K, Pietsch SA, Hasenauer H (2005) Testing mechanistic modeling to assess impacts of biomass removal. For Ecol Manage vol.207:37–57

Mohren GMJ, Burkhart HE (1994) Contrasts between biologically-based process models and management-oriented growth and yield models. For Ecol Manage vol.69:1–5

Novenko EY, Glasko MP, Burova OV (2009) Landscape-and-climate dynamics and land use in late Holocene forest-steppe ecotone of East European Plain (upper Don River case study). Quatern Int vol.203:113–119

Oh JH, Kim T, Kim MK, Lee SH, Min SK, Kwon WT (2004) Regional climate simulation for Korea using dynamic downscaling and statistical adjustment. J Meteorol Soc Jpn 82(6):1629–1643

Pacala SW, Canham CD, Saponara J, Silander JA, Kobe RK, Ribbens E (1996) Forest models defined by field measurements: estimation, error analysis and dynamics. Ecol Monogr 66(1):1–43

Petritsch R, Hasenauer H (2007) Interpolating input parameters for large scale ecosystem models. Aust J For Sci – Centralblatt für das gesamte Forstwesen vol.124:135–151

Petritsch R, Hasenauer H (2009) Tägliche Wetterdaten im 1 km Raster von 1960 bis 2008 für Österreich. Austrian Journal of Forest Science 126(4):215–225

Pietsch SA, Hasenauer H (2002) Using mechanistic modeling within forest ecosystem restoration. For Ecol Manage vol.159:111–131

Pietsch SA, Hasenauer H, Kučera J, Čermák J (2003) Modeling effects of hydrological changes on the carbon and nitrogen balance of oak in floodplains. Tree Physiol vol.23:735–746

Pötzelsberger E (2008) Assessing the productivity and water regime in the Schmittenbach catchment area. Master thesis at the University of Natural Resources and Applied Life Sciences, Vienna

Pötzelsberger E, Hasenauer H, Petritsch R, Pietsch SA, Quehenberger C (2009) Assessing the protection function of Alpine forest ecosystems using BGC modelling theory. In: European Geoscience Union (eds) Geophysical Research Abstracts, vol. 11, EGU 2009-12049

Rhee H, Fridley JL, Foltz RB (2004) Modeling erosion from unpaved forest roads at various levels of geometric detail using the WEPP model. Trans ASABE 47(3):961–968

Running SW, Hunt ER Jr (1993) Generalization of a forest ecosystem process model for other biomes, BIOME-BGC, and an application for global-scale models. In: Ehleringer JR, Field CB (eds) Scaling physiological processes: leaf to globe. Academic, New York

Saxton KE, Rawls WJ, Romberger JS, Papendick RI (1986) Estimating generalized soil-water characteristics from texture. Soil Sci Soc Am J vol.50:1031–1036

Sitch S, Smith B, Prentice IC, Arneth A, Bondeau A, Cramer W, Kaplan JO, Levis S, Lucht W, Sykes MT, Thonicke K, Venevsky S (2003) Evaluation of ecosystem dynamics, plant geography and terrestrial carbon cycling in the LPJ dynamic global vegetation model. Global Change Biol vol.9:161–185

Thornton PE (1998) Description of a numerical simulation model for predicting the dynamics of energy, water, carbon and nitrogen in a terrestrial ecosystem. Ph.D. thesis, University of Montana, Missoula, MT

Thornton PE, Running SW, White MA (1997) Generating surfaces of daily meteorological variables over large regions of complex terrain. J Hydrol vol.190:214–251

Thornton PE, Law BE, Gholz HL, Clark L, Falge E, Ellsworth DS, Goldstein AH, Monson RK, Hollinger D, Falk M, Dhen J, Sparks JP (2002) Modelling and measuring the effects of disturbance history and climate on carbon and water budgets in evergreen needle leaf forests. Agric For Meteorol vol.113:185–222

Wang E, Zhang L, Cresswell H, Hickel K (2006) Comparison of top-down and bottom-up models for simulation of water balance as affected by seasonality, vegetation type and spatial land use. In: Voinov A, Jakeman AJ, Rizzoli AE (eds) Proceedings of the iEMSs third biennial meeting: "summit on environmental modelling and software". International Environmental Modelling and Software Society, Burlington, USA, July 2006

Wemple BC, Jones JA, Grant GE (1996) Channel network extension by logging roads in two basins, western Cascades, Oregon. Water Resour Bull 32(6):1195–1207

Wilby RL, Charles SP, Zorita E, Timbal B, Whetton P, Mearns LO (2004) Guidelines for use of climate scenarios developed from statistical downscaling methods. Supporting material for the intergovernmental panel on climate change. http://www.ipcc-data.org/guidelines/dgm_no2_v1_09_2004.pdf. Accessed 10 June 2009

Wilks DS (1999) Simultaneous stochastic simulation of daily precipitation, temperature and solar radiation at multiple sites in complex terrain. Agric For Meteorol vol.96:85–101

Zeide B (2003) The U-approach to forest modeling. Can J For Res vol.33:480–489

Zhang L, Dawes WR, Walker GR (2001) Response of mean annual evapotranspiration to vegetation changes at catchment scale. Water Resour Res 37(3):701–708

Zhicheng Z (1995) Basic features of forest steppe in the loess plateau of China. Chin Geogr Sci 5(2):170–174

Chapter 23
A Small Catchment Scale Approach for Modeling Effects of Forest Management on Water Cycle in Boreal Landscape

Harri Koivusalo, Hannu Hökkä, and Ari Laurén

Introduction

Quantitative assessment of forest hydrological processes deals with a range of spatial and temporal scales. Mismatch of spatial scales between hydrological modelling, measurements, and processes is well recognized and studied in literature (e.g. Blöschl and Sivapalan 1995). Scaling approaches are introduced, for example, to link point measurements to area-specific predictions of hydrological models (Western and Blöschl 1999), to identify dependencies between remotely sensed information and hydrological variables within a catchment (Oldak et al. 2002; Scipal et al. 2005), and to recognize scale-dependencies in the runoff generation mechanisms (Vivoni et al. 2007). One of the scale issues is the spatial extent of forest management practices and their environmental effects. Forest treatments and water protection measures used in forestry cover only small part of the catchment, but have hydrological impacts at the catchment level. The question of temporal scales becomes important in forested areas, because the short time scales of hydrological processes greatly differ from the long time scale of the tree stand rotation.

Landscape in many regions in Europe is composed of mosaic structured land surface elements, including different land use forms, soils of different types, and areas with different topographic features and distances to water courses.

H. Koivusalo (✉)
Department of Civil and Environmental Engineering, Aalto University School of Science and Technology, FI-00076 Aalto, Finland
e-mail: harri.koivusalo@tkk.fi

H. Hökkä
Finnish Forest Research Institute, Rovaniemi Research Unit, FI-96301
Rovaniemi, Finland
e-mail: hannu.hokka@metla.fi

A. Laurén
Finnish Forest Research Institute, Joensuu Research Unit, FI-80101
Joensuu, Finland
e-mail: ari.lauren@metla.fi

M. Bredemeier et al. (eds.), *Forest Management and the Water Cycle: An Ecosystem-Based Approach*, Ecological Studies 212, DOI 10.1007/978-90-481-9834-4_23,

Classification of landscape patterns is often introduced to identify areas, which can be regarded as similar in terms of their geological, hydrological, or ecological characteristics (Becker and Braun 1999; Maréchal and Holman 2005; Schröder 2006). Distribution of soil types, soil depths, and landscape topography form the basis for the classification of geological properties (e.g. Klingseisen et al. 2008). The concept of hydrological similarity is applied to classify areas, where runoff generation mechanisms and hydrological response to rainfall or snowmelt events is comparable (e.g. Schneider et al. 2007). Classification of ecological characteristics can be based on the size, shape, and successional stage of communities such as forest stands composing of different tree species (e.g. Wu and Levin 1997). Based on the classification schemes, landscape is disaggregated into a set of basic units, which can be called, e.g., as geotopes, hydrotopes, ecotopes, patches, hydrologically similar areas, hydrological response units, or characteristic profiles. Quantification of hydrological processes can be implemented separately for each landscape unit using empirical, conceptual, or process-based hydrological models.

From the scale viewpoint the use of the classification approach means that integration of the landscape units within the catchment is necessary. Integration of the units through a channel network is customary in hydrological modeling (Ott et al. 1991; Karvonen et al. 1999; Maréchal and Holman 2005). The hydrological response from the landscape units forms runoff input to the network of channels and reservoirs, where conventional flow routing and delay procedures can be applied. This approach is well suited to boreal conditions, where relatively impermeable bedrock is typically covered with shallow layers of organic soil and till, and where the longest distance from a point in the landscape to a stream ranges from the order of tens of meters up to few thousands of meters. In such landscape the hydrological response units (HRU) can be assumed to be independent and the channel network is the only hydrological pathway connecting the areas within a catchment.

While managed forests with different soils, tree species, and succession stages form one set of land use types, agricultural areas, urban constructed areas, wetlands, and natural protected areas are additional land use types. The high diversity of land use is a challenge for hydrological modelling in densely inhabited regions in many parts of Europe. The disaggregation of the landscape into HRUs becomes difficult, when the mosaic pattern of the landscape is diverse and the patterns are hydrologically interconnected. While the HRU approach can be applied for characterizing hydrological processes across various land use types, the main focus of this presentation is on managed boreal forests.

Managed forests in the boreal region are growing on both peatlands and mineral soils. Hydrology plays an important role in the forest management from two different aspects. First, water storage in soil is a factor controlling growth and productivity of the tree stand, which is an important practical management issue in drained peatland forests. Second, environmental effects of forest management are realized in both peatland and upland forests through generation of runoff that carries excess loads of nutrients and sediment from treated forest areas to receiving water bodies. The typical forested area in the boreal region is a mosaic of tree stands with patches having relatively homogeneous species composition and age. Management of sufficiently large areas of well-maintained stands is a pre-requisite to economically sustainable forestry. The objective is to treat the entire stand and avoid subdivision of the stand into several management units. The forest growth, forest management practices, and environmental effects of forest treatments can be described in the HRU approach, where the scale of the HRU is matched with the scale of the management units.

While the management issues in the boreal region are mainly related to wood production and water quality of lakes and streams, forest management in other regions of Europe is motivated by different targets that are influenced by local interests and climatic conditions. For example, optimisation of water resources availability is a key objective of forest management in many dry areas in southern Europe (Callegari et al. 2003), restoration and conservation of riverine forests and peatlands are widely applied to improve microclimate, biodiversity, and biogeochemical cycles in floodplains (Naiman et al. 1998), and protection forests are introduced to prevent snow avalanches and rock falls in mountainous areas (e.g. Teich and Bebi 2009).

This section demonstrates how a forest growing on peatland or mineral soil in the boreal region can be parameterized as HRU, and how water cycle and environmental effects of forestry can be described. Specifically in peat soils both forest productivity and environmental impacts are directly regulated by changes in hydrology, which needs to be addressed in sustainable forest management. The aim is to outline a model structure that allows for computation of main hydrological variables in a scale comparable with the scale of the hydrological observations. Discussion is given on issues related to merging hydrological models and forest management models, which describe the forest growth over the stand rotation and the management impacts on forest development. The modeling approach introduced in this section is implemented in FEMMA, which is a hydrological and nitrogen transport model (Laurén et al. 2005; Koivusalo et al. 2006). The forest stand development is simulated with MOTTI, which is a stand simulator characterizing forest management impacts on forest development over the full rotation of the stand (Salminen et al. 2005; Hynynen et al. 2002).

Catchment Disaggregation in Hydrological Models

Hydrological models can be categorized on the basis of their process description or on the basis of their spatial description (Refsgaard 1997). Hydrological processes inherent in the water cycle include, e.g., precipitation, interception, evaporation, transpiration, snow accumulation and melt, soil and ground water movement, overland flow, and channel flow. The importance of each process depends on the catchment and climatic characteristics, as well as on the spatial and temporal scales.

In simulation models, the processes are always more or less simplified compared with the reality, and individual processes become often ignored or embedded with others. Differences in these choices lead to hydrological models that are different in their process description. From the process description viewpoint the models can be loosely classified as metric, conceptual, or physics-based (e.g. Koivusalo 2002).

Hydrological models are often categorized from the spatial description point of view as lumped or distributed models. Lumped models describe a catchment as a single unit, produce an average water balance of the catchment, and ignore spatial variability of hydrological processes. Distributed models account for spatial variability of hydrological state variables and fluxes, and are predominantly physics-based. In distributed models the spatial variability of hydrological processes can be described within the resolution of a finite grid, where the parameter values must be known in all grid cells. Such an extensive parameterization over a catchment is demanding in terms of computational burden and data acquisition (Grayson et al. 1992; Woolhiser 1996). When a gridded modelling domain is applied, the values of the model parameters and state variables depend on the grid resolution, which easily leads to a mismatch between modelling and observation scales.

Semi-distributed modelling approaches are introduced to account for spatial variability of hydrological processes and at the same time, alleviate the burden of the distributed models. The semi-distributed models share the assumption that spatial variability in key hydrologic properties can be described in a condensed form. Once the key hydrological properties and their drivers are identified, a catchment can be classified into sub-areas or units as introduced in earlier studies (e.g. Becker and Braun 1999; Flügel 1995; Kite and Kouwen 1992). In such a framework, a water balance model is implemented for each identified unit.

Ott et al. (1991) outlined a hydrological model that consisted of two components: the flood routing model of the stream network up to the subcatchment level, and a water balance model producing runoff that enters the stream network from each subcatchment. Gridded data at the subcatchment level was exploited to aggregate grid cells together to form ‘hydrologically similar units’ (HSU). The classification was based on physical subcatchment characteristics, such as vegetation cover, soil type, meteorological zone, elevation, slope, and exposition. In the methodology of Ott et al. (1991) a HSU can be non-contiguous, i.e., grid cells belonging to a certain HSU can be located in different unconnected parts of a subcatchment. The spatial disaggregation of a catchment has two steps: (1) from the catchment level to subcatchment level and (2) from subctachment level to the HSU level. Flügel (1995), Krysanova et al. (1998), and Karvonen et al. (1999) adopted a similar approach in their models.

The disaggregation of a catchment into subcatcments can be conducted on the basis of river network properties derived from a digital elevation model. Zhang and Savenije (2005) outlined a concept of Representative Elementary Watershed (REW), where the catchment subdivision was based on the Strahler stream order system. A hydrological model was formulated in each REW, and the REWs characterising subcatchments were allowed to be interconnected through both ground water aquifer and stream network.

The identification of patches, HRUs, or hydrotopes from gridded data within a subcatchment depends on the scale of the input data as pointed out by Lin et al. (2008). They studied the scaling effects on simulated land use patterns and hydrological components and their results demonstrated that changes in the grain size (resolution) of spatial input data were reflected to simulated hydrological components.

While the issue of scale is an important aspect in hydrological applications, the scaling between different metrics, processes, and levels remains a great challenge. Kummu (2008) presented hydrological impact assessments in large river basin of Mekong across different scales and noted that a multi-scale approach, where the processes and areas are concurrently explored in different scales, can often be a fruitful approach even without attempts to fully describe the linkages between different scales.

HRU for the Peatland Forest

Physics-based models have long been available for simulating water and energy exchange in a vertical one-dimensional soil column covered with vegetation (e.g. Feddes et al. 1978; Jansson and Halldin 1979; Karvonen 1988; Jansson and Karlberg 2001). A separate description of the tree canopy is embedded in these types of models, and it allows for quantification of canopy effects on main hydrometeorological processes such as radiation transfer, wind dissipation, evapotranspiration, and snow accumulation and melt (Wigmosta et al. 1994; Gustafsson 2002; Koivusalo 2002). The illustration in Fig. 23.1a depicts a soil column, which is subdivided into computation layers and overlain by overstorey vegetation (tree stand) and understorey vegetation (ground vegetation and field layer) or seasonal snow cover. The main vertical

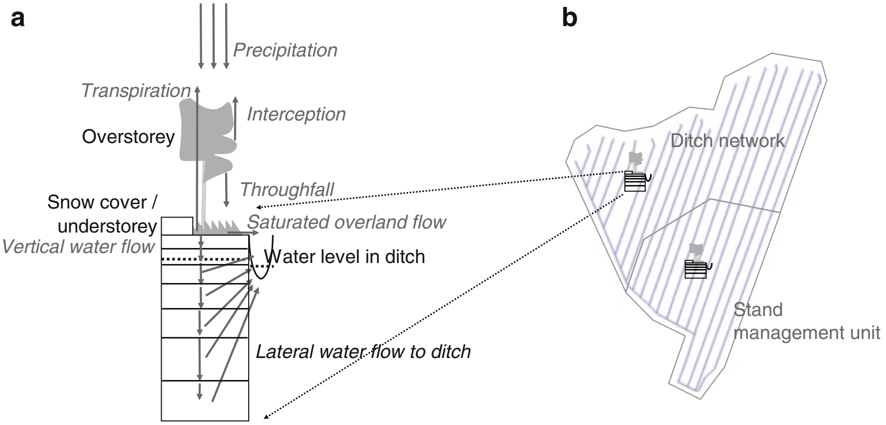


Fig. 23.1 Hydrological response unit (HRU) characterising ditch-drained tree stand management unit (**a**) and subdivision of ditch network area into stand management units (**b**). *Arrows* in HRU (**a**) characterise water fluxes and storages in drained peatland covered with overstorey (tree stand) and understorey, which can be covered with snow during winter. Ditch network area in (**b**) has two stand management units and each unit is represented with its own HRU

hydrological processes described are canopy interception and evapotranspiration in two vegetation layers, snow accumulation and melt, infiltration, and percolation in soil. The lateral soil water movement out of the soil column occurs in the saturated part of the soil and surface runoff out of the modelling domain is generated in situations when the soil column is fully saturated with water.

The vertical one-dimensional model can be coupled with a drainage flow scheme that accounts for the loss of water from the soil profile to the drainage ditch (see Fig. 23.1a). The lateral water flow to the drainage ditches is treated as a sink in the soil layers that are located in the saturated part of the soil column. In the parameterization depicted in Fig. 23.1a the hydrological model idealises the processes in a domain between two drainage ditches and forms the HRU for a homogenous forested patch located in a ditch-drained area with a flat topography (Fig. 23.1b). Each patch that is classified to have different soil and/or stand properties is assigned with its own HRU. Drainage flow and surface runoff into the ditch network are the two runoff components that form an input to a channel network routine characterizing the flow delay in the ditch/stream network.

Hydrological measurements conducted in isolated artificial catchments inside the ditch network can be exploited to calibrate and validate the peatland HRU (Fig. 23.2) (Koivusalo et al. 2008). Some hydrological measurements can be compared with the model results as such. For example, runoff aggregates the response of the total catchment area, whereas point measurements, such as throughfall gauge observations, snow cover observations, and sap flow readings need to be scaled to the level of the area between two ditches, which corresponds to the

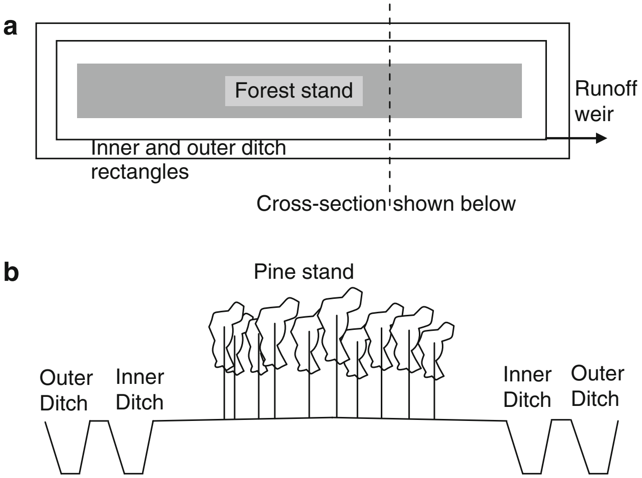


Fig. 23.2 Layout (**a**) and cross-section (**b**) of ditch-delineated artificial catchment for measuring water balance components in HRU of a ditch-drained forested area. The drained peatland forest in (**a**) is isolated with outer ditches from the surrounding area to collect and measure runoff from the stand area. Terrestrial hydrological variables, such as snow depth, throughfall and transpiration can be measured within the stand (Koivusalo et al. 2008)

HRU scale. The application of a process-based hydrological model binds point and catchment scale field measurements together and provides quantitative understanding of the hydrological processes at the level of the landscape unit.

The presented approach for simulating hydrological processes in drained areas has been applied in agricultural areas (e.g. Singh et al. 2007). Open ditches or subsurface drains and their impact on water table levels, soil moisture conditions, and runoff generation are processes occurring in the distance scale corresponding to the drain spacing. To some extent the ideas of peatland HRUs can be exploited in the models for peat extraction areas and urban areas, where sewage systems are applied to convey water and drain constructed sites. HRU for the drained area is less applicable to undrained areas, such as natural peatland areas, restored peatlands, and wetlands.

Water Cycle and Forest Management in Boreal Peatlands

Due to the climatic conditions in the boreal region, much of the practical management methods in peatland forests aim at reducing excessively high levels of soil moisture by improving tree growth with ditching and site preparation. Altering the site hydrological conditions by ditching has three long-term consequences: (1) decrease of water table in soil increases stand growth rate (e.g. Seppälä 1969), (2) improved soil aeration increases the rate of organic matter decomposition and changes the nutrient cycle in soil, and (3) ditching operations expose ditch sidewalls and bottom, as well as excavator tracks to erosion and detachment of soil particles and organic material.

The effect of ditching and ditch cleaning on tree growth is described in forest stand simulator MOTTI (Hynynen et al. 2002), which can be applied to simulate the production of stand-scale stem volume under a variety of forest management scenarios. MOTTI describes the stand growth responses to management practices – such as thinning and ditch cleaning – using relationships identified from empirical field inventory data. No feedback mechanisms between the actual hydrological conditions of a site and tree growth are included in the model (Hökkä and Salminen 2006).

Growth of trees in drained peatland may decrease or even increase without active management of site hydrology. Decreased growth may take place during a short period of time, e.g., when drainage network becomes blocked, or during a longer time span when ditches gradually deteriorate. In addition to runoff, evapotranspiration is an important loss term in the water balance especially during the growing season (Ahti and Hökkä 2006). Evapotranspiration as such alleviates wet soil moisture conditions and can therefore enhance tree growth if the evaporating biomass is large enough (Hökkä et al. 2008). These complex effects are not well accounted for in the present parameterisation of MOTTI (Hökkä and Salminen 2006).

Better understanding of peatland tree stand water use in the growth simulation models would permit segregation of sites and stands that need ditching operations to support their further development from those in which natural water use by the tree stand is sufficient to compensate for the deterioration of ditches. Conversely, the ditch cleaning operation may be unnecessary in highly stocked stands in spite of the poor condition of ditches. When unnecessary ditch cleaning is avoided, the impacts on the water environment become minimised. Joensuu (2002) notes that the sediment load caused by the maintenance of ditch networks is one of the main harmful environmental effects of forestry. The sediment loads primarily originate from the ditch network especially in the case when the ditches in peatlands reach the mineral soil beneath the surface peat layer. In the hydrological modelling framework, erosion, sediment transport, and deposition are processes related to the ditch network routing model, and not to HRU. The generation of nutrient loads that are induced by the changed soil moisture conditions after ditching, need to be modelled at the HRU level.

Since environmental loads from forestry are directly linked to the transport of water, understanding of water cycle is crucial in developing environmentally sustainable management practices in boreal peatlands. One step in achieving this goal is to combine models that describe the hydrological processes with models describing the stand growth process and effects of management on growth. This link can be built up at forest HRU levels and thus accounted for in catchment level calculations. Ultimately, improved description of within-stand water fluxes improves the predictions of hydrological models that are used to estimate the environmental consequences of forest management in peatlands.

Pristine peatlands are no longer drained for forestry in northern Europe. Presently, the impact of ditch network maintenance on tree stand and environment is an important management question in forestry. Apart from the drained peatlands there are different management issues arising in the restoration of unproductive forested peatlands, conservation areas, and isolated peatland areas used as treatment buffers between the managed forest and the water course. The peatland HRU presented in this section is not suited for characterising hydrology of these land use types. The distance to stream in undrained areas is longer than the typical scale of the ditch spacing, and therefore, HRU needs to account for lateral flow processes through longer distances and different site characteristics. HRU approach for characterising such areas is presented in the next section.

HRU for the Upland Forest

The one-dimensional model (Fig. 23.1a) is suitable for describing hydrological processes in an area where the distance to the closest surface water pathway (ditch or stream) is relatively short (in the order of 10–100 m). However, the one-dimensional system is not sufficient for simulation in a small catchment-scale, where the mean distance to a receiving water body is far higher than in the drained area (up to several 1,000 m) and where the topography of the terrain may vary. The simplest domain that accounts for lateral distance to a receiving water body is a hillslope that extends from a water divide to a stream. Kirkby (1985) outlined the concept of a hillslope for describing runoff generation mechanisms at different distances from a stream. A vertical two-dimensional hillslope can be used as a HRU for modeling a subcatchment area, where the hydrological characteristics primarily change as a function of distance to the stream (see Fig. 23.3b). The hillslope is subdivided laterally into computational columns (Fig. 23.3a), which are similar to the HRU for the drained area in Fig. 23.1a. Each predefined subarea within the catchment can be described as one HRU (Fig. 23.3c). Surface and subsurface runoff components discharged out of the HRUs form an input to the stream network, which separately accounts for the channel flow delays.

Kokkonen et al. (2001) and Laurén et al. (2005) outlined a methodology for identifying the shape of a hillslope from the spatial catchment data on elevation and stream network location. The identification of a typical hillslope was based on analysing digital elevation data and producing flowpaths from each grid cell to a receiving stream cell. The receiving grid cell (stream) was found by following the flow accumulation along the path with the steepest slope. The elevation of the grid cells above the receiving stream cell was averaged at different distances to the stream cell and the average elevation as a function of distance defined the topographic shape of the hillslope (see Fig. 23.4a).

Hydrological measurements conducted in small catchments can be exploited to calibrate and validate HRU. Measured runoff from a small catchment can be compared against the total runoff from the identified typical hillslope.

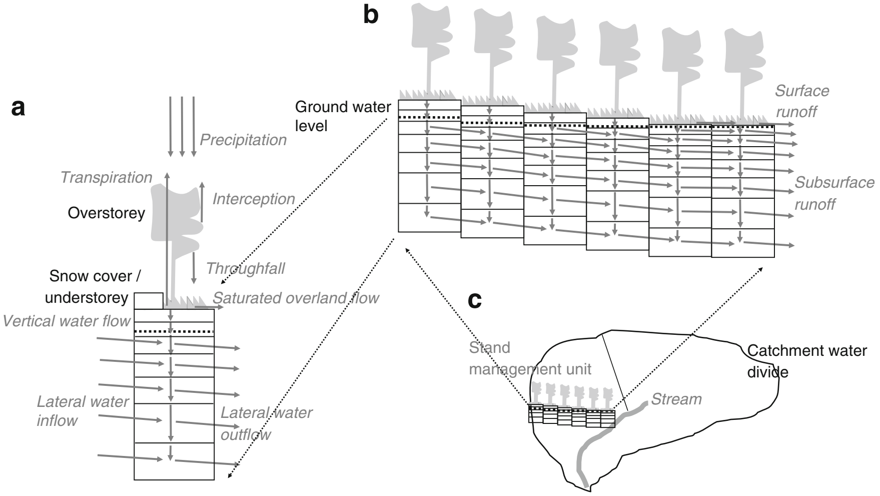


Fig. 23.3 Model column characterising tree stand and soil on upland site (**a**), hydrological response unit (HRU) characterising typical hillslope from water divide to stream (**b**), and subdivision of catchment into typical hillslopes that describe stand management units (**c**). *Arrows* in (**a**) and (**b**) characterise water fluxes in tree stand growing on upland mineral soil. The management unit in (**c**) is assumed to have soil and vegetation properties that vary as a function of distance to stream

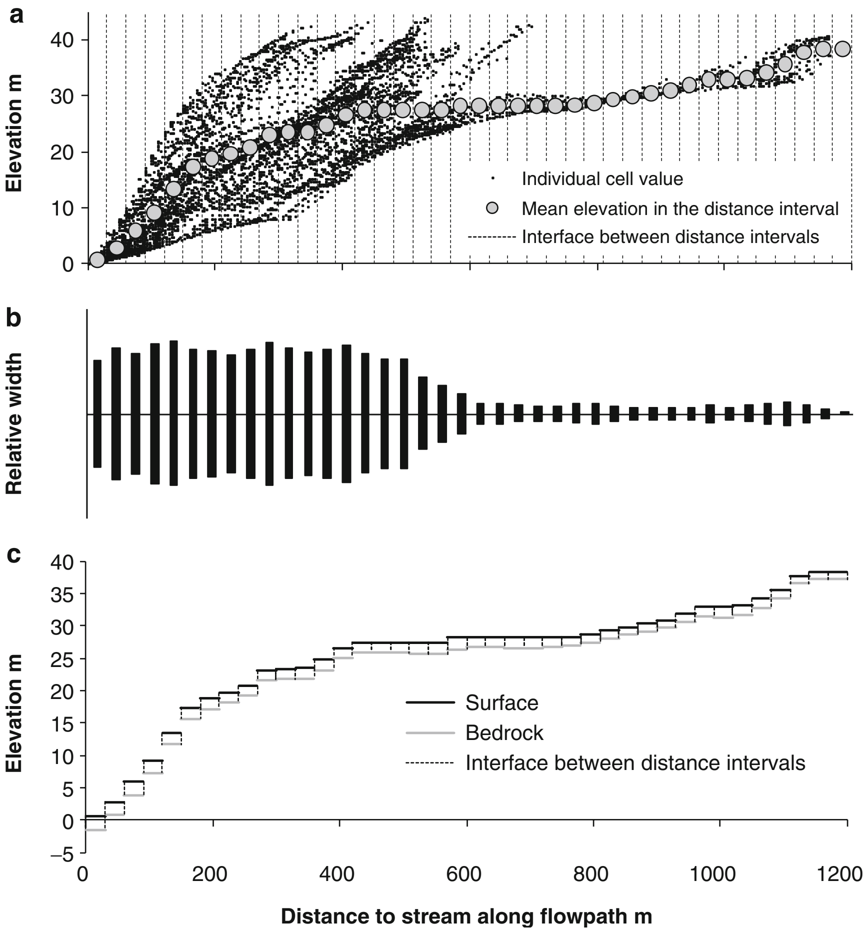


Fig. 23.4 Catchment grid data points prescribing elevation of a grid cell above a receiving stream cell as a function of the distance along the flow path to the receiving stream cell (**a**), relative hillslope width as a function of the distance along the flow path (**b**), and surface and bedrock topography of a typical hillslope (**c**) deduced from spatial data in Kangasvaara in eastern Finland (Kokkonen et al. 2006). The mean elevation within discrete distance intervals is also shown in (**a**) and it describes the surface elevation of the typical hillslope in (**c**). Relative width as a function of the flow path distance (**b**) is proportional to the number of pixels residing in the distance intervals shown in (**a**). The receiving stream cell is the cell where the DEM (digital elevation model)-based flowpath intersects the stream network

The measurements conducted at different distances to a stream, such as observation of water table level, can be compared to the model predictions, when the hillslope is parameterised according to the dimensions of an existing hillslope (see Fig. 23.4) where the measurements are conducted (e.g. Koivusalo and Kokkonen 2003). State variables and fluxes, such as snow water equivalent, throughfall, and transpiration, computed at the hillslope columns can be aggregated at the HRU level and compared against averaged measurements from small catchment.

The hillslope HRU can be applied to characterise natural and restored peatlands, where the distance to the receiving stream is longer than the ditch spacing in drained areas. The divergence of flow toward the stream can be described by using a hillslope parameterisation, where the width of the hillslope changes as a function of distance from stream (Fig. 23.4b). The buffer zones between the receiving stream and managed area need to be included in the HRU parameterisation. The columns next to the stream are assigned with soil and vegetation properties characterising the buffer area, and then the water discharged from the upslope area flows through the buffer zone (e.g. Laurén et al. 2005).

The key assumption in the hillslope HRU is that the catchment properties vary as a function of flow path distance to the stream. This assumption is often violated in catchments with large heterogeneity in soil properties, topography, and vegetation species. Kokkonen et al. (2006) studied how the simplifications made when representing the entire catchment by a single hillslope affected modelled water and nitrogen fluxes. The share of subsurface runoff from total runoff showed a non-linear relationship to changes in the hillslope length and slope, which suggested that representation of the entire catchment with a single profile leads to distortion of the computed runoff components and, as a result, nitrogen loads, when a catchment has sub-areas producing a large fraction of subsurface runoff and sub-areas generating low volumes of subsurface runoff.

Water Cycle and Forest Management in Boreal Upland Areas

Most productive forests in the boreal region are located in the upland mineral soil areas. In the humid boreal zone hydrological condition of upland site is typically not the main factor controlling the stand growth, although it may be important in the regeneration phase and thus influence the selection of site preparation method (Mäkitalo 2009). From the forest management viewpoint, hydrology is important in the transport of dissolved nutrients, such as nitrogen (N) and phosphorus (P), as well as sediments to lakes and rivers after forest treatments. Typical treatment practices in the upland forests include cuttings and soil preparation. Cuttings have hydrological impacts in terms of increased runoff and decreased evapotranspiration, and the main environmental impact is the increased nutrient transport to water courses. The N and P transport is caused by the decreased nutrient uptake by vegetation and increased release of nutrients from the decomposing logging residues. Site preparation changes the distribution and species composition of ground vegetation (Palviainen et al. 2007) and it exposes the soil surface to erosion. The environmental effects of forest management in upland areas need to be described at the HRU level.

The excess N and P loads from forestry areas are typically low, because the nutrient cycle in boreal forest ecosystem is nearly closed with minor leakage into receiving water bodies. Currently, new forest management practices including whole tree harvesting and stump removal can have more intensive environmental effects compared with the traditional practices of cuttings and site preparation. The management guidelines address the need to use water protection measures to reduce the impacts of forest treatments on lakes and rivers. The two-dimensional hillslope approach allows for the description of forest treatments as a function of distance to the receiving water body (e.g. Kokkonen et al. 2006). This is useful when the aim is to quantify how water protection methods, such as buffer zone near the streams, control the hillslope response to the upslope forest treatments. Laurén et al. (2005, 2007) used the HRU approach to describe a forested catchment as a typical hillslope and to model how the length of the buffer zone affects nitrogen export after clear-cutting. The model results were applied to make a reflection between the nitrogen load and the value of the tree stand left in the buffer zone. The study of Laurén et al. (2007) demonstrates how information on hydrological processes can be applied to support forest management decision making.

Remarks

Stream network and terrestrial domains in catchments are the two systems that can logically be separated from each other. The properties of the streams and tributaries can be exploited to disaggregate large catchments into small subcatchments. The concept of hydrological similarity can be then applied in the subcatchment level, where areas exhibiting similar runoff response to rainfall and snowmelt events can be grouped together. The grouped areas are the domains, which are referred to as HRUs, HSUs, hydrotopes, etc., and where hydrological models are parameterised to simulate the processes characteristic for the grouped domain. The HRU scale coincides with the mosaic pattern of forest management units found in the boreal region.

The HRU approach is well-suited to situations when HRUs are independent and not strongly affected by regional processes outside the HRU. The HRUs can be interlinked through a return flow occurring from stream network to the terrestrial domain or through lateral flow in a large groundwater aquifer. Recent studies have introduced schemes to simulate regional linkages between HRUs. The HRU parameterisation usually includes discretisation in the scale of the grouped terrestrial area and therefore, the scale mismatch between the model and observations remains an issue in the model analysis.

Blöschl and Sivapalan (1995) note that the characteristic water flow velocity increases when moving from the plot and hillslope scale to the catchment scale. This is due to the higher flow velocity in the stream network compared to the porous soil domain. The streamflow processes will ultimately dominate the hydrological response of a large catchment and then the main influence of land-surface processes is seen as a loss function through evapotranspiration. In a large catchment, the detailed HRU-scale process description easily becomes unnecessarily complex, whereas more detail is needed to simulate the routing of water in main streams and reservoirs.

There is a need to further develop present forest management practices to account for the link between stand development and water cycle specifically in sites where water has a crucial role. Integration of hydrological and forest management models provide several benefits in the silviculture of boreal forests: improvements in forecasting future stand development, improved long-term prediction of hydrological fluxes, and eventually better forest management. Good management practices can be achieved through identification of areas that are optimal for intensive treatments and areas that pose element load risks to surface waters. HRU approach provides an avenue for integrated modelling of hydrological processes, stand growth, and environmental impacts of forest management. The use of HRU for linking hydrology, management, and land-use is not merely bound to forests and boreal conditions. When data are available from small catchments dominated by different land use practices, HRU can be parameterised and calibrated to simulate hydrological behaviour of, e.g., agricultural and urban areas under different management scenarios.

Acknowledgments Finnish Forest Research Institute research projects ‘Hydrological nutrient losses and methods of water quality protection on peatlands drained for forestry purposes’ and ‘Prediction and mitigation of nutrient and sediment load from forested catchments’ are acknowledged for permitting the authors to participate in this COST action.

References

Ahti E, Hökkä H (2006) Effects of the growth and volume of Scots pine stands on the level of the water table on peat in central Finland. In: Proceedings of an International conference on hydrology and management of forested wetlands, 8–12 April 2006. New Bern, NC, pp 309–315

Becker A, Braun P (1999) Disaggregation, aggregation and spatial scaling in hydrological modelling. J Hydrol 217:239–252

Blöschl G, Sivapalan M (1995) Scale issues in hydrological modelling: A review. Hydrol Process 9:251–290

Callegari G, Ferrari E, Garfì G, Iovino F, Veltri A (2003) Impact of thinning on the water balance of a catchment in Mediterranean environment. Forest Chron 79:301–306

Feddes RA, Kowalik PJ, Zaradny H (1978) Simulation of field water use and crop yield. Simulation monographs, Pudoc, Wageningen, The Netherlands

Flügel WA (1995) Delineating hydrological response units by geographical information system analyses for regional hydrological modelling using PRMS/MMS in the drainage basin of the River Bröl, Germany. Hydrol Process 9:423–436

Grayson RB, Moore ID, McMahon TA (1992) Physically based hydrologic modeling, 2, is the concept realistic? Water Resour Res 28:2659–2666

Gustafsson D (2002) Boreal land surface water and heat balance – modelling soil-snow-vegetation-atmosphere behaviour. Stockholm: Land and Water Resource Engineering, TRITA-LWR. PHD, ISSN:1650-8602

Hökkä H, Salminen H (2006) Utilizing Information on Site Hydrology in Growth and Yield Modeling: Peatland Models in the MOTTI Stand Simulator. In: Proceedings of an international conference on hydrology and management of forested wetlands, 8–12 April 2006. New Bern, NC, pp 302–308

Hökkä H, Repola J, Laine J (2008) Quantifying the interrelationship between tree stand growth rate and water table level in drained peatland sites within Central Finland. Can J For Res 38:1775–1783

Hynynen J, Ojansuu R, Hökkä H, Siipilehto J, Salminen H, Haapala P (2002) Models for predicting stand development in MELA System. Metsäntutkimuslaitoksen tiedonantoja. Finnish Forest Research Institute, Research Papers 835. 116 pp ISBN 951-40-1815-X

Jansson PE, Halldin S (1979) Model for the annual water and energy flow in a layered soil. In: Halldin S (ed) Comparison of forest and energy exchange models. Society for Ecological Modelling, Copenhagen, pp 145–163

Jansson PE, Karlberg L (2001) Coupled heat and mass transfer model for soil-plant-atmosphere systems. Royal Institute of Technology, Department of Civil and Environmental Engineering, Stockholm, Sweden, 321 pp

Joensuu S (2002) Effects of ditch network maintenance and sedimentation ponds on export loads of suspended solids and nutrients from peatland forests, Finnish Forest Research Institute, Res. Papers, 868, Vantaa

Karvonen T (1988) A model for predicting the effect of drainage on soil moisture, soil temperature and crop yield. Ph.D. thesis, Helsinki University of Technology, Publications of the Laboratory of Hydrology and Water Resources Engineering, Otaniemi, 1, 215 pp

Karvonen T, Koivusalo H, Jauhiainen M, Palko J, Weppling K (1999) A hydrological model for predicting runoff from different land use areas. J Hydrol 217:253–265

Kirkby MJ (1985) Hillslope hydrology. In: Anderson MG, Burt TP (eds) Hydrological Forecasting. Wiley, Chichester, UK, pp 37–75

Kite GW, Kouwen N (1992) Watershed modeling using land classifications. Water Resour Res 28:3193–3200

Klingseisen B, Metternicht G, Paulus G (2008) Geomorphometric landscape analysis using a semi-automated GIS-approach. Environ Modell Softw 23:109–121

Koivusalo H (2002) Process-oriented investigation of snow accumulation, snowmelt and runoff generation in forested sites in Finland. Ph.D. thesis, Helsinki University of Technology, Water Resources Publications, TKK-VTR-6, Espoo, 2002

Koivusalo H, Kokkonen T (2003) Modelling runoff generation in a forested catchment in Southern Finland. Hydrol Process 17(2):313–328

Koivusalo H, Kokkonen T, Laurén A, Ahtiainen M, Karvonen T, Mannerkoski H, Penttinen S, Seuna P, Starr M, Finér L (2006) Parametrisation and application of a hillslope hydrological model to assess impacts of forest clear-cutting on runoff generation. Environ Modell Softw 21:1324–1339

Koivusalo H, Hökkä H, Laurén A, Nikinmaa E, Laine J, Ahti E (2008) Splitting the water balance of drained peatland forests into hydrological components. In: Farrel C, Feehan J (eds) Proceedings of the 13th International Peat Congress. After wise use – the future of peatlands, 8–13 June 2008. Tullamore, Ireland, International Peat Society, pp 485–487

Kokkonen T, Koivusalo H, Karvonen T (2001) A semi-distributed approach to rainfall-runoff modelling - a case study in a snow affected catchment. Environ Modell Softw 16:481–493

Kokkonen T, Koivusalo H, Laurén A, Penttinen S, Starr M, Kellomäki S, Finér L (2006) Implications of processing spatial data from a forested catchment for a hillslope hydrological model. Ecol Modell 199:393–408

Krysanova V, Müller-Wohlfeil DI, Becker A (1998) Development and test of a spatially distributed hydrological/water quality model for mesoscale watersheds. Ecol Modell 106:261–289

Kummu M (2008) Spatio-temporal scales of hydrological impact assessment in large river basins: the Mekong case. Ph.D. thesis, Water & Development Publications – Helsinki University of Technology, TKK-WD-04

Laurén A, Koivusalo H, Ahtikoski A, Kokkonen T, Finér L (2007) Water protection and buffer zones: How much does it cost to reduce nitrogen load in a forest cutting? Scand J Forest Res 22(6):537–544

Laurén A, Finér L, Koivusalo H, Kokkonen T, Karvonen T, Kellomäki S, Mannerkoski H, Ahtiainen M (2005) Water and nitrogen processes along a typical water flowpath and streamwater exports from a forested catchment and changes after clear-cutting: a modelling study. Hydrol Earth Syst Sci 9:657–674

Lin YP, Wu PJ, Hong NM (2008) The effects of changing the resolution and land use modelling on simulations of land-use patterns and hydrology for a watershed land-use planning assessment in Wu-Tu, Taiwan. Landsc Urban Plann 87:54–66

Mäkitalo K (2009) Soil hydrological properties and conditions, site preparation, and the long-term performance of planted Scots pine (Pinus sylvestris L.) on upland forest sites in Finnish Lapland. Dissert Forest 80:71, http://www.metla.fi/dissertationes/df80.htm

Maréchal D, Holman IP (2005) Development and application of a soil classification-based conceptual catchment-scale hydrological model. J Hydrol 312:277–293

Naiman RJ, Fetherston KL, McKay SJ, Chen J (1998) Riparian forests. In: Naiman RJ, Bilby RE (eds) River ecology and management. Springer-Verlag, New York, NY, pp 289–323

Oldak A, Pachepsky Y, Jackson TJ, Rawls WJ (2002) Statistical properties of soil moisture images revisited. J Hydrol 255:12–24

Ott M, Su Z, Schumann AH, Schultz GA (1991) Development of a distributed hydrological model for flood forecasting and impact assessment of land-use change in the international Mosel river basin. In: Hydrology for the water management of large river basins (Proceedings of the Vienna symposium, Aug 1991). IAHS Publ. no. 201, pp 183–194

Palviainen M, Finér L, Laurén A, Mannerkoski H, Piirainen S, Starr M (2007) Development of ground vegetation biomass and nutrient pools in a clear-cut disc-plowed boreal forest. Plant Soil 297:43–52

Refsgaard JC (1997) Parameterisation, calibration and validation of distributed hydrological models. J Hydrol 198:69–97

Salminen H, Lehtonen M, Hynynen J (2005) Reusing legacy FORTRAN in the MOTTI growth and yield simulator. Comput Electron Agric 49(1):103–113

Schneider MK, Brunner F, Hollis JM, Stamm C (2007) Towards a hydrological classification of European soils: preliminary test of its predictive power for the base flow index using river discharge data. Hydrol Earth Syst Sci 11:1501–1513

Schröder B (2006) Pattern, process, and function in landscape ecology and catchment hydrology – how can quantitative landscape ecology support predictions in ungauged basins? Hydrol Earth Syst Sci 10:967–979

Scipal K, Scheffler C, Wagner W (2005) Soil moisture-runoff relation at the catchment scale as observed with coarse resolution microwave remote sensing. Hydrol Earth Syst Sci 9:173–183

Seppälä K (1969) Kuusen ja männyn kasvun kehitys ojitetuilla turvemailla. English Summary: Post-drainage growth rate of Norway spruce and Scots pine on peat. Acta For Fenn 93:1–89 (In Finnish with English Summary)

Singh R, Helmers MJ, Crumpton WG, Lemke DW (2007) Predicting effects of drainage water management in Iowa's subsurface drained landscapes. Agric Water Manage 92:162–170

Teich M, Bebi P (2009) Evaluating the benefit of avalanche protection forest with GIS-based risk analyses-A case study in Switzerland. For Ecol Manage 257:1910–1919

Vivoni ER, Entekhabi D, Bras RL, Ivanov VY (2007) Controls on runoff generation and scale-dependence in a distributed hydrologic model. Hydrol Earth Syst Sci 11:1683–1701

Western AW, Blöschl G (1999) On the spatial scaling of soil moisture. J Hydrol 217:203–224

Wigmosta MS, Vail LW, Lettenmaier DP (1994) A distributed hydrology-vegetation model for complex terrain. Water Resour Res 30:1665–1679

Woolhiser DA (1996) Search for physically based runoff model - a hydrologic El Dorado. J Hydraul Eng – ASCE 122:122–129

Wu J, Levin SA (1997) A patch-based spatial modeling approach: conceptual framework and simulation scheme. Ecol Modell 101:325–346

Zhang GP, Savenije HHG (2005) Rainfall-runoff modelling in a catchment with a complex groundwater flow system: application of the Representative ElementaryWatershed (REW) approach. Hydrol Earth Syst Sci 9:243–261

Chapter 24
Large Scale Approach to Forest and Water Interactions

Gebhard Schüler, Laurent Pfister, Michael Vohland, Stephan Seeling, and Joachim Hill

Introduction

The larger the scale the more the sciences of the causes of floods admit uncertainty and imprecision on the prediction of land use impacts on the flood regime. It is also a well-known narrative that forests protect the groundwater quality. But forest management and a changing environment, such as global heating and air pollution may diminish the protection functions of forests with regard to the groundwater quality. The prediction of management and environmental impacts on the groundwater quality is a big challenge, not least because the scales of groundwater landscapes differ from the scale of forest – and land use - diversity. The interactions between forests and water as well as the impacts of forestry on groundwater quality and runoff remain a particular "grey" area in our knowledge. Getting better understanding of the interactions between forest, soil and water as well as of impacts of different management on floods remains an important scientific task.

G. Schüler (✉)
Research Institute for Forest Ecology and Forestry Rheinland-Pfalz, Hauptstrasse 16,
D-67705 Trippstadt, Germany
e-mail: schueler@forestclim.eu

L. Pfister
Centre de Récherche Public – Gabriel Lippmann (CRP-GL), 41, rue du Brill, L-4422
Belvaux, Luxembourg
e-mail: pfister@lippmann.lu

M. Vohland, S. Seeling, and J. Hill
University Trier, Environmental Remote Sensing, Behringstrasse, D-54286 Trier, Germany
e-mail: vohland@uni-trier.de; seelings@uni-trier.de; hillj@uni-trier.de

M. Bredemeier et al. (eds.), *Forest Management and the Water Cycle: An Ecosystem-Based Approach*, Ecological Studies 212, DOI 10.1007/978-90-481-9834-4_24,

Dealing with Ecohydrological Processes on Larger Scales

Precipitation, Soil Water and Discharge at Different Scales in Dependency upon Forestry and Land Cover Dynamics

Discussions – how to collect data and develop water budget models or runoff models -highlight the problem of scale (Calder 2005; Calder and Aylward 2006). The interactions between forests, soils and water can easily be detected at small spatial scale but the forest induced changes will range at a large temporal scale from years to thousands of years how soil properties will change with the development of vegetation cover and influence, e.g. by the root development and its soil pore generating impacts. Whereas, at small spatial scales increased water and sediment runoff due to deforestation can be identified and incorporated into hydrological models, increases in flood flows at larger scales are not clear at all due to the cumulative effects from an entire watershed of which only a specific percentage may have been affected by deforestation and due to hydraulic processes. Downstream flood peaks on larger scales are also the result of coincident peaks of different discharge flows. In addition discharge on larger scales is not only dependent upon forest or vegetation cover, it depends additionally on forestry management (e.g. forest road network, drainage, cultivation etc.) but also on competing land use and most important on the dominating physical site properties and on the particular meteorological situation (e.g. water saturation, frost, smelting snow etc.).

Approach of Hydrological Processes in Dependency of Forests and Forest Site Properties on Larger Scales

Water cycle components are influenced by a plethora of factors such as topography, geology, pedology, land use and climate conditions, the latter acting as an overarching external forcing. All these factors and forcings are characterized by a considerable spatial variability, causing in the end streamflow regimes to be proper to each individual catchment (Dunne 1978; Buttle 1998; Beven 2001). As a consequence of this complexity, contemporary hydrology is still investigating the same questions about the origin and fate of water that had already been phrased by Leonardo Da Vinci approximately 500 years ago (Pfister and Savenije 2006). Some 4 decades ago, Penman (1963), as well as Hewlett and Hibbert (1967), have formulated questions on where the water goes when it rains, the flowpaths it takes to the stream, as well as its residence times in a catchment. It appears that progress has been seriously hampered in hydrology as applied to real world problems since it is a highly uncertain (Beven 2006) and a difficult science, dealing with open systems that are very difficult to observe (Beven 2008).

Catchment hydrology thus experiences tremendous difficulties in providing unequivocal concepts for the description of the driving mechanisms behind first order controls of runoff generation. This situation has been summarized by Kirchner (2003) through the double paradox in catchment hydrology and geochemistry, where (a) the rapid hydrological response of catchments to precipitation during storm events, (b) produces storm runoff which is dominated by "old" pre-event water-like baseflows, (c) and exhibits different geochemical characteristics to baseflow.

As an ultimate consequence of these difficulties, hillslope models often turn out to "work", but unfortunately for the wrong process reasons (Seibert and McDonnell 2002). This leads in the end to the somewhat fatalist conclusion that each catchment is unique (Beven 2001), which consequently limits our ability to create generalizable or regionalizable hypotheses (Beven 2000).

The role of rainfall interception and evaporation is yet another example of a significant process in the water cycle which remains very difficult to grasp. This certainly is largely related to the fact that it is a process that is particularly difficult to measure (Lundberg et al. 1997; Llorens and Gallart 2000). It is still often considered as a minor flux in hydrological modelling, that can be disregarded or taken as a fixed percentage of incoming precipitations (Gerrits et al. 2007) and in the end compensated by another process after model calibration (Savenije 2004).

Despite these many difficulties, there has been in recent literature a considerable number of ideas and concepts that have tried to reconcile the variety of the sometimes equivocal insights that have been gained through the literally countless field studies. These attempts range from classification schemes for meaningful distinctions between catchments (McDonnell and Woods 2004), virtual experimental frameworks focusing on the quantification of first-order controls on flow pathways at hillslope scale (Weiler and McDonnell 2004), to calls for more catchments studies intercomparisons, as to better identify and define first order controls (Seibert and McGlynn 2007).

In this context, certain studies have highlighted the role of lithology, appearing as a major source of discrepancy in hydrological behaviour. This insight was then used to characterize and conceptualize each individual lithological substratum's hydrological behaviour into a parsimonious model structure (Van den Bos et al. 2006). Since data of high spatial and temporal resolution on the dominant runoff generating processes is often lacking at larger spatial scales, certain approaches attempt to simplify the process representations by relying mainly on substratum permeability and topography. The ultimate goal then consists in generating GIS-based maps of dominant runoff processes, providing the basis for rainfall-runoff modelling (Hellebrand et al. 2007a,b, c). Integrating the dominant transpiration and infiltration capacities of the forest cover allows to model the catchment scale effects using an upscaling procedure based on meta-modelling (Marshall et al. 2007).

However, so far none of the above ideas and concepts have been identified as being suited for tackling most of the remaining challenges. Recently nonetheless, Troch et al. (2008) have identified a clear paradigm shift in watershed hydrology, with the introduction of a more holistic view, which explicitly accepts landscape heterogeneity as a dominant control, rather than relying on the traditional view where catchment-scale processes can be modelled and understood from small-scale process understanding.

Hydrological System Behaviour at Catchment Scale

The first mainstream theories that had been formulated upon experience gained from experimentations in research catchments clearly underestimated the tremendous spatio-temporal variability of the processes taking place in the hydrological cycle. Horton's (1933) concept of a catchment averaged infiltration capacity, being constant both in space and time, with storm runoff being a fraction of those precipitations that are falling at intensities exceeding the soil's infiltration capacity, was questioned only three decades later by Betson (1964). Dunne and Black (1970) finally introduced the partial area contribution concept that would dominate in hydrological sciences for decades, guiding storm runoff and relying on restricted areas of the watershed concerned by saturation excess overland flow.

Despite the considerable progress that has been made over the past 3 decades in experimental hydrology, it remains tremendously difficult to generate generalizable hypotheses from the insights gained through small catchment studies. Until now, this situation has led quite systematically to a so-called one-model-fits-all view dominating hydrological model concepts (McDonnell 2003).

The introduction of chemical and isotope tracers (e.g. Pinder and Jones 1969; Sklash and Farvolden 1979) helped to eventually grasp some of the spatial and temporal variability of the origin of water in stream discharge. As a consequence, a major paradigm shift took place in hydrology, recognising the dominating role of pre-event water in storm runoff generation (McDonnell 2003).

Progress in experimental hydrology certainly suffered from the fact that for many decades, studies were rather focusing on temporal patterns of stream flow, rather than on spatial patterns as it is the case in other disciplines of earth sciences, such as geology or geomorphology (Grayson and Blöschl 2000). Here again, the tremendous complexity of combinations of physiographical basin characteristics is likely to have hampered investigations. Nonetheless, presumed non-random patterns in hydrological processes necessarily have to rely on some kind of underlying form of organisation.

Recent studies on first order controls on hydrological flow paths configurations, water storage, as well as residence time distributions, have helped to identify the dominating role of soils, via their integration of the catchment's physical characteristics (Uhlenbrook et al. 2004; Soulsby et al. 2004). While Soulsby et al. (2006) have highlighted the role of soil characteristics as first order control on flow path partitioning and residence time in seven nested sub-catchment of various sizes (10–230 km^2) of the Feshie catchment (Scotland), Pfister et al. (2002) and Hellebrand et al. (2008) have demonstrated the strong link existing between stormflow coefficients and geology in 18 sub-catchments of the Alzette river basin in Luxembourg.

Climate variability, both in time and space, acts as an overarching external forcing factor on most physiogeographical basin characteristics. Pfister et al. (2004) have shown the considerable influence of the spatial variability in rainfall trends over five decades on the trends that have affected winter peak discharge in several catchments located in the Grand-Duchy of Luxembourg. Likewise, Hellebrand et al. (2008) have documented the spatio-temporal variability of rainfall-runoff behavioral patterns in various meso-scale basins of the Rhineland Palatinate. In the same context, mean residence time estimates have been shown to be strongly sensitive to climate differences and variability (Tetzlaff et al. 2007; Hrachowitz et al. 2009).

As previously stated, despite tremendous progress made in investigative techniques in recent years, it still remains extremely difficult to generate generalizable concepts of first order controls in catchment hydrology. The non-linearity of hydrological systems stands as one of the major limiting factors with respect to hydrological predictability, causing Blöschl and Zehe (2005) to call for modesty when discussing the degree to which hydrological system behaviour can be represented.

Contribution of Remote Sensing Techniques to Assess Water and Forests Interactions at the Meso- and Macroscale

Due to a series of earth observation satellites launched during the last decades, remote sensing techniques are nowadays in the position to offer a broad repertory of spatial variables across different spatial scales. Systematic land surface observations provided by remote sensing, improved retrieval algorithms, and sensor advancements have enforced studies that use multi-sensor measurements in an integrated framework of data and models to quantify the terrestrial water cycle states and fluxes (Pan et al. 2008).

Classical approaches in this field are the mapping of different land use and forest stand types, or the detection of deforestation or afforestation respectively. Within the last decades marked success has been achieved in the field of land use mapping and change detection based on remote sensing imagery (Rogan and Chen 2004). Hydrological effects of large scale afforestation were assessed, e.g. by Archer (2003), Gimona and Van der Horst (2007) or Fenicia et al. (2008). Within transnational initiatives like SMOS (Grant et al. 2008) or HYDROPTIMET (Lacava et al. 2005), passive microwave data was used to derive the moisture content of landscape compartments with mixed land use including also forested areas. Moreover, microwave remote sensing techniques allow a direct quantification of dew (De Jeu et al. 2004) or intercepted water (De Jong 2001). Land use or, more specifically, forest stand parameters (e.g. tree composition, age class, leaf area index or biomass) are coupled with the magnitude of water balance components, and thus may be used for the parameterization of hydrological models; e.g., the leaf area index strongly determines the actual interception storage capacity or may be used to quantify canopy coefficients for the assessment of evapotranspiration terms (Andersen et al. 2002; Vohland 2003).

Based on spatial datasets as inputs, hydrological models may provide spatially distributed simulations of hydrological processes like ground water recharge, discharge or evapotranspiration (e.g. Andersen et al. 2002; Hochschild 1998; Seeling and Ahrends 2004). Nevertheless, validation of hydrological models is usually based on discharge measurements at the outlet of catchments, but this kind of validation with a lumped variable does not provide any information about an appropriate reproduction of the hydrologic cycle's spatial differentiation. Here, remote sensing can partly fill the gap, as thermal satellite imagery allows the assessment of spatial variations of surface temperature and therefore provides a key for evaluating heat flux partitioning. Thus, when combined with ancillary meteorological data, remote sensing data can be used to asses daily rates of evapotranspiration at pixel scale, and retrieved spatial patterns of this essential hydrological process variable allows the outputs of hydrological models to be evaluated (e.g. Casper and Vohland 2008). Existing methods to assess evapotranspiration from remote sensing data may be classified into two broad categories (Su 2002): (1) indices that can be defined in the scatter plot of vegetation indices versus land surface temperatures to estimate the relative evapotranspiration, and (2) approaches that calculate sensible heat flux first and then obtain the latent heat flux as the residual of the energy balance equation. Nonetheless, remote sensing approaches are restricted at the mesoscale due to the limited number of operating thermal satellite sensors with high to medium spatial resolution (e.g. Landsat TM, Aster) and their low repetition rates. At the macro- to global scale, e.g. MODIS or NOAA-AVHRR provide close time-series of reflective and thermal data which even allows for the calibration of hydrological models by remote sensing-based evapotranspiration estimates (e.g. Immerzeel and Droogers 2008).

A major difficulty in estimating water budget components is the complex interaction between the governing variables, which inter alia prevents a linear spatial up- or downscaling. With a view to evapotranspiration, El Maayar and Chen (2006) have shown for a forested study area, that distributed calculations that are based on a fine spatial resolution of all input data differ from lumped calculations with broad spatial units or large pixels. This limitation at the macroscale can partly be compensated by using corrective algorithm using sub-pixel information on land surface heterogeneity (El Maayar and Chen 2006). Thus, multi-sensor approaches and the application of downscaling techniques for remote sensing data with a coarse spatial resolution are important issues for a precise large scale assessment of water budget components. In this field, some studies already exist that deal with labeling techniques to derive sub-pixel fractions of land surface properties (e.g. Canters et al. 2002; De Fries et al. 1997).

Hydrochemical Interactions of Forest and Water and Its Impacts on Larger Scales

The functions of forests with regard to the groundwater and spring water quality are closely associated with element cycles, and they depend also on the amount and intensity of precipitation, on the site conditions as soil properties, buffer capacity and mineral composition of the soil series and geology, and also on the land cover at all including adjacent land-use types. It is expected that forests are the source of clean or at least less burdened groundwater and spring water. But the water quality is also influenced by the atmospheric input load of the seepage-water. Since the beginning of the industrialisation the increased atmospheric nitrogen and acid deposition continuously worsened the seepage water quality in forest soils (Wilpert 2007; Sucker et al. 2009). These depositions are concentrated in forests, caused by the filter effects of the tree canopies.

Large scale groundwater maps based upon investigations of drinking water resources showed a clear relation of nitrogen to agricultural land-use and of acidification, Al- and Mn-content to forested landscapes (Landesamt für Wasserwirtschaft 1993).

Additionally forest soils often have less buffer capacity than agricultural soils because forests were traditionally concentrated on the poorer soil substrates. v. Wilpert and Zirlewagen (2001) showed a significant acidification of the stream water discharge in forested catchments. Besides the acidification effects they pointed to an increasing charge of nitrate in the groundwater also in forested catchments. This threatens the quality of raw water for drinking water supply, but also the maintenance and diversity of hydro-biological biocoenosis. Stabilization measures as forest liming may improve biological cycle systems (Schüler 2002) and by this, they strengthen the filter function of forest soils for a good groundwater and spring water quality (Wilpert 2007). In consequence Wilpert et al. (2007) proved the positive effects of forest liming on the alkalinity of stream water discharge. The predominant topography of a catchment may have an important influence on the liming effects. Flat slopes reduce the runoff velocity and improve the liming effects, whereas the lime on steeper slopes had reduced effects due to an accelerated washout (Wilpert et al. 2007). Of course for qualifying the chemical characteristics of groundwater and spring water differences in hydrology must also be taken into consideration because flow path and retention time have significant influence on the chemical composition (Nebe and Abiy 2002). So, hydrologic flow paths exert a strong control on discharge chemistry. Water from soils and bedrock zones influence the dynamics of stream water quality but also depending on the differences in land-use and size in the catchments. (Martinez-Carreras et al. 2007).

Integrated Land and Water Management at Larger Scale

The Effects of an Integrated Forest and Watershed Management at Larger Scales – Myths and Realities

Reconciling an integrated forest and watershed management there is the confrontation between two broad narratives (Calder 2005): On the one hand there is the "engineering watershed management" narrative and in some advanced countries there is a strong movement towards engineering solutions especially in the field of flood management. On the other hand there is the "nature leads to an ideal world" narrative and often it is forgotten that, e.g. flooding is the natural way to discharge the water arising from an occasional large rainfall event. Floods caused no problems until people decided to use the natural flood plains for their own use. So, these narratives must be brought back to the last 50 years' results of scientific research.

Land cover and land-use impacts, particularly the impacts of forestry management on discharge peaks will be well percept at a small scale, because the initiated individual processes can be measured and modelled at plot scale and site scale. But at larger scales increases in discharge and in flood flows are not so easily discerned or modelled. This may be due to the integration of cumulative effects from an entire watershed, of which only a relatively small percentage my have been affected. Additionally the interacting processes of water, land-use and landscape morphology are very complex. The increase in scale reduces the likelihood of coincident flow peaks far downstream. Hence, the net effect downstream becomes increasingly difficult to predict at larger spatial scales (Calder and Aylward 2006). This hydraulic attenuation leads to the working hypotheses that as the scale and as the size of hydrological events, such as big floods, increase the effects of land-use would become less important. La Marche and Lettenmair (2001) showed after comprehensive field experiments in a 149 km^2 catchment that forest removal increases the mean annual flood by about 10%. And even the road effect could be quantified to an intensification of the mean annual flood about other 10%. The effect of extensively logging on the discharge decreased more and more for floods of greater magnitude with a longer return period. Grant (2005) discussed also the effects of forests on flood events. He found that forest harvesting and deforestation has only limited effects on big flood events at larger scales. Summarizing the results of a number of American studies on the effects of forestry on the watershed behaviour the USDA Forest Service (2000) stated that the effect of roads on the basin stream flow is generally smaller that the effect of forest cutting and that damaging floods increase less than 10% after forest clearance and less again the larger the spatial scales. Studies in Wales, UK (Robinson and Dupeyrat 2005) showed that forest cuttings increased annual flows and that peak flows were often attributed to soil compaction and soil disturbance reducing infiltration. Hence modern forest management guidelines should care to reduce soil damages by the use of brash mats on forest machine lanes. Thus, also the efficiency of precautionary water retention measures in forests must be discussed in the context of the observed scale, but also in the context of the magnitude and intensity of the rain fall. Forests tend to diminish flood peaks by interception, by supporting the infiltration in relatively high porous soils and by retarding overland flow with branches, fallen leaves and organic layers. But forests are not unlimited flood protectors. Just like sponges they will saturate more and more due to the crown interception capacity, to the antecedent soil moisture content, to the water storage and infiltration capacity of forest soils. Forest soils can affect its flow only so long as they have not taken up all the water they can hold. Water which falls after the forest and its floor is saturated runs into the basin streams almost as fast as it would run over bare ground. But the interactions between forests and soils and how they co-evolved over different time periods ranging from years to thousands of years and how soil properties will ultimately change with changes of the vegetation cover and forest management practices are still not clear in the scientific knowledge (Calder and Aylward 2006).

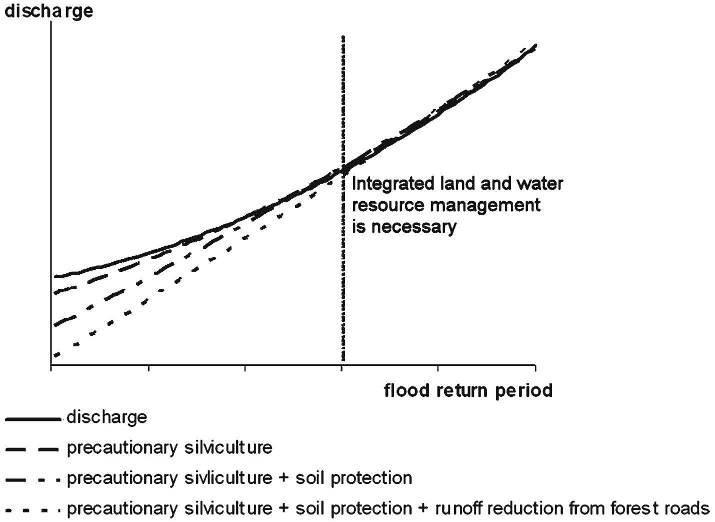


Fig. 24.1 Effects for discharge reduction by precautionary forestry management are shown in this unscaled graph. Different discharge curves vary according to the site and landscape structures. Intersection points and their assignment to a certain flood return period depend on the particular site conditions and on the site-dependent efficiency of precautionary land management measures (Schüler 2006)

It is beyond controversy, that each additional flood precautionary measure reduces or delays runoff, and by this making the flood discharge curves diverge from their starting points and meet on the flood frequency curve at a higher flood return period (Schüler 2006; Fig. 24.1). Land use and land management can tighten the runoff situation, if it doesn't consider runoff generation processes, e.g. by a precautionary forestry management in small scaled headwater catchments (Bredemeier and Schüler 2004; Schüler et al. 2007). A sustainable managed vegetation cover however levels runoff peaks. But it has to be considered, that the effects of reafforestation appear after a time lag. The soils beneath former grassland or agricultural use may take centuries to evolve after reforestation as well as the natural restoration of forest soils after compaction by heavy forest machinery my take so long time (Schneider and Schüler 2007). Hence, above a certain threshold of flood return period and at larger spatial scales only an integrated land and water resource management is effective. This threshold is defined by the point of intersection, when the reduced discharge curve meets the "normal" flood frequency curve. It depends strongly on the magnitude of the climatic event, and on the site, soil, geology, land-use and landscape characteristics. It should be the objective of spatial flood-protection planning to predict the particular threshold of danger, depending on the damage potential in catchment areas. The location where forestry makes a net positive contribution to flood control is in the actual floodplain itself (Nisbet 2002).

So, flood precaution measures should not only be restricted to forestry management concepts. Flood precautions require the cooperation of water, agriculture and viticulture sectors as well as land and infrastructural planning management in residential area and traffic systems, combined with spatial planning and domestic policy, all to be integrated into a truly eco-hydrological approach. All precautionary measures that effectively enhance water retention in small watersheds are suitable. So, if all small catchments in a larger watershed are managed with a view to water retention, the occurrence of damaging floods may be reduced (Schüler 2006). The keyword is reduced, because large scale climatic situations are responsible for the generation of infrequent, damaging floods.

The integrated land and water resource management includes (in extension to Calder and Aylward 2006):

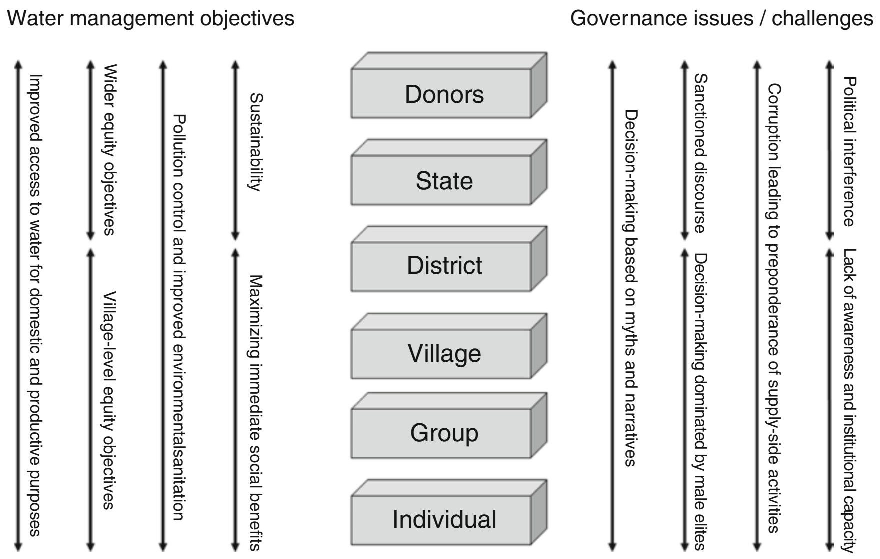
- Precautionary land-use and forestry measures.
- Structural measures aiming at keeping floodwaters away from the people (dams, polder etc.).
- Land-use planning measures aiming at keeping people away from the floodwaters (legal regulation of the most hazardous areas of floodplains).
- Flood preparedness measures aiming at getting people ready for floods before they come (flood forecasting systems).
- Flood emergency measures helping affected people to cope with floods.

Incorporating socio-economic developments to land-use and water management needs larger scaled models based on geographic information system linkages. Floods are natural processes in landscapes to discharge the water arising from one or several large and/or sustaining rainfall events. The generation of floods depends mainly on the storing capacity of landscapes and on their actual saturation. For example, forest cover depletes the soil water stores and it supports the infiltration of water into the soils. Hence, the forest soils remain longer capacious, but only as long as they are not saturated with water. The depletion by evapotranspiration of forests is not as fast as intensive rainfall can fill the soil pore system. Furthermore heavy rainfall exceeds the rate of evapotranspiration of forests. The larger the scale the more saturating processes may overlay and the danger of flood generation increases.

Land-Use and Water Management Governance: the European Water Framework Directive and the European Floods Directive

Land-use has considerably effects on the quality and structure of surface water and groundwater. Land-use contributes also to mitigate the effects of floods and droughts. Water management by precautionary land-use and forestry can support-protective objectives as well as promote the great variety and abundance of the water fauna and flora. An integrated land and water management has to consider the range of political, social, economic and administrative systems.

In Europe the core challenge in an integrated land and watershed management is that of land and water governance in relation to the political and societal founda-



Source: BATCHELOR 2005

Fig. 24.2 Water management objectives and governance challenges at different levels in society (Calder 2005)

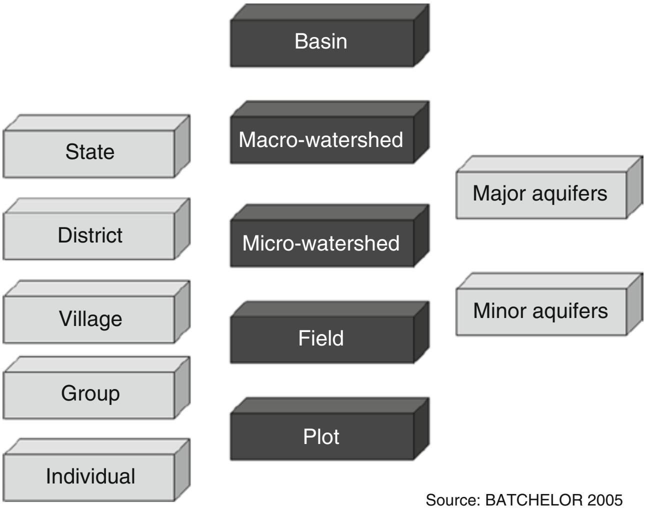
tions (Calder 2005). Recognition of the land-use and water services objectives at different levels in society and governance challenges they face at the different levels, may assist the task of identifying the correct entry points on which to initiate actions (Fig. 24.2).

Even if water management starts on a local level, it is a cross-border task. Precautionary land-use measures can be part of a good practice.

Recognition that the administrative boundaries of governance systems do not match spatially and identically with the physical boundaries of land and water systems is another issue that must be taken into account within an integrated land and water management (Calder 2005; Fig. 24.3).

The EU Water Framework Directive (2000/60/EC) establishes a legal framework to prevent further deterioration, to protect and enhance the status of inland surface water, transitional water, coastal water and groundwater on a large scale across Europe and ensure its long-term sustainable use. The directive approaches to a water and river basin management based on large scales, especially on the natural geographical and hydrological units.

To protect all water resources the Member States of the EU shall ensure that a river basin management plan is made for each river basin district lying entirely within their territory or in the case of an international river basin district falling entirely within the Community, the Member States shall ensure the coordination with the aim of producing a single international river basin management plan.



Source: BATCHELOR 2005

Fig. 24.3 Governance systems need to recognize the disjuncts that exist between hydrological and administrative boundaries (Calder 2005)

They are also responsible for the coordination of all programmes of measures in the river basin district. For international river basin districts the Member States concerned shall together ensure this coordination.

Besides other objectives the directive should

- Contribute to mitigating the effects of floods and droughts (art. 1d)
- Protect, enhance and restore all bodies of surface water (art. 4, 1a - i and ii, as well as art. 4, 5b)

The EU Water Framework Directive should be implemented by protecting, enhancing and restoring and by preventing the deterioration of the status of all bodies of surface water and groundwater (art. 4) and by monitoring of the surface water and groundwater status as well as protected areas (art. 8). The implementation of river basin management can be realized by "supplementary" measures (art. 11, 3) as codes of good forestry practice or as restoration of wetland areas.

In principle the entirety of a river basin is concerned (art. 4, 4 and art. 13), but especially those areas which are explicitly specified in the EU Water Framework Directive:

- Terrestrial ecosystems and wetlands directly depending on the aquatic ecosystems (art. 1a).
- Protected areas (art. 6 and annex IV), which are registered within each river basin district, and which have been designated as requiring special protection under specific Community legislation for the protection of their surface water

and groundwater or for the conservation of habitats and species directly depending on water.

- Surface water bodies (art. 4) as rivers, creeks and lakes, including the surrounding plains as far as the water morphology can be influenced.

Big European regions which are characterized by highlands and low mountain range structures are marked by two types of flood events. On a large scale, the flooding of larger rivers have always given rise to troubles, e.g. for the rivers Rhine, Moselle and Saar the last times in 1993 and 1995. In addition, flash floods of smaller and medium-sized tributaries are also of considerable significance. This was drastically demonstrated by the recent flood disasters in 2002, 2003 and even in 2005 in Bavaria and the Switzerland. These damages, summed up for several catchments, can exceed the flood damages caused on major streams. The warning times are short, as flooding occurs often unexpectedly. Protection measures in flood plains of smaller rivers, in particular the technical flood protection, are less optimized.

Flood protection knows no boundaries. The prevention of flood damages by precautionary measures is a transnational task that can only be adequately realized, if all abutting states of a river basin co-operate. In the Declaration of Arles in 1995 the competent EU environment ministers for Rhine, Moselle, Saar and Maas underlined, that not only measures of water management, but also measures in the field of spatial planning and land use are necessary for flood protection with regard to agriculture and forestry, the development of urban areas and the use of recreation. The EU Floods Directive (2007/60/EC) requires taking adequate and coordinated measures to reduce flood risks. This Directive shall be carried out in coordination with the Water Framework Directive, notably by flood risk management plans and river basin management plans. The flood risk management practices shall also be coordinated in large scale river basins and they shall in take into consideration long term developments, including climate change, as well as sustainable land use practices.

The Commission of the European Communities has stated in its communication to flood risk management (com. 2004, 472), that the way, how agricultural and forestry areas are used, is important for flood prevention and protection and that it is decisive for the capacity of soil and plants to retain water. One of the principles of flood risk management plans is an interdisciplinary approach: all relevant aspects of water management, physical planning, land use, agriculture, transport and urban development, nature conservation need to be considered at all levels (national, regional and local). To promote sustainable flood risk management measures it is necessary to have a list of cost-effective flood risk management measures which will be possible to support the objectives of water retention.

The political promotion of precautionary flood risk management by appropriate and efficient forestry measures must start at a local level. But flood prevention is a cross-border task. This task can be realized efficiently with the help of transnational management plans. The acceptance of precautionary forestry measures for flood prevention, especially if those “water services” exceed not only a duly land management but also measures of good practice, will be increased enormously if the recovery of costs will be regulated. This is possible with the help of the existing EU water directives.

Conclusions

At larger spatial scales, certain approaches to quantify the threat of bad groundwater quality, of drought or flood generation attempt to simplify the relevant processes by generating relevant GIS-based maps of dominating ecohydrological processes. There is a clear paradigm shift in watershed hydrology with the introduction of a more holistic view, which explicitly accepts landscape heterogeneity as a dominant control, rather than relying on the traditional view where catchment-scale processes can be modelled and understood from small-scale process understanding. Systematic land surface observations by satellites improved quantifying the terrestrial water cycle states, fluxes and processes.

Forests are often seen as the source of clean or at least less burdened groundwater. The groundwater and its aquifers at larger scales will be influenced by the diversity of plot-scaled seepage water qualities, whereas the seepage water quality is a result of the atmospheric nitrogen and acid deposition and micro-scaled ecohydrological processes in forests and forest soils.

Downstream flood peaks are the result of coincident peaks of different discharge flows. The larger scaled discharge depends on competing land use, on forestry management, on the very special forest or vegetation cover, and most important on the dominating physical site properties on plot scale as well as on the particular micro-scaled meteorological situation. That means, the big floods are developing in large spatial scale of big river basins. Forests and Forestry can tighten the runoff situation. A sustainable managed vegetation cover levels runoff peaks. But above a certain threshold of flood return period and only the integration of technical structural measures and other land and water resource management at larger spatial scale has effective impacts on floods and flood development.

The political promotion of precautionary water management by forestry starts in management scales, which means at a local level. But water management exceeds this local and even a regional level. It is a cross-border task. This task can only be realized within transnational activities. The acceptance of precautionary forestry measures for "water services" exceeds a duly land management. So, it needs the help of the existing European water and flood directives, but also of a "new" European management directive for sustainable and multifunctional forestry in Europe.

Acknowledgments Some of the research underlying this article received support from the European Union as part of the INTERREG III B NWE-program.

References

Andersen J, Dybkjaer G, Jensen KH, Refsgaard JC, Rasmussen K (2002) Use of remotely sensed precipitation and leaf area index in a distributed hydrological model. J Hydrol 264:34–50

Archer D (2003) Scale effects on the hydrological impact of upland afforestation and drainage using indices of flow variability: the River Irthing, England. Hydrol Earth Syst Sci 7:325–338

Betson RP (1964) What is watershed runoff? J Geophys Res 69:1541–1552

Beven KJ (2000) On uniqueness of place and process representations in hydrological modelling. Hydrol Earth Syst Sci 4:203–212

Beven KJ (2001) Rainfall-runoff modelling: the primer. Wiley, Hoboken, NJ

Beven KJ (2006) On undermining the science? Hydrol Process 20:3141–3146

Beven KJ (2008) On doing better hydrological science. Hydrol Process 22:3549–3553

Blöschl G, Zehe E (2005) On hydrological predictability. Hydrol Processes 19:3923–3929

Bredemeier M, Schueler G (2004) Forest ecosystem structures, forest management, and water retention. Ecohydrol Hydrobiol 4(3):255–266

Buttle JM (1998) Fundamentals of small catchment hydrology. In: Kendall C, McDonnell JJ (eds) Isotope tracers in catchment hydrology. Elsevier, Amsterdam, pp 1–43

Calder I (2005) Blue Revolution – Integrated Land and Water Resource Management, 2nd edn. Earthscan, London, Sterling, VA, p 353

Calder I, Aylward B (2006) Forest and floods: moving to an evidence-based approach to watershed and integrated flood management. Int Water Resour Assoc (Water Intl) 31(1):1–13

Canters F, Swinnen E, Eerens H, Van de Voorde T (2002) Estimation of land-cover proportions at the subpixel level from 1-km SPOT-VGT data. – Proc. of the 5th int. symposium on spatial accuracy assessment in natural resources and environmental sciences. Accuracy 2002:66–74

Casper MC, Vohland M (2008) Validation of a large scale hydrological model with data fields retrieved from reflective and thermal optical remote sensing data – A case study for the Upper Rhine Valley. Phys Chem Earth 33:1061–1067

Dunne T (1978) Field studies of hillslope flow processes. In: Kirkby MJ (ed) Hillslope Hydrology. Wiley, Chichester, pp 227–293

De Fries R, Hansen M, Steininger M, Dubayah R, Sohlberg R, Townshend J (1997) Subpixel forest cover in central Africa from multisensor, multitemporal data. Remote Sens Environ 60:228–246

De Jong J (2001) Remote sensing of wet forests. Ph.D. thesis, University of Groningen

De Jeu R, Heusinkveld B, Vugts H, Holmes T, Owe M (2004) Remote sensing techniques to measure dew: the detection of canopy water with an L-band passive microwave radiometer and a spectral reflectance sensor. – Remote Sensing for Agriculture, Ecosystems, and Hydrology VI. In: Owe M, D'Urso G, Gouweleeuw BT, Jochum AM (eds) SPIE Proceedings, vol 5568., pp 225–235

Dunne T, Black RD (1970) Partial area contributions to storm runoff in a small New England watershed. Water Resour Res 6:1296–1311

El Maayar M, Chen JM (2006) Spatial scaling of evapotranspiration as affected by heterogeneities in vegetation, topography, and soil texture. Remote Sens Environ 102:33–51

Fenicia F, Savenije H, Avdeeva J (2008) Anomaly in the rainfall-runoff behaviour of the Meuse catchment. Climate, land use, or land use management? Hydrology and Earth System Sciences Discussion 5:1787–1819

Gerrits AMJ, Savenije HHG, Hoffmann L, Pfister L (2007) New technique to measure forest floor interception – an application in a beech forest in Luxembourg. Hydrol Earth Syst Sci 11:695–701

Gimona A, Van der Horst A (2007) Mapping hotspots of multiple landscape functions: a case study on farmland afforestation in Scotland. Landscape Ecol 22:1255–1264

Grant GE (2005) The role of forests and forest harvest on floods: myths and realities. Presentation at the XXII IUFRO World Congress, Brisbane, Australia. Int Forest Rev Vol. 7(No. 5):311pp

Grant J, Saleh-Contell K, Wigneron J, Guglielmetti M, Kerr Y, Schwank M, Skou N, Van de Griend AA (2008) Calibration of the L-MEB Model Over a Coniferous and a Deciduous Forest. IEEE Trans Geosci Remote Sens 46:808–818

Grayson R, Blöschl G (2000) Spatial patterns in catchment hydrology: observations and modelling. Cambridge University Press, Cambridge, UK

Hellebrand H, Hoffmann L, Juilleret J, Pfister L (2007a) Assessing winter storm flow generation by means of permeability of the lithology and hydrological soil processes. Hydrol Earth Syst Sci 11:1673–1682

Hellebrand H, Juilleret J, Hoffmann L, Pfister L (2007b) Assessing hydrological information potential of lithology and soil for winter storm flow generation: a two-pronged approach to mesoscale regionalization. In: Schüler G, Gellweiler I, Seeling S (eds) Dezentraler Wasserrückhalt in der Landschaft durch vorbeugende Maßnahmen der Waldwirtschaft, der Landwirtschaft und im Siedlungswesen. Mitt. a.d. Forschungsanstalt für Waldökologie und Forstwirtschaft, Nr. 64/07. Trippstadt, pp 143–154

Hellebrand H, Müller C, Pfister L (2007c) Up-scaling of soil hydrological processes with respect to distributed rainfall-runoff modelling. In: Proceedings of the 1st scientific conference on integrated catchment management for hazard mitigation, Trier, pp 7–11

Hellebrand H, Van den Bos R, Hoffmann L, Juilleret J, Krein A, Pfister L (2008) Spatio-temporal variability of behavioral patterns in hydrology in meso-scale basins of the Rhineland Palatinate (1972–2002). Clim Change. doi:10.1007/s10584-008-9509-7

Hewlett JD, Hibbert AR (1967) Factors affecting the response of small watersheds to precipitation in humid areas. In: Sopper WW, Lull HW (eds) Proceedings of the international symposium on forest hydrology. Pergamon, Oxford, pp 275–290

Hochschil U (1998) Einsatz der Fernerkundung für die hydrologische Systemanalyse im südlichen Afrika. ZPF-Z Photogrammetrie Fernerkundung 66:98–107

Horton RE (1933) The role of infiltration in the hydrological cycle. Trans Am Geophys Union 14:446–460

Hrachowitz M, Soulsby C, Tetzlaff D, Dawson JJC, Dunn SM, Malcolm IA (2009) Using long-term data sets to understand transit times in contrasting headwater catchments. J Hydrol. doi:10.1016/j.jhydrol.2009.01.001

Immerzeel WW, Droogers P (2008) Calibration of a distributed hydrological model based on satellite evapotranspiration. J Hydrol 349:411–424

Kirchner JW (2003) A double paradox in catchment hydrology and geochemistry. Hydrol Process 17:871–874

La Marche J, Lettenmair DP (2001) Effects of forest roads on flood flows in the Deschutes River, Washington. Earth Surf Process Landforms 26:115–134

Lacava T, Greco M, di Leo EV, Martino G, Pergola N, Sannazzaro F, Tramutoli V (2005) Monitoring soil wetness variations by means of satellite passive microwave observations: the HYDROPTIMET study cases. Nat Hazards Earth Syst Sci 5:583–592

Landesamt für Wasserwirtschaft (1993) Grundwasserbericht 1992. LfW-Bericht, Nr. 225/93, 87pp

Llorens P, Gallart F (2000) A simplified method for forest water storage capacity measurement. J Hydrol 240:131–144

Lundberg A, Eriksson M, Halldin S, Kellner E, Seibert J (1997) New approach to the measurement of interception evaporation. J Atmos Oceanic Technol 14:1023–1035

Marshall MR, Francis OJ, Frogbrook ZL, Jackson BM, McIntyre NR, Reynolds B, Solloway I, Wheater, HS (2007) The Pontbren catchment study: a multi-scale experimental programme investigating the impact of UK upland land use on flood risk. In: Proceedings of the 1st scientific conference on integrated catchment management for hazard mitigation, Trier, pp 68–72

Martinez-Carreras N, Krein A, Iffly JF, Barnich F, Pfister L, Hofmann L, Gallart F (2007) Examining the spatial and temporal variations of erosion processes and hydrochemical response in mesoscale catchments – preliminary results from the Attert basin in Luxembourg. In: Schüler G, Gellweiler I, Seeling S (eds) Dezentraler Wasserrückhalt in der Landschaft durch vorbeugende Maßnahmen der Waldwirtschaft, der Landwirtschaft und im Siedlungswesen.

Mitt. a.d. Forschungsanstalt für Waldökologie und Forstwirtschaft, Nr. 64/07. Trippstadt, pp 125–134

McDonnell JJ (2003) Where does the water go when it rains? Moving beyond the variable source area concept of rainfall-runoff response. Hydrol Process 17:1869–1875

McDonnell JJ, Woods R (2004) Edilonal on the need for catchment classification journal of Hydrology 299(A-2):2-3

Nebe W, Abiy M (2002) Chemie von Quellwässern in bewaldeten Einzugsgebieten des Erzgebirges. Forstw Cbl 121:1–14

Nisbet T (2002) Implications of climate change: soil and water. In: Climate change: impacts on UK forests. Edinburgh: Forestry Commission, Edinburgh, chapter 5

Pan M, Wood EF, Wójcik R, McCabe MF (2008) Estimation of regional terrestrial water cycle using multi-sensor remote sensing observations and data assimilation. Remote Sens Environ 112:1282–1294

Penman HL (1963) Vegetation and hydrology. Commonwealth Bureau of Soils, Harpenden, UK, Technical communications 53

Pfister L, Humbert J, Iffly JF, Hoffmann L (2002) Use of regionalized stormflow coefficients in view of hydro-climatological hazard mapping. Hydrol Sci J 47:479–491

Pfister L, Drogue G, El Idrissi A, Iffly JF, Poirier C, Hoffmann L (2004) Spatial variability of trends in the rainfall-runoff relationship: a mesoscale study in the Mosel basin. Clim Change 66:66–87

Pfister L, Savenije HHG (2006) Leonardo da Vinci's scriptures as benchmark papers in hydrology. Hydrol Process 20:1653–1655

Pinder GF, Jones JF (1969) Determination of the ground-water component of peak discharge from the chemistry of total runoff. Water Resour Res 5:438–445

Robinson M, Dupyrat A (2005) Effects of commercial timber harvesting on streamflow regimes in the Plynlimon catchments, mid-Wales. Hydrol Process 19:1213–1226

Rogan J, Chen D (2004) Remote sensing technology for mapping and monitoring land-cover and land-use change. Prog Plann 61:301–325

Savenije HHG (2004) The importance of interception and why we should delete the term evapotranspiration from our vocabulary. Hydrol Process 18:1507–1511

Schneider R, Schüler G (2007) Impact of Heavy Forest Machinery on Physical Properties of Forest Soils. Proceedings of the Scientific Conference on Integrated Catchment Management for Hazard Mitigation, Trier, Germany, pp 33–37. http://ubt.opus.hbz-nrw.de/volltexte/2007/438

Schüler G (2002) Schutz versauerter Böden in nachhaltig bewirtschafteten Wäldern – Ergebnisse aus 10-jähriger interdisziplinärer Forschung. Allgem Forst- u Jagdztg 173:1–7

Schüler G (2006) Identification of flood-generating forest areas and forestry measures for water retention. For Snow Landsc Res 80(1):99–114

Schüler G, Gellweiler I, Seeling S (2007) Dezentraler Wasserrückhalt in der Landschaft durch vorbeugende Maßnahmen der Waldwirtschaft, der Landwirtschaft und im Siedlungswesen. Mitt. a.d. Forschungsanstalt für Waldökologie und Forstwirtschaft, Nr. 64/07. Trippstadt, 337 pp

Seeling S, Ahrends H (2004) Perspektiven der Verwendung multispektraler Fernerkundungsdaten im Rahmen flächen-distributiver Berechnungen der Grundwasserneubildungsrate im Mittelgebirgsraum. Jber Mitt Oberrhein Geol Ver 86:385–403

Seibert J, McDonnell JJ (2002) On the dialog between experimentalist and modeler in catchment hydrology: use of soft data for multicriteria model calibration. Water Resour Res 38(11):1241. doi:111029/2001WR000978

Seibert J, McGlynnn BL (2007) A newtrionjular multiple flow direction algorithm for computing upslopeareas from gridded digital elevotion models water Resources Research 43:wouson, doi:1029/2006WR005128

Sklash MG, Farvolden RN (1979) The role of groundwater in storm runoff. J Hydrol 43:45–65

Soulsby C, Rodgers Petry J, Hannah DM, Malcolm IA, Dunn SM (2004) Using tracers to upscale flow path understanding in mesoscale mountainous catchments: two examples from Scotland. J Hydrol 291:174–196

Soulsby C, Tetzlaff D, Rodgers P, Dunn S, Waldron S (2006) Runoff processes, stream water residence times and controlling landscape characteristics in a mesoscale catchment: an initial evaluation. J Hydrol 325:197–221

Su Z (2002) The surface energy balance system (SEBS) for estimation of turbulent heat fluxes. Hydrol Earth Syst Sci 6:85–99

Sucker C, Puhlmann H, Zirlewagen D, von Wilpert K, Feger KH (2009) Bodenschutzkalkungen in Wäldern zur Verbesserung der Wasserqualität – Vergleichende Untersuchungen auf Einzugsgebietsebene. Z Hydrol Wasserbewirtschaftung 53:250–262

Tetzlaff D, Malcolm IA, Soulsby C (2007) Influence of forestry, environmental change and climatic variability on the hydrology, hydrochemistry and residence times of upland catchments. J Hydrol 346:93–111

The European Parliament and the Council of the European Union (2000) Directive 2000/60/EC of the European Parliament and of the Council of 23 October 2000 establishing a framework in the field of water policy (EU Water Framework Directive). Off J Eur Commun L327:1–72

The European Parliament and the Council of the European Union (2007) Directive 2007/60/EC of the European Parliament and of the Council of 23 October 2007 on the assessment and management of flood risks (EU Floods Directive). Off J Eur Commun L288:27–34

Troch PA, Conllo GA, Heidbuchel L, Rajagopal S, Switonek M, Volhmann THM,Vager M (2008) Dealing with LandscapeHetrogenity in watershed Hydrology Geography Compass 3(1):375–392

Uhlenbrook S, Roser S, Tilch N (2004) Hydrological process representation at the meso-scale: the potential of a distributed, conceptual catchment model. J Hydrol 291:278–296

USDA Forest Service (2000) In: Gucinski H, Furniss M (eds) Forest roads: a synthesis of scientific information. United State Department of Agriculture, Forest Service, Washington DC, www.fs.fed.us/news/roads/science.pdf

Van den Bos R, Hoffmann L, Juilleret J, Matgen P, Pfister L (2006) Regional runoff prediction through aggregation of first-order hydrological process knowledge: a case study. Hydrol Sci J 51:1021–1038

Vohland M (2003) Optimierung modellgestützter Wasserhaushaltsbilanzierung durch fernerkundlich gestützte Parametrisierung landnutzungsabhängiger Größen. – Ph.D. thesis, University of Trier

von Wilpert K (2007) Waldbauliche Steuerungsmöglichkeiten des Stoffhaushalts von Waldökosystemen am Beispiel von Buchen- und Fichtenvarianten der Conventwald-Fallstudie. Freiburger Forstl. Forschung, 237 S. u. Anhang

von Wilpert K, Niederberger J, Puhlmann H (2007) Fallstudien zur Bewertung und Entwicklung forstbetrieblicher Optionen zur Sicherung der Wassergüte in bewaldeten Einzugsgebieten. Forschungsber. FZKA/BWPLUS (Förderkennz. BWR 22014), 43 S

Von Wilpert V tirlewagan D (2001) Boden Vesanerning and Entwickling derWasser qualitot imbewaldeten Einzngsgebiet der Contwald Fallstudie Benchte Feiburger Foistliche Forstriching 33:123–138

Weiler M, McDonnell JJ (2004) Virtual experiments A new approach for imporing process conceptnalization in hillscope hydrology, fournal of Hydrology, 285:3–18 doi:1016/50022–1694(03) 00271–3

Chapter 25

Model-Based Assessment of Forest Land Management on Water Dynamics at Various Hydrological Scales – A Case Study

Andreas Wahren and Karl-Heinz Feger

Introduction

The European Flood-Directive points out the need of flood risk ma-ps. Such maps will be the basis of flood risk management plans focusing on prevention, protection, and preparedness. This effort has to be done river-basin-oriented and directly linked to the EU-Water-Framework-Directive. Thus, emphasis has to be put on the detection of synergy effects between a good ecological status (e.g. minimize technical impacts) of the water bodies and flood protection. As a result of the disastrous floods during recent years, especially the Elbe River flood in Saxony (East Germany) in August of 2002, the novel water law of Saxony (SächsWG) contains regulations concerning flood originating areas (German: *Hochwasserentstehungsgebiete*). Such areas were defined by the flood protection authority. For the flash flood areas in Saxony, which are mostly located in the Northeastern Ore Mountains, the novel law addresses the conservation and improvement of the natural water retention. The soils should be unsealed or afforested if possible. In case of an unavoidable loss or reduction of the natural water retention in these areas a suitable compensation is required (e.g. afforestation).

In comparison to other low mountain range areas the Ore Mountains especially the eastern part have a very low percentage of forest cover (partially <20%). A main reason for this landscape characteristic is the mining history of the area. Afforestation has been promoted in order to increase decentralized flood retention, in addition, e.g. the demographic change background and increasing demand for

---

A. Wahren (✉)
Technische Universität Dresden, Institute of Soil Science and Site Ecology, Dresden Water Centre, 01735, Tharandt,Germany
and
Dr. Dittrich & Partner Hydro-Consult GmbH, Gerlinger Str. 4,01728 Bannewitz,
e-mail: wahren@forst.tu-dresden.de; wahern@hydro-consult.de

K.-H. Feger
Technische Universität Dresden, Institute of Soil Science and Site Ecology, Dresden Water Centre, 01735 Tharandt, Germany
e-mail: fegerkh@forst.tu-dresden.de

M. Bredemeier et al. (eds.), *Forest Management and the Water Cycle: An Ecosystem-Based Approach*, Ecological Studies 212, DOI 10.1007/978-90-481-9834-4_25,

wood energy stress out that an increase in the percentage of forest cover may be necessary in the future. Moreover, the catchments in the Ore Mountains also traditionally provide drinking water from reservoirs and are known for the lower specific runoff under forest. Competitive goals of future land-use planning like flood protection, profitable food/wood production, water supply, water protection show the conflicts dealing with land-use change and the fundamental difficulty of making decisions which affect an uncertain future (land-use change, climate change, demographic change etc.).

Reliable integrated model calculations are essential for successful implementation of measures. In FLOODsite (IP 6th EU-FP) the impact of land-use on runoff generation and runoff concentration was investigated on different scales. One should be aware that the soil plays a key role as an interface that compartmentalizes the runoff in the different runoff types (surface, subsurface, base flow, and deep seepage). A portion of the water is stored in the soil. The soil water fluxes and related runoff formation depend on the soil hydraulic properties which are highly related to the land cover and the type of land management. In one study the change of soil hydraulic properties caused by an afforestation measure was investigated. This work was done by analyzing a 'false chronosequence' – different aged tree stands with comparable initial pedogeological conditions (Wahren et al. 2009).

To quantify the impact of land-use changes on the behavior of flood generation in river basins spatially-distributed rainfall-runoff-models were used. To scale-up the results of the plot investigations and to show potentials and limitations of afforestation measures for the 129 km² large catchment of the Schwarze Pockau River (central part of the Ore Mountains) a complete afforestation scenario (potential natural vegetation) was compared with the present land-use for different flood events.

However, a scenario of total afforestation of the catchment is not realistic. Therefore, based on the results of the spatially-distributed rainfall-runoff-model (AKWA-M®), the possible consequences of socio-economically founded afforestation scenarios on the specific runoff and water retention in changing climatic conditions were simulated and discussed. Thus, the objective was (1) to describe scenarios that reflect a close-to-reality future development with different boundary conditions which could be guided by political authorities; (2) to consider climate change in general and especially assessing the site conditions before a flood generating rainfall event. Our approach combines socio-economic methods with state-of-the-art hydrological modeling and the available information about climate-change.

Soil Properties

The revitalization and increase of natural water retention potentials – notably in headwater catchments – is a major component of sustainable flood prevention strategies. It is logical that changes in land-use patterns (e.g. expansion of settlements including road-construction, deforestation, distinct practices in arable and grassland management) contribute to an increased frequency and severity of flood generation. For forest land-use, it has been stated that afforestation and the promotion of close-to-nature silviculture will considerably increase the water retention in landscapes. However, there is a controversial debate on the quantitative role of such non-structural flood risk management measures with respect to event size and scale-based physical conditions. Modeling approaches very often neglect important aspects when rainfall-runoff-models are parameterized. Hence, many models just consider vegetation parameters (root depth, leaf area index (LAI), canopy height, etc.). Some more advanced models (AKWA-M®, WASIM-ETH, etc.) also include pre-event soil water content by calculating land-use specific evapotranspiration. In addition, one should also be aware that changes in the vegetation cover (e.g. conversion from arable land into grassland or forest) in the mid- to long-term will also result in distinct changes in soil hydraulic properties (infiltration, percolation, retention).

Our investigation in the catchment of the upper Mulde River, a pilot-area of the EU-IP FLOOD*site*) explored the impact of afforestation measures on the soil hydraulic properties. 'False chronosequences' were used to quantify the time-dependent dynamical character of such changes. Four plots were identified at a test area with comparable pedological start conditions and relief situation. There, a set of tree stands of different age: (1) arable field (initial state); (2) 6-year-old afforestation; (3) 50-year-old afforestation; (4) ancient natural forest ('target' stocking) were investigated.

Hood infiltrometer field tests, in conjunction with time-domain reflectometry (TDR) measurements, were carried out to measure the effect of afforestation on soil hydraulic properties. For this purpose, the UGT infiltrometer was used. This hood infiltrometer enables the measurement of hydraulic properties from saturation up to the bubble point of the soil. Detailed information on the instrumental method and data analysis is given by Schwärzel and Punzel (2007). The target is to describe the saturated matrix flow as well as the macropore flow. For every land-use plot 5 replicate measurements were performed.

Upon completion of the infiltration experiments, undisturbed soil cores were extracted from the centre of every infiltration site to enable the measurement of bulk density and unsaturated hydraulic properties (i.e. the water retention curve and the conductivity curve using the multistep outflow method). The unsaturated soil hydraulic properties were determined in the laboratory during transient conditions using the evaporation method (Wendroth et al. 1993) and during steady-state conditions using the hanging water column method and a pressure apparatus (Dane and Topp 2002). Hydraulic conductivities were calculated using the approach of Wind (1968). For every soil horizon three undisturbed samples for soil physical lab measurements were taken.

Distinct differences were detected. Especially the top soil layers revealed an increased conductivity and a higher portion of coarse/middle pores causing an increased infiltration and soil water retention potential (Table 25.1).

Hydraulic conductivities measured in the field are presented in Table 25.2. Distinct differences between the four land-uses in parallel to the pore size distribution (cf. Table 25.1) were observed.

Table 25.3 shows an example from sites in the Mulde catchment (area: Zellwald). The model (AKWA-M®) simulates the runoff behavior influenced by afforestation for

Table 25.1 Pore distribution (vol%), related field capacity (mm) and unsaturated hydraulic conductivity cm day^{-1} in the top layers (30 cm)

		Pore diameter, μm				Hydraulic
Land-use	Horizon	>50	50–10	10–0.2	Field capacity	conductivity cm day^{-1} at pF 2.5
Arable land	Ap	6	5	3	106	0.011 ± 0.02
Young afforestation	Ah	6	9	7	149	0.026 ± 0.05
Old afforestation	Ah	5	14	10	179	0.043 ± 0.10
Ancient forest	Ah	5	11	8	129	0.032 ± 0.10

Table 25.2 Saturated hydraulic conductivity (cm day^{-1}) at in the top layers

Land-use	Horizon	Saturated hydraulic conductivity
Arable land	Ap	360 ± 180
Young afforestation	Ah	710 ± 300
Old afforestation	Ah	1,100 ± 400
Ancient forest	Ah	1 200 ± 450

different pre-event soil moisture conditions. The runoff peaks were calculated comparing two model parameterizations ('soil properties unchanged' and 'soil properties changed') for two flood generating rainfall events. If the 'changed soil' the hydraulic behavior is neglected, the peak reduction for HQ (25) and HQ (100) is smaller (4–8%) than in the other case (with HQ(25) – flood event with a return period of 25 years). The reason for that difference is the increased conductivity causing a higher infiltration, and the increased field capacity which enables the soil to store more water.

Our study underlines that rainfall-runoff models should consider changes both in vegetation and soil properties. Only such a combined approach ensures to address land-use effects in an appropriate way. The lack of relevant data on changes in soil properties should not lead to the conclusion, that land-use measures are ineffective. It is quite clear that their impact is limited, but the sustainability and the synergy effects to nature protection and soil conservation should keep the considerations about that non-structural measures vital.

Targeted measures to improve the natural retention in the watershed are long-term challenges and, thus, the appearance of the benefits may comprise a time period of several decades. As a consequence, reliable model calculations are imperative to estimate the actual potential of land-use strategies and their limitations. Beside the use of state-of-the-art models to predict effects for the present planning and building of flood protection measures, also tools to predict key parameters should be improved continuously in order to minimize the uncertainties.

Upscaling

Two different land-use parameterizations have been modelled in order to show the potential impact of land-use on the hydrological response of local subareas and the total catchment, respectively. Especially the soil water budget is considered in this

Table 25.3 Peak runoff from an afforested site for initial state and two model parameterizations (‘soil properties unchanged’ and ‘soil properties changed’) for the ‘target state’ land-use (old forest)

Land-use	P(T = 25 years, D = 2 h), mm	HQ(25 years), mm			Reduction,%			Soil properties	P(T = 100 years, D = 2 h), mm	HQ(100 years), mm			Reduction,%		
		Pre-event soil moisture								Pre-event soil moisture					
		High	Middle	Low	High	Middle	Low			High	Middle	Low	High	Middle	Low
Arable land	45	30	27	17	–	–	–		56	41	39	27	–	–	–
Forest (target state)	45	17	16	13	–43	–41	–24	Unchanged	56	26	24	22	–37	–39	–19
	45	15	14	11	–50	–48	–35	Changed	56	24	22	21	–41	–43	–23
Difference					7	7	11						4	4	4

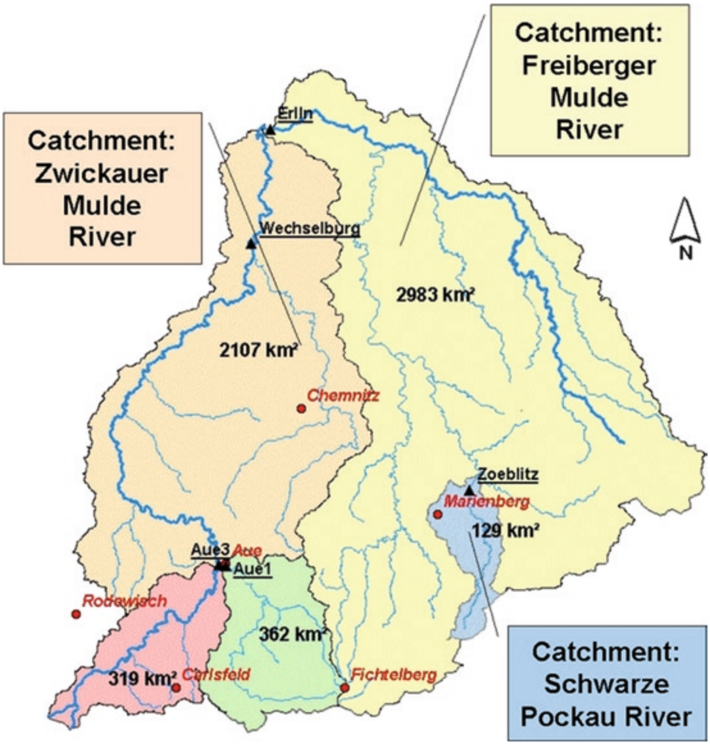


Fig. 25.1 Catchment of the Schwarze Pockau River with gauging station Zöblitz (Mulde catchment, Ore Mountains-Elbe basin)

investigation (change of storage, conductivity, etc.). The catchment of the Schwarze Pockau River as a subcatchment of the Mulde River basin was chosen to compute land-use effects (Fig. 25.1). This mountainous catchment is a ‘flash flood area’ (according to SächsWG) and belongs to basin of the Elbe River, where the most serious flood damages ever in Europe occurred in August 2002. The choice of the two parameterizations (Fig. 25.2) should represent afforestation to demonstrate the maximum forest caused change in water retention like the novel water law of Saxony advises. The used rainfall-runoff model AKWA-M® is based on the water budget model AKWA-M (Golf and Luckner 1991; Münch 1994). The model was advanced by Dittrich & Partner Hydro-Consult GmbH (Münch 2004). This water balance and rainfall-runoff model simulates the water balance and flood runoff in watersheds and transforms the different processes from a site to a larger area. It contains physically based components as well as a conceptual background. As a big advantage AKWA-M® allows to calculate the pre-event situation with the water budget model (time step 1d) and then to increase the time discretization for the flood event using actual conditions (storage, water contents, etc.) as initial state.

Two events were calculated in order to estimate the effects of land-use changes in their spatial distribution a more frequent event (~60 mm day^{-1}) and the highly infrequent event from August 2002 (~230 mm within 2 days).

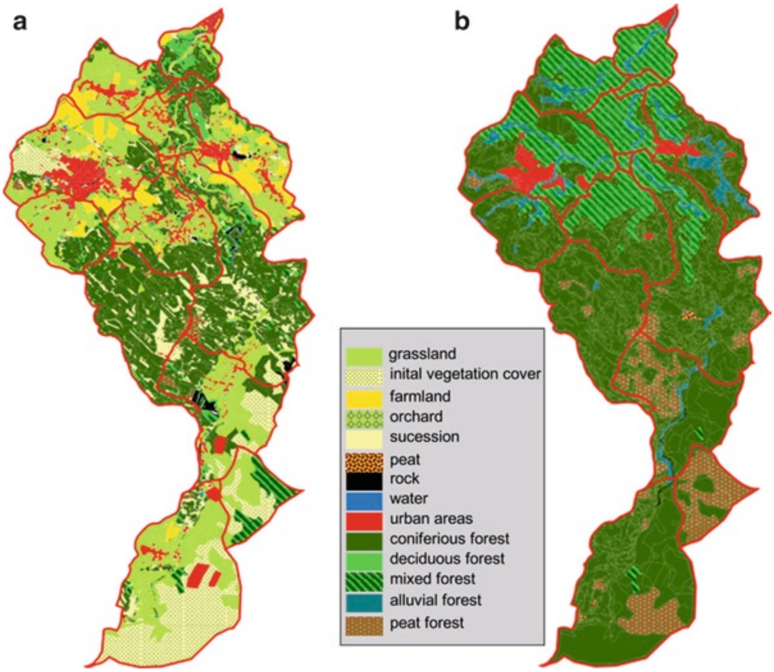


Fig. 25.2 Land-use parameterizations: (**a**) present land-use given by CIR-data (LfUG 2004): 41% forest; 47% grassland; 7% farm land – calibration; (**b**) potential natural vegetation (PNV) except urban areas given by PNV-data (Schmidt et al. 2002): 97% forest predominant oak–beech mixed forest and spruce forest

Figure 25.3 shows for these two events the change in flood effective rainfall. The flood effective rainfall is the portion of the storm rainfall which is neither retained on the land surface nor which infiltrates into the soil – it is that part of the rainfall which is transformed into fast runoff components. It is obvious that there is nearly no change for the very seldom flood event because the storage capacity is limited. It appears that the role of land-use during such heavy rain events is negligible. However, for the more frequent event the afforested areas yield a decrease in flood effective rainfall from 5% up to 20%. Thus, the impact of the land-use on flood formation decreases with the increase of rainfall intensity. Figure 25.4 depicts the corresponding runoff hydrographs for both events. The peak discharge for the very infrequent event from August 2002 is nearly unchanged. For the more frequent event the decrease of the peak flow is ~20%.

Integrated Modeling

The historical as well as the present-day vegetation pattern has always been affected by human activities. Today's land-use in the Ore Mountains is the outcome of hundreds (or even thousands) of years of natural and man-made landscape conversion. Therefore,

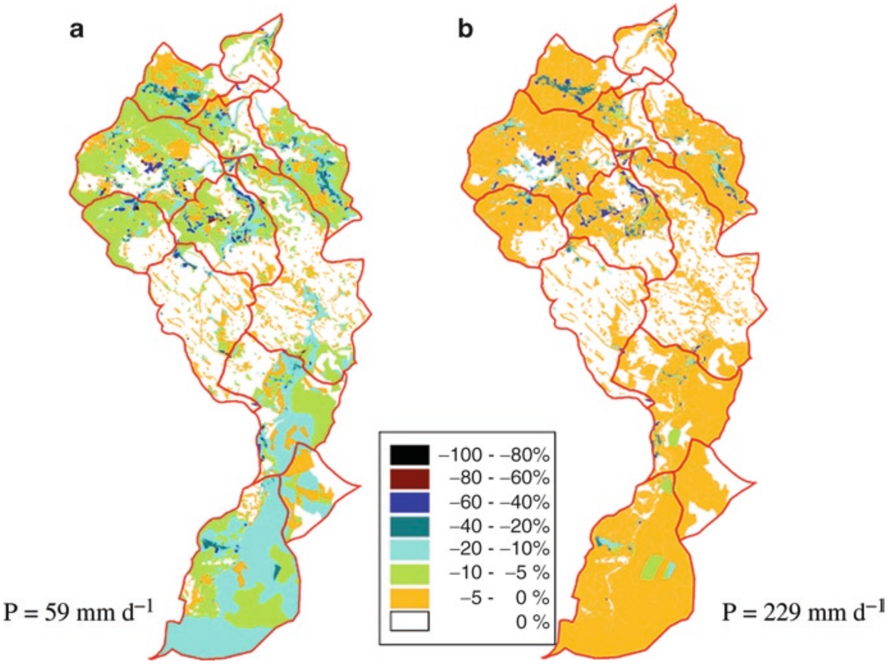


Fig. 25.3 Difference of flood effective rainfall scenario (**b**) – scenario (**a**) (percent) (cf. Fig. 25.2) for two rainfall events: *left* – 59 mm day^{-1}, *right* – 229 mm 2 day^{-1}

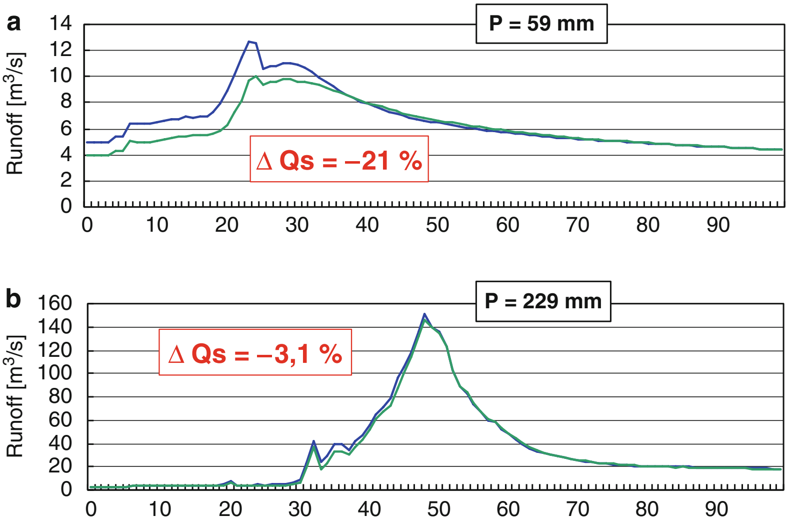


Fig. 25.4 Changed peak flow (Qs) and runoff hydrograph due to changed land-use (*blue* – scenario (**a**) – CIR, *green* – scenario (**b**) – PNV) for two different rainfall situations; P = precipitation

integrated model approaches are needed to forecast the spatial land-use pattern considering socio-economic changes as well as climate change. The effects of such changes on water budget and flood generation were also calculated with the spatially-distributed rainfall-runoff model AKWA-M® and are based on the investigations above.

The investigated area for this research is a subcatchment of the Schwarze Pockau River (Mulde catchment) called 'Schlettenbach'. It is located in the central part of the Saxon Ore Mountains in the south of the town Marienberg. The catchment covers an area of 6.8 km² from which 52% consists of grassland. The forest portion is ~35%, the arable land covers 7%. A portion of 6 % of the catchment area is sealed (urban and traffic areas). The soils are dominated by Cambisols (sandy loam – Ls3) and Stagnic Cambisols (Ls3). Along the rivers Gleysols were mapped.

In cooperation with socio-economic researchers the generally accepted Foresight approach (EVANS et al. 2004) was chosen to develop future scenarios. These socio-economic scenarios are intended to suggest possible long-term futures, exploring alternative directions in which social, economic, and technological changes may evolve within coming decades. Under the Foresight Futures, one future axis depicts the scale of governance from global to local, while the other reflects values from those that are community orientated to individual consumerism (Table 25.4).

Thus, five land-use scenarios were developed for the investigation area. A far-reaching afforestation or tree species adaptation (considering the scenario boundary conditions) and the maintenance of the village 'Wüstenschlette' was valid for all scenarios. The potential natural vegetation – PNV (Schmidt et al. 2002) – a nearly complete forest cover with oak-beech-stands in the lower altitudes and spruce/fir-beech mixed stand in the higher regions is used as a "maximum retention scenario" (Tables 25.5 and 25.6).

The climate future scenarios are outcomes from an investigation by the Umweltbundesamt (Enke et al. 2006). The results base on the climate model run ECHAM 5 which takes into account two different changes CO_2 concentration until 2100. The ECHAM 5 results were downscaled with the WETTREG-model, which produces daily climate data for every meteorological observation station in Germany. The feedback between the socio-economic future and the climate future was neglected in this investigation.

The climate prognosis forecast for the research area (station Marienberg, 50°39 00 N, 13°09 30 E) a precipitation reduction between 1% (IPCC scenario B1) and 7% (IPCC scenario A2) and a temperature increase between 1.8°C (B1) and 2.5°C (A2). Table 25.7 shows the results from the present water budget simulation (16 years: 1985–2001) for socio-economic scenarios. It is obvious, that the increase of the forest percentage leads to higher simulated evapotranspiration especially by increasing the interception storage but also by a higher transpiration. The runoff decreases with the growth of the forest area. Scenario SZ1 demonstrates, that a forest management, that deals with clear cuts and monocultural plantations can lead to a loss of water retention although the portion of forest area in the catchment was increased. Table 25.8 summarizes the water budget components under the changed climate situation (IPCC Scenario B1). The increase of the evapotranspiration due to an increased temperature leads to a distinct reduction of the runoff. It is logical, that the average soil storage decreases as well under these changed conditions.

Table 25.4 Summary of foresight futures according to Evans et al. (2004)

Scenario	Aims	Consequences
1 – World markets	Globalization; personal independence; material richness; Policy: Minimal, enabling markets	Multinational trusts, high productivity and mobility, rural migration, income differentials increase
2 – National enterprise	Personal independence and material richness influenced by a national identity Policy: State-centered, market regulation to protect key sectors	Decline of small and medium-sized enterprises slower than in Scenario 1; growth in service sector; specialisation on products for the regional market and luxury goods; agriculture subsidized
3 – Global sustainability	Common welfare, equal opportunities and intact environment realized by communities of values Policy: Corporatist, political, social and environmental goals	World-wide regulation of economical growth; international trade with high social and ecological standards; agriculture balances yield, biodiversity and sustainability; renewable energy; income decreases
4 – Local stewardship	Sustainable standards of common welfare on a local scale, shelter of local markets Policy: Interventionist, social and environmental	Increment of small and medium-sized enterprises; high social, ecological and quality standards; production site specific fragmented; agriculture for self-sufficiency; mobility and income differentials decrease

If these changes are going to happen in the catchment of a drinking water reservoir serious gaps in water supply could be caused by the changed land-use pattern. It is evident from the different soil storage distribution functions (empirical non-exceedance probability) for the present weather situation (1984–2001) with respect to flood protection, that the effect of the land-use increases with decrease of the soil moisture (Fig. 25.5). If the weather situation before a heavy rain event has been dry the difference between the land-use types (interception, evapotranspiration, root depth) leads to different water consumption. In the case of a wet pre-event situation the limited refillable porosity is filled.

It can be stated that the climate change will influence the provided storage much more than the infiltration behavior. Figure 25.6 shows for the present land-use and scenario 4 (LS) the empirical non-exceedance probability of soil storage for both

Table 25.5 Applied foresight futures

Scenario	Consequences for the investigation area
World markets (WM):	Agriculture unprofitable. Multinational timber trust use nearly the whole catchment for spruce production. Landscape shows an alternation of clear-cut areas, spruce stands of different ages (10, 30, and 70 years) and grassland around the buildings at 'Wüstenschlette'.
National enterprise (NE):	Private agriculture focuses on intensive pasture production. The woodside areas, which are difficult to till, are afforested with spruce. The other forested areas maintain. Around the village 'Wüstenschlette' fish and fruit production is established. All activities are considering national or "home-" production aspects.
Global sustainability (GS):	Natural water retention is a general European interest. Afforestation and soil conserving agriculture is subsidised by the EU. Crops for energy production are planted. Forestry develops species-appropriate mixed stands. Connected forest stripes along riverside forests are established. Products are exploited with "according to flood protection"-sign.
Local stewardship (LS):	Small sized and site specific soil conservation tillage (pasture, crops). Close-to-nature forestry and afforestation. Continuous riverside forests are established.

the present weather conditions (1984–2005) and the future conditions (2100) given from the more moderate B1 scenario. There, it is assumed that in Saxony (southern part of East Germany) it becomes drier and warmer, but the abundance of heavy rain events will increase. The differences in water storage at different forms of land-use will become more pronounced as the number of drier days will increase. Therefore, it should be placed greater importance on land-use considering water retention.

For a number of rain events between 1984 and 2001 the runoff was simulated. Three of these rainfall-runoff calculations of analyzed events are shown here with different precipitation sums and different pre-event soil moisture conditions. Figure 25.7 shows the average pre-event soil moisture situation (present land-use) (Table 25.9).

However, not only the additional storage in the soil provided by different forms of land-use is important. Another dominant factor is the ability of the soil to conduct the water from the surface into deeper layers. The infiltration routes water to free storage in the deeper soil or brings it to subsurface pathway which normally decelerate the runoff. The superposition of these processes causes the intended peak reduction. The peak reduction for the event with the highest pre-event soil moisture (1996) is very small in each scenario. The runoff for the 1995 event shows an observable peak reduction in scenario 3 and 4 and a notable reduction for the PNV

Table 25.6 Land-use portions % for all scenarios

	Forest												
Scenario	Total	Clear cut	Spruce, 10a	Spruce, 30a	Spruce, 70a	Wet	Intensive grassland	Extensive grassland	Conventional arable land	Soil conserving arable land	Settlements	Hedges with fruit-trees	Sealed
IST (present land-use)	35						41	11	7		4		2
1 (WM)	81	11	11	19	34	6		12			4		3
2 (NE)	44						41				4	9	2
3 (GS)	53									42	3	1	1
4 (LS)	63							15		18	2	1	1
PNV	99										1		–

Table 25.7 Water budget components for all socio-economic future scenarios (mm), where IST…present land-use, SZ 1, SZ 2, SZ 3, SZ 4…land-use scenarios (cf. Tables 25.5 and 25.6 and PNV…potential natural vegetation

1985–2001	IST	SZ 1	SZ 2	SZ 3	SZ 4	PNV
Precipitation	972	972	972	972	972	972
Actual evapotranspiration (ETR)	472	480	505	509	534	593
Transpiration	193	178	212	212	222	244
Interception	190	238	227	221	252	296
Runoff	507	501	476	473	448	386
Runoff-difference to present land-use (IST), (%)	–	−1	−6	−7	−12	−24

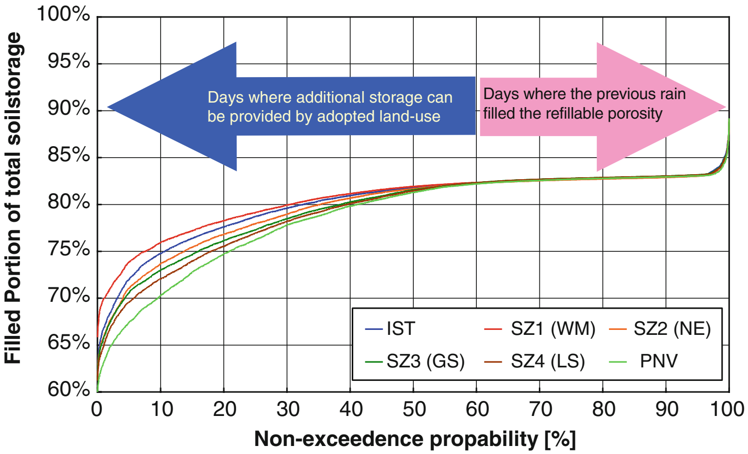


Fig. 25.5 Empirical non-exceedance probability of average soil storage content for all scenarios

Table 25.8 Water budget components for all socio-economic future scenarios under future climate conditions IPCC scenario B1 mm, IST…present land-use, SZ 1, SZ 2, SZ 3, SZ 4…land-use scenarios (cf. Tables 25.5 and 25.6) and PNV…potential natural vegetation

B1 Scenario (2100)	IST	SZ 1	SZ 2	SZ 3	SZ 4	PNV
Precipitation	964	964	964	964	964	964
Actual Evapotranspiration (ETR)	501	512	532	550	581	653
ETR-Difference to present climate (mm) and (%) (Table 25.7)	+29 +6%	+32 +7%	+27 +5%	+41 +8%	+47 +9%	+60 +10%
Runoff	481	472	439	414	387	349
Runoff-Difference to present climate (%) (Table 25.7) (Runoff IST = 507 mm)	−5%	−7%	−13%	−18%	−24%	−31%

scenario. The rainfall characteristics in 2002 cause pronounced two-peak behavior. The resulting runoff points out the effects of the lateral components and the limited soil storage. There is a high reduction of the first peak especially in scenario 3 and 4 the peak is reduced by of more than 40% of the IST-runoff. The second peaks do not show such strong reductions. PNV-runoff for the second peak of this event

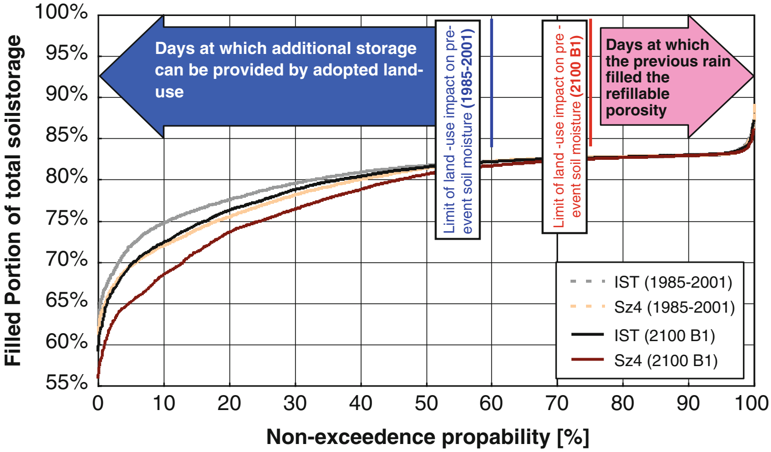


Fig. 25.6 Empirical non-exceedance probability of soil storage content today and 2100 for IST and scenario SZ4

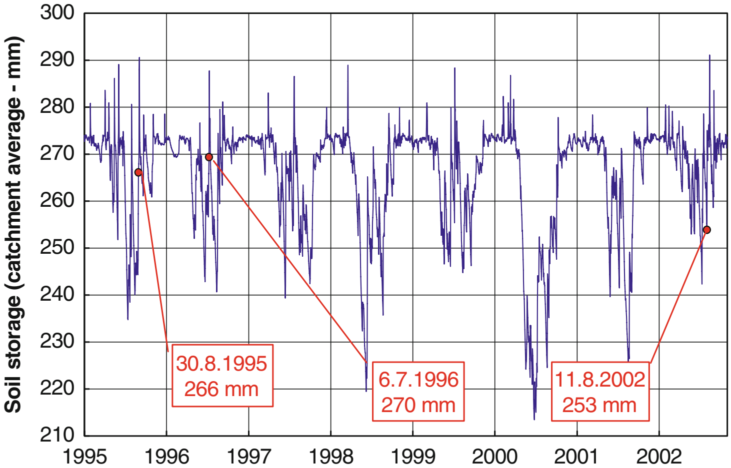


Fig. 25.7 Calculated soil storage and initial points for the rainfall-runoff calculation

(70% reduction for the first peak) is even higher than in scenarios 3 and 4 because the retained runoff from the first peak is still part off the runoff when the second peak reaches catchments outlet. Accordingly, the second peak equals a flood event with high pre-event soil moisture. Table 25.10 summarizes the peak reductions. The results demonstrate that the different afforestation scenarios combined with conserving and improving the hydraulic functions of the soil also by non-intensive pasturing, preservative tillage and depaving measures can provide a distinct contribution to flood prevention. But especially forest cover results in an increased water consumption due to higher interception and transpiration rates and in case of conifers

Table 25.9 Rain events and their statistical placement (Barthels et al. 1997) with precipitation (P, mm), duration (D, h) and return period (T, years)

	D							
	1		4		24		72	
Date	P	T	P	T	P	T	P	T
31.08.95–1.09.95 P = 155 mm	8	<0.5	27	0.5	108	~100	124	~20
7.07.96–11.07.96 P = 77 mm	6	<0.5	15	<0.5	67	5–10	77	2–5
11.08.02–15.8.02 P = 200 mm	22	1–2	50	~10	173	>100	198	>100

Table 25.10 Calculated runoff peaks (p, mm/h) and percent differences for all scenarios (pRed, %)

	1995		1996		2002 Peak 1		2002 Peak 2		
Scenario	p	pRed	p	pRed	p	pRed	p	pRed	Peak reduction for all investigated rainfall events, %
IST	1.95	–	0.84	–	4.0	–	3.3	–	–
1 (WM)	1.92	1	0.84	0	3.0	23	3.3	0	From 1 to 23
2 (NE)	1.90	2	0.84	0	3.6	9	3.1	5	From 0 to 15
3 (GS)	1.85	5	0.76	9	2.4	40	2.8	14	From 2 to 40
4 (LS)	1.74	11	0.76	9	2.2	46	2.8	17	From 9 to 46
PNV	1.41	26	0.74	11	1.2	70	3.0	8	From 11 to 70

with an all-year high LAI. In sum, all these forest effects lead to situations of a lower pre-event soil moisture. However, effects are limited by the available soil storage. As a consequence, the ‘forest effect’ on floods can be split into two general parts: retention by additional provided storage (1) and decelerating runoff by shifting water into slower pathways (2).

Land-use change measures are mostly long-term challenges and the appearance of the benefits may take decades. Another major restriction of adopted land-use as a component in an integrated flood risk management results from the fact that benefits of water retention in the landscape are mostly not directly noticeable at the point a measure is implemented. In the investigated region of Saxony the local authorities and the land owners are convinced that they do the possible according to their financial situation. Therefore, societal/legal constraints to improve the situation are indispensable. It has been shown that the more community-oriented scenarios are able to establish conditions for land-use types that consider water retention in conjunction with other targets of a sustainable development. That means that flood protection in the field (blanket coverage of the entire catchment area) as announced in the European Flood Directive must find its way to both the law and the subsidy policy. In that context it makes sense to identify synergy effects addressed in EU-Water Framework Directive.

Conclusions and Outlook

The land-use pattern in a landscape is complex, socio-economic driven, and long-term. Therefore, simple approaches (e.g. PNV, total afforestation) are not realistic since they only show general tendencies. Socio-economic methods have to be combined with state-of-the-art hydrological modelling approaches.

Integrated model approaches should deal with all competitive requirements of futures land-use and the prognosticated climate change. Consequently, synergies and feedbacks have to be detected and defined.

Benefits of land-use for an optimized flood protection are (mostly) not directly noticeable at the point a measure is implemented.

EU subsidy policy regarding land-use changes focuses on agricultural policy (CAP) followed by structure and cohesion policy. Hence, flood protection is at the best an additional benefit, but not a target!

Flood protection as announced in the EU Flood Directive must find its way to existing legal provisions and policies on subsidies. Therefore, synergy effects to the EU-Water Framework Directive have to be identified.

Acknowledgments Research was supported within the framework of the projects "Analysis, Assessment, and Implementation of Measures for Preventive Flood Protection in the Jeseniky Mountains/Czech Republic" (financed by the German Foundation for the Environment, DBU) and "FLOODsite – Integrated Flood Risk Analysis and Management Methodologies" (Integrated Project 6th EU-FP).

References

Barthels H, Malitz G, Asmus S, Albrecht FM, Dietzer B, Günther T, Ertel H (1997) Starkniederschlagshöhen für Deutschland – Kostra. Deutscher Wetterdienst, Offenbach am Main, Germany

Dane JH, Topp GC (eds) (2002) Methods of soil analysis: Part 1, Physical methods, 3rd edn. Soil Science Society of America, Madison, WI

Enke W, Kreienkamp F, Spekat A (2006) Neuentwicklung von regional hoch aufgelösten Wetterlagen für Deutschland und Bereitstellung regionaler Klimaszenarien mit dem Regionalisierungsmodell WETTREG 2005 auf der Basis von globalen Klimasimulationen mit ECHAM5/MPI – OM T63L31 2010 bis 2100 für die SRES – Szenarien B1, A1B und A2. Projektbericht im Rahmen des F+E-Vorhabens 204 41 138 "Klimaauswirkungen und Anpassung in Deutschland – Phase 1: Erstellung regionaler Klimaszenarien für Deutschland", 94 S

Evans E, Ashley R, Hall J, Penning-Rowsell E, Saul A, Sayers P, Thorne C, Watkinson A (2004) Foresight. Future flooding. Scientific summary: Volume I Future risks and their drivers. Office of Science and Technology, London, UK

Golf W, Luckner K (1991) AKWA – ein Modell zur Berechnung aktueller Wasserhaushaltsbilanzen kleiner Einzugsgebiete im Erzgebirge. Acta Hydrophys 32:5–20

LfUG (2006): Sächsisches Landesamt für Umwelt und Geologie – ColorInfraRed (CIR) – biotop type and Land-use mapping (in German). LfUG Abteilung 4 (Natur, Landschaft, Boden). Time reference: 1992/1993

Münch A (1994) Wasserhaushaltsberechnungen für Mittelgebirgseinzugsgebiete unter Berücksichtigung einer sich ändernden Landnutzung. Ph.D. thesis, TU Dresden, Fakultät für Forst-, Geo- u. Hydrowissenschaften

Münch A (2004) AKWA-M® – Teilflächen basiertes Wasserhaushalts- und Hochwassermodell. Dr. Dittrich & Partner Hydro-Consult GmbH, Bannewitz, Germany

Schmidt PA, Hempel W, Denner M, Döring N, Gnüchtel A, Walter B, Wendel D (2002) Potentielle Natürliche Vegetation Sachsens mit Karte 1:200 000. In: Materialien zu Naturschutz und Landespflege 2002. Sächsisches Landesamt für Umwelt und Geologie (Hrsg.), Dresden

Schwärzel K, Punzel J (2007) Hood infiltrometer – a new type of tension infiltrometer. Soil Sci Am J 71:1438–1447

Wahren A, Schwärzel K, Feger KH, Münch A (2009) Land-use effects on flood generation – considering soil hydraulic measurements in modeling. Adv Geosci 21:99–107

Wendroth O, Ehlers W, Hopmans JW, Kage H, Halbertsma J, Wösten JHM (1993) Reevaluation of the evaporation method for determining hydraulic functions in unsaturated soils. Soil Sci Soc Am J 57:1436–1443

Wind GP (1968) Capillary conductivity data estimated by a simple method. In: Rijtema PE, Wassink H (eds) Proceedings of the UMESCO/IASH symposium. Water in the unsaturated zone, June 1966, vol 1., pp 181–191

Chapter 26
Typological Up-Scaling of Wooded Peatlands

Elve Lode, Jüri Roosaare, and Margus Pensa

Introduction

About 5% (i.e. 292,700 km^2) of Europe is covered by peat and peat-topped soils (0–30 cm) with a threshold of >25% organic carbon, and they have a strong northern bias in their distribution (Montanarella et al. 2006). Based on the national level estimations (Vasander et al. 2003; Oleszczuk et al. 2008; Finnish statistical 2006; Hånell and Magnusson 2005; Forest Service 2007; Paavilainen and Päivänen 1995a; Adermann 2007) it could be concluded that about 29%, or 82,310 km^2, of the peat-covered areas of the eight European peat rich countries (Finland, Sweden, UK, Norway, Ireland, Estonia, Germany) have been drained for forestry. Already since 1773 the lowering of the groundwater level, via a man-made drainage system, was the main approach to improve forest growth on waterlogged peatlands (Paavilainen and Päivänen 1995b), and this method reached the highest level in northern and eastern Europe, the British Isles, and some parts of North America during the 1960s and 1970s (Laine et al. 2006). The long term practice of peatland forest drainage has shown that the necessary depth of the groundwater (GW) level for peatland forest growth is 35–55 cm below the soil surface (Heikurainen 1967; Toth and Gillard 1988), which traditionally is achieved with average distances of

---

E. Lode (✉)
Tallinn University, Institute of Ecology, Uus Sadama 5, 10120 Tallinn, Estonia
and
Department of Soil and Environment, Swedish University of Agricultural Sciences,
SE-75007 Uppsala, Sweden
e-mail: elve.lode@tlu.ee

J. Roosaare
Institute of Ecology and Earth Sciences, University of Tartu, Vanemuise St 46, 51014
Tartu, Estonia
e-mail: juri.roosaare@ut.ee

M. Pensa
Tallinn University, Institute of Ecology, Uus Sadama 5, 10120 Tallinn, Estonia
e-mail: margus.pensa@tlu.ee

M. Bredemeier et al. (eds.), *Forest Management and the Water Cycle: An Ecosystem-Based Approach*, Ecological Studies 212, DOI 10.1007/978-90-481-9834-4_26,

25 or 50 m between ditches in drainage systems (Valk 2005). From this, it could be estimated that for growing forest on 29% of European peat and peat-topped soils the total length of ditches to be created would be between 1.6–3.3 × 10^6 km (!).

Several recent study results of mire surface dynamics show a widespread *“natural”* forest expansion on the Boreal peatlands due to the complex interactions of human-induced drainage, climate change, fire events and isolation of peatlands within an agricultural plain. In Pellerin and Lavoie (2003) it was summarised that even the peatlands which were known to be resistant to disturbances may overgrow rapidly, and experience major changes when they are subjected to multiple disturbances, and especially to human influences.

Plant macrofossil analyses of Canadian peatlands show that, prior to European colonization (ca. eighteenth century), the peatlands of Canada were predominantly open landscapes, dominated by *Sphagnum* mosses which subsequently became forested with little or no *Sphagnum*. Analysis of aerial photographs over the south Quebec region indicated that widespread forest expansion occurred in all mires between 1948 and 1995, and the percentage of the total peatland area occupied by the forest increased from 22.5% to 56.5% (Pellerin and Lavoie 2003).

The study of *Sphagnum*-dominated mire ecosystems in southern and central Sweden show a high degree of 40-year plant dynamics on natural peatlands, and increment of tree coverage (mainly *Pinus sylvestris*) on virgin mires in east Sweden, at least for the last 50 years (Åberg 1992, and Gunnarsson and Rydin 1998 ct. in: Gunnarsson et al. 2002). An increment of tree cover and hummock-type microforms over the Åkhultmyren mire (southern Sweden) was studied from aerial photographs from the period 1947 to 1977 (Ihse et al. 1992 ct. in: Gunnarsson et al. 2002).

By 1953 it had already been stated that the structural studies of peatland or mire surfaces, which were based on recognition of the plant cover and/or micro-topographical features on aerial photographs, was an important approach for the identification of water dynamics and mire hydrology on water-saturated landscapes (Ivanov 1953).

Modern *spatial monitoring* has opened up several possibilities to investigate both the peatland surface topography and the plant cover via aerial photographs and satellite images. Data quality improvement has been achieved by producing high quality *multiple spectral images* via the use of multispectral sensors for land surface scanning. Different sets of *spectral images* have been adapted for recognition of certain types of surface patterns on the ground, where the *near infrared* (*NIR*) and *red* spectral regions were used for interpretation of the plant cover.

Improvement of image resolutions, increment of number of used colour channels (e.g. hyperspectral images) or number of sensors used – these are the continuing contemporary technical progress toward improvement of quality and quantity of remote sensing databases (Lillesand et al. 2008).

Aerial photography is the oldest method of remote sensing. In the course of time the first panchromatic (black-and-white) photos, sensitive to visible light, were replaced by colour photographs, – with stereo pairs and *colour infrared* (*CIR*) images. Analogue cameras have been increasingly superseded by digital cameras. In *CIR* images the *NIR*, *red*, and *green channels* have been applied in forestry, agriculture and other environmental fields.

In several cases the ortophoto *shadows* (i.e. tree shadows) were used for the determination of tree species, their heights and other forest parameters (Groom et al. 2006; Ihse 2007). The shadows of Landsat TM images are often blurred, and indirect database calculations have to be used for their extraction from the image datasets (Liira et al. 2006). In high resolution imagery a single tree crown contributes several pixels with varying illumination across the crown, and therefore separation of individual tree crowns is possible (Brandtberg and Walter 1998; Gong et al. 1999; Quackenbush et al. 2000; Leckie et al. 2003). In this case the quality of quantified tree cover interpretation is forest type and land level calibration dependent (Leckie et al. 2003), and the image based interpretation results could be almost equivalent to the ground level surveyed results (Brandtberg and Walter 1998).

The aim of this chapter is to introduce the GIS-based case study results of the small wooded Selisoo mire, located in the Eastern part of Estonia and belonging to the *Ramsar*'s Muraka mire complex. This has been formulated as follows:

1. To summarise the principles of scale-dependent mire surface pattern mapping
2. To study the reflection of different mire disturbances on ortophoto-delimited mire ecotopes
3. To study the reflection of different mire disturbances on the tree stands
4. To introduce the GIS based typological up-scaling of remote sensing datasets to the wooded peatland ecotopes

Features of Aerial Photo Mapping of the Peatlands

Features of Peatland Mapping

There are three important features of peatland mapping which can be identified: (a) the mire boundaries (i.e. "zero" depth of the peat soil limits; in: Ivanov 1975); (b) the surface water bodies (i.e. streams, ditches, pools and lakes; in: Eggelsmann et al. 1993), and (c) the *horizontal* and *vertical structure* of the patterned mire surfaces (Masing 1982). While the horizontal structure of the mire surfaces is formed by the *positive* (e.g. hummocks, ridges) and *negative* (e.g. lawns, carpets, mud-bottoms) microrelief elements called *microforms* (e.g. Ivanov 1975) then the vertical stratification depends on the mire plant cover, which could be described either by the *vegetation layers* i.e. tree, shrub, field and moss layer or *vegetation horizons* i.e. tree canopy, tree stem, shrub canopy, herb leaves, moss surface horizon (Masing 1982). In the latter case the peat layer below the vegetation horizon forms a *litter peat horizon* (Malmer 1988).

Distinct patterns of peatlands are often formed by the ridges and depressions that alternate in ladder-like or net patterns, and they can be found on bogs, fens, or a combination of the two (Rydin et al. 2006). The bogs with patterned tree cover were divided into (a) *completely opened bogs*, (b) *wooded on the margin and open in the centre*, (c) *completely wooded*, and (d) *wooded in the centre and opened on the margin*.

The bogs with opened centres might be *centre wet* (dominated by the *Sphagnum* lawns) or *centre dry* (dominated by dwarf shrubs of *Cladonia*, and *Sphagnum* hummocks). Wooded centre bogs could be formed into *wheel spoke bog*s (patterned trees with radiating lines like spokes in a wheel) and *feather bogs* (patterned trees with a linear crest of trees with lines coming off like veins of a feather) (Ibid).

Both the climate and anthropogenic drainage increase the plant cover changes on peatlands, and the extension of forests to these. Tree growth on peatlands causes profound changes in peatland ecosystems via altering the hydrological conditions (Ohlson et al. 2001). Wooded conditions on peatlands influence the amount of water stored in the soil by intercepting rain fall, withdrawing water from peat, and influencing evapotranspiration rates (Anderson et al. 2000; Frankl and Schmeidl 2000; Van Seters and Price 2001) together with changes to the hydro-physical properties in the underlying peat column (Eggelsmann et al. 1993; Paavilainen and Päivänen 1995a).

In general a *patterned surface* of peatlands is a reflection of integrated ecological processes of mire living plant production, peat accumulation, and water dynamics (Rydin et al. 2006). It takes place due to comparably shallow aquifers of the mires, called *acrotelm* (Ivanov 1975, 1981; Van der Schaaf 1998, 1999).

In the course of mire ecology research development the structure for mire mapping scale levels were formed and demonstrated in many studies (Table 26.1). In ecohydrological studies the *macro*, *meso* and *micro* scale spatial levels were defined for different mire research areas (e.g. restoration or protection of mire ecology), where the mire biotic and abiotic factors were equally lionised (Van der Schaaf 2002). Since the difference in scales between the meso-level and the micro-level is substantial (km^2 vs dm^2 m^2) intermediate levels, i.e. *ecotope* and *community-complex* levels were introduced, whereby the *ecotope-scale* level was defined as a compromise between the larger plant cover zone (*biotic* conditions) with a certain uniformity in *abiotic* conditions, i.e. a specific range in mean groundwater levels, a specific range of water table fluctuations or a specific water type e.g. atmospheric, lithospheric or mixed (Ibid).

Table 26.1 Generalised levels and scales for empirical mire research and mapping (Adapted from Masing 1998, Rydin et al. 2006)

Masing (1998)		Rydin et al. (2006)
Levels	Scales	Scales
Clonal and population	–	
Ceonotic	1:10–1:100	
Microstructural	1:100–1:1,000	0.25–100 m^2
Microtope or coenocomplex	1:1,000–1:10,000	100 m^2 to few km^2
Mesotope or mesostructural	1:10,000–1:100,000	0.5–20 km^2
Macrotope or macrostrucutural	1:100,000–1:10,000,000	Few km^2 to >20 km^2
Regional	1:10,000,000–1:100,000,000	Thousands of km^2

Aerial Photo Mapping of Peatlands

In remote sensing, peatlands were easily distinguishable in near-infrared satellite images with wave length of 0.75–0.9 µm of the Landsat TM (Aaviksoo et al. 2008). High spatial resolution of remotely sensed data (4 m pixel size) has been shown to be ideal for the study of bog landscapes (Milton et al. 2005), whereas the short-wave infra-red region of 1.55–1.75 µm had a good resolution for the study of *wetness index* distribution over the peatlands (Hardisky et al. 1983).

Aaviksoo et al. (2008) stated that the aerial photos as well, from *Ikonos* and *QuickBird* images, were suitable for mapping *mire microforms* (with a spatial resolution of 0.6–4 m pixel size in images), and SPOT HRV, Landsat 5 TM and Landsat 5 ETM+ (pixel size of 10–30 m) – for mapping of *mire microtopes* and *mire complexes*. The AVHRR sensor data with a ground resolution of ≈1.1 km (NESDIS 2009) was useful for mapping individual *mire massifs* or entire *mire landscapes*. In this way, from historically established seven *mire mapping ground levels* (Table 26.1), the five lower levels were covered by the remotely gained data systems named above (Aaviksoo et al. 2008).

Advanced technologies enabled the production of remote sensing data for the first two mire mapping levels, whereas the different airborne laser scanners (e.g. LIDAR – LIght Detection And Ranging) were used for the identification of different vertical and horizontal peatland tree parameters, such as the tree density and changes in their height and growth (Genc et al. 2004; Hopkinson et al. 2004; Waser et al. 2006).While the Landsat images were the most suitable for the European landscape ecology study (Groom et al. 2006) the structural studies of the peatland surfaces were still favourable from the *aerial photos* (Langanke et al. 2007) and/or *QuickBird* images (Milton et al. 2005).

Aerial photos have been used for reconstruction of temporal changes in bog forests (Cameron et al. 2000; Gunnarsson et al. 2002; Linderholm and Leine 2004) and they are still used in *temporal* studies of peatland vegetation changes (Ihse 2007; Langanke et al. 2007). In Jauhiainen et al. (2007), the study of the *tone* and *texture* values, extracted from the aerial photographs of Finnish peatlands, were stated to be useful for differentiation of forested site types of drained peatland from undrained and open site types. Using canonical correspondence analysis, some correlations were found between tone values and peatland site types, but they were not identified at an accurate level from aerial photographs for the mosaic like structure of vegetation inside the site type.

From the peatland forestry management point of view it is important to know that many gullies and channels can be clearly identified, measured, and mapped onto aerial photographs in open areas. The LIDAR technique can be used to locate and map gully systems (down to approximately 3 m top width) that lie beneath thick canopy (James et al. 2007). This mapping capability is important for gully inventories for hydrological modelling (e.g. flow accumulation models), determination of land use treatments, and estimation of soil erosion and sedimentation volumes (Ibid).

Linderholm and Leine (2004) summarised that aerial photographs are a powerful tool for detecting spatial and temporal changes in vegetation coverage of peatlands, and especially at the regional/country level, but there were difficulties for separation of the climatological causes of vegetation changes from anthropogenic causes, especially if signs of ditches or logging are not visible on analysed images. Using information from aerial photographs was limited also by the gap between the real beginning of the forest expansion and its photographic evidence, since samplings have to reach a minimal height before being detectable on aerial photographs. In that sense land survey dendrochronology was an excellence tool to determine significant changes in the growth environment and reflection of these to the tree growth rates, magnitude of tree coverage, and timing of crucial changes in peatland conditions (Ibid). However in Waser et al. (2007) the combination of remote sensing data with regression analysis and fractional cover approaches was shown to be appropriate for fractional tree/shrub cover mapping and assessing changes of forest area in a peatland biotope. Both the accuracy of the forest *masks* and the fractional tree/shrub covers strongly depend on the accuracy of digital surface model (DSM) data. It was stressed by Waser et al. (2007) that the usage of a dense and accurate DSM and digital terrain model (DTM) were absolute prerequisites for the derivation of accurate topographic parameters, which in turn were used to derive the forest masks and the fractional tree/shrub covers.

An object-based image analysis (OBIA), using *eCognition* software, also gave promising results for the highly patterned, but less tree covered, peatland surfaces (Burnett et al. 2003), which in principle was called the *multi-scale segmentation/ object relationship modelling* approach (Burnett and Blaschke 2003).

Analyses of Wooded Peatlands

Ortophoto and Ground Level Parameterisation of Wooded Peatlands

Ortophoto-based identification of mire ecotopes for mire landscape in forestry management conditions were carried out on the small Estonian Selisoo mire. The digital ortophoto of the Estonian Land Board that was used is a processed aerial photograph with distortions caused by terrain relief and camera tilt relative to the ground at the moment of exposure and camera central projection removed. The Selisoo mire ortophotos were produced with an analogue camera (Rc20) during a routine remote land survey flight period from 14th May 2005 to 25th August 2005. Digital ortophotos in Estonia are for *regular* use, and they are available as *true colour images* in *MrSID* format with a pixel size of 0.5 m.

Manual delimitation of the mire ecotopes took place in the *MapInfo* environment, based on clear differentiation of the microforms in the ortophoto, their orientation pattern and relationship to tree coverage (Fig. 26.1). Each of the ecotopes

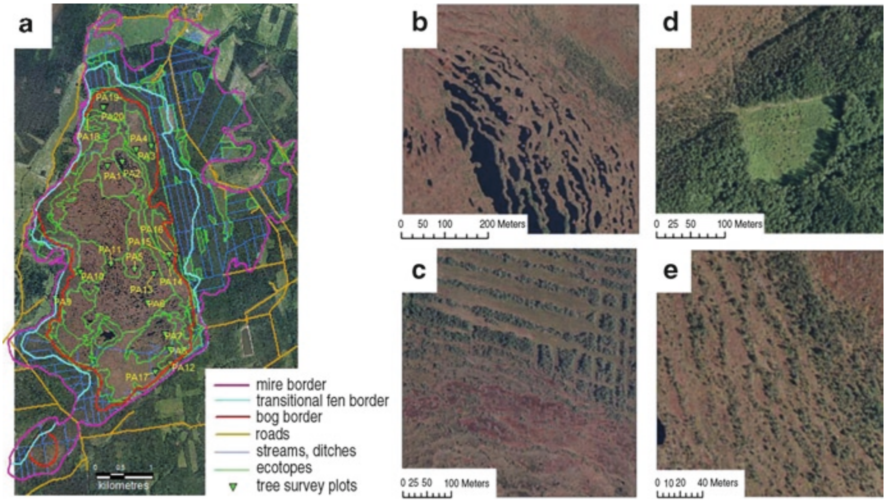


Fig. 26.1 Ecotopes of Estonian Selisoo mire, manually identified from the standard ortophoto of the Estonian Land Board 2005, where roads, streams and ditches were identified from the Estonian Basic Map and mire borders from the Estonian Soil Map (**a**) together with excerpts from the same ortophoto layer: (**b**) oriented pool-ridge surface pattern on the bog marginal slope (marked PA2 in **a**), (**c**) hollow-ridge surface pattern beside the self-recovering peat pit surface pattern with treed peat beds between (marked PA19 in **a**), (**d**) mineral soil covered forest transition toward the mire lagg on the fen peat extension (close to PA17 in **a**), and (**e**) oriented treed ridge pattern beside the forest band (marked PA14 in **a**)

was facilitated with the ecotope ID (ID_{ecot}), the surface pattern descriptions (incl. the tree coverage) and visually estimated total tree coverage on the ecotope (Tables 26.2 and 26.3). Beside the ecotope layer the digitised pool layer was created and the *MapInfo Query* of *Calculate Statistics* was used for calculation of the ecotope areas, their coverage with the pools and for calculation of the length and density of linear elements on ecotopes (i.e. ditches and roads).

In parallel to mire ecotope identification, the survey of the ground level tree stands was associated with some environmental parameter measurements that were carried out on 20 *tree plots* during the growing periods of 2007–2008 (Fig. 26.1 and Table 26.4).

Tree cover parameterisation was the main objective of the field survey. It was conducted within the circular plots of 100 m^2 (i.e. radius of plots of 5.6 m), where the total number of trees was counted and the stem diameter of the trees with tree heights of $h > 1.3$ m were measured at a stem height of 1.3 m. Trees with tree heights of $h > 1.3$ m and stem diameters of $d > 5$ cm were cored for tree ring analysis.

After the tree stem diameter measurements, the *basal area* of the trees ($BA_{1.3}$) was measured and expressed in cm^2 per 100 m^2. Tree ring analysis (i.e. the width of the tree rings formed in a given year) was carried out using the measurement system of *WinDENDRO* (Regent Instruments of Canada Inc., Nepean, Ontario).

Table 26.2 Description of Estonian Selisoo mire ecotopes, manually delimited from the ortophotos, where: ID_{ecot} = code for the ecotope type; n = number of the ecotope polygons

		Pools in ecotope	Tree/forest coverage in ecotope					
			Total coverage	Pattern description, %				
[a]ID_{ecot}	n	%	%	Single trees,	Trees in groups	Trees on ridges	Scattered trees	Forest
2	1	7.2	20	70	30			
10	2	[b]0.2;[c]5.2	[b]10;[c]5	50		50		
13	1	2.1	50			65	35	
9	1	31.6	20	10	10	80		
12	2	[b]12.0;[c]11.2	40			50	50	
4	2	[b]1.5;[c]0.4	15	80		20		
3′	4		50			100		
7	7		95					100
6	1		30				100	
3″	2		50				100	
99	1		95					100
5	1	3.4	50	1		99		
14	3	[b]0.6;[c]0.3;[d]0.3	50			100		
8	2	[b]7.4;[c]7.3	75				10	90
15	3	[b]1.8;[c]0.2	[b]5;[c]20	10		90		
11	1	3.5	20	50	50			
1	4		75				100	
77	1	0.1	30		50	10	40	

[a]Ecotope surface pattern description seen in Table 26.3 [b, c, d]Corresponding values, if more than one ecotope under the certain ecotope type

Table 26.3 Ecotope surface pattern description for the Table 26.2

ID_{ecot}	Surface pattern description
2	Open bog with weir-oriented hollows and some pools
10	Hollow oriented, hollow-lawn microtope with some trees
13	Hollow-lawn microtope with some pools
9	Pool-ridge microtope with well oriented ridges
12	Pool-lawn-ridge microtope with trees
4	Lawn type open bog
3′	Terminated peat cutting, trees in ridges
7	Edge drained bog forest
6	Managed open peat field
3″	Terminated peat cutting, scattered trees
99	Drained peatland forest
5	Lawn-ridge microtope with pools and well oriented ridges
14	Lawn-ridge microtope with well oriented ridges
8	Forested pool-ridge microtope
15	Hollow microtope with some tree covered ridges
11	Hollow-ridge microtope with some pools
1	Lawn type microtope with scattered tree growth
77	Edge-drained bog with scattered trees, some in ridges and groups

Table 26.4 Results of the ground level tree survey of the Selisoo mire ecotopes during the growing periods of 2007 and 2008, where: ID_{ecot} = code for the ecotope; ID_{tree} = code for the tree plot; D_{trees} ($Nr_{all}/Nr_{h>1.3\ m}$ per 100 m^2) = density of the trees, where Nr_{all} = count of all trees; $Nr_{h>1.3\ m}$ = count of trees with the tree height of $h > 1.3$ m; $BA_{1.3}$ ($cm^2/100\ m^2$) = tree basal area, A_{mean} (years) = mean tree age; H_{peat} (m) = mean peat depth

ID_{ecot}	ID_{tree}	Mire site type	D_{trees}’	$BA_{1.3}$	A_{mean}	H_{peat}
2	PA1	Wooded lawn-ridge	20/6	15.6	22	5.55
9	PA2	Wooded pool-ridge	38/8	28.9	69	5.50
12	PA13	Ridge-hollow	25/0	0.0	35	5.80
	PA6	(Wooded) ridge-hollow	60/6	38.9	36	6.89
	PA11	Ridge-hollow	41/1	38.5		5.80
4	PA5	(Wooded) lawn-ridge	37/2	60.5	37	6.53
3′	PA18	Wooded ridge - terminated peat cutting	36/27	57.0	74	3.10
3′	PA19	Wooded ridge - terminated peat cutting	24/9	161.0	43	3.05
	PA20	Wooded ridge - terminated peat cutting	86/8	3.9	41	3.60
7	PA12	Drained peatland forest	36/28	28.6	83	1.20
7	PA16	Drained peatland forest	32/26	40.9	59	2.76
	PA4	Drained peatland forest	34/21	92.3	56	3.43
6	PA15	Wooded hummock	61/19	8.4	18	4.71
3″	PA3	Wooded hummock – terminated peat cutting	33/16	26.5	41	3.91
99	PA9	Drained peatland forest	13/13	121.5	55	2.87
14	PA14	Bog forest	79/33	19.9	29	6.41
8	PA7	Bog forest	64/48	21.1	41	4.69
11	PA10	Hollow-ridge	49/2	4.41	94	4.27
1	PA8	Wooded hummock	40/6	18.3	26	4.10
	PA17	Bog forest	30/22	36.0		3.38

Results of the tree ring measurements were standardised by using the negative exponential or linear function. The means of the series of indexed height increment were calculated for all collected tree rings over the Selisoo mire area, as well as for each ecotype separately, by dividing the measured radial increment with the results from exponential or linear function. Standardisation of the tree-ring time series was made with *ARSTAN* software.

During the tree stand survey the mire *site type* and the depth of the peat deposit were determined and the mire groundwater (GW) level was measured in the plot location. The mire *site type* for the studied plots was determined according to the classification of Estonian bog sites proposed by Paal et al. (1998).

GIS Based Analysis of Wooded Peatland Ecotopes

Ortophoto polygons of manually delimited ecotopes formed the framework for a subsequent GIS based Selisoo mire data analysis. Performing this at the ecotope level, instead of pixel-by-pixel calculations overall mire extension, enabled the limitation of the size of the analysed data sets and calculation time, following a general principle of object-based image analysis (Burnett et al. 2003; Blaschke et al. 2008).

The main objectives of the GIS based analyses of remote sensed ecotope datasets in the current case study were:

1. To study the best classification schemas for the Selisoo mire ecotopes by scaling up procedure in wooded bog condition
2. To establish possible GIS based relationships between the land surveyed tree plots with pixel-based classification schemes

There are two classification approaches for the interpretation of remote sensed images – *supervised* and *unsupervised* classification of pixels, or their associations (segments). In the case of *supervised classification* the *land truth statistical etalons* should be used for the desired pixel classes. The aim of data processing in this case is to sort each *spatial-unit* to the most suitable *land-truth* class. In the *unsupervised-classification* the set of the land truth *pixel* classes are not predefined, but will be chosen subsequently from a series of corresponding software *iteration runs* by man examination afterward. *Cluster analysis* is the most frequently used method of unsupervised classification. The main goal of this approach is to calculate the series of *clustered* or *classified* datasets, together with corresponding functional relationships of analysed images, suitable for mapping the *land truth* coverage pattern (Lillesand et al. 2008; Eastman 2006).

Cluster analysis of the Selisoo mire spatial datasets started from the decomposition of true-colour spaced RGB ortophoto into seven-colour channel images, namely *blue* (B), *green* (G), *red* (R), *yellow* (Y), *hue* (H), *lightness* (L) and *saturation* (S) colour channel images (later here and after – channel image(s)). In fact, the RGB ortophotos are already presented by the *red, green* and *blue* coloured channel images, but the HLS (*Hue-Lightness-Saturation*) colour system may uncover additional qualities of the high resolution aerial or satellite images (Komura et al. 2005). This was demonstrated with the Selisoo mire ecotope images by the high linear correlations of $r > 0.9$ between datasets of R, G, and B channel images and much lower r values between the H, S, and L channel datasets (e.g. $r_{SH} = 0.34$).

The channel image transformations for the Selisoo mire were performed by the ESRI's *ArcGIS* and Clark Lab's *Idrisi Andes edition* (*IAe*) software, and the programme *LSTATS* (2009). The *IAe* was used for the spatial data analysis of the mire ecotope images and the *ArcGIS* software – for the integration of spatial datasets, including visualization and metadata documentation.

The size of each channel image throughout the bog extension in Selisoo mire was 10,000 × 20,000 pixels and transformed ecotope channel images (extracted from the ortophotos) varied from 500 × 600 to 3,700 × 7,500 pixels (see in Fig. 26.2a). All land surveyed 20 tree plots (PA's in Table 26.4) were also subjected to the channel image transformation. The size of each of the tree plot images was 400 pixels (see Fig. 26.2b and c). From all the images the descriptive statistical parameters (minimum, maximum, mean, range, and standard deviation of the pixel values) were extracted by the *IAe* spatial queries, and they were used for the

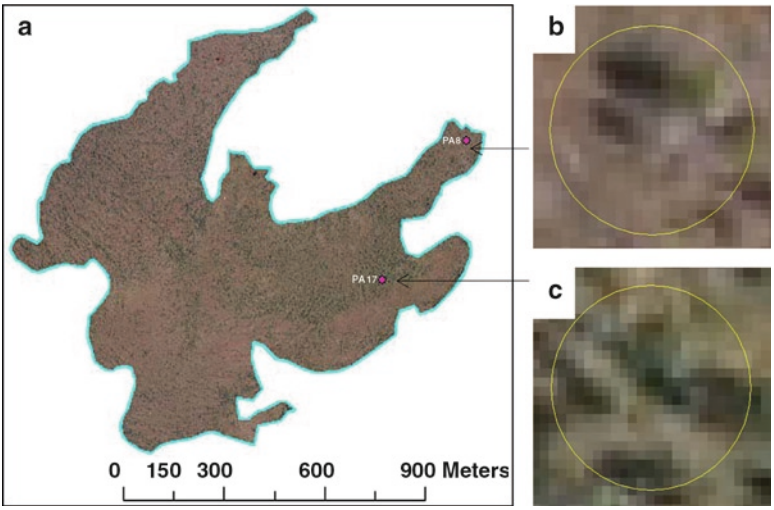


Fig. 26.2 Example of (**a**) delimited ecotope polygon on the ortophoto, where the ecotope is with semi-natural or secondary disturbances following by the scattered tree coverage on low oriented microforms ($\text{ID}_{ecot}=1$ in Table 26.2); (**b** and **c**) corresponding land surveyed tree plot images (PA8 and PA17 in Table 26.4)

constructing of correlation relationships in *Excel* environment with the field data from the tree plots survey. All decomposed channel images of surveyed tree plots constituted the empirical GIS database of the tree plots for the following *cluster* analyses performed by the *IAe*.

As a first step of cluster analysis a “broad classification” (Eastman 2006) was performed. A meaningful number of clusters/classes for the following cluster analysis was defined by the *histogram peak technique.* Thereafter the *Maximum Likelihood* procedure of an iterative self-organizing module of *ISOCLUST* (Eastman 2006) was used to find the best clustering combination for a predefined number of classes (C_k). The *runs* of the *ISOCLUST* were repeated independently for the each set of ecotope channel images with the class number changing from the range C_{10} to C_3 included. The class number decreasing procedure from the C_{max} downward C_{min} in the sequence of the *ISOCLUST* runs can be regarded as a *typological up-scaling* of remote sensed datasets of mire ecotopes and tree plots. The final procedure of classified images was interpretation of the derived formal classes and their renaming according to expert knowledge and corresponding *ortophoto land cover* conditions.

The *ISOCLUST* runs with different numbers of iterations and *C* sets resulting in a number of different *image clustering schemas* for clustered image formation, the so-called *clustering hierarchy*. The strength of the clustered schemas in the clustering hierarchy (e.g. classes of the C_6 and C_4 images) was estimated by the cross-tabulated *Cramer’s V* values between the chosen clustering schemas, where *V* values closer to “1” indicate strong cluster *associations,* and *vice versa* for *V* values closer to “0” (Bishop et al. 1975).

Results

Ortophoto and Ground Level Parameterisation of Wooded Peatlands

- Ortophoto-based ecotope identification of Selisoo mire resulted in 18 types of ecotopes with 39 polygons (Fig. 26.1a, Table 26.2) and they were classified as follows:
  1. Ecotopes with natural microforms, where
     (a) Microforms were oriented (ID_{ecot} = 2; 10; 13; 9; 12)
     (b) No orientation in microforms (ID_{ecot} = 4)
  2. Ecotopes with man-made disturbances, where
     (a) Microforms were oriented (ID_{ecot} = 3′)
     (b) No orientation in microforms (ID_{ecot} = 7; 6; 3; 99)
  3. Ecotopes with semi-natural or secondary disturbances, where
     (a) Microforms were oriented (ID_{ecot} = 5; 14; 8)
     (b) No orientation in microforms (ID_{ecot} = 15; 11; 1; 77)

Results of the *MapInfo* based ecotope datasets of the 2,051 ha Selisoo mire showed that at the current stage more than half of the mire (i.e. 65% or 1,333 ha) is covered by the drained forest. Wooded tree coverage constituted 15% (i.e. 308 ha) of the mire extension, mainly on the bog part of the mire, and only about 19% (i.e. 396 ha) of the total mire extension remains, probably, in its natural state. Almost 90% (100 km) of the ditches and 100% (8 km) of the roads were located on the drained-for-forestry fen and transitional fen peat extension, where the mean density of the *line* objects were about 100 m/ha.

The areal coverage of ecotopes with oriented microforms constituted about 90 % of the presumed natural part of the bog area (Table 26.5), where the visually well distinguishable microform orientations accompanied with comparably large numbers of pools (about 1,700 pools on 11% of the ecotope areal coverage, Table 26.5) and low tree coverage (between 10–25% of the ecotope areal coverage, Table 26.2). Ecotopes with no microform orientation constituted 95% of the ecotope areal coverage with man-made disturbances. Usually these ecotopes were accompanied with high tree coverage (between 50% and 95% of the ecotope areal coverage, Table 26.2). Ecotopes with well oriented and wooded ridge microforms constituted 44% of the semi-natural or secondary-disturbed mire ecotopes, where the tree coverage were estimated to be 50–70% of the ecotope area. Other semi-natural ecotopes had visibly no distinguishable microforms orientation and usually they were covered with no oriented single trees, or trees in groups (Tables 26.5 and 26.2). During the last 50 years the open water area of the mire lake been decreased by 67% (Table 26.5).

The ground level tree layer density and the basal areas of the trees ($BA_{1.3}$) showed a great variation across the study plots over the whole Selisoo mire area (Table 26.4), partly following the mire GW levels measured on the days of the tree survey (Fig. 26.3).

Table 26.5 Areal distribution of classified ecotopes (F_{ecot}) and pools (F_{pools}) identified on Selisoo mire ortophoto (see also Table 26.4)

Ecotope class	F_{ecot}		
	ha	%	F_{pools}, ha
1. Ecotopes with natural microforms			
(a) Microforms were oriented	355.5	17.3	39.2
(b) No orientation in microforms	40.3	2.0	0.5
2. Ecotopes with man-made disturbances			
(a) Microforms were oriented	56.4	2.8	–
(b) No orientation in microforms	1,123.1	54.8	–
3. Ecotopes with semi-natural or secondary disturbances			
(a) Microforms were oriented	138.3	6.7	4.7
(b) No orientation in microforms	173.5	8.5	1.2
Lake	8.9[a]/2.9[b]	0.4	
Non forested lagg area	155	7.6	–
Total	2,051	100	45.6

[a] Lake with open water area in 1950
[b] Current lake with open water area

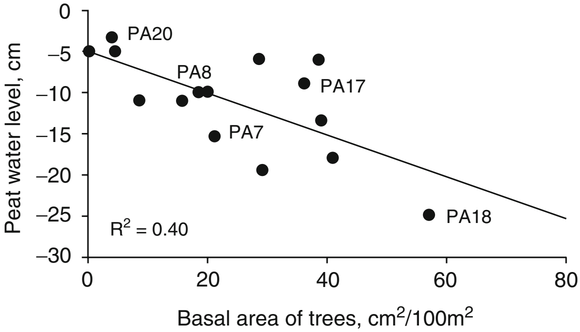


Fig. 26.3 Relationship between stem basal area and mire groundwater, measured during the Selisoo mire land survey in June–Sept 2008

The time span of the series of all analysed raw tree-ring data was from the years 1872 to 2008, with a radial increment ranging from 0.2 to 2.0 cm (Fig. 26.4). The high peak of the radial increment of the trees from the terminated peat cutting plots was in the 1940s, indicating with a high probability the influence of the lowered mire GW level, due to peat-cutting activity. The increase in radial growth in all tree study plots since the 1970s indicates improved tree growth conditions on the plots, and improved possibilities for the establishment of a new generation of trees, presumably due to a high level of alkaline air pollution transported from the oil-shale industry (Liblik et al. 2003), and also a lowered and stabilized GW level on the mire.

The correlation results of the standardized chronologies of the tree radial increments across the whole Selisoo mire indicate that the growth of the trees

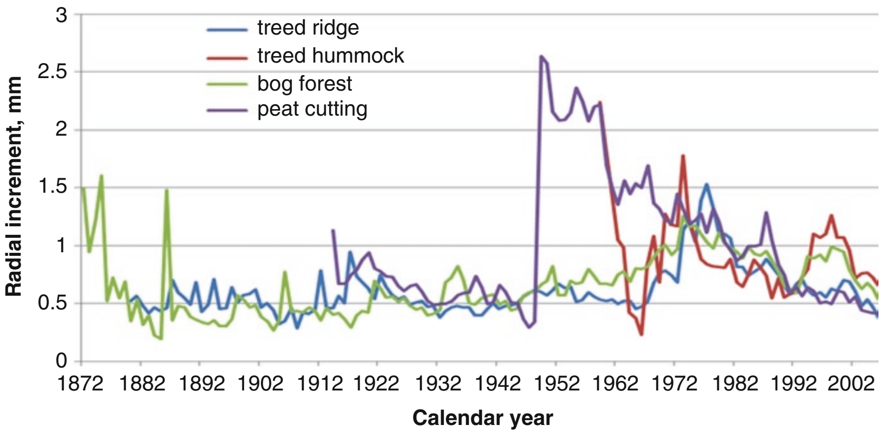


Fig. 26.4 Radial increment chronology of *Scots pine* tree stems from all land surveyed tree plots on Selisoo mire, where the plots were classified according to the mire site type of the tree plot, i.e. treed ridge and treed hummock, bog forest and terminated peat cutting (Table 26.4, see also Fig. 26.1)

Table 26.6 Results of the cross-tabulated Pearson's correlations between the tree radial increments measured in all of the plots of the Selisoo mire, where the plots were classified according to the mire site types (Table 26.4); significant correlation ($p = 0.05$) is marked with an asterisk

Mire site type	Wooded ridge	Wooded hummock	Bog forest
Wooded hummock	0.16		
Bog forest	0.21	0.45*	
Peat cutting	−0.15	0.29	0.28

over all time span between 1872 and 2008 was independent of tree plots. The highest correlation of the tree radial increment was between the trees which were sampled on the *bog forest* and the *wooded hummock* site types (Table 26.6). However, there were some years that indicated increased tree growth in all the studied ecotopes (e.g. 1986 in Fig. 26.4). The highlighted chronologies of the tree radial increment of the ecotope trees at terminated peat pit cutting ("ortophoto" in Fig. 26.5), pool-ridge ("ortophoto" in Fig. 26.6) and lawn type (Fig. 26.2a) mire sites showed a clear improvement of tree growth conditions since the 1950 when the beds/ridges of the pit type peat cutting area were especially suitable for tree growth following peat cutting termination (Fig. 26.7). It should be stressed that the tree density of the young trees of PA20 were higher due to overgrowing of the neighbouring peat pit while it was comparably low on PA19 plot on the pit bed (Table 26.4).

In general, the number of young trees was comparably low in older drained forest areas, but there was a significantly higher number of the young trees classified in this study as *ecotopes with semi-natural or secondary disturbances,* which is

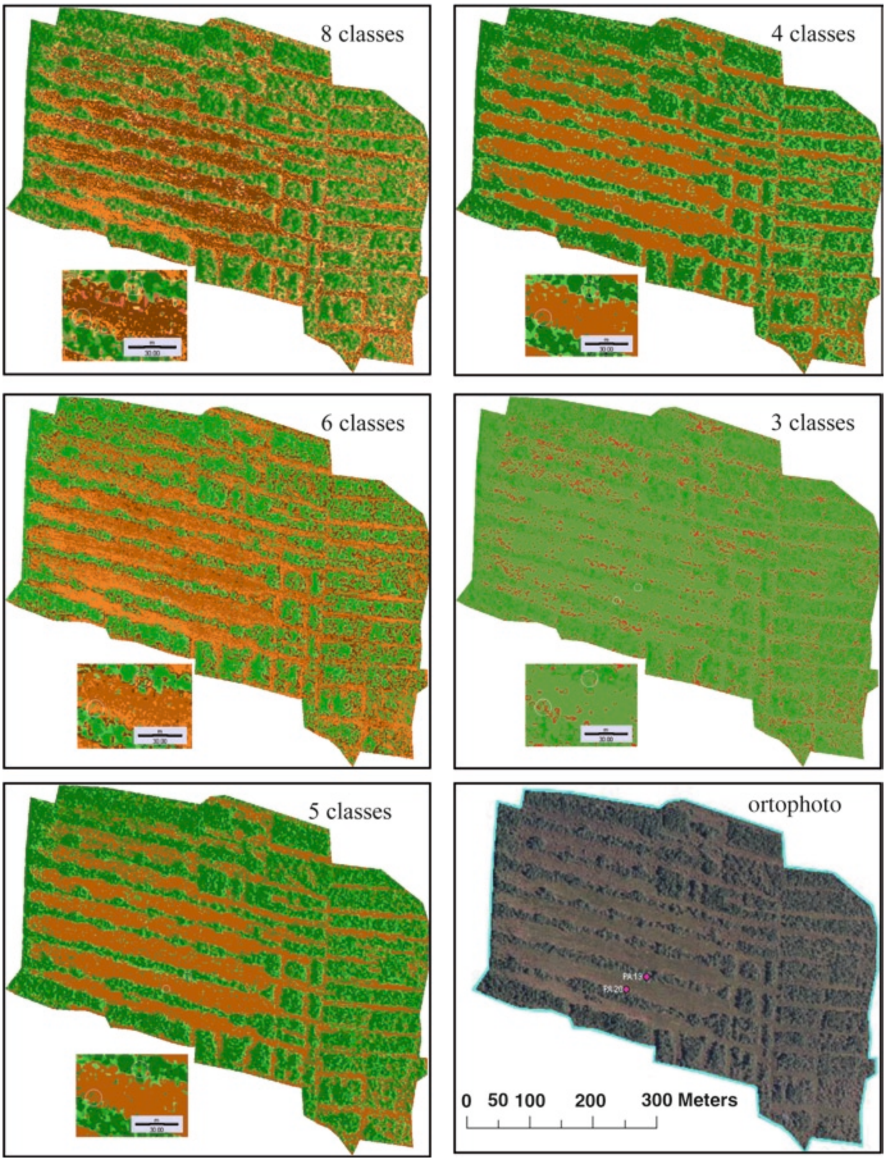


Fig. 26.5 Visualized *typological* up-scaling results of the Selisoo mire ecotope with well oriented and wooded ridges on terminated peat cutting pit (ID_{ecot}= 3′ in Table 26.2), where the up-scaling was made from the C_{max} = 8 toward the C_{min} = 3 included. Ground level surveyed tree plots (PA19 and PA20 in Table 26.4) are visualised on a delimited ecotope ortophoto image

reflected in the stem radial increment of the PA7 and PA17 tree plots and especially those of PA8 (Table 26.4, Fig. 26.7). Thus, the radial increment of *Scots pine* may reveal important changes of environmental conditions in mire landscapes general, and particularly in bogs.

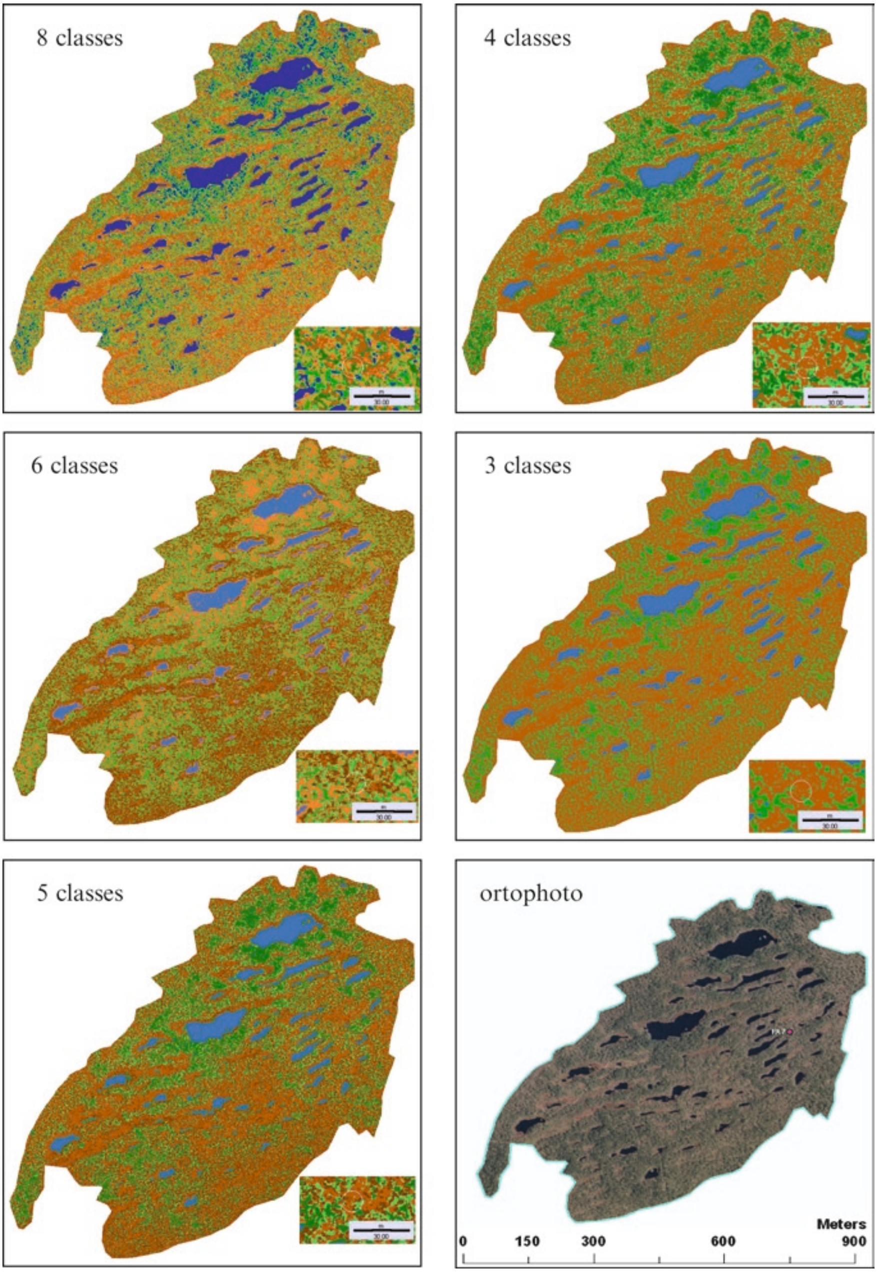


Fig. 26.6 Visualized *typological* up-scaling results of the Selisoo mire ecotope with well oriented wooded ridges on pool-ridge microtope ($ID_{ecot} = 8$ in Table 26.2), where the up-scaling were made from the maximum number of the clustered classes $C_{max} = 8$ toward the $C_{min} = 3$ included. Ground level surveyed tree plot (PA7 in Table 26.4) is visualised on a delimited ecotope ortophoto image

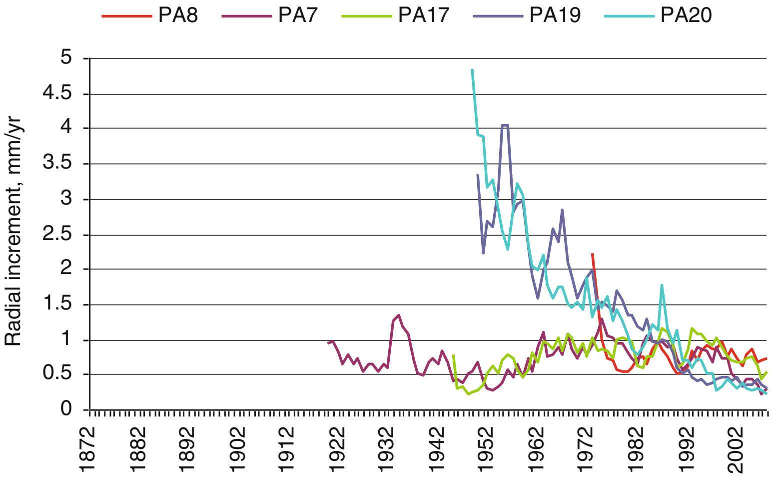


Fig. 26.7 The radial increment chronology of *Scots pine* tree stems from land surveyed tree plots at Selisoo mire ecotope of terminated peat cutting (PA19 and PA20 in Fig. 26.1), pool-ridge (PA7 in Fig. 26.1) and lawn type (PA8 and PA17 in Fig. 26.1) mire microforms

There is a promising relationship between surveyed tree basal areas and mire GW levels on the tree plots (Fig. 26.3), although with a low correlation ($R^2 = 0.4$) in our case, – probably influenced by a very high precipitation growth period in 2007 and correspondingly high GW levels over the whole mire landscape. The 10–20 cm lower GW levels recorded in the summer of 2007 (Table 26.2) show a need for longer period GW level recordings on ecotopes.

GIS Based Analysis of the Wooded Mire Ecotopes

Multiple iterative and independent *ISOCLUST* runs were performed and cross-tabulation correlations between results were analysed. *Typological* up-scaling results of Selisoo bog ecotopes revealed visibly better results for a *distinct* mire micro-topographical surface pattern with different wooded conditions in comparison with *no distinct* pattern. However, the best match of up-scaled images with the ortophoto took place in different scales i.e. in different clustered *C* levels (Figs. 26.5, 26.6 and 26.8). The biggest clustering mismatch with ortophoto images was produced by the *shadows* of the trees with different heights and crown densities and, in several cases, some shadowed areas coincided with the *field and moss layer* of the specified wetness conditions, named *wet*, *moderate*, or *dry* – in the case of the Selisoo mire ("Clustering legend" in Figs. 26.9 and 26.1b–e).

As an example, the clustered scheme of the *C* = 8 classes of the ecotope of *wooded ridges on terminated peat pit cutting* conditions classified *well* for the *tree layer* and *shadowed tree layer* classes, but the scheme of the *C* = 6 classes classified *well* for the *tree layer* and *enlightened tree layer* classes (Figs. 26.5 and 26.9a). At the same

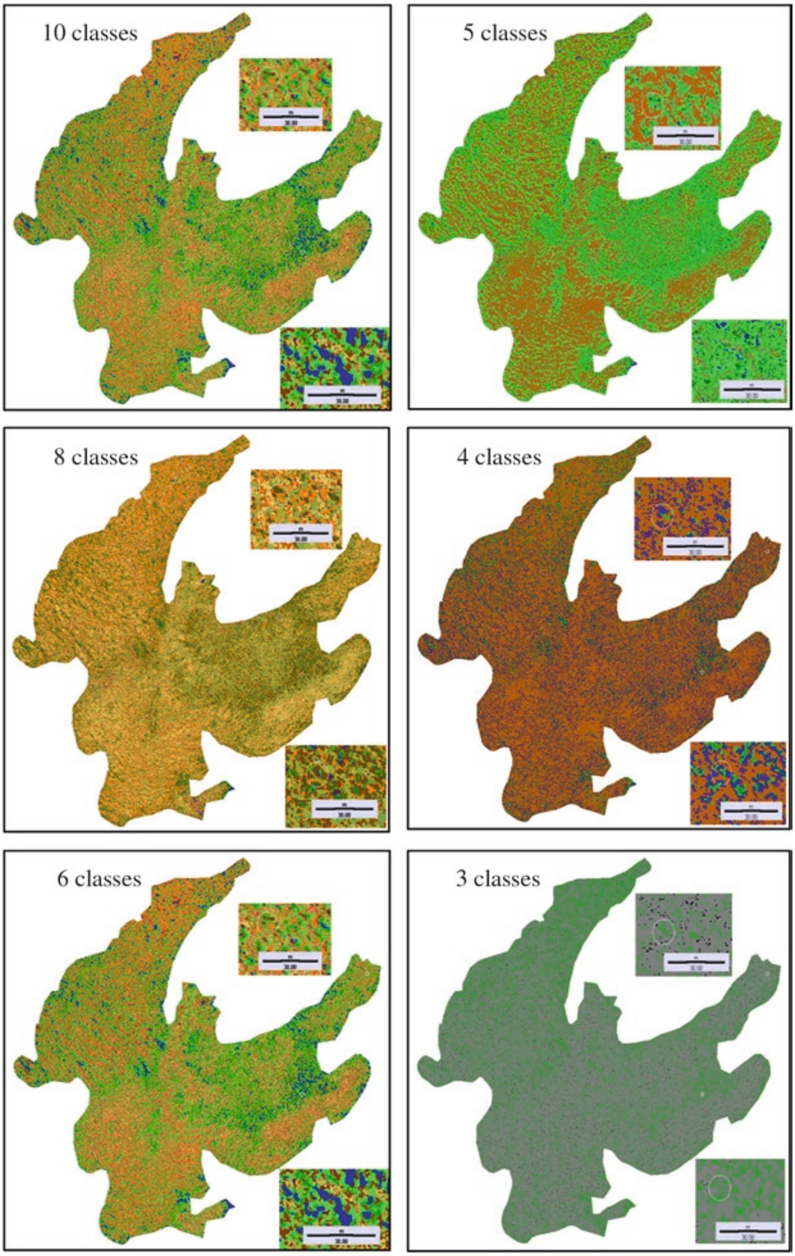


Fig. 26.8 Visualized *typological* up-scaling results of the Selisoo mire ecotope with scattered, low oriented and increased tree coverage on lawn type microtope ($ID_{ecot} = 1$ in Table 26.2), where up-scalings were made from the maximum number of the clustered classes $C_{max} = 10$ toward the $C_{min} = 3$ included. Ground level surveyed tree plots (PA8 and PA17 in Table 26.4) are visualised on the delimited ecotope ortophoto image in Fig. 26.2

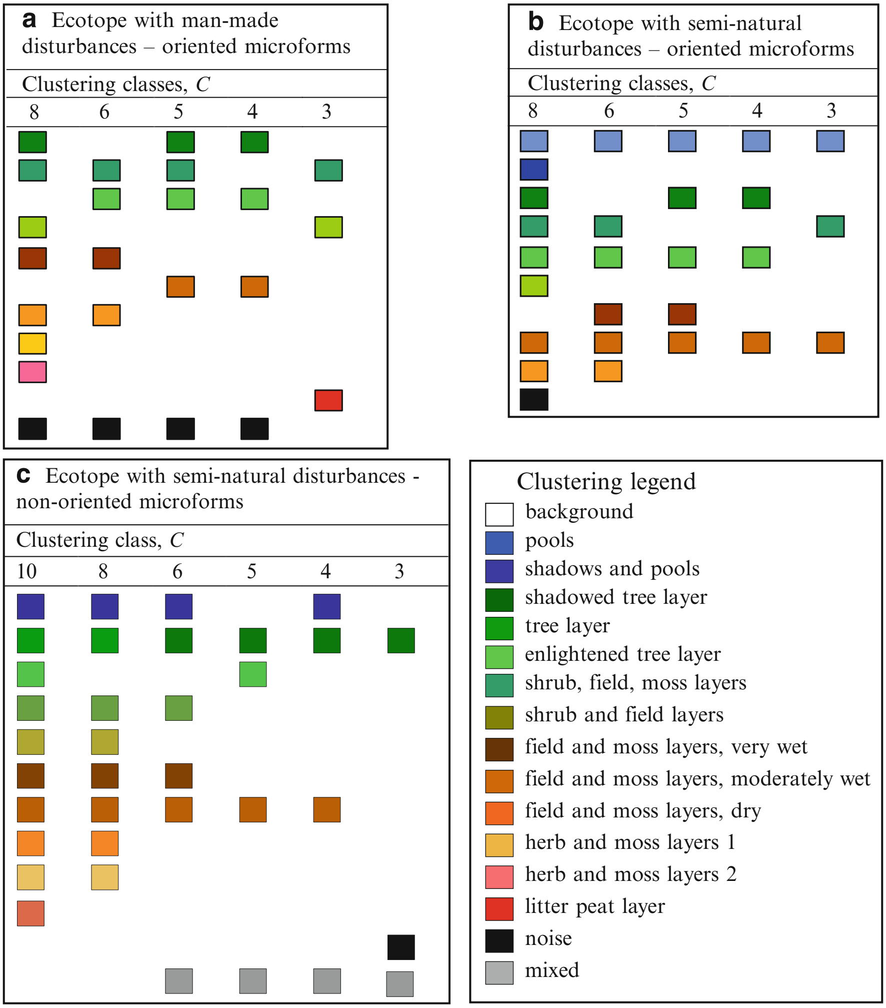


Fig. 26.9 Hierarchies of typological up-scaling of Selisoo mire ecotopes (i.e. *ISOCLUST* runs from the level of C_{max} toward the C_{min} = 3 included) according to identified and renamed classes in generalised clustering legend (*Clustering legend* in fig.), where: (**a**) hierarchy of ecotope of *man-made disturbances with oriented microforms* (Fig. 26.5), (**b**) and (**c**) hierarchies of ecotopes of *semi-natural disturbances* with oriented and non-oriented microforms respectively (Figs. 26.6 and 26.8)

time the clustered scheme of the $C = 6$ classes of the ecotope of *well-oriented wooded ridges on pool-ridge microtope* enabled to the easy discernment of the *field and moss layer* classes under different moisture conditions (Figs. 26.6 and 26.9b). The most complicated clustered results were gained from the ecotope of the *lawn type microtope with prevailing scattered tree coverage* where, from the whole clustered hierarchy in four clustered schemes one class was named *mixed* class, meaning that the ortophoto based *tree layer* were clustered together with the field or moss layer, which was impossible to identify from the ortophoto (Fig. 26.8 and 26.9c). In the latter case the clustered scheme of the $C = 5$ and $C = 4$ classes were more robust and the meaningful interpretation was lost totally for the $C = 3$ classes.

Since the different clustered schemes for the ecotope were calculated independently, a strong association between them (*Cramer V* value) was regarded as evidence for existing *hierarchical relationships* between the described classes of clustered datasets. Results of typological transitions of hierarchical clustered schemes of analysed ecotopes (Figs. 26.5, 26.6 and 26.8) are depicted in Fig. 26.8a–c with corresponding *Cramer's V* values in Table 26.7. The *Cramer's V* values of hierarchical relationships of *wooded ridges on terminated peat pit cutting* ecotope ranged from 0.69 to 0.81, whereas it was lowest ($V = 0.46$) between the lower C_4 and C_3 levels. The range of the *Cramer's V* value of the *well-oriented wooded ridges on pool-ridge microtope* ecotope was from 0.56 to 0.81, whereas it was lowest ($V = 0.53$) between the higher C_8 and C_6 levels. The hierarchical relationships for the *lawn type microtope with prevailing scattered tree coverage* were low at almost all C levels ($V = 0.17–0.53$).

In spite of the small datasets, the high linear relationship between up-scaled tree layer coverage with surveyed tree basal areas ($R^2 = 0.95$; $p = 0.005$) showed the correct clustered schemes choice for the areal quantification of the tree plot land cover (Fig. 26.10d). Hence it could be summarised that the most tree covered tree plot

Table 26.7 Results of cross-tabulated *Cramer V* values between clustered classes schemas within the topologically up-scaled ecotope hierarchy (Fig. 26.9)

	Cramer V				
Clustering classes, *C*	8	6	5	4	3
(a) Ecotope with man-made disturbances-oriented microforms (*Wooded ridges on terminated peat pit cutting*)					
10	0.81				
8		0.69			
6			0.75		
5				0.80	
4					0.46
(b) Ecotope with semi-natural disturbances-oriented microforms (*Well-oriented wooded ridges on pool-ridge microtope*)					
8		0.56			
6			0.70		
5				0.81	
4					0.81
(c) Ecotope with semi-natural disturbances – non-oriented microforms (L*awn type microtope with prevailing scattered tree coverage*)					
10	0.53		0.48		
8		0.40			
6			0.21		
5				0.17	
4					0.26

a Ecotope of man made disturbances - oriented microforms

C_5	GIS coverage, m^2	
	Tree plots	
	PA19	PA20
🟩	32	18
🟩	37	3
🟩	25	4
🟧	5	75
⬛	1	

b Ecotopes of semi-natural disturbances - oriented microforms

C_5	GIS coverage, m^2
	Tree plot
	PA7
🟩	5
🟩	19
🟫	22
🟧	54

c Ecotope of semi-natural disturbances - non-oriented microforms

C_{10}	GIS coverage, m^2	
	Tree plots	
	PA8	PA17
🟦		25
🟩	31	20
🟩	4	28
🟩	7	1
🟨	12	3
🟫	11	18
🟧	4	
🟧	22	0
🟨	1	5
🟥	8	

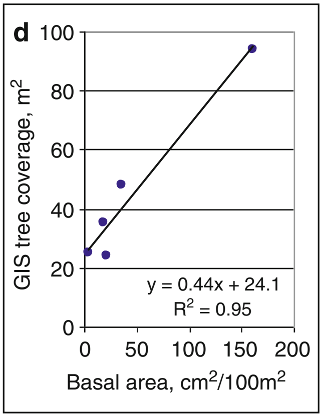


Fig. 26.10 Areal coverage results from Selisoo mire tree plot images extracted from hierarchies of typological up-scaling (**a**–**c**), where in: (**a**) PA19 and PA20 = tree plots of man-made disturbances with oriented microforms ecotope (Fig. 26.5), (**b**) and (**c**) PA 7, and PA8 and PA17 = tree plots of semi-natural disturbances ecotopes with oriented and non-oriented microforms respectively (Figs. 26.6 and 26.8), (**d**) linear relationship between the GIS based tree coverage (m^2) and land surveyed basal area (cm^2/100 m^2). The tree plots clustering legend followed the ecotope generalised legend ("Clustering legend" in Fig. 26.9), and the highest *C* level schemas from the *C* pairs with the highest *Cramer V* values was chosen for compiling of linear relationship of (**d**) graph in figure

was located on the *wooded ridges* of the terminated peat cutting pits (i.e. PA19 = 94 m^2) followed by the *bog forest* plot (PA17 = 48 m^2) on the lawn type microtope of the semi-naturally disturbed ecotope. The least tree coverage was found for the *bog forest* PA7 plot (24 m^2) on the well-oriented pool-ridge microtope of semi-naturally disturbed ecotope and *wooded ridge* plot (PA20 = 25 m^2) on terminate peat cutting, and *wooded hummock* plot (PA8 = 35 m^2) of lawn type microtope of semi-naturally disturbed ecotope (Fig. 26.10).

The relationships between the calculated GIS statistics of the tree plot images with parameterised tree plot values surveyed throughout the mire area confirmed a reservation about GIS analysis limitations derived from the so-called *universal* approach of up-scaling of spatial datasets for the whole Selisoo bog area (graph – Fig. 26.11a). In this case the highest linear correlation of $r = -0.5$ was found between the *range* of pixels' values of the *hue channel* (H) images, and the mean age of the trees. Much stronger relationships were found in relation to the ecotope images, although the datasets were too small to achieve statistical significance

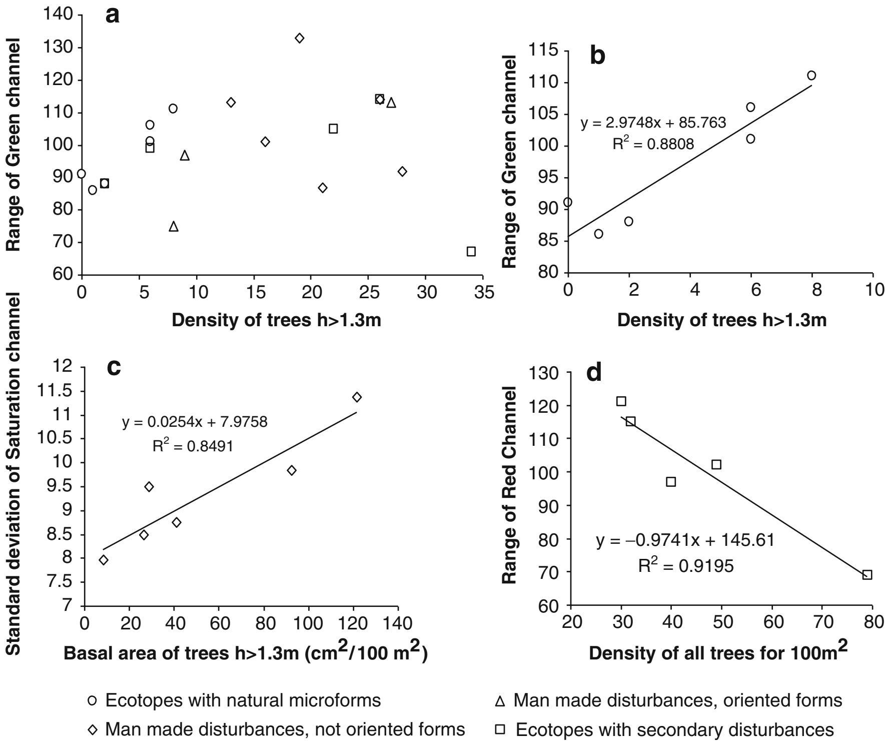


Fig. 26.11 Examples of identified stronger relationships between different tree plot image statistics with different land surveyed tree plot parameters subdivided into ecotope classes (Table 26.5), where: (**a**) universal linear relationship between the range of pixels' values of G channel and the density of the trees with heights of h > 1.3 m from the land surveyed tree plots over the whole of the Selisoo bog extension (i.e. PA1…PA20 in Table 26.5); (**b**) linear relation between the range of pixels' values of G channel and the tree basal area from natural mire ecotopes; (**c**) linear relationship between the standard deviation of pixels' values of the S channel and the tree basal area from the ecotopes with man-made disturbances, (**d**) linear relationship between the range of pixels' values of the *R* channel and tree density from the ecotopes with secondary disturbances

(graphs – Fig. 26.11b–d). It should also be stressed here that the high correlation values were gained between different image statistical parameters with different land surveyed tree parameters of different ecotopes.

Conclusions

Although modern spatial monitoring has opened up several possibilities to investigate both the vertical and horizontal structure of wooded peatlands or mire landscapes, the spatial datasets derived from ortophotos are still favourable due to: (a) the technical progress of dataset quality and quantity improvement, and (b) user-friendly availability both from the technical and the financial point of view.

However, the results of mire ecotope spatial datasets showed that in spite of delimitation of the mire landscape to the different eco-hydrologically influenced ecotopes, the surface structure reflection in ortophotos caused different up-scaling quality at different levels. In general, the visibly well oriented and wooded microform patterns in ecotope ortophoto images reflected reasonably well in different clustered schemas, and a clustered hierarchy formed from them was characterised with comparably strong associations between different clustered levels. In that sense there were no great differences between the surface pattern reflections in ortophotos from the natural or anthropogenically formed ecotopes. Due to the clear horizontal surface pattern, and following that the wooded coverage, it was comparatively easy to choose the most "land truth" classification scheme for estimating the tree areal coverage from the up-scaled hierarchical schemas.

The complicated boundary lines of limited ecotopes showed slow but visibly significant changes of the mire surface pattern related to the different surface transforming factors in peatland. Comparatively well distinguished tree pattern, visible both from the ortophotos and typologically modelled from the spatial databases, gave promising results for the development of tree expansion studies to mire landscapes from the spatial datasets. However there is no doubt about a need for a ground level parameterisation.

The ecotope scale level also seems to be a reasonable unit for continuing study of mire landscapes under different natural and anthropogenical conditions. These units are also important from the GIS based 3D ecohydrological modelling point of view where the mire surface fragmentation ought to be generalised into hydrologically unified parts, and the surface pattern classification might be a key for modelling of water movement for a whole landscape.

Typologically up-scaled results of the current study also developed a reliable database or strategic base for following field studies, pointing out the "bottlenecks" both with modelling and ground level surveys.

Acknowledgments This work was prepared in the framework of the COST Action FP0601 "Forman" project, a collaboration of the University of Tartu (Department of Geography), Estonia, the Swedish University of Agricultural Sciences (Department of Soil and Environment) and the Institute of Ecology of Tallinn University, Estonia. Financial sources were; the Estonian basic research foundation of SF0280009s07 project of "Impact of disturbances on wetland ecosystems in Estonia" and SF0180127s08 "Material cycling of landscapes in changing climate and land use conditions and ecotechnological control thereof" integrated with FP0601.

References

Åberg E (1992) Tree colonisation of three mires in southern Sweden. In: Bragg OM, Hulme PD, Ingram HAP, Robertsson RA (eds) Peatland ecosystems and man: an impact assessment. Dept of Biological Science, University of Dundee, Dundee, Scotland, pp 268–270

Aaviksoo K, Leivits A, Leivits M (2008) Kaug- ja linnuseire Nigula rabas. In: Väljataga K, Kaukver K (eds) Kaugseire Eestis, artiklikogumik. Keskkonnaministeeriumi info- ja Tehnokeskus, Tallinn, pp 106–122

Adermann V (2007) Eesti metsad 2006. Metsavarude hinnang statistilisel valikmeetodil. Metsakaitse ja Metsauuenduskeksus. SMI osakond. http://www.keskkonnainfo.ee/publications/16270_PDF.pdf 05-09-2010

Anderson AR, Ray D, Pyatt DG (2000) Physical and hydrological impacts of blanket bog afforestation at Bad a'Cheo, Caithness: the first 5 years. Forestry 73:467–478

Bishop YMM, Fienberg SE, Holland PW (1975) Discrete multivariate analysis. MIT Press, Cambridge, MA

Blaschke T, Lang S, Hay G (eds) (2008) Object-based Image Analysis. Spatial concepts for knowledge-driven remote sensing applications. Springer, Berlin, Heidelberg, XVII+817+CD

Brandtberg T, Walter F (1998) Automated delineation of individual tree crowns in high spatial resolution aerial images by multiple-scale analysis. Mach Vis Appl 11:64–73

Burnett C, Aaviksoo K, Lang S, Langanke T, Blaschke T (2003) An object-based methodology for mapping mires using high resolution imagery. In: Järvet A, Lode E (eds) Ecohydrological processes in northern wetlands. Selected papers. Tartu University Press, Tallinn-Tartu, pp 239–244

Burnett C, Blaschke T (2003) A multi-scale segmentation/object relationship modelling methodology for landscape analysis. Ecol Model 168:233–249

Cameron ED, Miller DR, Ramsay F, Nikolaou I, Clarke GC (2000) Temporal measurement of the loss of native pinewood in Scotland through the analysis of orthorectified aerial photographs. J Environ Manage 58:33–43

Eggelsmann R, Heathwaite AL, Grosse-Braukmann G, Küster E, Naucke W, Schuch M, Schweickle V (1993) Physical processes and properties of mires. In: Heathwaite AL, Göttlich Kh (eds) Mires: process, exploitation and conservation. Wiley, Chichester, UK, pp 171–262

Eastman JR (2006) IDRISI Andes. Guide to GIS and image processing. Clark Labs, Worcester, MA, 327 pp

Finnish statistical (2006) Finnish statistical yearbook of forestry 2006. Finnish Forest Research Institute, Helsinki, 435 pp

Forest Service (2007) National Forest Inventory – Republic of Ireland – results. Forest Service, Department of Agriculture, Fisheries and Food, Johnstown Castle Estate, Co., Wexford

Frankl R, Schmeidl H (2000) Vegetation change in a South German raised bog: ecosystem engineering by plant species, vegetation switch or ecosystem level feedback mechanisms? Flora 195:267–276

Genc L, Dewitt B, Smith S (2004) Determination of Wetland Vegetation Height with LIDAR. Turk J Agric Forestry 28:63–71

Gong P, Biging GS, Lee SM, Mei X, Sheng Y, Pu R, Xu B, Schwarz K-P, Mostafa M (1999) Photo econometrics for forest inventory. Geogr Inform Sci 5(1):9–14

Groom G, Mücher CA, Ihse M, Wrbka T (2006) Remote sensing in landscape ecology: experiences and perspectives in a European context. Landsc Ecol 21:391–408. doi:10.1007/s10980-004-4212-1

Gunnarsson U, Malmer N, Rydin H (2002) Dynamics of constancy in *Sphagnum* dominated mire ecosystems? A 40-year study. Ecography 25:685–704

Gunnarsson U, Rydin H (1998) Demography and recruitment of Scots pine on raised bogs in eastern Sweden and relationships to microhabitat differentiation. Wetlands 18:133–141

Hardisky MA, Klemas V, Smart M (1983) The influence of soil salinity, growth form and leaf moisture on the spectral radiance of Spartina-Alterniflora canopies. Photogramm Eng Remote Sensing 49:77–83

Heikurainen L (1967) On the possibilities of optimum drainage in peat lands. 14th IUFRO Cong Papers 4:264–277

Hopkinson C, Chasmer LE, Zsigovics G, Creed IF, Sitar M, Treitz P, Maher RV (2004) Errors in LiDAR ground elevation and wetland vegetation height estimates. Int Arch Photogramm Remote Sensing Spatial Inf Syst XXXVI(8/W2):108–113

Hånell B, Magnusson T (2005) An evaluation of land suitability for forest fertilization with biofuel ash on organic soils in Sweden. Forest Ecol Manage 2009:43–55

Ihse M (2007) Colour infrared aerial photography as a tool for vegetation mapping and change detection in environmental studies of Nordic ecosystems: A review. Nor Geogr Tidsskr 62(4):170–191

Ihse M, Malmer N, Alm G (1992) Remote sensing and image analysis for study of small changes of vegetation and microtopography, applied on mires in southern Sweden. In: Bragg OM, Hulme PD, Ingram HAP, Robertsson RA (eds) Peatland ecosystems and man: an impact assessment. Dept of Biological Science, University of Dundee, Dundee, Scotland, pp 283–286

Ivanov KE (1953) Gidrologija bolot. In: Sokolov AA (ed) Gidrometeorologitcheskoe Izdatelstvo, Leningrad, 295 pp

Ivanov KE (1975) Vodoobmen v bolotnykh landshaftakh. Gidrometeoizdat, Leningrad

Ivanov KE (1981) Water movement in Mirelands. Academic, London

James LA, Watson DG, Hansen WF (2007) Using LIDAR data to map gullies and headwater streams under forest canopy: South Carolina, USA. Catena 71:132–144, www.elsevier.com/locate/catena, Doi:10.1016/j.catena.2006.10.010

Jauhiainen S, Holopainen M, Rasinmäki A (2007) Monitoring peatland vegetation by means of digitized aerial photographs. Scand J Forest Res 22:168–177

Komura R, Kamata N, Kubo M, Muramoto K (2005) Identification of dead tree of Japanese oak wilt (JOW) using high spatial resolution satellite imagery. Proc IEEE Int Geosci Remote Sensing Symp (IGARSS) 1(8):4351–4354

Laine J, Laiho R, Minkkinen K, Vasander H (2006) Forestry and boreal peatlands. Ecol Stud 188:331–357

Langanke T, Burnett C, Lang S (2007) Assessing the mire conservation status of a raised bog site in Salzburg using object-based monitoring and structural analysis. Landsc Urban Plann 79:160–169

Liblik V, Pensa M, Rätsep A (2003) Air pollution zones and harmful pollution levels of alkaline dust for plants. Water Air Soil Poll: Focus 3(5–6):193–209

Liira J, Püssa K, Peterson U (2006) The radiance contrast of forest-to-clearcut edges on a medium resolution Landsat Enhanced Thematic Mapper satellite winter image. Int J Remote Sensing 27(13):2753–2766

Lillesand TM, Kiefer RW, Chipman JW (2008) Remote sensing and image interpretation, 6th edn. Wiley, New York, 768 pp

Linderholm HW, Leine M (2004) An assessment of twentieth century tree-cover changes on a southern Swedish peatland combining dendrochronology and aerial photograph analysis. Wetlands 24(2):357–363

Leckie DG, Gougeon FA, Walsworth N, Paradine D (2003) Stand delineation and composition using semi-automated individual tree crown analysis. Remote Sensing Environ 85:355–369

LSTATS (2009) http://www.geo.ut.ee/LSTATS/2009-09-10

Malmer N (1988) Patterns in the growth and the accumulation of inorganic constituents in the *Sphagnum* cover on ombrotrophic bogs in Scandinavia. Oikos 53:105–120

Masing V (1982) Structure and productivity of the bog plant cover. In: Masing V (ed) Peatland ecosystems. Researchers into the plant cover of Estonian bogs and their productivity. Tallinn, Valgus, pp 50–92

Masing V (1998) Multilevel approach in mire mapping, research, and classification. Contribution to the IMCG Classification Workshop, March 25–29, Greisfswald. http://www.imcg.net/docum/greifswa/masing.htm 2009-05-15

Milton EJ, Hughes PD, Anderson K, Schulz J, Lindsay R, Kelday SB, Hill CT (2005) Remote sensing of bog surfaces. JNCC Report No. 366: 166 pp

Montanarella L, Jones RJA, Hiederer R (2006) The distribution of peatland in Europe. Mires and Peat 1: Article 1. Int Mire Cons Group and Int Peat Society. http://www.mires-and-peat.net 2009-08-22

NESDIS (2009) Advanced Very High Resolution Radiometer – AVHRR. http://noaasis.noaa.gov/NOAASIS/ml/avhrr.html 2009-09-18

Ohlson M, Økland RH, Nordbakken J-F, Dahlberg B (2001) Fatal interactions between Scots pine and *Sphagnum* mosses in bog ecosystems. Oikos 94:425–432

Oleszczuk R, Regina K, Szajdak L, Höper H, Maryganova V (2008) Impacts of agricultural utilization of peat pails on the greenhouse gas balance. In: Strack M (ed) Peatlands and climate. International Peat Society, Jyväskylä, Finland, pp 70–97

Paal J, Ilomets M, Fremstad E, Moen A, Børset E, Kuusemets V, Truus L, Leibak E (1998) Estonian wetland inventory 1997. Publication of the project "Estonian wetlands conservation and management strategy". Eesti Loodusfoto, Tartu, 166pp+xxviii

Paavilainen E, Päivänen J (1995a) Peatland forestry. Ecology and principles. Ecol Stud 111:248

Paavilainen E, Päivänen J (1995b) Utilisation of peatlands. Ecol Stud 111:15–29

Pellerin S, Lavoie C (2003) Reconstruction the recent dynamics of mire using a multitechnique approach. J Ecol 91:1008–1021

Quackenbush UJ, Hopkins PF, Kinn GJ (2000) Developing forestry products from high resolution digital aerial imagery. Photogramm Eng Remote Sens 66(11):1337–1346

Rydin H, Jeglum J, with contribution Hooijer A, Clarkson BR, Clarkson BD, Mauquoy D, Bennet KD (2006) The biology of Peatlands. Biology of habitats. Oxford University Press, Oxford, 343 pp

Toth J, Gillard D (1988) Experimental design and evaluation of a peatland drainage system for forestry by optimization of synthetic hydrographs. Can J For Res 18:353–173

Valk U (2005) Eesti rabad. Ökoloogilis-metsanduslik uurimus. Eest Põllumajandusülikool, Metsanduslik Uurimisinstituut, Tartu, 314 pp

Van der Schaaf S (1998) Self regulation of acrotelm transmissivity and discharge in two Irish midland raised bogs. In: Malterer T, Johnson K, Stewart J (eds) Proceedings of the international peat symposium, Peatland restoration and reclamation – techniques and regulatory considerations, Duluth, Minnesota, USA, 14–18 July 1998, pp 161–169

Van der Schaaf S (1999) Analysis of the hydrology of raised bogs in the Irish Midlands. A case study of Raheenmore Bog and Clara Bog. Ph.D. Dissertation, Wageningen University, The Netherlands, 375 pp

Van der Schaaf S (2002) Relationships between biotic and abiotic conditions. In: Schouten MGC (ed) Conservation and restoration of raised bogs; geological, hydrological and ecological studies. Department of the Environment and Local Government, Ireland; Staatsbosbeheer, The Netherlands, pp 186–209

Van Seters TE, Price JS (2001) The impact of peat harvesting and natural regeneration on the water balance of an abandoned cutover bog. Que Hydrol Proc 15:233–248

Vasander H, Tuittila E-S, Lode E, Lundin L, Ilomets M, Sallantaus T, Heikkilä R, Pitkänen M-L, Laine J (2003) Status and restoration of peatlands in northern Europe. Wetl Ecol Manage 11:51–63

Waser LT, Baltsavias E, Eisenbeiss H, Ginzler C, Gruen A, Kuechler M, Thee P (2007) Change detection in mire ecosystems: assessing changes of forest area using airborne remote sensing data In: Schaepman ME, Liang S, Groot NE, Kneubühler M (eds) The 10th international symposium on physical measurements and spectral signatures in remote sensing, International archives of the photogrammetry, remote sensing and spatial information sciences, vol XXXVI, Part 7/C50. Davos, Switzerland, pp 313–318

Waser LT, Ecker K, Ginzler C, Küchler M, Schwarz M, Thee P (2006) Extraction of forest parameters in a mire environment using airborne spectral data and digital surface models. Workshop on 3D Remote Sensing in Forestry, 14–15 Feb 2006, Vienna – Session 1:15–23. http://www.wsl.ch/staff/lars.waser/wien_2006_lars.pdf 2009-09-20

Chapter 27

Difficulties of Scaling in Forest and Water Management in Urban Areas: Social and Institutional Dimension

Sultan Bekiroglu and Ömer Eker

Introduction

In recent decades scientists and policy makers have been intensively focused on adapting new strategies both at national and international level to diminish the negative effects of climate change and unconventional weather conditions. Growing environmental degradation and health problems have also accelerated to take urgent policy and management measures to keep the natural resources and environment in balance. In this respect concerns in understanding the interrelationships between water and forest have come into the forefront.

The sustainable management of forests has a key role to play in protecting global water supplies. Forests help maintain healthy aquatic ecosystems and provide reliable supplies of clean freshwater. But not only do they filter and clean water forests also help prevent soil erosion, reduce sedimentation in reservoirs and mitigate the risks of mudslides and floods, all problems that can threaten downstream water supplies. And while forests themselves consume water, they also improve infiltration rates, thereby helping recharge underground aquifers (FAO 2008).

In many countries, forest and water policies, legislation and administration have long been shaping forest rehabilitation programs; this has been the case in European countries such as France, Italy and Switzerland since the eighteenth century. Only in the past few decades, however, has the emphasis on theory and practice of hydrology been replaced by a more comprehensive approach embracing environmental issues, land use and watersheds (Zingari and Achouri 2007).

S. Bekiroglu (✉)
Forest Engineering Division, Department of Forestry Economics, İstanbul University,
34473 Bahçeköy, İstanbul, Turkey
e-mail: sulbekir@istanbul.edu.tr

Ö. Eker
Forest Engineering Division, Department of Forest Economics, Kahramanmaraş Sütçü İmam
University, Bahçelievler Yerleşkesi, 46060 Kahramanmaraş, Turkey
e-mail: omereker@ksu.edu.tr

M. Bredemeier et al. (eds.), *Forest Management and the Water Cycle: An Ecosystem-Based Approach*, Ecological Studies 212, DOI 10.1007/978-90-481-9834-4_27,

The Shiga Expert Meeting on forest and water which was held in Japan, in 2002 identified the following key issues: Promoting the development and the wider adoption of holistic approaches to forest and water management that integrate the needs of people and the environment; improving understanding of the bio-physical interaction between forests and water; improving understanding of the cultural and socio-economic impacts of different forest and water policies and management practices; developing better mechanisms for managing upstream/downstream linkages and interactions; and enhancing knowledge and information sharing (Anonymous 2002).

The European Union Water Framework Directive (EU WFD), adopted in 2000, is designed to take an integrated approach to water management. The directive's overall objective is to achieve good water status for all waters in Europe by 2015 (Nilsson 2006). However the implementation of the WFD is at the beginning phase in Europe. One possible explanation is connected to problems of spatial fit and institutional interplay. In short, spatial fit refers to the overlap between the territorial borders of political and management institutions, and the biogeophysical resource to be managed (Nilsson 2006). The idea is that creating better fit between responsible institutions and the resource to be managed reduces spatial externalities, which otherwise may benefit free riders and harm others beyond the spatial extent of the management institution. Institutional interplay, on the other hand, refers to the idea that the success or effectiveness of institutions is dependent not only on their own performance, but also to a large degree on their interactions with other institutions (Nilsson 2006).

Such points are important in establishing a successful water management system at different scales (watershed, river basin, basin) under the integrated water management umbrella. That also requires coordination and cooperation between water management and other sectors in achieving other environmental goals.

This study is mainly aimed at determining the effects of social, institutional, political and physical factors on water and forest management in urban areas and describes how these factors affect scaling and management decisions in watershed and basin areas.

Urban Growth and Industrialization

Half the world's population today lives in urban landscapes and it is estimated that the urban population will increase to five billion by 2030 (United Nations (UN) 2004). The level of urbanization in Europe is estimated to be 74.6% with an expected annual growth of 0.3% per year between 2000 and 2015 (United Nations Centre for Human Settlements (UNCHS) 2002). As was represented in Fig. 27.1 the level of urban population in Western Europe is high and it is expected that Europe will stabilize at an urbanization level of about 82%. Currently, one-half of the population of Europe lives in small towns of 1,000–50,000 people, one-quarter in medium-sized towns of 50,000–250,000 people and one-quarter in cities of more than 250,000 people (UNCHS 2002). Many parts of Europe do not have as much

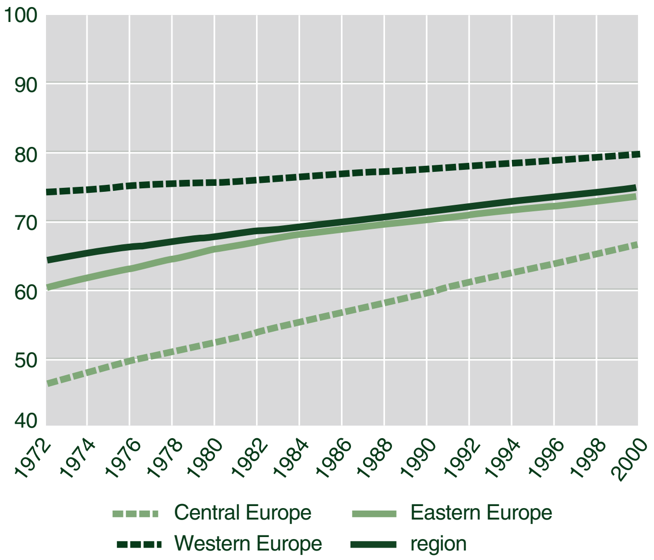


Fig. 27.1 Urban population (UNCHS 2002)

water as they need. In 60% of European urban and industrial areas, water resources are overexploited. The problems of urban development and industrialization and their impacts on the environment have been challenging for European policy-makers. They also affect the management of water and forest resources in a negative manner at different scales. Land degradation, green space loss, decrease in water amount and quality are the major impacts of urban growth and industrialization.

As in the Seine River Case a high level of industrialization and urbanization has a major impact on water quality in the watershed, especially in its centre where analyzed results range between ‘bad’ and ‘adequate’ ratings. Water quality is directly altered by intensive agriculture, water treatment plant effluents and the presence of one of the biggest mega cities of Europe (25% of the French population lives here on 12% of the national territory) (Anonymous 2007). The diversion of waters from the Seine River Basin for economic development and local flooding protection has negatively impacted the river’s water level and quality. During years with low precipitation the river runs dry and river habitat and fish populations suffer. The diversion of water largely takes place outside of the city, beyond the jurisdiction of the municipal government. Efforts to change existing diversions require support from the provincial government and need to be pursued if the water conditions in the river are to be further improved (GPPArthitecture 2007).

The Volga River Basin, which covers 10% of the total territory of the Russian Federation, has a population of over 60 million. The biggest environmental problems stem from major industrial complexes, big dams, large cities and maintaining navigability. Just three percent of the surface water in the Volga River Basin is considered an environmentally safe source of drinking water. Some 42 million tons of toxic waste pile up each year in the Basin, causing immense health problems (UNEP 2004). 4.7 Million residents are estimated to be vulnerable to floods and pollution challenges in the region. Although an integrated water management system was put into practice in Russia the priorities in water usage were given to the development of growing regional agriculture and industries without any attention being paid to the needs of the environmental flows and forest ecosystems. The growing concern over environmental aspects of watershed management has been difficult to feed into practical water management plans due to, among other reasons, lack of transboundary cooperation. Many countries of the former Soviet Union do not have agreements on transboundary water management. Though some agreements do exist, they are of a superficial character and not fully implemented (Lagutov 2008).

In the case of the Tagus River which rises in east-central Spain and flows through Portugal, the main environmental pressures are pollution from industrial and municipal point sources and diffuse sources. The water demand in the river basin has doubled in 24 years to 500 million cubic meters due to the increasing requirements for irrigation and tourism. The increase in irrigated land and tourist activities has led to the destruction of thousands of hectares of protected natural areas. Water uses are partially uncontrolled. More than 100 million cubic meters of transferred water "disappears" illegally to supply tourist resorts and golf courses. Social imbalances have increased as the transfers mainly benefit big agro-businesses and construction companies, marginalizing traditional farmers. Illegal immigration and exploitation of immigrants are increasing with about 30% being undeclared labour (UNEP 2004).

As has been mentioned earlier, by the end of 2000 The European Commission and Parliament approved and published the Water Framework Directive (WFD: 2000/60/EC) which intended to provide a framework for a common approach to the management of water in all European Union member states (UNEP 2002). The aims of the WFD are to prevent deterioration and enhance the status of aquatic ecosystems, including groundwater; to promote sustainable water use; to reduce pollution and to contribute to the mitigation of floods and droughts. However, implementation of the directive is quite difficult in terms of social and ecological flexibility limitations in many highly urbanized and industrialized European landscapes. It requires adaptation of multi-scale management approaches among physical, ecological, social and institutional characteristics of regions at different scales.

Transboundary Waters

The pressure on transboundary water (and land) resources can lead to potential conflicts between users and states sharing the same river, lake, sea or groundwater basin (UNDP 2008). There are 150 major transboundary rivers in Europe that form or cross borders between two or more countries, some 25 major transboundary lakes, and some 100 transboundary aquifers. The numbers of international basins and the nations which they traverse change over time in response to alterations in the political situation. For example, the break-up of the Soviet Union in 1991 led to the internationalization of several basins (e.g. Dnieper, Don and Volga) and to a change in the political composition of existing international basins. Although most of the shared water resources in Europe are covered by numerous transboundary agreements, some tensions and disputes still exist. These include water shortage problems during the summer, dam construction, pollution and boundary problems, as water bodies are often used as international boundaries (UNEP 2004). In some cases agreements have not yet been established between countries. If there are agreements, these are not always fully implemented and may not be effective tools to tackle the relevant issues, nor address social, economic and environmental aspects in a holistic way (EUWI 2008).

In transboundary river basins the mismatch between political boundaries and natural river basins and aquifer systems also creates conflicts in scaling. This requires tailoring water management to the specific conditions of the many transboundary catchment areas in the region; establishment of joint bodies such as river commissions that would provide coordination among various governmental entities and determination of actors such as local institutions, non-governmental organizations (NGO's) and stakeholders which are involved in cooperation on transboundary waters.

Institutional and Organizational Framework

In addition to technical and ecological considerations, social, economic and institutional issues are also important preconditions for sustainable water management at the national and transboundary levels. However, incomplete knowledge of ecosystem dynamics and institutional constraints often leads to institutional management frameworks that do not match the scale of ecological patterns and processes (Borgström et al. 2006). Another issue of major importance is the inclusion of stakeholder groups and the public at large in the development and implementation of water resource management schemes. This explicitly requires the involvement of the public in the development of a river basin management plan. The complexity and uncertainty of current water and river realities urge for new forms of governance. These new forms should replace the traditional hierarchical systems, oriented to control, by participatory and flexible systems, based on experimenting between multiple actors (Wolters et al. 2006). Nilsson (2006), for instance, emphasizes that all countries have established river basin districts according to Article 3 of the WFD in Europe. However, no country has radically changed their administrative water management structures to better fit these new spatial management units. Instead all countries have – to a greater or lesser extent – chosen to manage the districts by coordinating the work among "old" administrative water management organizations. In none of the countries are the geographical borders of the established river basin districts exactly the same as the geographical borders of the competent authorities appointed as responsible for the management of the districts. Only in Estonia and Sweden can the competent authorities be regarded as – at least partly – coincident with river basin district/sub-basin borders. In these countries, one county environmental department (Estonia) and one county administrative board (Sweden) have been specifically designated as competent authority for a river basin district/sub-basin, taking an overall responsibility for the whole district/basin.

In best-practice examples, the institutional framework is focused on the local level, with clear arrangements for integration within permanent agencies and for interagency collaboration, as seen in the Turkey Eastern Anatolia Project. Government commitment to the program and simplicity and clarity of responsibilities are also important factors in success: the interface between local government, technical agencies, and community organizations needs to be carefully defined and managed, and capacity building at all levels is essential (Darghaut et al. 2008).

However, adaptation of this approach in urban lands is affected by social pressures and political priorities, as seen in the Istanbul case.

It is well known that expanding human settlement areas restrict and endanger water and forest resources. In Istanbul investigations show that the 158,000 illegal constructions counted in watershed areas in 2003 had grown to 200,000 by 2007. As was shown in Fig. 27.2 the most dramatic example of green space loss has occurred with the illegal establishment of a settlement area called Sultanbeyli District in Istanbul.

In 5 years (1985–1990) its population increased 2100%. This district expanded by exploiting natural resources on its periphery and has become one of most crowded regions in Istanbul. In this region forested areas were considerably reduced and some watersheds were polluted. Historically politicians have tolerated these illegal settlements for political expediency and eventually introduced legislation to legalize these illegal settlements. Amnesties have been declared for illegal forest area occupations

Fig. 27.2 Sultanbeyli District in Ömerli Watershed, Istanbul

four times through laws prepared by the government. A proposed fifth amnesty for forestland squatters has been an important subject of debate on the political agenda for the last 5 years. These facts are continuously encouraging illegal settlement activities on forest and watershed lands. The government has failed to make the link between the laws they propose and the resulting severe effects on forest and water lands.

In Turkey, with regard to the development, management and conservation of water resources a great number of institutions with governmental and non-governmental organizations such as The Ministry of Environment and Forest, The Ministry of Tourism and Culture, The Ministry of Energy and Natural Resources: The General Directorate of State Hydraulic Works and General Directorate of Electrical Power Resources Survey and Development Administration, The Ministry of Agriculture and Rural Affairs: General Directorate of Rural Services, The Ministry of Public Works and Inhabitance: The General Directorate of Agrarian Reform and the water and sewage administrations connected to the various metropolitan municipalities have responsibilities and duties. Although these institutions work well in their own fields of expertise uncertainties concerning the distribution of responsibilities and gaps in the institutional framework cause waste of time and financial sources as well as irrational use of water resources. The application of laws such as Groundwater Law Code No: 167, for example, may be confusing or incomplete in terms of what level it is performed at. According to Law all groundwater works come under the authorization of The General Directorate of State Hydraulic Works but in practice The Ministry of Environment and Forest, The General Directorate of Rural Services and The General Directorate of Bank of Provinces also undertake some work as well.

In some forested areas there may be more than one organization responsible for the protection of water resources on the same piece of land. In Istanbul, for example, The General Directorate of Forests has reserved some areas of water production with a steady protected land status. The General Directorate of Nature Conservation and National Parks is responsible for protecting these areas while Istanbul Water and Sewage Works is responsible for the management of dams and lakes on these protected forestlands. As a rule these two organizations must be in collaboration. However, there is no agreement between these organizations on water production and water protection. The identification of these problems and rearrangement of water institutions and regulations are important measures that would form a better and more effective water resource management. The severity of these problems in Turkey is likely to increase if the need for a collaborative approach among the various institutions and organizations and the need for new management approaches are neglected.

Jurisdictional Boundaries

The boundaries of administrative entities (communities, municipalities, districts, provinces, regions, or countries) normally do not coincide with the natural limits of basins. In such cases, water management platforms require the collaboration of entities from different political and administrative zones, the coordination of which generates problems of competence and jurisdiction. There is often jurisdictional interference among local institutions, national agencies, and international assistance or investment programs. Many water use actions in basin areas are taken without consulting local authorities, especially if there are no visible basin-level water management bodies (Bruns et al. 2005). Many river basins in Europe are located in more than one municipality's border. In this case decisions concerning the same river basin require involvement of all water related municipalities, stakeholders and other interest groups. This will entail adoption of different scale approaches in the watershed management areas. According to Lovell et al. (2002) specifying jurisdictional zones is, nevertheless, easier said than done, not least because administrative boundaries, infrastructural links, community limits, and informal networks seldom correspond with physical resource boundaries, to the extent that these can be agreed upon. To complicate matters further, integrated natural resource management involves the integrated management of a multitude of common-property, open-access, and privately owned resources such as cropland, pastures, forests, and water. Each has an associated complex of often-conflicting interests held by stakeholders both inside and outside the particular resource boundary.

Discussion

The social and institutional dimensions of water management in urban lands force governments to find new management options and policies incorporating related national sectors while also supplying demand for water in Europe. Although each nation has developed its own water laws and policies according to their national program these laws and policies need to be reformulated and harmonized by taking collective common interests and environmental protection measures into consideration. The biggest challange related to water management in European countries arises from efforts to keep water resources clean.

As seen in the Seine, Volga, Tagus and Ömerli river basins a high level of industrialization and urbanization are the most important threats accelerating water and environmental pollution. The Tagus and Ömerli river basin cases have similarities in terms of being open to illegal settlements by immigrants. Therefore the starting point to find the best water management options needs to focus on strict national policy plans and measures first. The new water management plans should consider the long term impacts of human activities on water and should be harmonious with the EU-WFD. Although some individual provisions were made in past decades, coordinated efforts are still required to ensure safe and clean water for Europe's population.

The incorporation of forest hydrology knowledge in water policies and inclusion of forest-sector contributions in integrated water resource management policies are key elements in resolving water related problems.

Adapting successful water management experiences from various regions is of course an important tool in finding solutions to problems and at the same time speeding up the process. However, social, political and institutional framework need to be rearranged before the required modifications are fulfilled. Without these innovations problems among the hydrological boundaries and institutional and social aspects of water management will continue.

Conclusion

In conclusion hydrological boundaries have mismatches with institutional and socio-political issues. Water management is affected by urban growth and industrialization; the transboundary state of water resources, institutional and organizational frameworks and the jurisdictional boundaries of municipalities and districts in many European Regions. During the application of water management practices at micro-watershed and local levels the range of difficulties and bottlenecks (such as stakeholder participation and collaboration among organizations) may be small. However when it comes to larger landscapes with river basins the dimension of problems increases with the necessity of international involvement and the priorities and expectations of each country about the water resources. These issues require developing new management strategies with cross-scale approaches among the social, ecological, economic and institutional parameters at local, national and international levels.

Acknowledgments We would like to give special thanks to the European Cooperation in Science and Technology (COST) for giving us the opportunity to get involved in this research.

References

Anonymous (2002) International expert meeting on forests and water: Shiga decleration on forests and water. http://www.rinya.maff.go.jp/faw2002/shiga.html Accessed: 10 December 2008

Anonymous (2007) Major European watersheds. http://www.grid.unep.ch/product/publication/freshwater_europe/seine.php. Accessed: 12 Oct 2008

Borgström ST, Elmqvist T, Angelstam P, Alfsem-Norodom C (2006) Scale mismatches in management of urban landscapes. Ecol Soc 11(2):16

Bruns BR, Ringler C, Dick RM (2005) Water rights reform: lessons for institutional design. International Food Policy Research Institute Publications (IFPRI), Washington DC

Darghaut S, Ward C, Gambarelli G, Styger E, Roux J (2008) Watershed management approaches, policies, and operations: lessons for scaling up, vol 11, Water sector board discussion paper series. The World Bank, Washington DC

European Union Water Initiative (EUWI) (2008) EECCA Component: Eastern Europe, Caucasus and Central Asia. http://euwi.jrc.it/index.php?main=1&sub=1&id=128#56. Accessed 30 Oct 2008

FAO (2008) Forests and freshwater: vital connections. http://www.fao.org/english/newsroom/focus/2003/wfc2.htm. Accessed 10 Dec 2008

GPPArthitecture (2007) Seine River Greenway: Reach 1 Redriver to Happyland Park. http://www.winnipeg.ca/ppd/planning/pdf_folder/ SeineRiverPlan.pdf. Accessed 10 Oct 2008

Lagutov V (2008) Rescue of sturgeon species in the Ural River Basin, NATO science for peace and security series – C: Environmental Security. Springer, The Netherlands

Lovell C, Mandondo A, Moriarty P (2002) The question of scale in integrated natural resource management. Conserv Ecol 5(2):25

Nilsson S (2006) Managing water according to river basins: Information management, institutional arrangements and strategic policy support – with focus on the EU Water Framework Directive. TRITA-LWR. Ph.D. Thesis, Sweden

United Nations (UN) (2004) World urbanization prospects: the 2003 revision. U.N. Department of Economic and Social Affairs/Population Division.http://www.un.org/esa/population/publications/wup2003/2003WUPHighlights.pdf. Accessed 30 Oct 2008

United Nations Centre for Human Settlements (UNCHS) (2002) State of the environment and policy retrospective: 1972–2002. http://www.unep.org/geo/geo3/english/pdfs/chapter2-8_urban.pdf

United Nations Environment Programme (UNEP) (2002) Global environment outlook 3: past, present and future perspectives. Earthscan Publications, London

United Nations Environment Programme (UNEP) (2004) Freshwater in Europe – facts, figures and maps. DEWA-Europe. Switzerland.http://www.grid.unep.ch/product/publication/freshwater_europe/intro.php. Accesed 24 November 2008

United Nations Development Programme (UNDP) (2008) Water governance in Europe and CIS: Transboundary water management. http://europeandcis.undp.org/environment/wg/show/FA499AD7-F203-1EE9-BF1DD1F9F83C486F. Accessed 20 Oct 2008

Wolters H, Ridder D, Mostert M, Otter H and Patel M (2006) Social learning in water management: Lessons from the harmoniCOP Project. E-W Ater http://www.ewaonline.de/journal/2007_01.pdf. Accessed 25 Oct 2008

Zingari PC, Achouri M (2007) Five years after Shiga: recent developments in forest and water policy and implementation. http://www.fao.org/docrep/010/a1598e/a1598e13.htm Accessed 18 Dec 2008

Chapter 28
Synthesis and Outlook

Shabtai Cohen and Michael Bredemeier

Introduction

A popular paradigm for understanding fine art photography is that photographs are "Mirrors and Windows". It turns out that how the photographer looks through his lens can teach us about how we look at the world around us. The dichotomy between an anthropocentric-utilitarian view of nature and a holistic naturalist view is especially relevant when we talk about forests. Do forests have their own lives or do they exist only for our use? Sometimes it is hard to separate the views, since ours is tainted by a little silver, which turns our lenses into mirrors, and at other times our viewpoint results from having experienced the powers of nature in a tragedy (e.g. a flood or forest fire) that might have been prevented by appropriate forest management. Documents that discuss forests and water usually focus initially on the increasing demand for good quality water by the increasing population of the earth and the role that forests play in supplying that water. Forests in Europe are almost exclusively managed by man. Yet they can still be seen as "near natural" systems and we would argue that they have their own lives independent of our requirements for their services.

In the European context of forests and water, how much are we concerned for the forest and its wellbeing or are we only interested in forest services? But we would contend that sustainability of those services depends on continuity of the forest and its hydrological system. Put another way, it's clear that whatever our viewpoint vis-à-vis the forest, our challenge is to maintain forest ecosystem health and integrity (however nebulous those terms may be), which is the best path to

---

S. Cohen (✉)
Institute of Soil, Water and Environmental Sciences, ARO Volcani Center, Bet Dagan, 50250, Israel
e-mail: vwshep@agri.gov.il

M. Bredemeier
Forest Ecosystems Research Centre, University of Göttingen, Büsgenweg 2, D-37077 Göttingen, Germany
e-mail: mbredem@gwdg.de

M. Bredemeier et al. (eds.), *Forest Management and the Water Cycle: An Ecosystem-Based Approach*, Ecological Studies 212, DOI 10.1007/978-90-481-9834-4_28,

sustainability, while ensuring the quality and quantity of hydrological and other forest services. Forest degradation can then be viewed as a 'disease' of the forest ecosystem which can be 'healed' by proper management. Forest degradation will most certainly lead to a deterioration of forest water resources, forest services and forest–water related risks (Hamilton 2008). We will most certainly incur considerable real or latent costs as forests are degraded. In the face of future climate risks and environmental challenges, it is important to understand these costs, but more important to be prepared to sustain our ecosystem resources. And we contend that one key to sustainability is scientific investigation.

Forest Management and Science

The utter existence of some of the forests, e.g. in arid regions, and the sustainability of forests in other regions, depends on how they are managed. This in turn depends on scientific input in understanding the processes that hold the forest systems together and monitoring these processes in the forest.

Forest management, including that related to the water cycle, depends on actions at several levels. These might be seen as:

- Scientific understanding of forests, the water cycle and their interactions with management
- Monitoring of forest health, integrity and processes, and of related water supply and quality
- Government policy decisions, directives at the political level and ongoing governance
- Forest management by forestry organizations

This volume focuses on the scientific side of the issues and is heavily weighted toward the first aspect listed above. Little attention has been given here to monitoring issues and even less to government policy. The importance of science in understanding forest systems is basic, although its monetary value cannot be quantified from the chapters in this volume since the economic importance of the science is not included. The monetary value of forests and ecosystem services is a relatively new topic that demands more attention (Costanza et al. 1997). Obviously, the omission of these aspects in the current book indicates that in the context of scientific study of forests and water in Europe, inclusion of these aspects is rare to non-existent. This is not an oversight, and may call for action. As capitalism overtakes our economies, it is vital to build a widespread awareness of the economic benefits of forests before the axe of thriftiness falls on their trees. This may require more economists in forest science departments.

The book deals with many aspects of forests and water from a wide range of viewpoints, e.g. that of the physiologists and atmospheric scientists studying trace gas exchanges between forest and atmosphere, soil scientists studying chemical and physical composition of forest soils and water infiltration and runoff, forest geo-botanists studying genetic diversity and modellers trying to integrate our understanding of forests in order to make management decisions. It is not easy at first sight to find the common denominator between the different chapters other than that they are a representative sample of the range and breadth of forest and forest-hydrology science in Europe today.

However, if we consider forest services and costs, it is clear that they together make a considerable contribution to the category of ecological services within the domain of overall ecosystem services.

Forest Services and Costs

Lindberg et al. (1997) identified four categories of forest services: ecological, economic, socio-cultural and 'scenic and landscape'. Forest ecological services (after Sousson et al. 1995 and the World Bank 1997) can be separated into the following headings:

1. The regulation of water regimes and water quality by intercepting rainfall and regulating its flow through the hydrological system. This service provides drinking water to much of our population.
2. The maintenance of soil quality and the provision of organic materials through leaf and branch fall.
3. The limiting of erosion and protection of soil from the direct impact of rainfall. The soil conservation function of forests cannot be over-valued, since the productivity of future agricultural systems may depend on these soils, since our current agricultural land will be degraded to the point of marginality.
4. Modulating climate and air quality through direct and indirect interactions with the atmosphere. This aspect includes the considerable role that forests have in the global climate system and mitigation of future climate change.
5. Comprising key components of biodiversity both in themselves and as a habitat for other species.

Economic services of forest comprise the long list of goods that we take out of the forest, which range from wood and wood products (e.g. paper, corks and fuel), rubber and fruits to game, fuel, medicine and herbs. Additional economic services are derived from grazing and agro-forestry. The socio-cultural services of the forest are more nebulous and depend on the particular religious or cultural beliefs of a given society. But the 'scenic and landscape' services of the forest are obvious to city-dwellers who search out to the forests for their recreational value at various levels.

Many of the services listed above depend on or are intertwined with forest management and the water cycle. Several have been dealt with directly and indirectly in this book.

Forests costs, on the other hand, include the price that we pay to maintain the forest. These costs might include the cost of forestry organizations, forest research, and, in arid regions, the cost of water used by the forest as compared to that of fallow land. In regions prone to forest fires, the cost includes the cost of these destructive events, although proper forest management should minimize the incidence of fires and their cost.

Holism and Reductionism in Forest Management and the Water Cycle

The topic of forest and water or forest management and the water cycle turns out to be so large that we are confronted with several levels of abstraction in this book. The first section looks into forest and forest–water processes at the molecular to organism level. The second section sees the forest as a collection of individuals and services. The third section includes several chapters on the concept and approaches to scaling our knowledge to the scale (or level) at which most forest and forest–water management decisions are made, i.e. community, region and national scales. Some of the chapters of the third section may echo previous ones, but a careful reading shows that they each constitute a specific viewpoint to the scaling issue. The resulting emphasis on approaches to scaling our understanding of forest processes to the level of whole forests, watersheds and beyond is the main challenge facing us today in forest hydrology (Anonymous 2008).

In holistic thinking the whole is more than the sum of its parts and therefore the whole cannot be fully described by its component parts alone. Reductionism, which is sometimes considered to be the opposite of holism, contends that the whole can be explained by a reduction of the systems to their component parts and understanding those parts (Wikipedia). Although many of us might have a holistic view of the forest and its many aspects, the current volume is certainly reductionist in its approach to understanding forests, their management and the water cycle. The book builds from the basic tree and leaf processes in the first section to whole tree, species level in the second section, and finally uses these building blocks to scale up to the landscape level – to arrive at problems and solutions usually encountered in many high level forest management decisions.

An example of the reductionist approach is seen in Fig. 28.1. In order to deal with the landscape level issues of water supplies (services) and water risks (e.g. floods) we must integrate our understanding of the forest and its components from the basic level and up. This bottom-up approach is a thread that leads from the chapters on below ground and above ground processes in the first section through species and stand structure issues in the second section and finally to the scaling issues discussed extensively in the third section.

More holistic scientific investigation of forest–water interactions is difficult because of the large scale of the problem, which requires catchment and larger scale experimentation. An example of more holistic study is the classic Hubbard–Brook

Fig.28.1 An example of a reductionist view of forest hydrology, it's study and objectives

project in the US (http://www.hubbardbrook.org/). Holistically oriented, large scale studies in the European context are discussed here by several of the contributions in Part III (e.g. Chapters 20 and 24). Hopefully upscaling of the reductionist studies and comparison with the more holistic studies will show where there are gaps in our understanding of the forest–water system.

Conclusion

In conclusion, the current volume began as a collection of papers from the COST FORMAN action participants, who each present their own views and research reviews on their particular subject. As the project developed, the three sections of the text, which came from each of the three working groups of the project took shape. These as a whole give a good overview of the state of scientific research on forest hydrology and related management today in Europe. We hope that the readers will appreciate this addition to the literature.

References

Anonymous (2008) Hydrologic effects of a changing forest landscape. Report by the Committee on Hydrologic impacts of forest management, National Research Council. The National Academies Press, 180 pp

Costanza R, d'Arge R, de Groot R, Farber S, Grasso M, Hannon B, Limburg K, Naeem S, O'Neill RV, Paruelo J, Raskin RG, Sutton P, van den Belt M (1997) The value of the world's ecosystem services and natural capital. Nature 387(6630):253–260

Hamilton LS (2008) Forests and water. FAO Forestry Paper 155. FAO, Rome, p 78

Lindberg K, Furze B, Staff M, Black R (1997) Asia-Pacific forestry sector outlook study: ecotourism and other services derived from forests in the Asia-Pacific region: outlook to 2010. USDA Working Paper No: APFSOS/WP/24

Sousson J, Shestra B, Uprety L (1995) The social dynamics of deforestation: a case study from Nepal. Parthenon, Casterton Hill

World Bank (1997) 25 years of improvement. Chapter VII. Forestry. From website http://www.worldbank.org/html/cgiar/25years/forest.html

Taxonomic Index

Subject Index