FORESTRY SCIENCES

The Economics of Forest Disturbances

Wildfires, Storms, and Invasive Species

Thomas P. Holmes, Jeffrey P. Prestemon
and Karen L. Abt
editors

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FOREWORD

by Peter J. Roussopoulos, Director, Southern Research Station

The world and its ecosystems are repeatedly punctuated by natural disturbances, and human societies must learn to manage this reality. Often severe and unpredictable, dynamic natural forces disrupt human welfare and alter the structure and composition of natural systems. Over the past century, land management agencies within the United States have relied on science to improve the sustainable management of natural resources. Forest economics research can help advance this scientific basis by integrating knowledge of forest disturbance processes with their economic causes and consequences.

As the twenty-first century unfolds, people increasingly seek the goods and services provided by forest ecosystems, not only for wood supply, clean water, and leisure pursuits, but also to establish residential communities that are removed from the hustle and bustle of urban life. As vividly demonstrated during the past few years, Santa Ana winds can blow wildfires down from the mountains of California, incinerating homes as readily as vegetation in the canyons below. Hurricanes can flatten large swaths of forest land, while associated floods create havoc for urban and rural residents alike. Less dramatic, but more insidious, trees and forest stands are succumbing to exotic insects and diseases, causing economic losses to private property values (including timber) as well as scenic and recreation values. As human demands on public and private forests expand, science-based solutions need to be identified so that social needs can be balanced with the vagaries of forest disturbance processes.

Forest economics and allied disciplines can help provide solutions to natural resource management problems by linking policy questions to valuation frameworks. Utilizing the insights from biological, sociological, physical, and atmospheric sciences, economists can add value to forest policy decisions by identifying the trade-offs implicit in alternative policy scenarios. And, as economists are ever cognizant of the importance of budget constraints in making decisions, economic analysis provides insights into the efficient allocation of scarce resources to satisfy the needs of society.

Given the preponderance of natural disturbances currently affecting forests and human communities, *The Economics of Forest Disturbances: Wildfires, Storms, and Invasive Species* is a timely book. Its impact derives both from its presentation of a unifying framework for conducting economic analyses and through its careful explanations of the latest research advances. It is my hope that this book will contribute to an appreciation for the scientific issues raised by the study of forest disturbances and the techniques used by resource economists to understand them. Furthermore, I hope that these chapters stimulate new thinking about the means by which landowners, communities, and governments may become more efficient and effective stewards of the forests they treasure.

PREFACE

As Hurricane Ivan bore down upon the cozy mountain setting of Asheville, North Carolina in late September, 2004, a dedicated team of resource economists gathered to pool their knowledge about the measurement and management of forest based disturbances. Barely one week after Hurricane Frances drenched the region and, anticipating the potential chaos of downed trees, flooding, power outages, food and water shortages, and closed facilities, a decision was made to evacuate to a more hospitable location. In the end, the city was spared significant damage, and our flight appears to have been more precautionary than essential. Our disrupted meeting, however, provides a cogent example of the challenges faced by managers who must make forest protection decisions before the ultimate state of nature, ranging from brutal to benign, is revealed.

Forest protection efforts attempt to reduce the probability and/ or consequences of forest disturbances. Management interventions are costly, requiring significant financial outlays for activities such as aerial surveys, insect trapping, forest thinning, fuel reduction, fire suppression, insect and disease eradication, biological control, timber salvage, and ecosystem restoration. Decisions regarding when and where to incur these expenses are complicated by the fact that the timing and spatial extent of forest disturbances are highly stochastic and difficult to predict. Although the economic and ecological impacts of forest disturbances can be catastrophic, economically significant disturbance events typically occur with low probabilities in locations that are not well known in advance.

During the past decade, resource economists in government and academic institutions have made significant progress in defining and understanding the economic dimensions of forest disturbance processes, and the *raison d'être* for this book is to synthesize the most recent advances in this emerging field of applied economics. It is our premise that microeconomic theory provides a natural foundation for the integration of disturbance ecology with an array of empirical methods that can be used to illuminate the often subtle linkages between forest protection efforts and social welfare. As evidenced in many chapters of this book, this integration requires forays into econometrics, statistics, operations research, market and non-market valuation, and institutional analysis. The authors of this book have individually published in many of the premier peer reviewed journals in natural resource economics, forestry, and atmospheric science, and their work collectively represents hundreds of years of experience in characterizing and analyzing forest disturbances. The book that we have jointly created will, we hope, stimulate thought and further research.

This book was written so that policy-makers, managers, researchers, and students of natural resource economics could rapidly gain familiarity with this field of study. While some of the chapters are quite technical, and some sections of various chapters demand familiarity with advanced concepts, each chapter contains an introduction and conclusion that we hope are accessible to interested readers, and provide the essential messages.

For researchers interested in natural disturbances, this book provides both conceptual approaches and empirical methods that could be applied and advanced in future work. We strongly encourage contact with chapter authors when questions or new ideas arise, and encourage collaborative efforts in new research projects.

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SECTION I

THE ECONOMICS AND ECOLOGY OF FOREST DISTURBANCES

CHAPTER 1

AN INTRODUCTION TO THE ECONOMICS OF FOREST DISTURBANCE

Thomas P. Holmes, Jeffrey P. Prestemon, Karen L. Abt

1. FOREST DISTURBANCES AND ECONOMIC SYSTEMS

Increasing severity of recent wildfires, storms, pest outbreaks, and biological invasions has intensified concern among governmental agencies, private enterprises, and the general public regarding the future of forest resources. Economic analysis can help decision-makers understand the causes and consequences of forest disturbances, as well as evaluate trade-offs, and set priorities. It is the premise of this book that similarities existing among forest disturbances permit the development of a unified framework for economic analysis. This book sketches out how this framework could be constructed, provides an overview and summary of current research in the economics of forest disturbances, and illustrates how economic theory and empirical methods can be applied to address specific disturbances.

From an economic perspective, a forest disturbance can be defined as an event that interrupts or impedes the flow of goods and services provided by forest ecosystems that are desired by people. This definition parallels the ecological definition of a natural disturbance as "…any relatively discrete event in time that disrupts ecosystem, community, or population structure and changes resources, substrate availability, or the physical environment" (White and Pickett 1985, p. 7). Although timber harvesting and land use change are forest disturbances according to this definition, in this book we address large-scale natural disturbances that can be mediated and modified by human actions. Examples of such large-scale natural forest disturbances occurring during the past century include the chestnut blight which largely eliminated chestnut trees from hardwood forests in the eastern United States, Hurricane Katrina which blew down large swaths of forest in the United States Southeast, and the 1988 fires in Yellowstone National Park and surrounding forest ecosystems that burned more than one-quarter million hectares.

Catastrophic disturbances affect both public and private land, and the management interventions applied to mitigate damages will vary depending on the

T. P. Holmes et al. (eds.), The Economics of Forest Disturbances: Wildfires, Storms, and Invasive Species, 3–14.

objectives of the land manager. Management interventions are typically made with at least one of two goals in mind: (1) reducing the probability (risk) of an unwanted state of nature, and/or (2) reducing the negative consequences if an unwanted state of nature occurs. Interventions can be made prior to (prevention), during (suppression/eradication), or subsequent to (restoration/recovery) a natural disturbance event. Although interventions can be viewed as an optimal capital management problem from the perspective of a private timber manager (chapter 3), this book focuses attention on the economic analysis of interventions by public land managers and policy makers in forest disturbance processes, keeping in mind that public forest health protection programs are often designed to alter the behavior of private forest owners (chapter 19).

Although the use of economic analysis to inform decisions about the optimal level of public investment in forest protection is not new (chapters 16 and 18), the modern practice of forest disturbance economics continues to pose challenges. First, inference and prediction regarding future forest disturbances are characterized both by high variability (fluctuations not explained by deterministic processes) and uncertainty (limited knowledge of model parameters). Rigorous mathematical, statistical, and econometric models are required to address these special characteristics of forest disturbance production. Second, many of the effects of forest disturbances fall upon non-timber goods and services, such as outdoor recreation or aesthetic views. Because few non-market economic studies of disturbances have been conducted to date, and the current level of understanding of these impacts is limited, existing estimates of economic damages from forest disturbances may be severely biased by the lack of information on non-market impacts. Third, understanding the linkages between the costs of management interventions and changes in the net economic benefits provided by forest ecosystems is challenging because of the time lag between interventions and changes in the provision and value of ecosystem services. Consequently, empirical examination of the effects of protection investments requires long time spans, large data gathering efforts, and careful and innovative scientific enterprise.

To motivate subsequent analysis, the following section presents a broad characterization of various classes of forest disturbances and describes how specific disturbance characteristics constrain the set of management interventions that can be employed to mitigate economic impacts. Section 3 then provides an overview of the major lessons learned and presented throughout the remainder of the book. The chapter concludes in section 4 by offering suggestions for future research.

2. FOREST DISTURBANCE CHARACTERISTICS

Forest disturbances can be classified in three broad categories (table 1.1): abiotic events (storms, landslides, volcanoes, droughts, and floods), biotic events (insects, diseases, and invasive plants), and wildfires (a mix of abiotic and biotic disturbances). We further characterize forest disturbances using four key variables that

Table 1.1. Characteristics of forest disturbances and list of book chapters containing applied economic analysis.

Disturbance Type	Disturbance Sub-Type	Rate of Spread	Maximum Spatial Scale	Endogenous or Exogenous	Forest Management Strategy	Book Chapter
Abiotic						
Storms	Tornado	Hours	Small	Exogenous	None	—
	Hurricane	Hours → days	Very large	Exogenous	Salvage	11
	Straight-line wind	Hours→ days	Large	Exogenous	Salvage	—
	Ice	Hours→ days	Very large	Exogenous	Remove hazard trees	—
Volcanoes	n/a	Hours→ days	Large	Exogenous	Restore	—
Floods	n/a	Hours→ months	Medium	Exogenous	Restore	—
Drought	n/a	Months→ Years	Very large	Exogenous	Restore	—
Biotic						
Invasive plants	n/a	Years→ centuries	Very large	Exogenous	Eradicate	—
Pests	Exotic Insects and Diseases	Years→ centuries	Very large	Exogenous	Eradicate; slow spread; pre-emptive harvest; bio-control; salvage	2, 3, 11, 19
	Native Insects and Diseases	Months→ years	Large	Endogenous	Shorten rotation; genetic selection; chemicals; salvage	2, 3, 19
Mixed						
Wildfire	All Ignition Sources	Hours→ months	Large	Endogenous and exogenous	Suppress; reduce fuels; salvage; restore	2-18

influence economic costs and losses: rate of spread, spatial scale, whether the source of the disturbance is endogenous (inside) or exogenous to (outside) the forest, and forest management strategies employed to mitigate impacts. Forest disturbances are of interest to economists when their frequency and size are consequential enough to induce a management or policy response, and economic analysis of forest disturbance generally seeks to identify the optimal level of intervention in disturbance processes by balancing costs and losses. Our classification scheme recognizes that the scope and type of interventions available are a function of their biotic or abiotic nature, spread rate, and sources.

2.1 Abiotic Disturbances

Abiotic disturbances are characterized as deriving from energy sources originating outside of forests, and include climatic and geologic disturbances. Abiotic disturbances are stochastic and difficult to predict far in advance. Neither the probability of occurrence nor the magnitude of effects on forests can be significantly influenced by forest management. Although manipulation of stocking density or species selection may have some effect on reducing damage from abiotic events (Wilson and Baker 2001), the main forest management strategies are to salvage timber and restore landscapes and ecosystems.

2.1.1 Climatic events

Long-term climate change, acting as a slowly changing parameter that conditions the dynamic behavior of fast moving variables, can affect the entire constellation of forest disturbance processes including fire, drought, introduced species, insect and disease outbreaks, hurricanes, windstorms, ice storms, and landslides (Dale et al. 2001). Recognizing this, here we review the fast climatic events that affect forests—tornadoes, ice storms, hurricanes, droughts, and floods.

Tornadoes may damage tens to hundreds of hectares of forest cover (Glitzenstein and Harcombe 1988, Harcombe 1988, Peterson and Pickett 1995), and therefore may have a substantial impact on individual forest owners. However, they are generally too small and infrequent to have an impact on aggregate economic welfare. In contrast, straight-line winds occasionally have large-scale, catastrophic impacts, such as the July 4, 1999, blowdown that damaged 57,000 hectares of forest in the Boundary Waters Canoe Area in northern Minnesota (Schulte and Mlandenoff 2005). Other recent examples can be found in Europe (Nilsson et al. 2004). Tropical cyclones (typhoons and hurricanes) can also cause catastrophic forest damage, as was the case with Hurricane Hugo, which destroyed 1.8 million hectares of forest in South Carolina (Sheffield and Thompson 1992). Large scale climatic events can have substantial economic impacts on timber markets (chapter 9) and, additionally, can cause non-market economic losses to residential properties, public parks, and rural landscapes.

Large, damaging ice (or glaze) storms are infrequent, although they can occasionally cause tree damage over millions of hectares of urban and rural forests (Smith 2000). Management interventions typically focus on the decision of whether or not to remove damaged trees.

Major, infrequent floods can cause tree mortality if soils are saturated long enough to create anoxic conditions, which cause tree roots to die. This was the case in the Midwest flood of 1993 on the upper Mississippi River, which caused extensive mortality to trees and shrubs in the floodplain (Sparks et al. 1998).

Severe droughts can also induce economic costs and losses on forested properties, and impacts on urban forests and residential properties can be particularly severe. For example, the drought of 1934 killed about 25 percent of the trees and injured another 25 percent of the trees in Manhattan, Kansas (Stiles and Melchers 1935). The loss of aesthetic value, shading, and other non-market benefits of trees due to drought is compounded by the costs of removing and replacing dead and dying trees.

2.1.2 Geological events

Geological events are similar to climatic events, releasing large amounts of energy over a short time period. The two types of geological events that are most consequential to forests are volcanoes and landslides. Landslides occur in forested regions with steep topography and can be triggered by heavy rain or seismic events such as earthquakes or volcanoes. Earthquake caused landslides can be a major disturbance in tropical forests, and landslides ranging from 5,000-10,000 hectares have been observed (Veblen and Ashton 1978). Smaller landslides of less than 1 hectare may be quite common in tropical forests with steep slopes (Guariguata 1990). In the United States, landslides not associated with volcanoes are not known to influence forests to an economically significant extent.

Perhaps the best known volcanic eruption-related forest disturbance in the United States was the eruption of Mount St. Helens in southwest Washington in 1980. This eruption affected an area exceeding 70,000 hectares, including a variety of disturbances due to pyroclastic flow, tree blowdown, scorched trees, mudflows, and debris avalanches (Turner et al. 1997). Such occurrences in the volcanoes around the Pacific Rim are anticipated to occur every 100-1,000 years.

2.2 Biotic Disturbance

Biotic forest disturbances result from the propagation, growth, and spread of biological organisms that depend on forest resources to complete their life cycle. These disturbances include a diverse array of native and exotic insects, diseases, and invasive plants. Biotic disturbances are endogenous, and thus have a different suite of interventions available to affect the probability of occurrence and the extent of damages.

Invasive forest plants compete with native vegetation and can reduce the biological diversity of forest ecosystems. The growth and spread of invasive forest plants is relatively slow and predictable, and the primary control strategy is eradication followed by rehabilitation with fast-growing native plants (Miller 2003). A common invasive forest plant is kudzu (*Pueraria montana*), which is thought to cover 3 million hectares in the eastern United States and is expanding at the rate of 50,000 hectares per year (Forseth and Innis 2004).

Forest insects and diseases attack selected tree species, and pest outbreaks typically do not cause mortality to all trees in an infested area. However, population growth and spread can result in damages to public and private goods and services across broad landscapes. Because native trees have not co-evolved with exotic pests, they are particularly vulnerable to successful attack over the entire range of host species. Population growth of forest insects and diseases may follow non-linear or chaotic dynamics (Turchin 2003) and may be triggered or synchronized by atmospheric processes (Williams and Liebhold 2000, Liebhold et al. 2004). Insect and disease outbreaks may also interact with wildfire, complicating predictions of the timing, location, or intensity of biotic disturbances (Castello et al. 1995, McCullough et al. 1998).

The spatial spread of biotic disturbances occurs on time scales of years to centuries (e.g., gypsy moth), which is slow relative to the rate of spread of abiotic disturbances. This slower time scale, together with their host dependencies, permits a greater number of management strategies to be developed to combat biotic disturbances. Timber management strategies are based on the idea that the amount of timber at risk of damage or loss can be reduced by actions such as shortening timber rotations, pre-emptive harvesting of timber in anticipation of an imminent (actively spreading) insect or disease outbreak, and selection or propagation of trees with natural resistance to the pest (Cubbage et al. 2000). Other strategies can be used to protect the aesthetic and non-market values of trees and forests, such as tree removal, the application of chemicals to eradicate or slow the spread of insects or reduce the rate of disease progress on particular trees, and biological control. In the wake of biotic disturbances, timber salvage and ecosystem restoration strategies can be used to minimize short term economic impacts and restore long term economic values.

2.3 Wildfires

The temporal scale of wildfires is intermediate between biotic and abiotic disturbances—wildfires are briefer in duration than biotic disturbances but can be longer than abiotic disturbances. On a spatial scale, wildfires span more than four orders of magnitude (assuming that the smallest wildfire is in the order of 0.1 ha). Large wildfires can equal or exceed the size of most abiotic forest disturbances (except hurricanes) and yet are smaller in area than the most severe biological invasions.

As with biotic disturbances, wildfires are dependent on the availability of sufficient host material, and their extent and spread are limited by weather and climatic conditions. This dependency on host materials—fuels—provides the rationale for management strategies such as prescribed fire and mechanical fuel reduction which are applied with the goal of reducing wildfire spread and intensity. Because wildfires spread over hours to months, and because they often spread in relatively predictable directions, fire suppression can be used to limit fire sizes. Additionally, because the destructive character of large wildfires is patchy, substantial areas of forest may be killed while other areas remain relatively unharmed. Consequently, timber salvage following fire is often a viable management option. Restoration of areas burned by wildfires is also possible, mitigating negative impacts on watersheds and other future ecosystem values (Kent et al. 2003). Finally, it should be recognized that wildfires can convey benefits to fire dependent ecosystems, and the practice of letting some wildfires burn (referred to as "wildland fire use" in the United States) is becoming a more commonly accepted tool for public forest management (Doane et al. 2006).

3. OVERVIEW OF CHAPTERS

The structure of this book reflects our view that: (1) economic analyses of forest disturbance is enhanced by its congruence with ecological understanding (chapter 2); (2) forest disturbances can be modeled as stochastic economic production processes (section II); (3) consistent accounts of market and non-market economic effects of forest disturbances are pre-requisite to planning and decision-making (section III); and (4) economic models can be used to improve decisions taken to mitigate the negative economic consequences of forest disturbances and to set priorities (section IV). Below, we provide an overview of the contents of individual chapters.

3.1 Forest Disturbance Processes

From an economic perspective, forest disturbances are stochastic events that can be modeled as production processes. Some inputs into disturbance production are free (such as drought, lightning, or wind) and other inputs are purchased (such as capital and labor). The stochastic nature of disturbance processes suggests that disturbance outputs can be measured using probability distributions for metrics such as area burned or the number of large fires (chapters 2-7). By conducting statistical and econometric analyses, the economic consequences of management interventions can be identified as shifts in the stochastic distribution of disturbance events that occur in response to the application of purchased inputs.

Forest disturbances are characterized by high variance in the scale of physical and economic impacts (chapters 2-5) which can be explained by a number of factors. First, favorable site conditions for disturbance establishment and spread vary irregularly over time and space. Second, prior disturbances condition the landscape for subsequent events. Third, stochastic exogenous factors such as weather strongly influence the size of individual forest disturbances (chapters 4-6). Fourth, disturbances may be highly nonlinear in their responses to managerial and free inputs, resulting in discontinuous and catastrophic ecosystem behavior (chapter 2).

The processes that govern forest disturbances also include human caused wildfire via unintentional (e.g., campfires and debris burning escapes) and intentional behavior (arson). Arson wildfires can be understood as a production process involving a combination of weather and climate-dependent fuel conditions, economic variables, penalties, and psycho-social phenomena (chapter 7). Consequently, law enforcement and public education campaigns may be effective at reducing the frequency of arson and accidental fires. Managers may be able to mitigate the impacts of arson and other human caused fires through fuels management and pre-positioning of suppression resources.

3.2 Valuing the Economic Impacts of Forest Disturbances

The perspective presented in this book is that a full accounting of the costs and economic losses due to forest disturbances is prerequisite to effective planning and priority setting. The first step is to establish a consistent accounting and data collection framework (chapter 8). Economic systems are connected over time and space—many goods and services are substitutes and complements in consumption, and many inputs are substitutes and complements in production—and economic assessments are sensitive to spatial scale (geographic area to be assessed), temporal scale (time span used to assess impacts), and sectoral scale (economic sectors to be included). Economic assessments need to be conducted across multiple scales, and decision-makers need to be informed of the sensitivity of economic measures to the scale at which economic models are applied.

Forest disturbances such as insect epidemics, hurricanes, and wildfires can have extreme impacts on markets for goods obtained from forests. In timber markets, timber losses and damages affect economic equilibria, both through the pulse of timber salvaged from an event and through reductions in stocks of standing timber (chapter 9). Economic welfare is redistributed after a catastrophic forest disturbance, with some economic agents gaining (e.g., consumers of wood products in the short-term) while others lose (e.g., producers of damaged timber). Timber salvage policies instituted by governmental agencies should be sensitive to the redistributional impacts of such policies.

Forest disturbances can induce a significant loss of wealth for private property owners in the wildland-urban interface. For example, changes in risk perceptions resulting from nearby catastrophic wildfires can induce private property value losses reaching millions of dollars in a single community (chapter 11). Similarly, tree mortality caused by an exotic forest insect can cause losses in property values exceeding a million dollars in an individual community due to the loss of aesthetic values and the costs associated with tree removal (chapter 11). Because wildfires reduce the value of private residential properties, private homeowners have a substantial willingness to pay for public programs designed to protect residences and communities from wildfires (chapter 12).

Wildfires can destroy recreational infrastructure and can alter the quality of outdoor recreation sites. Although few studies have been conducted to evaluate the impact of wildfires on the demand for outdoor recreation, preliminary evidence suggests that wildfires may increase the number of Wilderness visitors in the short-run, due to an influx of curiosity-seekers (chapter 10). Over the span of several decades, however, the economic value of wilderness areas that have experienced large wildfires may decrease because of visitation reductions brought about by the loss of mature forests and the presence of less desirable forest conditions. More research is needed to understand the impacts of wildfires, storms, and invasive species on all forms of outdoor recreation.

3.3 Decision Making in Response to Forest Disturbances

Forest disturbances typically involve an element of surprise, and forest protection decisions must be made before the ultimate state of nature is revealed. A general approach to forest protection is to reduce the risk (probability) that an unwanted state of nature will occur and to take steps that would reduce negative consequences in the event that an unwanted state does in fact occur (chapter 19). One example of this approach is evidenced by the various state and local governmental agencies that have established programs to reduce wildfire hazards in high risk areas through regulations on land-uses and vegetation management (chapter 14). Another example is provided by fuel management programs implemented by private and public forest landowners, which have been shown to reduce both damages and subsequent suppression costs (chapter 13). Much may be learned by examining the successes and shortcomings of existing programs and policies.

Another approach to managing uncertainty about future conditions is to construct forecast models using the best available data. Econometric forecasts of future wildfire suppression costs provide a rigorous means of establishing budget requests by federal land management agencies (chapter 17). Econometric models can also quantify the degree of uncertainty about parameter values and test hypotheses about proposed driving variables. Loss functions can be used to compare the performance of various models and allow managers to use planning tools in ways that reflect their priorities and risk perceptions.

Economists are cognizant of the role that incentives play in decision-making (chapter 16). Incentives regarding wildfire suppression and overall fire program management influence the costs and benefits of high profile suppression efforts by federal agencies. For example, funding wildfire suppression with emergency funds provides little incentive for cost containment (chapter 16). Further, because wildfires can produce ecological benefits, recognition by incident managers of the fuel treatment or other benefits of fire could facilitate improvement of management approaches and reduce associated costs (chapters 15 and 16).

Programs designed to protect forest ecosystems are complex and include many interacting components. For example, governmental programs for fire management include components for fire prevention, detection, fuels management, suppression, and post-fire site rehabilitation. Because of the linkages and feedbacks between components, economic efficiency is compromised when analysis is conducted component-by-component (chapter 18). The development of integrated forest protection programs will likely be worthwhile but present significant challenges because they require models and tools that accurately describe the trade-offs among alternative program inputs.

Forest health protection from invasive species is a public good, in that the benefits from forest protection are shared by other members of the community. This context provides the justification for government intervention. Further, forest health protection is a weakest-link public good. The weakest-link character of forest health protection relegates the level of forest protection attained by a community to the weakest members of the community. Consequently, effective forest health protection programs require that the weakest links be strengthened by targeting information to those most likely to engage in risky behavior. In particular, information describing the weakest-link nature of forest protection should be targeted at private landowners to enhance the likelihood that they will participate in forest protection programs (chapter 19). Weakest links can be identified using economic surveys of household behavior.

4. RESEARCH NEEDS

Economic models need to account for the complexity of disturbance processes so that the efficiency and efficacy of management interventions can be realistically assessed. Nonlinear dynamics and spatial diffusion are challenging attributes of forest disturbances, and further development of statistical, econometric, mathematical, and simulation models that address management interventions across various temporal and spatial scales are needed. In particular, we suggest that research is needed that would enhance the ability to predict catastrophic changes in ecological and economic variables.

Preliminary evidence suggests that the non-market economic impacts of forest disturbances are substantial, but few studies have been conducted. Further studies of the economic damages caused by forest disturbances to private property values, to ecosystem service values provided by public and private forests, and to human health (e.g., smoke from wildfires, wildland use fires, and prescribed fires; dermatitis from caterpillars) are needed. A more comprehensive understanding of non-market economic impacts would illuminate the severity of these threats and provide a larger knowledge base for improved management decisions.

Fire and forest health protection programs need to be evaluated as integrated systems, rather than being evaluated in isolation. The wildfire program is an example, where analysis has focused on the market effects of timber damages from wildfire and wildfire suppression costs. Yet wildfire programs also encompass the outcomes from fuels management and the potential positive impacts of restoring ecosystem function and reducing future wildfire. More generally, forest disturbances such as wildfire, insect and disease outbreaks, and biological invasions interact across broad spatial and temporal scales. Economic and ecological models for integrating the various components of fire and forest health protection programs are needed and will likely lead to lower program costs and greater benefits to society.

The time lag between the imposition of a management intervention and the occurrence of a catastrophic event creates uncertainty about the efficacy of management actions. Models of decision-making under uncertainty is a key topic for future research, and models that incorporate learning as new information is revealed are needed.

Finally, data required for economic analyses of forest disturbances still need improvement. Although economists have developed specialized econometric methods for analyzing non-experimental data, the data available for analyzing forest disturbances is often inconsistent, fragmentary, or unavailable over the time spans at which disturbance processes operate. Improved coordination between economists and the data collection operations conducted within land management agencies would enhance the ability for economists to evaluate trade-offs and provide meaningful and timely information to policy-makers.

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CHAPTER 2

FOREST ECONOMICS, NATURAL DISTURBANCES AND THE NEW ECOLOGY

Thomas P. Holmes, Robert J. Huggett, Jr., and John M. Pye

1. INTRODUCTION

The major thesis of this chapter is that the economic analysis of forest disturbances will be enhanced by linking economic and ecologic models. Although we only review a limited number of concepts drawn generally from mathematical and empirical ecology, the overarching theme we present is that ecological models of forest disturbance processes are complex and not particularly well-behaved from an economic perspective. We discover that standard concepts in the economists' tool kit, such as asymptotic equilibrium and convex production, may not adequately represent the dynamic behavior of forest disturbances. Consequently, other tools for economic analysis will be required.

This chapter proceeds by first sketching out the economic problems deriving from the peculiar temporal and spatial dynamics associated with forest disturbances (section 2). Then we provide a brief overview of select topics in ecological literature supporting the view that some important forest disturbances exhibit multiple- or non-equilibrial processes and that, additionally, stochastic factors induce high variation in the spatial pattern of disturbance production (section 3). These themes are illustrated by reviewing two models: (1) the classic spruce budworm model of pest outbreak, demonstrating how the interaction of slow and fast ecosystem variables cause multiple equilibria (section 4), and (2) a cellular automata model of forest fires, which demonstrates how the local interaction of stochastic processes can generate the emergence of unconventional spatial signatures at larger spatial scales (section 5). The chapter ends with a summary of the main points and some suggestions for future research (section 6).

2. ECONOMIC EQUILIBRIUM, NON-CONVEX PRODUCTION, AND SPATIAL SCALE

Since the early decades of the twentieth century, the concepts of equilibrium and comparative static analysis (the qualitative change in equilibrium conditions in

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response to a change in a structural parameter) have been central in the development of neoclassical economic theory. Much credit for this development is due to Samuelson (1947) who emphasized that comparative static analysis needs to correspond with an underlying, asymptotic dynamic model. In the standard market model, for example, excess demand is usually thought to cause an increase in price until equilibrium is restored. This result can be found as the solution to an ordinary differential equation describing price dynamics, for which the root of the characteristic equation for the complementary function is negative (Chiang 1974, p. 472-473).[1] The resulting equilibrium is said to be asymptotically stable (Tu 1994, p. 33).

Of particular relevance to this chapter, Samuelson (1947) further recognized that some economic processes move rapidly relative to other, slow long run processes and that it is often convenient to treat slow processes (such as changes in the stock of capital) as fixed parameters while concentrating on the fast processes of economic interest (such as the level of investment, income, or employment). He goes on to note that due recognition needs to be given to the evolution of the slow variables in order to study the development of the economic system over time.[2]

In this chapter, we propose that some economically important forest disturbance processes, such as pest outbreaks and fires, result from the interaction of variables across fast and slow timescales, and that policy-relevant economic models need to recognize the impacts of long-term ecosystem dynamics on the fast behavior of economic variables. Because movement in a slow ecosystem variable (e.g., forest foliage, fuel accumulation) can induce a sudden, catastrophic eruption in a fast variable (e.g., area infested by pests, area burned) which is linked, in turn, to various economic variables (e.g., pest eradication costs, fire suppression costs, economic damages), simple comparative static analysis may provide uninformative predictions of changes in economic variables. This more complex situation arises when the root(s) of the characteristic equation describing system dynamics are non-negative, and the Implicit Function Theorem breaks down (Tu 1994,

[1] It may be recalled that the general solution to a first-order differential equation is of the form $p(t) = Ae^{rt}$ where p (say, price) is a function of time (t) and r is the root of the characteristic equation of the complementary function describing the deviation of p(t) from asymptotic equilibrium. If $r < 0$, then p(t) will asymptotically converge to the particular integral describing equilibrium as $t \rightarrow \infty$.

[2] This decomposition into slow and fast variables was also suggested by Simon and Ando (1961) regarding the aggregation of variables in a dynamic macroeconomic system. They argued that aggregation could be accomplished by classifying the variables of an economic system into a small number of sectors. Because the dynamic interactions within a sector reach equilibrium relatively rapidly, an index representing the equilibrium condition for each sector could be established and then the slower interactions between sectors could be studied.

p. 241). Intuitively, the equilibrium path is not asymptotically stable and may suddenly jump to a different domain.[3]

A recent Symposium held by the Beijer Institute of Ecological Economics in Stockholm focused attention on the implications of discontinuities in ecosystem dynamics for economic analysis, and emphasized the importance of understanding Nature's non-convexities (Dasgupta and Mäler 2004).[4] One of the themes of the Symposium was that bifurcations in equilibrium paths, representing ecological thresholds, manifest across time and therefore require dynamic analysis. Non-convexities in ecosystem production due to discontinuities are consequential for economists because, under these conditions, a decentralized price system cannot reliably guide the economy to an optimal solution and other institutions are required for efficient resource allocation (Dasgupta and Mäler 2003).[5] Fortunately, when the economic planner is confronted with discontinuous ecosystem production, optimal economic programs can be evaluated using optimal control methods (Brock and Starrett 2003, Crépin 2003, Dasgupta and Mäler 2003, Mäler et al. 2003).

Although economists are generally familiar with dynamic processes operating over time, they are less familiar with dynamic processes operating over space. Spatial dynamics have been extensively studied by ecologists who have recognized that characteristic spatial patterns in complex adaptive systems can emerge purely from interactions at the local level (Levin 2002, Chave and Levin 2003, Hastings 2004, Pascual and Guichard 2005), and the use of statistical analysis for detecting complex patterns of spatial dynamics is an emerging discipline in ecology (Gumpertz et al. 2000, Turchin 2003, Liebhold et al. 2004).

Statistical models have been productively employed in the economic analysis of management interventions to control wildfires (Davis 1965, Ward et al. 2001, Prestemon et al. 2002, Bridge et al. 2005) by recognizing that, if wildfire occurrences converge to a statistical distribution, then interventions can be evaluated by identifying corresponding changes in the parameters of the statistical distribution. Some spatial patterns associated with forest disturbances are not well-behaved in

[3] The case of the backward-bending supply curve provides a good example of an unstable equilibrium separating two stable equilibria. Small shifts in demand can cause catastrophic jumps in price and quantity (Clark 1976).

[4] A standard assumption of economic analysis is that production sets are convex, where a set is convex if the line joining any two points of the set is also entirely within the set. Non-convexities in forest production have been studied for the case of multiple local optimal solutions in a continuously differentiable multiple-use benefit maximization problem (Swallow et al. 1990) and for the case of multiple-use forest production with bifurcations occurring in the production possibility set (Crépin 2003).

[5] Standard comparative static analysis of forest protection programs that equate the marginal benefit of a management intervention with the marginal input cost may likewise provide inadequate guidance for optimal economic decisions if forest disturbance production is non-convex.

that they are scale invariant (i.e., they display self similar patterns across scales of measurement) as typified by power law relationships (Malamud et al. 1998, Chave and Levin 2003, Malamud et al. 2005). In such cases, innovative statistical methods are required to conduct economic analysis (chapter 4).

3. DISTURBANCE ECOLOGY AND THE LOSS OF BALANCE

The balance of nature paradigm has a long-standing tradition both in Western culture and in the development of ecological theory (Egerton 1973). A quasi-scientific foundation for the balance of nature perspective is found in the essay "The Oeconomy of Nature" (1749), written by the famous Swedish biologist Carl von Linné. In this article, Linneaus presents a view of nature that is divinely ordered and functions like a well-oiled machine (Worster 1994). This perspective was echoed throughout the 19th century, and can be found in the works of George Perkins Marsh (who authored the widely cited conservation classic *Man and Nature* in 1864) and Charles Darwin, both of whom accepted the view of nature as fundamentally orderly and maintaining a permanent structure (Wu and Loucks 1995).

More modern statements of the balance of nature paradigm are found in mathematical-ecological concepts such as equilibrium, stability, steady-state and homeostasis (De Angelis and Waterhouse 1987). Separation of the mathematically tractable concept(s) of equilibrium from the more vague notions of balance-of-nature has allowed ecologists to test equilibrium theories and models, at least in principle. However, even fundamental mathematical models of population equilibrium, such as density dependent regulation of population size, are often empirically untestable because the scale at which density dependence operates may be much broader than the scale at which observations are typically made (DeAngelis and Waterhouse 1987). Notably, when models of static ecosystem stability have been tested, they often fail (Wu and Loucks 1995).

Much interest in ecology has focused on thresholds and alternate stable states in ecosystems (May 1977). More than three decades ago, a critique of the equilibrium perspective of nature was advanced by Holling (1973) who argued that the classical equilibrium concept cannot account for the transient behavior observed in many ecological systems. As an alternative, he proposed a model based on the idea of resilience, which he defined as a measure of the ability of an ecosystem to absorb disturbance before flipping over to an alternative domain of attraction. In particular, Holling (1973) argued that random disturbances such as wildfires and pest outbreaks can drive ecosystems from one domain of attraction to another and he proposed that research should focus on locating the domain boundaries.

A second approach to thinking about ecosystem stability that does not rely on asymptotic equilibrium was provided by Botkin and Sobel (1975). By examining the fire history of the Boundary Waters Canoe Area (BWCA) in northern Minnesota as described by Heinselman (1973), they concluded that static stability was an inappropriate concept either for the analysis or management of fire-dependent ecosystems. They proposed a definition of stability based on θ-persistence which characterizes the bounds attained by ecosystem states (characteristics of interest such as biomass or population). In their view, the trajectory of an ecosystem is θ-persistent about state x_0 if $|x_0 – x_t| \leq \theta$ for all $t \geq 0$. Here, x_0 does not connote a state of equilibrium, but rather a state within the system. By emphasizing the bounds attained by ecosystem states, this perspective is consistent with natural variability concepts that are currently applied by resource managers to maintain biological diversity and understand human impacts on forests (Landres et al. 1999).

Along the trajectory of an θ-persistent ecosystem, various ecological states can be repeated, and thus represent recurrent states. Botkin and Sobel (1975) argue that management interventions should focus on maximizing the size of the state space that is recurrent and that minimizes the recurrence time of desirable states. They go on to argue that the satisfaction of these two conditions "is equivalent to ensuring the aesthetically desirable wilderness status—an ecosystem having maximal structural (species) diversity" (p. 636). We prefer to view this conjecture as a hypothesis and suggest that forest ecosystems in continual flux offer opportunities for economists to evaluate public preferences for dynamic, time-varying ecosystem characteristics.[6]

The shift away from a focus on asymptotic dynamics in ecology can also be found in Hastings (2004) who proposed that transient ecosystem dynamics may hold the key to long-term ecological understanding, where the term "transient" implies rapid changes in the state variable(s) of interest. An illuminating example of transient dynamics is the study of epidemics by Kermack and McKendrick (1927) who, employing a system of nonlinear differential equations, demonstrated that the outbreak and termination of an epidemic depends upon a particular set of infectivity, recovery, and death rates and a threshold population density.[7] The key to this approach was to focus attention on the time course of an epidemic and not on the asymptotic state (which is, of course, the state where the epidemic dies out and may occur where only a small proportion of the susceptible members of the population have been infected). Further, the timescale of an epidemic in humans is shorter than the average human lifespan, and it is this juxtaposition of timescales that has been identified as the essential element for understanding transient dynamics in ecosystems (Rinaldi and Muratori 1992, Carpenter and Turner 2001, Rinaldi and Scheffer 2001, Hastings 2004).

[6] See chapter 10 for recent empirical evidence of post-wildfire wilderness demand.

[7] For an application of epidemiological methods to an invasive pathogen of trees, see Swinton and Gilligan (1996).

An alternative perspective argues that because ecosystems are open systems under the influence of stochastic processes, they are best characterized as nonequilibrium systems (DeAngelis et al. 1985, DeAngelis and Waterhouse 1987). This view is supported by historical evidence on wildfires and pest epidemics. For example, fire history data reconstructed from tree rings sampled in giant sequoia groves in the Sierra Nevada Mountains suggest that fire patterns are a nonequilibrial process synchronized by weather events (Swetnam 1993). This view is additionally supported by long-term fire history data from the Yellowstone sub-alpine ecosystem (Romme 1982). Stochastic meteorological phenomena have also been identified as key variables affecting outbreak dynamics for several forest insect pests (Peltonen et al. 2002).

The statistical analysis of forest disturbances has been enhanced by recent developments in phenomenological time series analysis that integrates deterministic nonlinear ecological models of population dynamics with stochastic variables due to exogenous factors. Berryman (1992) shows how to identify models for analyzing ecological time series using the autocorrelation and partial autocorrelation functions familiar to economists, and Berryman and Turchin (2001) later modified the standard time series model by introducing the partial rate correlation function. Turchin (2003) provides a detailed analysis of complex population dynamics and demonstrates that, for the economically important case of the Southern Pine Beetle, population fluctuations exhibit chaotic behavior.[8]

In sum, this review finds substantial evidence in the ecology literature that "the equilibrium view of ecological systems, which has always had a fair number of skeptics, now seems unsatisfactory to a large fraction, perhaps a majority, of ecologists" (DeAngelis and Waterhouse 1987, p.1). A pressing challenge for forest economists is to incorporate complex ecosystem dynamics into economic analyses of forest disturbances and, ultimately, to integrate economic analysis with decision-making (Pielke, Jr. and Conant 2003) and adaptive ecosystem management.

4. SLOW-FAST ECOSYSTEM PROCESSES: TEMPORAL DIMENSION

At an intuitive level, wildfires and biotic forest disturbances such as insect and disease outbreaks must rely to some degree upon the forest resource. This intuition has been formalized in ecological models by viewing forest disturbances as resulting from the interaction of variables across time scales. For example, the change in forest biomass during wildfires takes place on the scale of hours to months, while the growth of trees occurs on the scale of centuries. Models

[8] For a general reference on the evidence for chaos in ecology, see the work of Hastings et al. (1993). Turchin and Taylor (1992) provide an accessible overview of complex dynamics in ecological time series.

designed to describe the evolution of a forest ecosystem over time in the presence of wildfire would have to simultaneously integrate the equations of motion for the slow and fast variables which, for practical purposes, is not possible. However, mathematicians have developed special methods for solving this type of problem, known as singular perturbation theory (Kokotovic 1984). In this section, we provide a simple example that demonstrates how singular perturbation methods can be used to characterize the temporal dynamics of an important forest pest, the spruce budworm, which causes severe mortality in boreal forests in eastern Canada and the northeastern United States on roughly 40 year cycles (Boulanger and Arseneault 2004).[9]

Simply stated, the singular perturbation method separates the dynamic variables into slow and fast categories which allow the fast and slow dynamics to be studied sequentially rather than simultaneously (Simon and Ando 1961, May 1977, Rinaldi and Muratori 1992, Rinaldi and Scheffer 2000). In the spruce budworm model, spruce budworm is a fast variable f(t) and forest foliage is a slow variable s(t):

$$\begin{aligned} \dot{f}(t) &= F(f(t), s(t)) \\ \dot{s}(t) &= \varepsilon S(f(t), s(t)) \end{aligned} \tag{2.1}$$

where the dot notation is used to represent the rate of change over time, and ε is a constant representing the ratio of the slow and fast time scales. For example, if forests grow on the scale of centuries and budworms grow on an annual scale, then $\varepsilon = 0.01$. Since the budworm dynamics occur much faster than forest growth, the quasi-equilibrium position for budworms can be evaluated by treating s(t) as a fixed parameter s(0):

$$\dot{f}(t) = F(f(t), s(0)) \tag{2.2}$$

which is equivalent to the singular case $\varepsilon = 0$.

Ludwig and others (1978) showed that budworm dynamics f(t) result from the interaction of the per capita rate of budworm growth and the per capita rate of budworm death, due to predation by birds. Avian predation is limited at low levels of budworm density because budworms are scarce and predators are not rewarded for specializing on that prey. At higher budworm levels, predation is limited by satiation—a relatively fixed population of birds can eat only a limited number of budworms. This behavior gives rise to a non-convex per capita death rate function (fig. 2.1). When the per capita budworm growth rate is greater (less) than the predation rate, budworm density increases (decreases). Thus, steady-

[9] We note that the model we present is deterministic while recent research on forest pest dynamics emphasizes the importance of stochastic factors (Peltonen et al. 2002). The importance of the model is that it provides a simple demonstration of non-convex ecosystem production with multiple steady-states.

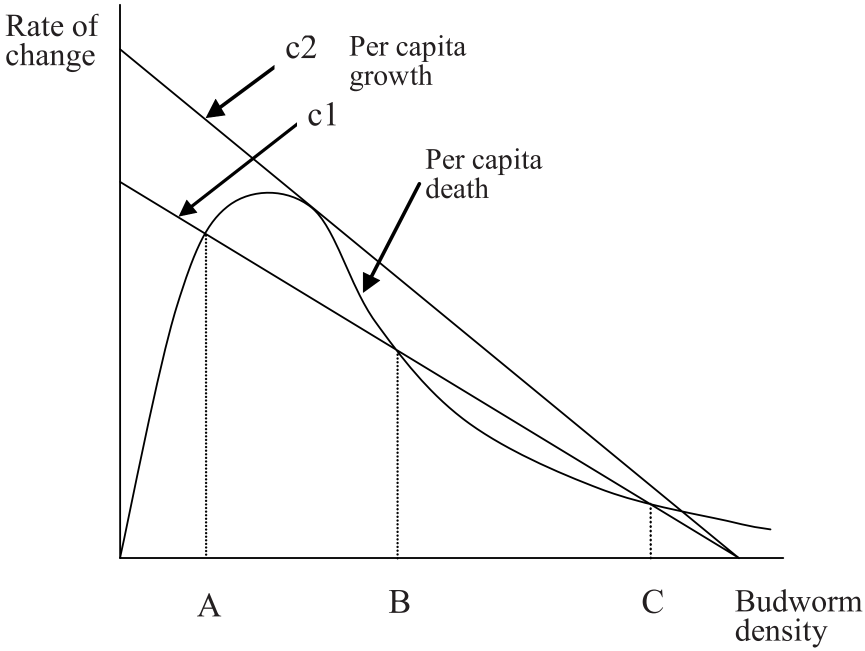

Figure 2.1. Spruce budworm dynamics demonstrating multiple steady states and catastrophic outbreak. Adapted from Ludwig, et al. (1978), with permission.

state positions for the budworm, holding forest growth constant, are found at the intersections of the per capita growth and predation curves. Considering an initial budworm growth curve (c1), three equilibrium points can be identified, for which two population levels are stable (A and C) and one is unstable (B).

The next step in singular perturbation analysis is to examine what happens to the equilibrium positions of the fast variable for any given value of the slow variable, $f^e(s)$. In the case of the spruce budworm, this can be represented by an upward rotation of the budworm growth function as the forest foliage parameter increases (fig. 2.1). Assume that budworm populations are initially at a low level (A). As forest foliage increases, the lower equilibrium converges with the unstable equilibrium. When these two equilibria become coincident (c2), budworm populations jump to the upper equilibrium and an outbreak is underway.

However, this is not the end of the story. Changing the time unit from 1 (for the fast variable) to 1/ε (for the slow variable), and substituting $f^e(s(t))$ for f(t), the dynamics of the slow variable are:

$$\dot{s}(t) = S[f^e(s(t)), s(t)]. \tag{2.3}$$

As forest foliage is consumed by budworms, the slow parameter (the amount of forest foliage) decreases and the budworm growth function rotates downwards.

At first, the unstable equilibrium re-appears and slowly moves towards the upper equilibrium. However, when forest foliage is at a low level, avian predation can again regulate the budworm population, and the population will crash. In figure 2.1, this occurs when the per capita budworm growth function lies nearly along the horizontal axis, and only the lower, stable equilibrium remains. As forest foliage regrows, the pattern is repeated and the cycle of forest growth followed by a rapid release of accumulated capital recurs.[10]

Although this model of ecosystem dynamics was presented in a heuristic fashion, it provides a qualitative illustration of the complexity of designing forest protection policies that maximize economic welfare.[11] Because the threshold for spruce budworm outbreak is not the same as the threshold for population collapse, the behavior of the system is history dependent (i.e., it exhibits hysteresis), and optimal policies depend upon the system memory. For example, historical evidence illustrates that forest-wide insecticide spraying in areas with high budworm densitites and imminent severe tree mortality can keep budworm populations in a perpetual outbreak condition (Ludwig et al. 1978). An alternative approach is to spray early when budworm egg masses are in isolated areas and at low densities (Stedinger 1984). Instead of focusing on the dynamics of the fast variable (budworms), alternative management strategies focus attention on managing the slow variable (trees) by harvesting live trees (Shah and Sharma 2001). Such a strategy may help prevent an outbreak, but once an outbreak is underway, vast amounts of timber would need to be harvested to cause a population collapse. In such a situation, the optimal policy may focus on salvaging dead and dying timber (Irland 1980). A complete economic analysis of the spruce budworm problem would thus need to evaluate the trade-offs between a suite of economic variables including spraying costs, public welfare impacts of increased use of insecticides, timber market impacts of pre-emptively harvesting green timber and timber salvage, and the non-market economic impacts of changes in forest health.

The slow-fast interaction leading to spruce budworm outbreaks suggests that management strategies may need to simultaneously address both pest and forest dynamics rather than focusing exclusively on the dynamic behavior of a single variable. This approach is evidenced in the recent paradigm shift in fire management (Dombeck et al. 2004). The long standing “10 a.m.” policy that sought to control all wildfires by the morning after they were first detected focused on direct control of the fast variable (fire) to protect lives and property and ensure a predictable supply of timber. However, suppression or exclusion of the fast vari-

[10] Technically, this ecological process is described as a cusp catastrophe because the dynamics can jump back and forth between states, and is therefore reversible. For the application of a cusp catastrophe to wildfires, see (Hesseln et al. 1998).

[11] Grimsrud and Huffaker (2006) demonstrate the complexity of finding the solution to an economic optimization problem that is subject to constraints incorporating slow-fast dynamics.

able (fire) can lead to a critical change in the slow variable, forest growth/fuel accumulation, resulting in larger and more intense fires (GAO 1998, GAO 1999, Stephens and Ruth 2005). Increased prescribed burning, wildland fire use, and mechanical fuel reduction programs are evidence of the resulting paradigm shift away from a policy of fire suppression and exclusion toward one that recognizes fire as a vital ecosystem process. Unfortunately, it is not yet known what effect fuel reduction efforts will have on wildfire dynamics or fire suppression costs. Until the linkages between these slow and fast ecosystem variables are understood, a full economic analysis of fire protection programs will be incomplete.

The long-term periodicity in the spruce budworm example provides another lesson. Data sets spanning decades or centuries may be required to understand slow-fast ecosystem dynamics (Holling and Gunderson 2002). Data that do not incorporate evidence of the feedback between fast and slow variables would likely yield misleading analyses and inadequate policy prescriptions.

Finally, we note that climate change might alter slow-fast ecosystem dynamics for some important forest disturbances (Dale et al. 2000, Logan et al. 2003). Westerling et al. (2006) identified a statistically significant change in the annual frequency of large (> 400 ha) western United States wildfires after 1987 that was correlated with mean March through August temperatures, suggesting that climatic thresholds may be important for fire dynamics. Others (Logan and Powell 2001, Logan et al. 2003) have suggested that global warming may be an important factor in widespread insect epidemics such as the recent Mountain Pine Beetle outbreak in British Columbia.

5. SLOW-FAST ECOSYSTEM PROCESSES: SPATIAL DIMENSION

In the previous section, we demonstrated how the interaction of slow and fast ecosystem variables can give rise to transient dynamics and rapid changes in ecosystem states. Our goal in this section is to show how the interaction of slow and fast variables can give rise to characteristic spatial patterns that are amenable to statistical analyses. Because the ecological literature on spatial spread and spatial pattern is extensive and succinct reviews are available elsewhere (Hastings 1996, Hastings et al. 2005), we are not compelled to review the entire span of this literature. Rather, we focus our attention on a recent innovation in spatial modeling, cellular automata, that utilizes Monte Carlo simulation to analyze spatial pattern. A more focused review of this literature reveals that some unconventional statistical distributions are associated with forest disturbances. Consequently, novel statistical methods may be required for economic analysis of interventions into these processes (chapter 4).

Cellular automata have been developed to model a variety of abiotic phenomena, including fire and wind damage in forests (Pascual and Guichard 2005). These models consist of a grid of cells on which discrete system dynamics unfold according to rules that specify the consequences of interactions between cells in a local neighborhood. Iteration of these models over many time steps simulates how characteristic patterns of disturbance can develop across forested landscapes, and these spatial patterns are characterized by spatial power laws (Malamud 1998). This spatial signature is not pre-determined by the specific rules governing local interactions. Rather, it is a self-emergent property resulting from many interactions across the entire system. Cellular automata Monte Carlo simulations generate simulated wildfire size distributions similar to those observed in fire data recorded in temperate and boreal forests (Ricotta et al. 1999, Cumming 2001, Song et al. 2001, Zhang et al. 2003, Malamud et al. 2005).

A cellular automaton uses a d-dimensional lattice with L^d regularly spaced cells to represent the spatial organization of the ecosystem. During the simulation, the value of each cell is updated in discrete steps according to deterministic or probabilistic rules, and rules governing cell behavior are applied equally to all cells. Thus, there is no local heterogeneity governing system behavior. Given a set of rules describing nearest neighbor interactions, the system is simulated over many time steps and the spatial pattern of disturbed areas is analyzed.

Drossel and Schwabl (1992) describe a prototypical forest fire model where each site (cell) is either empty, occupied by a living tree, or occupied by a burning tree. The system is updated in discrete steps using the following rules: (1) empty site → living tree with probability p, simulating regeneration that is well-mixed across the forest matrix; (2) living tree → burning tree with probability f, simulating an ignition source such as lightning; (3) living tree → burning tree if at least one immediate neighbor is burning, and (4) burning tree → empty site. Simulation of this model over many time steps results in a fire size-frequency density f(·) characterized by a power-law (Malamud et al. 1998, Malamud et al. 2005):

$$f(Area_i) = \alpha Area_i^{-\beta} \tag{2.4}$$

where $Area_i$ is the area burned in normalized fire class i, and α and β are parameters.[12] As emphasized by Pascual and Guichard (2005), the power-law spatial pattern that results from many iterations of this model depends on a double separation of time scales. Fire spread is a fast variable, forest growth is a slow variable, and the rate of fire ignition (lightning strikes per unit area) is a very slow variable.

One convenient aspect of a power law is that a plot in log-log space results in a straight line. Figure 2.2 illustrates this result for the empirical size-frequency

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[12] Following Malamud et al. (2005), frequency density is defined as:

$$f(Area_i) = \frac{N_F}{\delta Area_i}$$

where N_F is the number of fires in a bin width of $\delta Area_i$, and bin widths increase with fire size.

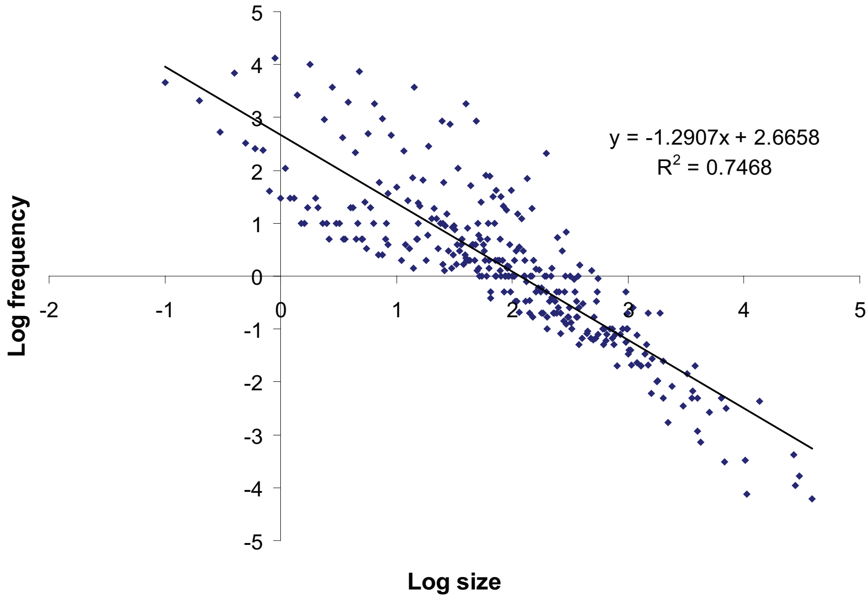

Figure 2.2. Frequency density for wildfires in Florida caused by lightning, showing power law behavior on a log-log scale.

distribution for lightning fires in Florida, U.S.A. A linear function fitted to the Florida data shows that a power law representation fits the data well across 4 orders of magnitude. The fitted function over-predicts fire frequency for fires exceeding about 5,000 acres, perhaps due to the fragmentation of forest fuels on the landscape (Ricotta et al. 2001).

Power law functions have heavy tails—most of the disturbance occurs in a small number of large events. Power laws are unconventional statistical distributions as they have infinite variance and may have an infinite mean. However, robust statistical procedures are available for analyzing spatial power law distributions (chapter 4). Power laws have also been used to describe the spread of plant pathogens (Shaw 1994), so their use in economic modeling of forest disturbance may become more common as their properties become more widely understood.

6. CONCLUSIONS

Over the past few decades, the view that nature is balanced and tends to return to a stable equilibrium following a natural disturbance has been challenged by alternative paradigms. Accompanying this change has been a shift in perspective regarding the role of forest disturbances such as wildfires, insect outbreaks, disease epidemics and storms. No longer are disturbances viewed as nuisance variables that temporarily perturb ecosystem equilibrium. Rather, disturbances are now generally regarded as key processes driving the temporal and spatial structure of landscapes. In this chapter we have highlighted how the interaction of slow and fast variables contributes to forest disturbance processes across temporal and spatial scales.

The literature we reviewed demonstrated that forest disturbance production functions represent the complex, transient behavior of ecosystems. Certain ecosystem processes such as wildfires, pest outbreaks and storms can be characterized as stochastic, nonlinear dynamic processes which induce a variety of temporal and spatial signatures including multiple steady-state cycles and non steady state dynamics. Given this evidence, we suggest that forest economists can utilize two general approaches to incorporate ecological models in the economic analysis of forest disturbances. First, ecosystem dynamics can be included in the specification of an economic welfare maximization problem. Notably, this bioeconomic approach to analysis has recently been applied to the economics of biological invasions (Sharov and Liebhold 1998, Leung et al. 2002, Olson and Roy 2003, Leung et al. 2005, Perrings 2005), and a call for the development of explicit bioeconometric analysis has been articulated (Smith 2006). Second, complex ecosystem dynamics can be summarized using statistical distributions. Taking advantage of the stochastic behavior of forest disturbance systems allows economists to investigate how statistical distributions shift in response to abiotic, biotic, and economic variables (Davis 1965, Prestemon et al. 2002, Mercer et al. 2007, chapters 3-5 of this book).

This chapter is necessarily incomplete and has not addressed some topics in ecology relevant to economic modeling of forest disturbances. These include the problem of aggregation across scales, the explicit spatial modeling of fires, insects, and pathogens in heterogeneous environments, and understanding the interactions among multiple forest disturbances. However, we hope that we have provided insight into the complexities associated with modeling forest disturbances and guidance into how ecological analysis can be incorporated into economic analysis. In sum, we think that economic analysis of forest disturbances will be improved by its congruence with ecological understanding and that, ultimately, joint economic-ecologic analysis will provide more relevant information for use in adaptive ecosystem management.

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SECTION II

FOREST DISTURBANCE PROCESSES

CHAPTER 3

NATURAL DISTURBANCE PRODUCTION FUNCTIONS

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1. INTRODUCTION

Natural disturbances in forests are driven by physical and biological processes. Large, landscape scale disturbances derive primarily from weather (droughts, winds, ice storms, and floods), geophysical activities (earthquakes, volcanic eruptions, even asteroid strikes), fires, insects, and diseases. Humans have always been affected by these processes and have invented ways to harness such processes or manipulate vegetation to enhance the values obtained from nature or reduce their negative impacts on human societies. For example, humans have cleared brush using fire to reduce pest[1] populations and encourage forage for animals (Pyne 1995). Historically, humans have relied on traditions, rules of thumb, and trial and error to predict how their actions may affect disturbance probabilities and characteristics. More recently, economic assessment tools have helped gauge the consequences of natural disturbances on forests.

As the availability of science, technology, and environmental data have improved, scientists and economists have been able to quantify disturbances as production processes that emanate from a combination of biological, physical, and (or) human-initiated inputs. Ecologists have long recognized that disturbances lead to changes in ecological communities, which subsequently affect human societies. Economists, on the other hand, have been focused on understanding how humans can intervene to alter both the frequency and severity of natural disturbances. Improving scientific and economic assessment tools, and experience using them, have in turn helped us to appreciate the many consequences of natural disturbances. The objectives of this chapter are to (1) define disturbances and their stages, (2) discuss how mathematical expressions of disturbance processes, disturbance production functions, may differ from the production functions defined in neoclassical economics, (3) identify the stages of disturbances, (4) provide a typology of production functions relevant to forest

[1] We define a "pest" in this chapter as a plant, animal (especially an insect), or disease that potentially causes damages to, or reduces output of, a valued good or service.

T. P. Holmes et al. (eds.), The Economics of Forest Disturbances: Wildfires, Storms, and Invasive Species, 35–58.

disturbances, and (5) conclude with a discussion of management and science implications of recent research. Our focus is to understand how disturbances are produced and how they may be affected by intentional managerial actions. We show that quantitative characterization of disturbance processes is required to understand how management interventions into disturbances can lead to net societal gains. Throughout the chapter, we provide examples of how information about disturbances can be used to better achieve management and policy goals.

2. DEFINITIONS OF DISTURBANCES

2.1 Natural Disturbances as Production Processes

A natural disturbance is a process that results in significant changes in ecosystem structure, leading to alterations in function and the goods and services that humans derive from nature. Disturbances, or their outcomes, may be affected by human-mediated inputs. Forest disturbances can be small or large—e.g., affecting a few plants in the forest or areas the size of a continent. Disturbances often have multidimensional implications for ecosystems and society. For example, fires can be described by the area that they burn, the quantity or value of the timber that is damaged, or the heat that they release. The disturbance process is also multi-staged. It proceeds from introduction to establishment, spread, and an endpoint. Although disturbances require non-human mediated natural inputs at every stage, human-mediated inputs can affect any or all stages. For example, a pest can be spread by people but requires suitable weather and hosts to survive and reproduce. As well, fire can be started by a match but driven by wind and fueled by native vegetation.

Human and natural inputs into disturbance processes may also be temporally defined and sequence–dependent. For example, above normal rain last year followed by a drought this year would produce a different wildfire output this year than if the sequence of rain and drought were reversed. Disturbances are stochastic—their outputs are in part randomly determined, even given the same level and temporal sequence of all inputs. In mathematical notation, $h = H(Y,Z)+\varepsilon$, where h is a disturbance output such as acres burned, H is a function describing how the variables Y and Z affect the disturbance output, and ε is a random shock added to $H(\bullet)$. Finally, disturbances may have short- and long-run consequences for ecosystems and the societies that depend on them. For example, a large wildfire in one location consumes fuels and vegetation today, altering how future fires in the same and neighboring locations may develop.

Economists have developed models that account for disturbances when making production and investment decisions. The models incorporate natural disturbances in two ways: (1) in a commodity production objective function and (2) in a management objective function. In the commodity production approach, disturbances have been viewed as either an exogenous (nuisance) or an endogenous process.

When viewing disturbances as nuisances, management decisions do not affect the probability that a disturbance will occur. Disturbances such as ice storms and hurricanes are suited to the nuisance approach in forestry, for example, because their occurrence is not affected by how land is managed. The nuisance approach in forestry was first described by Martell (1980) and Routledge (1980) and then by Reed (1984) and extended by several others (Yin and Newman 1996). In this approach, the likelihood of a nuisance disturbance leads to lower optimal stand densities and shorter optimal rotation lengths. In agricultural and resource economics, economists have long recognized that exposure to a production hazard of any sort lowers optimal investment levels (Just 1975, Pope and Kramer 1979), especially when decision makers are risk averse (Friedman and Savage 1948, Babcock and Shogren 1995).

More recent research has recognized that human actions can affect the probability, extent, duration, or severity of many disturbance events. For example, Shogren (1991) described an economic model that included disturbances as "endogenous risks" in the production process. Here, actions that individuals take can affect the probability of the disturbance and therefore individual utility. Shogren and Crocker (1991), recognizing work by Erlich and Becker (1972), describe the problem as a joint decision on how much effort to expend in self protection and reducing the probability of loss. In general, then, an aggregate objective function could be described that maximizes welfare by allocating spending across efforts that reduce the value lost if a disturbance occurs and the probability that a disturbance related loss occurs.

At its simplest, disturbance enters an endogenous risk objective function as a probability of occurrence, expressed as a function of a single action taken by a manager. An example is construction of a firebreak to reduce wildfire arrival rates. More complex are actions that can affect multiple features of the disturbance. In this case, human interventions affect not only the probability of occurrence but also qualitative features (e.g., severity) of the disturbance affecting the commodity in question.

When the time and space dimensions of disturbances are important considerations in production of desired goods and services from nature, then decisions on how to intervene in the disturbance process may increase in complexity. For example, actions taken today to reduce damages from a pest invasion in one forest may affect the future risks faced by other forests (Gumpertz et al. 2000). In wildfire management, reducing fuels levels in one location can affect fire arrival rates in other locations and may have effects that last several years. These spatio-temporal effects of management can sometimes limit the scope of action for managers: management decisions in location A are subject to the conditions in locations B, C, and D and to the decisions made in the past in location A.

Additionally, human attempts to reduce the probability of occurrence or damages resulting from a disturbance may affect the probabilities of other forest disturbances occurring in the same location (Meyers and van Lear 1998). For example, forest thinning to reduce fuels for wildfires might increase the probability of insect or disease infestation; and salvaging burned timber to reduce net economic damages can raise the probability of other disturbances such as exotic species invasions (McIver and Starr 2000, 2001).

Because disturbances themselves can affect many economic sectors (Butry et al. 2001, Kent et al. 2003), it is possible that actions in one sector have spillover consequences for other sectors. For example, forest thinning to reduce damages to timber from a potential forest fire may affect features important to recreators in that same forest. Somewhat more complicated still is when an intervention helps improve one value obtained from a forest but worsens other values. For example, prescribed burning can reduce wildfire severity and extent but can also worsen air quality.

In contrast to the endogenous risk approach, some economists have placed the disturbance production process at the center of economic decision making, especially when values produced are dispersed, public, or multi-sectoral, and one example is the "cost plus loss model" (Headly 1916, Sparhawk 1925). This model describes the wildfire suppression resource allocation decision as choosing the quantities of wildfire intervention inputs that minimize the sum of expected net damages from wildfire (the losses) and expenditures on the intervention inputs (the costs). Davis (1965) outlines a method of minimizing the sum of costs and expected losses from wildfires occurring over a wildfire season by manipulating the amounts of fuels and other inputs into wildfire management in the management unit. The cost plus loss framework is not the only one available for managing disturbances directly. For example, the optimal set of inputs to wildfire management can be chosen so as to maximize averted damages minus input costs (see chapter 18). Sharov and Liebhold (1998) describe how to optimally slow the spread of an exotic forest insect by identifying the best width and location of buffer zones. The disturbance-centered approach requires understanding not only of the disturbance production process but also how the disturbance creates losses. In other words, the nature of the loss function must be known. In the case of wildfire, the loss function's value must be identified for every possible or feasible combination of disturbance management inputs.

2.2 Disturbances as Damage Processes

Research has shown that managerial actions can influence the scale of losses from disturbances (Holmes 1991, Butry et al., Kent et al. 2003, Prestemon and Holmes 2004). One way to capture how a disturbance causes economic losses is to define a damage function, a mathematical expression that quantifies how variables influencing a disturbance result in damages to valued goods and services.

In agricultural economics, much research has focused on understanding how to optimally use pesticides to reduce the damages to agricultural commodities (Lichtenberg and Zilberman 1986, Carpentier and Weaver 1997, Kuosmanen et al. 2006). Mathematically, a damage function may be described as $G(X,Y,Z)$, where X are inputs intended to increase the output of the desired good (i.e., the purchased inputs into good production), Y are inputs intended to decrease the damages caused by the disturbance process (i.e., the purchased inputs into the damage process), and Z are the natural inputs into the damage and good production processes (i.e., the free inputs). In terms of forests or landscapes, the desired good Q could be the flow of goods and services provided by an "undamaged" forest or landscape in a given time period. In terms of a country, the desired good Q could be total economic welfare produced by the economy of the country in a given time period. Q is directly affected by X and Z but potentially also by Y, and it is reduced by the damage process, G: $Q=f[X,Y,Z,G(X,Y,Z)]$. For example, inputs applied to change the amount of fine fuels on a landscape (part of Y) can also lead to changes in the growing conditions faced by trees whose timber may be the desired output, Q. Free inputs, such as rain, contained in Z, can affect the productivity of fuels management efforts and the growth rate of trees.

If we define a disturbance process as the more general description of a phenomenon that can damage a commodity or reduce the quality or quantity of a value produced by, say, a forest, then the damage function is a transformation of the disturbance process: $G(X,Y,Z) = g[H(X,Y,Z)]$. The function g is a transformation of a disturbance function, $H(\bullet)$. In the case of wildfire, this transformation could be a summation of the amount of area burned by multiple wildfires in a specific region in a given fire season, divided by the total area of the region (Davis and Cooper 1963, Prestemon et al. 2002). H could also combine two kinds of disturbance functions, one describing the aggregate area affected by a pest in a given year in a specific landscape, and another defining the degree (severity) of damage by that pest within the area affected.

2.3 Disturbances as Probability Distributions

Disturbances can be defined in various forms, and each form has its own uses in for addressing questions in science and strategies for management. Disturbances can be discrete events or collections of events; they can be qualitative measures; or they can be ordered aggregations, or size-frequency distributions, of events produced in a landscape during a specified period of time. In other words, disturbance processes operate at multiple spatial and temporal scales, and recognition of such scaling issues can inform how to intervene in the process to achieve a desired outcome. For example, wildfires ignite at specific points in a landscape, and their timing and locations in that landscape can be measured as counts of events and related statistically to hypothesized driving factors.

Disturbances also often produce multiple outputs, creating scientific and statistical challenges for capturing the effects of inputs on each of their outputs. For example, wildfire output can be measured as area burned, the number of structures lost or threatened, or the average intensity of fire over time. Inputs such as suppression resources and fuels management can simultaneously affect many outputs—in this case, all of the three listed measures.

The stochastic nature of disturbance processes has implications for predicting and managing disturbances across landscapes and over time. Davis (1965) recognized that disturbance management meant managing the landscape to shift the probability distribution of future disturbance outcomes. For example, constructing firebreaks across a large management unit may reduce the expected total area of fire observed during the fire season in the management unit by altering fire spread and affecting suppression input productivities. Although other factors besides firebreaks would also affect the expected total area of wildfire, building more firebreaks in the landscape could shift the probability distribution of the total area of wildfire observed in a fire season. Figure 3.1 illustrates how alternative probability distributions (disturbance probability density functions) may be affected by a change in an input. Part A of figure 3.1 shows how a Normally distributed measure could exhibit a reduction in variance without a change in the mean, or a reduction in the mean without a change in variance, in response to a change in an input to the disturbance process. Of course, probability distributions do not have to be statistically Normal. Parts B, C, and D of figure 3.1 show the effects of input changes on the positions and shapes of Poisson- Exponential- and Gamma-distributed measures.

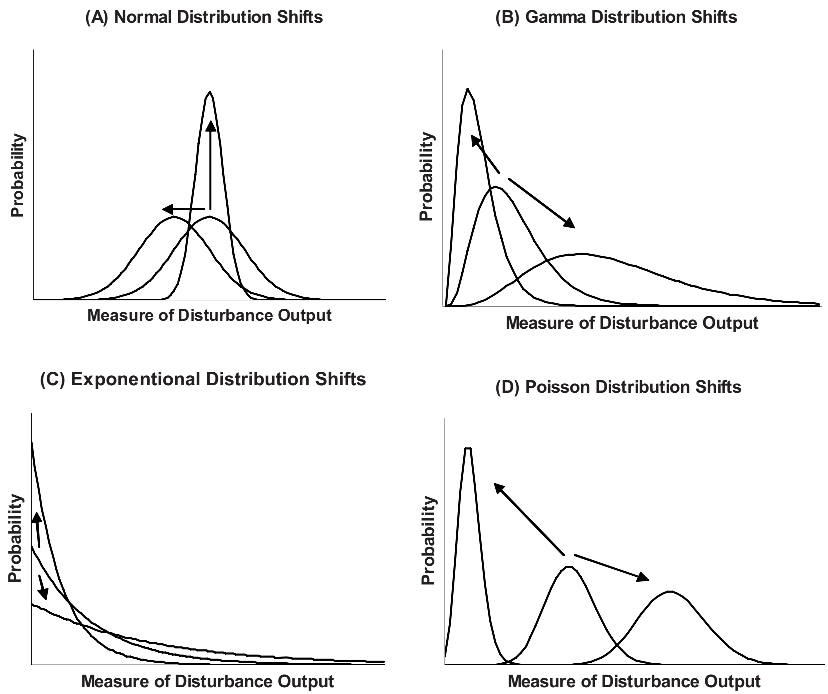

Figure 3.1. Hypothetical probability distribution shifts under alternative distributional assumptions for a measure of a disturbance process, as affected by an input that is intended to affect the disturbance process.

2.4 Disturbances as Production Processes in Economics

Disturbances are far more complicated than those implied by the classical production function (Chambers 1991). Below, we describe how disturbance production functions may differ from classical production functions, as defined in economics. Appendix table 4.1 provides a concise listing of these differences.

Although a principal characteristic of a classical production function is that output increases with increased amounts of a purchased input (monotonicity), this may not apply to disturbances. Inputs to disturbance functions may be intended to decrease some negative aspect of the disturbance. Free inputs, such as those associated with rain or human activities not intended to affect the disturbance, may have any direction of effect. Lastly, the effect of additional inputs may not be describable as an "increase," such as when an ecosystem changes from state A to state B.

A second characteristic of classical production functions is that each additional unit of purchased input should produce no more than the previous unit of additional input (quasi-concavity). In disturbances, concavity may not be relevant, as in the case of discrete or qualitative output measures. Alternatively, it may be true only in the negative sense, such as where each additional unit of input yields an equal amount or smaller reduction in output than the previous input; in essence, disturbance production functions may be quasi-convex.

A third characteristic of classical production is that if any or all purchased input quantities are zero, then output is zero (essentiality or weak essentiality). Many disturbance outputs occur without active intervention by humans. That is, they can operate with free inputs provided by nature or society. Thus withholding purchased inputs does not set outputs to zero.

A fourth characteristic of classical production is that the set of possible outputs is closed for all levels of output. In other words, it is feasible to produce any desired level of output. In the case of disturbances, if the process is defined as a collection of discrete events, then production is discontinuous and therefore not a closed set. This is especially true when disturbance production can be measured qualitatively.

A fifth characteristic of classical production is its nonstochasticity—a specific quantity of input always yields the same quantity of output. With natural disturbances, randomness can yield a different quantity of output for the same quantity of input.

Lastly, classical production functions are continuous and twice-differentiable (Chambers 1991, p. 9). In other words, to identify optimal input amounts, it is necessary for production functions to be increasing at a decreasing rate across some region of economical output. Because disturbance production can be discrete, qualitative, or discontinuous, it is clear that disturbance functions can sometimes not be continuous or twice-differentiable. As we shall see later, however, there are ways to identify optimal inputs into disturbance production functions that yield desired outputs, even while the disturbance process itself may not conform to all the classical assumptions of production. Nonetheless, a primary implication of the discontinuities, discreteness, and other features of disturbances is that it may not be economically optimal to intervene. In other words, the best choice may be to set purchased input levels to zero.

3. STAGES OF DISTURBANCE PRODUCTION FUNCTIONS

Accurately modeling disturbances and their damages requires understanding how physical, biological, and human mediated inputs affect key processes. Typical forest disturbances proceed in four stages (Williamson 1996): introduction, establishment, spread, and post-disturbance. Between spread and post-disturbance is a point called extinction or outbreak cessation. Humans can intervene productively in some or all stages. Figure 3.2 traces out these stages and indicates where interventions may be possible. In the case of insects, diseases, and wildfires, the first stage is the introduction or the ignition. The second stage, establishment, occurs when introduction is successful—that is, the disturbance takes hold or survives. In the case of pests, establishment means that the pest invader carries out a life cycle and reproduces. In the third stage, spread, the disturbance spreads spatially

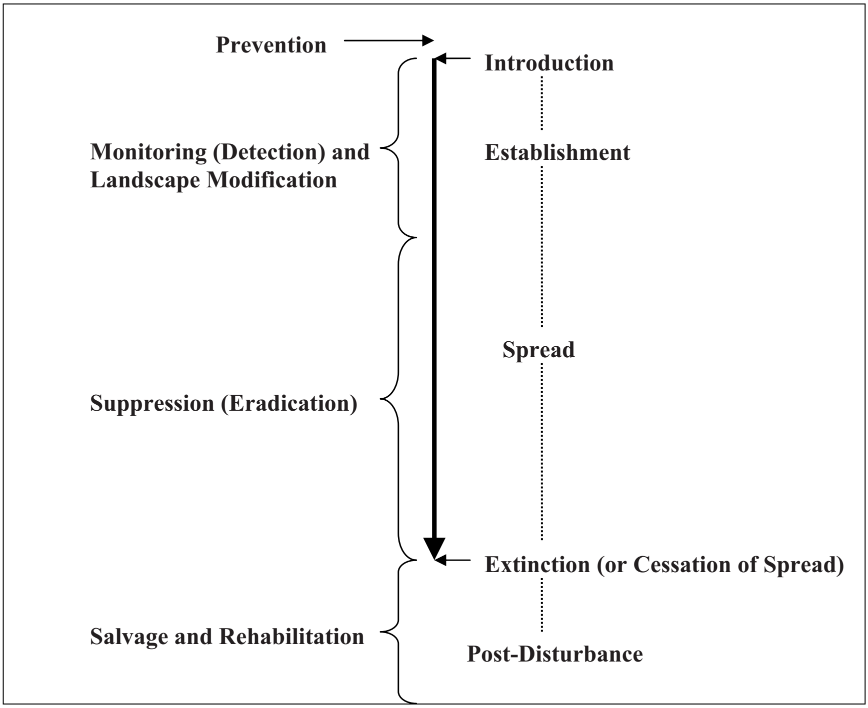

Figure 3.2. Stages of disturbances and intervention points.

until extinction or, in the case of some pests, returns to innocuous or endemic population levels (i.e., outbreak cessation occurs). Finally, "post-disturbance" follows extinction or outbreak cessation, which lasts indefinitely and may be characterized by ecosystem changes from the disturbance.

We define occurrence as the appearance of a new instance of a disturbance, possibly deriving from a distant or exogenous source but not as a result of a spatially connected spread process. Introduction and establishment can therefore be combined into one stage called "occurrence." The distinction between stages is often indefinite or fuzzy. For example, spreading to a neighboring point is the same as occurrence at that neighboring point.

3.1 The Introduction Stage

Introduction is the placement, through some process, of the disturbance into the landscape. An introduction could be an ignition of a wildfire by escape from a campfire ring or the appearance of an exotic pest in a new landscape by release from a shipping container. Introductions can be prevented by many kinds of actions. For wildfire, these can be banning of campfires or open debris fires, which are typical sources of accidental wildfire ignitions. In the case of pests, humans introduce exotic plants and animals intentionally and unintentionally through international trade or through (unintentional) long distance transport (di Castri 1989, Mack et al. 2000). Sometimes, these exotics become invasive pests. Prevention measures for exotic pest introductions, then, could include the banning of trade in certain, potentially infested commodities or shipping containers, or it could mean inspection of recreational boats for pests before they are moved between lakes. In wildfire, law enforcement efforts have been linked to reduced wildland arson ignitions (Prestemon and Butry 2005) (see chapter 7 for additional details and support). Prevention is not currently possible, of course, for many kinds of natural disturbances affecting forests—e.g., volcanic eruptions, hurricanes, and ice storms.

3.2 The Establishment Stage

Establishment of a natural disturbance means that the disturbance has moved past mere introduction. In terms of insects and diseases, establishment could be defined as the successful reproduction *in situ*. A wildfire is "established" when an ignition is sustained long enough so that further spread is possible. (This stage may only be brief and defined only *ex post*, if spread actually occurs.) For many disturbances, establishment depends on the collocation of sufficient quantities and qualities of host materials (or fuel) and favorable weather or other site conditions. Because establishment requires favorable conditions for propagation or survival, managers can alter the probabilities of successful establishment by modifying the landscape. A pest whose potential host is not present cannot become established, even if introduced. Research shows that non-establishment is the most frequent outcome following introduction. Pest managers have estimated that introductions average five to twenty times the rate of establishment (Williamson 1996). Successful establishment may be defined as an "event" in empirical analyses and then related to measures designed to limit introductions or establishment.

3.3 The Spread Stage

Widespread ecologically and economically significant changes are produced during the spread stage. Some disturbances, such as ice storms and hurricanes, are exogenous and rapid, so that features of a forest, for example, may not significantly affect its overall extent. In these cases, actions taken to reduce losses of valued goods or services are applied either *ex ante*, by removing or reducing values at risk in anticipation of a potential disturbance, or *ex post*, in the post-disturbance stage. Note that *ex post* interventions are possible for all disturbance processes, not just fast ones. For slower spread processes, such as those of insects, diseases, and fires, limiting spread is often possible.

Variables affecting the rate and ultimate extent of spread of slower disturbance processes such as fire and pests also often affect establishment: the quantity of available host material in a landscape, weather, climate, geographical features, and the amounts and timing of efforts to suppress the disturbance. Manipulation of potential host material and placement of suppression inputs are *ex ante* actions that can be taken to reduce the spread of a disturbance. During active spread, suppression primarily involves manipulating (wetting, burning) or removing host material.

Once a disturbance is established and detected, the final extent of disturbance spread may depend on the speed of application of suppression resources (Butry 2006). For example, in wildland fire management in the United States, the so-called "10 a.m. policy" focuses on extinguishing fires as quickly as possible following detection of an ignition. This kind of suppression guideline is based on the notion that fire area can increase exponentially (Donovan and Rideout 2003a), and this exponential rate of spread is often higher later in the day, after temperatures rise and humidity falls. Fire managers often credit the policy with the successful suppression within 24 hours of 98 percent of all wildfires on federal lands. For insects, efforts to control or slow the spread (Sharov et al. 1998) involve taking quick action to suppress establishments occurring beyond the advancing front of a spreading pest. Managers therefore exploit the Allee effect (Leung et al. 2004), which involves keeping insect populations low on the spreading front, which reduces the reproduction rate of the invasive insect.

The economics of spread management (or suppression in wildfire terminology) is the subject of extensive theoretical development and modeling. Elaborate strategies and infrastructures have been developed to manage the spread of insects and diseases (Sharov and Liebhold 1998b, Mack et al. 2000) and wildfire (Sparhawk 1925, Donovan and Rideout 2003b).

3.4 The Post-Disturbance Stage

The post-disturbance stage is defined spatially as the area of influence of the disturbance, which can extend beyond the boundaries of the actual area directly affected. Although the length of time of the post-disturbance stage is indefinite, the timing of human actions may be important in determining the short- and long-run implications of the disturbance. In post-disturbance, landowners and managers often quickly assess the effects of the disturbance, sometimes salvage part of the affected timber or other valued products, take actions that reduce long-run negative side effects of the disturbance, and often work to restore some of the features of the ecosystem present before the disturbance. Human actions taken following the disturbance are often termed "rehabilitation and recovery." Rapid assessment of the effects of a disturbance is important for planning further actions. One action, timber salvage, has been shown to yield significant economic returns and be time sensitive (Prestemon et al. 2006). Removal of some of the killed timber and erosion control following a disturbance may alter risks of additional damage (McIver and Starr 2000, 2001, Kent et al. 2003). Although the specification of a meta-model that describes these types of feedbacks is beyond the scope of this chapter, a disturbance production function for one type of disturbance might include a set of variables that derive from other disturbance types. This approach would allow for joint modeling of production functions for a variety of disturbances (Hyde et al. 2006).

4. TYPES OF DISTURBANCE FUNCTIONS AND FUNCTIONAL FORMS

Disturbance functions can be classified into at least the following five broad classes: (1) event, (2) individual extent, (3) aggregate extent, (4) effect, and (5) joint (combinations of the other classes). Each class describes the stages of the disturbance across varying spatial and temporal scales or aggregates, and each may be useful in economic analysis. The five classes of disturbance production functions are briefly discussed below.[2] Also offered are examples or guidance on the statistical methods that could be used to identify the relative economic importance and direction of influence of free and purchased inputs to the disturbance processes defined in each class of model. We also suggest how simulation methods can be used to identify these influences, especially in cases where information about disturbance inputs are not available or are available at a different spatial or temporal scale than the output variable of interest.

[2] Mercer and Prestemon (2005) discuss a similar typology for wildfire production and provide empirical examples.

4.1 Event Models

Disturbance events can be modeled in at least three ways: (a) discrete event models that explain whether the disturbance occurred or the number of occurrences of the disturbance; (b) point process models, which describe the spatial and temporal distribution of occurrences; and (c) continuous models, which describe the rate of arrival or elapsed time between occurrences. An example of a discrete event approach is a binary choice (logit, probit) model that predicts the occurrence of a disturbance with particular characteristics. For example, in a wildfire event model, each point on a landscape each day might have a certain ignition probability, hypothesized to be a function of weather variables, vegetation features, and terrain. A logit or probit model could be used to estimate the probability that a fire would occur, given the measured levels of these causal variables. Data required to estimate the model would include occurrence data in many locations across a landscape as the dependent variable, coded to indicate whether a fire occurs at a given location during a specified time period, along with measures of the hypothesized causal variables for each location. Scales of analysis should be fine grained enough that more than one event does not occur in the same time and place. An example of this kind of modeling is found in Pye et al. (2003).

Count data models are extensions of the binary choice event models. In count models, the measure of observation is a count of the occurrences within a given time period and spatial unit. For example the unit of observation in a count model might be the number of fire starts in a county in a year, rather than the probability of a single ignition at a specific time and location. Poisson-type models are a common choice for relating the count to hypothesized causal variables (Martell et al. 1987, Gill et al. 1987, Vega Garcia et al. 1995, Prestemon and Butry 2005, Lee et al. 2006).

Point process models (Ripley 1976) are used to describe the spatial or temporal dispersion of events observed across a landscape within a given time period—for example, whether or not the pattern is random or non-random. The degree of randomness could inform the analyst about the effectiveness of spatially targeted interventions. An example is an analysis by Genton et al. (2006), who evaluate the clustering of wildfire ignitions in Florida.

Duration or survival and hazard type models relate hypothesized explanatory variables to the amount of time elapsed until an event occurs (Cox and Oakes 1984, Collett 1994). Duration modeling could use time series data on individual fire starts to relate the amount of time between fire starts to a variety of weather, ecosystem, management, and socio-economic variables. Survival models are common in analyses of treatment efficacy to reduce mortality from pest attacks (Woodall et al. 2005) and could also be used to evaluate time to events or occurrence probabilities of disturbances.

4.2 Individual Extent and Spread Models

Individual extent models relate explanatory variables to the amount of a resource or commodity affected by a single event. Many of the variables influencing establishment also help explain the extent of a particular disturbance, although an additional set of variables to include would be those associated with suppression or cessation of spread. Individual extent models that include suppression strategies can aid in tactical decision making aimed at slowing or stopping the spread of the disturbance.

Spread models focus on the spatial and temporal dynamics of an individual disturbance process after establishment but before cessation. Spread models may or may not include variables related to suppression efforts. Spread models may describe the arrival rate and direction of spread, and they are often used to compare the effects of alternative suppression tactics. Wildfire spread models have been embedded in fire simulation tools used by wildfire managers (Andrews and Bevins 1999). Tools such as FARSITE (Finney 1998, Finney and Andrews 1999) allow simulation of the effects that simple suppression strategies have on fire spread. Repeated runs of wildfire spread simulation models can show how a particular strategy affects the probability distribution of burned areas under operational or experimental conditions. Pest management makes similar use of simulated spread processes to compare the effects of alternative control strategies. Such experimentation can help managers and policy makers understand the trade-offs and economic returns of alternative suppression strategies (Sharov and Liebhold 1998a,b,c, Sharov et al. 1998).

Sharov and Liebhold (1998a,b,c) illustrate how spread models can answer important economic and management questions about barrier zone suppression strategies. The European gypsy moth (and many other pests) spreads in a stratified dispersal process (Liebhold 1998c), where spot outbreaks appear randomly or chaotically some distance beyond the zone of infestation. Spots continue to grow until they coalesce with other spots, merge with the infested zone, or are eradicated. Control actions consist of using aerial surveillance or pheromone traps to monitor the transition zone, an area of land surrounding the completely infested zone that encompasses the range of potential spread. Spot eradication measures are applied when a colony spreads into the transition zone.

The spread process described in these studies of the gypsy moth can be defined mathematically as a traveling wave equation for every cell (spatial unit) in the actual or potentially invaded range. Once a cell's population reaches a carrying capacity, the cell is considered a part of the colony in the infested zone. The population of any particular cell is determined by the probability of a new spot invading the cell and the population in the colony. Invasion probability for any cell is a negative function of distance from the infested zone. The colony's population is a positive function of the colony's age. The spread rate slows as the number of spots in the transition zone is reduced. However, spread can continue in a wave even without any successful spotting. In this case, slowing the spread rate requires the eradication of all individuals in the transition zone. Because spots spread at a rate that increases with spot age, more intense monitoring of transition zones and quicker response times once a spot is identified typically produce greater control benefits. As such, monitoring and eradication are production substitutes under most conditions.

Calibrating a model of pest spread as a function of monitoring effort and eradication efforts requires data on spread rates with and without eradication efforts and how time since initiation of eradication affects its success. In empirical analysis, the success of barrier zone management can be quantified and potentially compared to a "no-action" alternative by simulating how the average spot size changes in response to differing levels of pest monitoring or lags before initiation of eradication.

4.3 Aggregate Extent Models

Aggregate extent models relate the amount of a resource or commodity affected by disturbance events occurring over a defined area and time. Statistical models of aggregate extent often rely heavily on long run and spatially aggregated measures of weather, climate, host materials, and suppression. An example is a model of the likelihood of beetle outbreak in a county, as related to the amount of host forest in the county, seasonal average precipitation and temperature levels in the county, the amount of National Forest lands in the county, and measures of spatial autocorrelation (Gumpertz et al. 2000).

The increased spatial and temporal aggregation of these models allows analysis of large and long scale disturbance patterns and dynamics. Because natural disturbances are stochastic in both location and timing, this broader scale analysis can help reveal the overall effects of management and suppression strategies across wider scales. Such broad analyses may also more effectively capture the underlying effects of free inputs to disturbance processes, especially when these other inputs may vary little within a small location or a short time period but more widely when viewed across broad landscapes and long time horizons. For example, the area burned in a county in a year could be expressed as a function of areas burned in that county in previous years, aggregate amounts of fuel treatments in the county in the current and previous years, county level annual measures of socioeconomic variables, and broad scale weather patterns such as a measure of ocean temperature oscillations. Barnett and Brenner (1992), Keeley et al. (1999), Prestemon et al. (2002), Westerling et al. (2002), Norman and Taylor (2003), and others have developed empirical aggregate extent models of wildfire in different parts of the United States.

Statistical methods are not always available for quantifying the impacts of disturbances at broad spatial and temporal scales. In these cases, it still may be possible to quantify their impacts by using simulation approaches. For example, the aggregate amount of wildfire in a landscape in a given fire season could be simulated using statistical models of individual fire occurrence (event models) and spread, simulated weather, and imputation of known vegetation and landscape features. If the fire occurrence and spread models are specified as functions of fuels, weather, and suppression variables, then repeated simulations can reveal the effects of altering assumed levels of each of these, producing a picture of the broad spatial and temporal effectiveness of fuels management and fire suppression efforts.

An example from wildfire illustrates how the wildfire disturbance process exhibited at broad spatial and temporal scales can be used to identify the effects of free and purchased inputs into wildfire management. Prestemon et al. (2002) develop a model relating wildfire probability in a county in a year as a function of both non-purchased inputs (climate measures and historical wildfire) and purchased inputs (prescribed fire and small diameter timber removals). Using a cross-sectional time series empirical model, the area of wildfire (W_{it}) relative to the area of county i's forest (f_{it}) in year t, (W_{it}/f_{it}) = π_{it}, is specified as a function of prescribed fire area (x_i) relative to forest area, (x_t/f_i)=y_{it}, in that same year and one previous year (y_{it}, $y_{it\text{-}1}$) =$\mathbf{y}_i$, small diameter timber removals in that county in the three previous years ($h_{it\text{-}1}$, $h_{it\text{-}2}$, $h_{it\text{-}3}$)=$\mathbf{h}_{it}$, historical proportions burned by wildfire in that county for the previous twelve years ($\pi_{it\text{-}1}$, $\pi_{it\text{-}2}$,…, $\pi_{it\text{-}12}$)= π_{it}, the El Niño-Southern Oscillation Niño-3 sea surface temperature anomaly in degrees centigrade (E_t), a dummy measuring a Super El Niño cycle (D_t) in 1998, and the county's housing density (U_{it}). The proportion of forest area burned is assumed stochastic, such that

$$\ln(\pi_{it}) = \alpha_i + \boldsymbol{\beta}' \ln(\boldsymbol{\pi}_{it}) + \boldsymbol{\gamma}' \ln(\mathbf{y}_{it}) + \boldsymbol{\delta}' \ln(\mathbf{h}_{it}) + \mu_1 E_t + \mu_2 D_t + \mu_3 \ln(U_{it}) + \varepsilon_{it} \quad 3.1$$

Equation (2) is estimated with weighted least squares and a heteroscedasticity correction, using a short panel (1994-1999) and 37 cross-sections. Mercer and Prestemon (2005) and Mercer et al. (2007) estimate similar models with longer and wider panels of data. Prestemon et al. (2002) found that prescribed fire can have an effect on wildfire activity, but that its effect is not large relative to long run climatic patterns and historical wildfire activity.

4.4 Effects Models

Effects models describe how independent variables influence the characteristics of a particular event. For example, the species diversity of a forest might be altered as a result of successful invasion of an exotic species. The effect could be measured in terms of changed species diversity levels observed following an invasion. Another example is timber quality changes following a storm. Because damages to timber quality might take years to manifest following a storm, an effects model would relate the presence or absence of storm damage in each forest stand some number of years following the storm to features of the storm in that location, site conditions, and vegetation conditions before the storm.

For a wildfire example, the proportion of fire-killed timber per unit area or the soil temperatures observed during a wildfire in each location might be related to wind, humidity, temperature, and the amounts of fuels of different sizes in each location. If forest fuels can be manipulated by a land manager and are known to affect the intensity of wildfires that burn in the forest, then a statistical model relating the degree of wildfire-related losses of goods or services provided per unit area of wildfire area burned would describe how purchased inputs into fuels management would directly affect these losses.

4.5 Combined Models

Any version of at least two of the above models can be combined to yield another class of disturbance model. For example, size-frequency distribution models, which quantify the parameters of a statistical distribution of wildfire across size classes, summarize disturbance activity across broad landscapes and long time scales. Research has shown that size-frequencies of many natural phenomena including disturbances are distributed in log-linear fashion (Strauss et al. 1989, Li et al. 1999, Holmes et al. 2004). Extreme value functions (Moritz 1997) are models increasingly used in insurance applications, can describe how the number of events of different ordered classes are distributed in probability (see chapter 4). As with aggregate extent models, size-frequency distribution and extreme value models could be used to identify the effects of long-run or large-scale changes in free and purchased inputs. For example, estimates of the parameters of size-frequency distributions of wildfires occurring in simulated or otherwise identical landscapes with and without fuels management could reveal the effect of efforts to reduce negative outcomes of wildfires in the landscape.

In another wildfire example, a measure of overall damages by wildfire in a season across a landscape can be constructed by combining both the intensity and the aggregate extent of wildfires in a landscape over a fire season. This measure of damages can then be related to variables hypothesized to influence the effect and the aggregate extent of damages. For example, Mercer et al. (2007) relate an aggregate of the product of wildfire intensity (an effect) and area burned by all the fires occurring in one year in one county (aggregate extent) to several hypothesized explanatory variables, including prescribed fire and relate historical data on intensity-weighted area burned to the economic damages associated with wildfire in the State of Florida. In their economics application, the benefits of wildfire economic damages averted by intense wildfires trade-off with the costs of to identify economically preferable fuels management rates. A variation on the Mercer et al. (2007) and Holmes et al. (2004) approaches would be to identify a family of wildfire size-frequency distributions, a distribution for each fire intensity level. Similarly, one might use combination models to analyze whether spot sizes of southern pine beetle infestations possess the kinds of spatial dynamics identified by Gumpertz et al. (2000).

Another kind of combined model is of spatio-temporal point processes (STPP). These models describe how a collection of events is distributed across space and time. The empirical manifestation of a STPP is a spatio-temporal point pattern.

A primary focus of STPP analyses is to evaluate whether the pattern observed differs significantly from a random distribution of events across space and time. Examples of such patterns might be the occurrences of disease outbreaks, wildfire ignitions, and pest infestations. STPP's could be of use to wildland managers if analysts were able to link the patterns to variables that managers can affect, or if optimal planning for a disturbance depends on the amount of clustering of events. For example, wildfire managers might want to understand the STPP's to understand wildfire suppression resource needs. Examples of published research include Podur et al. (2003), who use STPP's to analyze lightning fires in Canada, and Genton et al. (2006), who apply STPP's to analyze wildfires produced by all major ignition categories in the United States.

5. IMPLICATIONS FOR MANAGEMENT, POLICY, AND SCIENCE

This chapter has sought to explain what disturbance production processes are, describe how they differ from classical economic production processes, characterize the various forms of disturbance processes, and briefly describe how analysts have modeled them. The availability of large and long term datasets on natural disturbances and improvements in software and computing power have led to advances in science and management. These advances include a better understanding of the long-run, broad scale effects of human interventions and free inputs into disturbance processes (e.g., societal variables not intended to affect the process but nevertheless do affect it, climate, weather), quantification of the long-run economic net benefits and effects of various kinds of interventions into these processes, and revelations about previously unidentified spatial and temporal patterns in disturbances. We anticipate that application of the kinds of modeling approaches outlined here could lead to advances in questions of current and future importance to society, including those associated with large scale spending on fuels management to reduce the net economic damages from wildfire.

An avenue for further study involves examining how agents of disturbances respond to actions to limit the agents' effectiveness. Research into agent-based disturbance modeling would focus on how humans and pests respond to interventions to mitigate the effectiveness of the interventions. For example, little is known about how arsonists might change their behavior in response to stepped up law enforcement (Prestemon and Butry 2005). Research should focus on how greater enforcement in one area could lead to simple shifts of arson activities in space and time. Similarly, controls on the importation of invasive species could create averting actions by importers to get around rules and regulations. In terms of invasive species spatial processes, barrier zone management might induce changes in the aggregate spread behavior of populations. Alternatively, pesticide use may, in the long-run, lead to increased pesticide resistance in the population, requiring more complex models of pest spread and control (Carpentier and Weaver 1997).

A better understanding of these kinds of feedbacks may reveal important limitations and open up new approaches to forest and landscape management with disturbances.

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Appendix Table 3-1. Comparisons between classical economic production functions and disturbance production functions.

Characteristics of Classical, Single-Output Production Functions (Chambers 1991, p. 9)	Disturbance Production Function	Notes
1a. Monotonicity: if $x' > x$ then $f(x') > f(x)$	Negative monotonicity for purchased inputs (x) of single output disturbance functions: if $x' > x$ then $f(x', \mathbf{z}) \leq f(x,z)$; non-monotonic for free inputs ($\mathbf{z}$)	(Negative) Monotonicity not required for individual outputs respect to purchased individual outputs of multioutput functions
1b. Strict monotonicity	Negative strict monotonicity for purchased inputs of single output disturbance functions: if $x' > x$ then $f(x', \mathbf{z}) < f(x, \mathbf{z})$	Non-monotonic when purchased inputs are zero: $f(0,z) \geq 0$
2a. Quasi-concavity	Quasi-concavity possible in purchased inputs; no assumption for free inputs	Quasi-concavity assumption undermined by possible sequencing in temporally defined input sets, interaction of purchased and free inputs in production
2b. Concavity	Concavity is possible in purchased inputs; no assumption for free inputs	Concavity assumption undermined by possible sequencing in temporally defined input sets, interaction of purchased and free inputs in production
3a. Weak essentiality: $f(0)=0$	No essentiality for purchased inputs: $f(0,\mathbf{z}) \geq 0$	Purchased inputs are not required for nonzero output, due to free inputs

(continued)

Appendix Table 3-1. Comparisons between classical economic production functions and disturbance production functions. ***(continued)***

Characteristics of Classical, Single-Output Production Functions (Chambers 1991, p. 9)	Disturbance Production Function	Notes
3b. Strict essentiality: $f(x_1,x_2,\ldots x_{i-1},0,x_{i+1},\ldots,x_n)=0$	No strict essentiality for purchased inputs: $f(x_1,x_2,\ldots x_{i-1},0,x_{i+1},\ldots,x_n,z) \geq 0$	No single purchased inputs is required for nonzero output, due to free inputs
4. V(y) is a nonempty and closed set for all y>0	V(y) is nonempty, but discontinuous production is possible, implying a non-closed set	Discontinuity is possible when disturbance is a discrete process (e.g., an event occurrence or count process) or is categorical or qualitative
5. f(x) = y is finite, nonnegative, real valued, and single-valued for a finite x	Because of stochasticity, f(x) may not be single-valued	
6a. f(x) is continuous	Not assumed	
6b. f(x) is twice-differentiable	Not assumed	
Other Characteristics		
No free inputs	Free inputs exist (z)	
Non-stochastic	Stochastic	
Single-output	Sometimes multioutput	

(continued)

Appendix Table 3-1. Comparisons between classical economic production functions and disturbance production functions. ***(continued)***

Characteristics of Classical, Single-Output Production Functions (Chambers 1991, p. 9)	Disturbance Production Function	Notes
Previous inputs are nonessential: for any constant, a, $f_t(x_t,x_{t-k}) = f_t(x_t,ax_{t-k})$ for all x_{t-k} and all $k \neq 0$	Possibly temporally defined input set: for any constant, $f(x_{1,t},\ldots, x_{i-1,t}, x_{i,t},x_{i+1,t},\ldots,x_{n,t},x_{1,t-1},\ldots, x_{i-1,t-1}, x_{i,t-1},x_{i+1,t-1},\ldots x_{n,t-1},\ldots,\mathbf{z}) \leq \geq f(x_{1,t},\ldots,x_{i-1,t}, x_{i,t},x_{i+1,t},\ldots, x_{n,t},x_{1,t-1},\ldots,x_{i-1,t-1},ax_{i,t-1}, x_{i+1,t-1},\ldots x_{n,t-1},\ldots,\mathbf{z})$	Temporally defined input set for any purchased or free input
Not sequence-dependent with respect to inputs (guaranteed by an assumption of static with respect to inputs)	Possibly sequence-dependent with respect to inputs: for any nonzero and non-unitary constants a and c, $f(x_{1,t}=c,x_{2,t}, \ldots x_{n,t},x_{1,t-1}=ac,x_{2,t-1},\ldots x_{n,t-1}, \ldots,\mathbf{z}) \leq \geq f(x_{1,t}=ac,x_{2,t},\ldots x_{n,t}, x_{1,t-1}=c,x_{2,t-1},\ldots x_{n,t-1},\ldots,\mathbf{z})$	Potentially sequence-dependent with respect to inputs for any purchased or free input
Previous outputs are nonessential: for any constant, a: $f_t(x_t,y_{t-k}) = f_t(x_t,ay_{t-k})$ for all $k \neq 0$	Potentially dynamic production: for any constant, a, $f_t(x_t,y_{t-k},\mathbf{z}) \leq \geq f_t(x_t,ay_{t-k},\mathbf{z})$ for any $k \neq 0$	

CHAPTER 4

STATISTICAL ANALYSIS OF LARGE WILDFIRES

Thomas P. Holmes, Robert J. Huggett, Jr.,
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1. INTRODUCTION

Large, infrequent wildfires cause dramatic ecological and economic impacts. Consequently, they deserve special attention and analysis. The economic significance of large fires is indicated by the fact that approximately 94 percent of fire suppression costs on U.S. Forest Service land during the period 1980-2002 resulted from a mere 1.4 percent of the fires (Strategic Issues Panel on Fire Suppression Costs 2004). Further, the synchrony of large wildfires across broad geographic regions has contributed to a budgetary situation in which the cost of fighting wildfires has exceeded the Congressional funds appropriated for suppressing them (based on a ten-year moving average) during most years since 1990. In turn, this shortfall has precipitated a disruption of management and research activities within federal land management agencies, leading to a call for improved methods for estimating fire suppression costs (GAO 2004).

Understanding the linkages between unusual natural events, their causes and economic consequences is of fundamental importance in designing strategies for risk management. Standard statistical methods such as least squares regression are generally inadequate for analyzing rare events because they focus attention on mean values or typical events. Because extreme events can lead to sudden and massive restructuring of natural ecosystems and the value of economic assets, the ability to directly analyze the probability of catastrophic change, as well as factors that influence such change, would provide a valuable tool for risk managers.

The ability to estimate the probability of experiencing a catastrophic event becomes more advantageous when the distribution of extreme events has a heavy-tail, that is, when unusual events occur more often than generally anticipated. Heavy-tail distributions have been used to characterize various types of catastrophic, abiotic natural phenomena such as Himalayan avalanches (Noever 1993), landslides, and earthquakes (Malamud and Turcotte 1999). Several studies also indicate that wildfire regimes have heavy-tails (discussed in section 2 below). For decades, economists have been interested in heavy-tails appearing in the distribution of income (Mandelbrot 1960), city sizes (Gabaix 1999, Krugman 1996), commodity prices series (Mandelbrot 1963a, Mandelbrot

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1963b), financial data (Fama 1963, Gabaix et al. 2003), and insurance losses (Embrechts et al. 2003).

Despite the fact that heavy-tail distributions have been used to characterize a variety of natural and economic phenomena, their application has been limited due to the fact that heavy-tail distributions are characterized by infinite moments (importantly, mean and variance). Reiss and Thomas (2001) define a distribution function *F(x)* as having a heavy-tail if the j^{th} moment $\int_0^\infty x^j dF(x)$ is equal to infinity for some positive integer j (p. 30). Note that a moment is infinite if the integral defining the statistical moment is divergent (it converges too slowly to be integrated)—therefore, the moment does not exist.

Recognizing that standard statistical tools such as the Normal distribution and ordinary least squares regression are not reliable when moments are infinite, Mandelbrot (1960, 1963a, 1963b) suggested that the Pareto distribution be used to analyze heavy-tail phenomena. The Pareto distribution is extremely useful because, in addition to the capacity to model infinite moments, it has an invariant statistical property known as stability: the weighted sum of Pareto-distributed variables yields a Pareto distribution (adjusted for location and scale). Other commonly used long-tail distributions, such as the log-normal, do not share this stability property. More recently, Mandelbrot (1997) refers to distributions with infinite variance as exemplifying a state of randomness he calls "wild randomness".

Over the past few decades, special statistical methods, known as extreme value models, have been developed for analyzing the probability of catastrophic events. Extreme value models utilize stable distributions, including the heavy-tailed Pareto, and have been applied to problems in ecology (Gaines and Denny 1993, Katz et al. 2005), finance, and insurance (Reiss and Thomas 2001, Embrechts et al. 2003). The goals of this chapter are to: (1) show how extreme value methods can be used to link the area burned in large wildfires with a set of explanatory variables, and (2) demonstrate how parameters estimated in the linkage function can be used to evaluate economic impacts of management interventions. In doing so, we provide a brief, somewhat technical overview of the statistical analysis of extreme events and discuss previous applications of these models to wildfire analysis (section 2). A major contribution of this chapter is the discussion of how extreme value models can be parameterized to include covariates such as climate or management inputs as explanatory variables (section 3). To clarify the presentation, the statistical methods are applied to an empirical analysis of nearly a century of fire history in the Sierra Nevada Mountains of California (section 4). A summary of the major points, and implications of the empirical analysis for risk managers, are discussed (section 5).

2. HEAVY-TAIL DISTRIBUTIONS AND WILDFIRE REGIMES

The idea that much can be learned about economic costs and losses from wildfires by recognizing the special significance of large fires can be traced to an article published by Strauss and colleagues (1989) titled "Do One Percent of the Forest Fires Cause 99 Percent of the Damage?" In that study, the authors provided a statistical analysis of wildfire data from the western United States and Mexico that showed the underlying statistical distribution of fire sizes was consistent with the heavy-tailed Pareto distribution. Several subsequent studies, spanning a wide array of forest types in the United States (Malamud et al. 1998, Malamud et al. 2005), Italy (Ricotta et al. 1999), Canada (Cumming 2001), China (Song et al. 2001) and the Russian Federation (Zhang et al. 2003), also concluded that wildfire regimes are consistent with the heavy-tailed Pareto distribution. The Pareto wildfire distribution may be truncated (Cumming 2001) or tapered (Schoenberg et al. 2003) to account for the finite size that can be attained by fires within forested ecosystems.

To fix ideas regarding the nature of the heavy-tailed Pareto distribution and the consequence of such a data generation process for the analysis of large wildfires, it is necessary to introduce some notation. To begin, a cumulative distribution function of the random variable X, denoted by F(x) = P(X ≤ x), is said to be heavy-tailed if x ≥ 0 and

$$\lim_{x\to\infty} P(X > x + y \mid X > x) = \lim_{x\to\infty} \frac{\bar{F}(x+y)}{\bar{F}(x)} = 1,\ y \geq 0 \tag{4.1}$$

where $\bar{F}$= 1-F(x), referred to as the "tail distribution" (Sigman 1999) or "survivor function" (Miller, Jr. 1981). Intuitively, equation (4.1) states that if X exceeds some large value, then it is equally likely that it will exceed an even larger value as well. The Pareto distribution is a standard example of a heavy-tailed distribution: $\bar{F}(x) = x^{-\alpha}$ where $x \geq 1$ and $\alpha > 0$. If $\alpha < 2$, then the distribution has infinite variance (the distribution converges so slowly to zero that it cannot be integrated), and if $\alpha > 1$, the distribution has infinite mean.

Extreme value models focus attention on the tail of a statistical distribution of events rather than imposing a single functional form to hold for the entire distribution. It is important to understand that the family of extreme value statistical models does not impose a heavy-tail upon the data. Rather, the extreme value parameter estimates indicate whether the data have a light, moderate or heavy-tailed distribution (Coles 2001). The classical method used in the statistics of extremes, known as the Generalized Extreme Value (GEV) method focuses attention on the statistical behavior of the maximum value attained by some random variable during each time period (or "block"):

$$M_n = \max\left(X_1, X_{2,} \ldots, X_n\right) \tag{4.2}$$

where $X_1,\ldots,X_n$ is a sequence of independent random variables each having an underlying distribution function F. If n represents the number of wildfire observations recorded in a year, then M_n is the largest wildfire recorded that year. Classical extreme value theory shows that there are three types of distributions for M_n (after linear renormalization): the Gumbel (intermediate case), Fréchet (heavy-tail) and Weibull (truncated at a maximum size) families. These three families are described by Coles (2001).

Using extreme value theory, Moritz (1997) fitted a GEV distribution using wildfire data from two geographic divisions within the Los Padres National Forest in southern California. He found that the percentage of years in which the single largest fire burned more than one-half the annual total was 65 percent and 81 percent for the two study areas, and that the size distribution of the largest annual wildfires between the years 1911 and 1991 was heavy-tailed. This result is important because it is consistent with empirical studies showing that the entire range of fire sizes is Pareto distributed. Further, based on graphical evidence, he speculated that "extreme weather" might create conditions such that large wildfires are "immune to suppression" (p. 1260). Thus, a possible linkage between very large wildfires, environmental conditions, and fire suppression technology was suggested.

Although the GEV model provides a theoretical foundation for the analysis of extreme events, data use is inefficient in model estimation because only a single observation per time period is utilized. A second approach to extreme value analysis overcomes this limitation by using observations which exceed a high threshold value, often referred to as the "peaks over threshold" method. Again let $X_1, X_2, \ldots$ represent a sequence of independent and identically distributed random variables with distribution function F, and let u represent some high threshold. The stochastic behavior of extreme events above the threshold is given by the conditional probability

$$\Pr\left(X > u + y \mid X > u\right) = \frac{\bar{F}(u+y)}{\bar{F}(u)}, \quad y > 0 \tag{4.3}$$

which clearly bears a strong resemblance to equation (4.1). It can be shown that, by taking the limiting distribution of equation (4.3) as u increases, the distribution function converges to a Generalized Pareto distribution $G_{\xi\sigma}(y)$ (Coles 2001):

$$G_{\xi\sigma}\left(y\right) = \Pr\left(X - u \le y \mid X > u\right) = \begin{cases} 1 - \left(1 + \dfrac{\xi}{\sigma} y\right)^{-\frac{1}{\xi}} & \text{if } \xi \ne 0 \\ 1 - e^{-\frac{y}{\sigma}} & \text{if } \xi = 0 \end{cases} \tag{4.4}$$

where $y = x–u$. The parameter ξ is called the shape parameter and σ is the scaling parameter. When $\xi < 0$, the distribution has a finite upper endpoint at $–\sigma/\xi$; when $\xi = 0$, the distribution is an exponential (light-tail) distribution with mean σ; when $\xi > 0$, the distribution has a heavy-tail (or Fréchet distribution) with mean

$\sigma/(1-\xi)$, given that $\xi < 1$ (Smith 2003). If $\xi \geq 1$ the mean of the distribution is infinite, and if $\xi > 1/2$ the variance is infinite ("wildly random").

Parameters of the Generalized Pareto model were estimated by Alvarado and colleagues (1998) for large wildfires between 1961 and 1988 in Alberta, Canada. Using alternative threshold values (200 hectares and the upper one percentile of fire sizes) they concluded that the data were Fréchet (heavy-tail) distributed. In fact, the fire data were so heavy-tailed that the fitted distributions were found to have both infinite means and infinite variances.

The various findings reported above—that wildfire size distributions are heavy-tailed—represent an important statistical regularity. However, economists are generally interested in conditional probabilities, that is, factors that induce non-stationarity in statistical distributions (Brock 1999). In the following section, we describe how covariates can be introduced into models of heavy-tailed statistical distributions and show how hypotheses about covariates can be tested in a "regression-like" framework. These methods provide a powerful tool for researchers to investigate factors that influence the generation of large wildfires.

3. INCLUDING COVARIATES IN EXTREME VALUE THRESHOLD MODELS

As mentioned above, Generalized Pareto models are more efficient in the use of data than classical extreme value models because they permit multiple observations per observational period, such as fire year. The main challenge in the Generalized Pareto model is the selection of a threshold for data inclusion. Statistical theory indicates that the threshold u should be high enough to be considered an extreme value, but as u increases less data is available to estimate the distribution parameters. Although rigorous methods for determining the appropriate threshold are currently receiving a great deal of research attention, graphical data exploration tools are typically used to select an appropriate value for u using a plot of the sample mean excess function (Coles 2001). In particular, the threshold is chosen where the sample mean excess function (i.e., the sample mean of the values that exceed the threshold) becomes a linear function when plotted against the threshold value.

Having determined a threshold value, parameters of the Generalized Pareto distribution can be estimated by the method of maximum likelihood. For $\xi \neq 0$, the likelihood function is

$$L(\xi,\sigma) = \prod_{i=1}^{m} \frac{1}{\sigma}\left[1 + \xi\frac{(x_i - u)}{\sigma}\right]^{-\left(1+\frac{1}{\xi}\right)} \tag{4.5}$$

and the log-likelihood is

$$\ln L(\xi,\sigma) = -m\ln\sigma - \left(1 + \frac{1}{\xi}\right)\sum_{i=1}^{m} \ln\left[1 + \frac{\xi}{\sigma}(x_i - u)\right] \tag{4.6}$$

where m is the number of observations, x_i is the size in acres of fire i, and u is the threshold fire size in acres. Note that equation (4.6) can only be maximized when $\left(1+\sigma^{-1}\xi\left(x_i-u\right)\right)>0$ for all i = 1,…,m. If this is untrue, it is necessary to set $lnL(\xi,\sigma)$ = -∞ to assure convergence. For the special case where ξ = 0, the log-likelihood is

$$ln\,L(\sigma)=-m\,ln\,\sigma-\sigma^{-1}\sum_{i=1}^{m}\left(x_i-u\right) \tag{4.7}$$

and the model is a member of the exponential (non-heavy-tailed) family of distributions.

If the underlying stochastic process is non-stationary, then the simple Generalized Pareto model can be extended to include covariates such as time trends, seasonal effects, climate, or other forcing variables. Non-stationarity is typically expressed in terms of the scale parameter (Smith 2003). For example, to test for a time trend, the scale parameter could be expressed as a function of time, where the scale parameter for observation i is $\sigma_i=\beta_0+\beta_1 t_i$, where t represents time. More generally, a vector of covariates can be included in the model by expressing the scale parameter as a linear function of the product of a vector of explanatory variables and parameters (β) to be estimated:

$$\sigma=\left[1,z_1,\ldots,z_n\right]\begin{bmatrix}\beta_0\\ \beta_1\\ \vdots\\ \beta_n\end{bmatrix} \tag{4.8}$$

where n is the number of covariates included in the model.

The Generalized Pareto model is asymptotically consistent, efficient, and normal if ξ > –0.5 (Coles 2001, Smith 2003), allowing for the derivation of standard errors for the parameter estimates using either the bootstrap method or the inverse of the observed information matrix (Smith 2003). Having obtained estimates of standard errors, hypotheses regarding the statistical significance of the covariates can be tested.

The statistical model can be used to estimate the expected value (average size) of large fires during a fire season given values for the set of covariates and estimates of the parameter vector [β_0, …, β_n]. In the simplest case, the value for a covariate may represent an updated value for a time trend. Or, the value may represent the forecasted value of a covariate such as a climate indicator. For the Generalized Pareto model, the expected value of an event that exceeds the threshold has a simple expression:

$$E(Y)=\frac{\sigma(Z)}{1-\xi} \tag{4.9}$$

given that $\xi < 1$ (recall that if $\xi > 1$ the mean is infinite), Y is the amount by which an observation exceeds the threshold ($Y–\mu > 0$), and Z is a vector of covariates. In terms of wildfire sizes, $E(Y) + \mu$ provides an estimate of the expected (or average) size of a large wildfire given that a wildfire size has exceeded the threshold value.

Economic metrics can be calculated using information on the economic values associated with the expected area burned. For example, the expected value of timber at risk of loss to a large wildfire could be estimated by multiplying the expected number of acres burned in a large wildfire by an average per acre estimate of stumpage value. Expected suppression costs associated with large wildfires could be estimated in a similar fashion. Or, if information were available on the non-market economic values of resources related to recreation, watersheds or wildlife habitat, then economic estimates of non-market values at risk could be computed as well. If statistically significant covariates associated with management interventions are identified that alter the production of large wildfires, then the parameter estimates on the covariates can be used to estimate the economic benefits of interventions. An illustration is presented in the following empirical example.

4. LARGE WILDFIRES IN THE SOUTHERN SIERRA NEVADA MOUNTAINS

The Southern Sierra Nevada Mountains (SSNM) provide a useful case study for illustrating the application of extreme value analysis to wildfire modeling. Nearly a century of fire data are available for land management units located in this region, allowing us to investigate factors influencing wildfire production over short, medium and long time scales. The fire data analyzed in this chapter come from the Sequoia National Forest (SQF) which sits at the southern extension of the SSNM and comprises 5,717 km^2, or 27 percent, of the federally managed lands in the SSNM (fig. 4.1). The northern and western reaches of SQF have the most forest cover, with substantial area at lower elevations in the southwest in grassland and in the southeast in chaparral. Giant sequoia groves are a small, but important, component of the fire-adapted ecosystems in SQF.

Fire history data for SQF were derived from fire perimeter records (fig. 4.2) for the years 1910–2003, obtained from the California Department of Forestry Fire and Resource Assessment Program. A histogram of the fire size distribution for SQF (fig. 4.3) clearly shows that the distribution is not normal or lognormal, is highly skewed, and has a long right-hand-side tail (note that fire sizes above 10,000 acres have been combined for graphical convenience). At a first approximation, the distribution of fire sizes for SQF appears as though it may be Pareto distributed (heavy-tailed) or, perhaps, distributed as a negative exponential (light-tailed). Fortunately, statistical methods can be used to test whether the distribution is light- or heavy-tailed (Reiss and Thomas 2001).

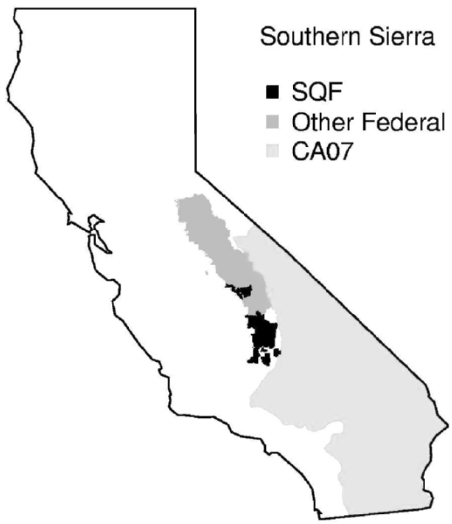

Figure 4.1. Location of Sequoia National Forest (SQF, black) in relation to other Federal Forest and Park land in the Southern Sierra Nevada Mountains (grey) and California Climate Division number 7 (CA07, light grey).

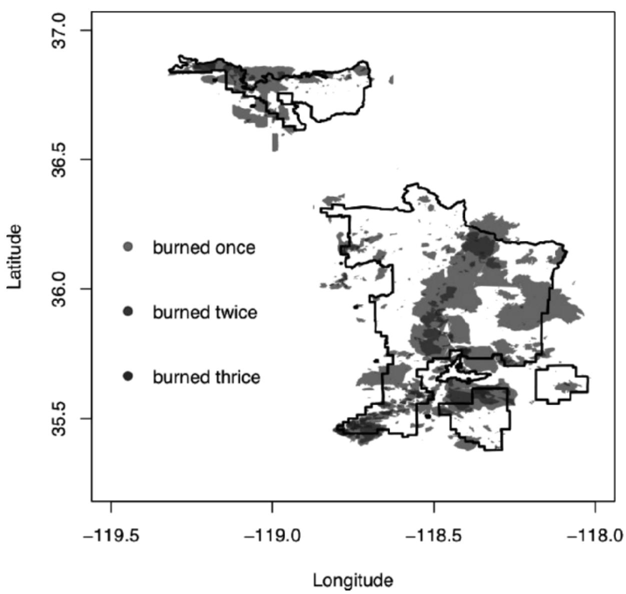

Figure 4.2. Map showing areas burned since 1910 (shaded) and Sequoia National Forest boundary.

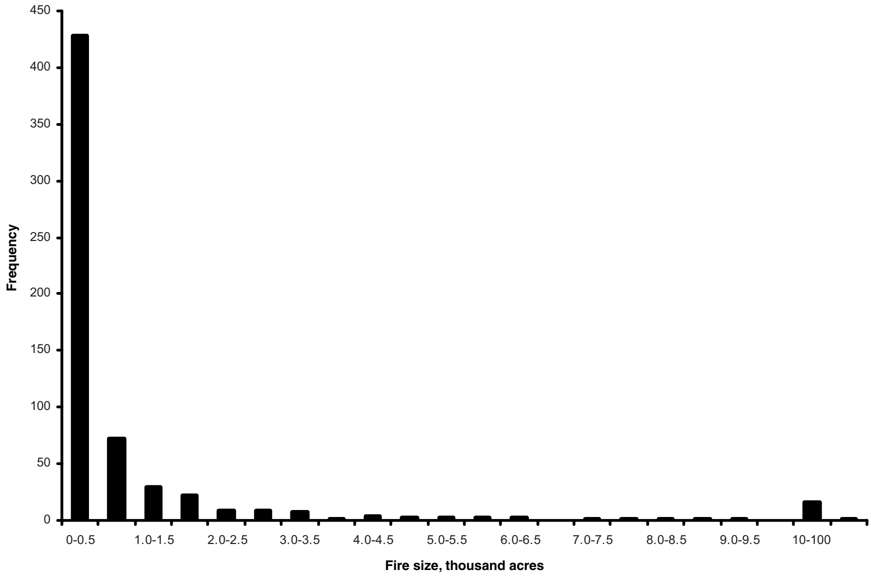

Figure 4.3. Fire size distribution, Sequoia National Forest, 1910-2003

4.1 Model Specification

The fire history for SQF permits us to test a variety of hypotheses, including whether or not a long-term trend can be identified in the occurrence of large fires. Additionally, the time-series allows us to investigate whether shorter-run trends, such as changes in fire suppression technology, and seasonal influences, such as climatic effects, have influenced the production of large wildfires. Although the covariates discussed below are included in the model specification, the results should be viewed as illustrative. Because this model is the focus of ongoing research, it should be understood that alternative model specifications may (or may not) yield somewhat different results.

4.1.1 Time trend

In the SSNM, the combined influence of livestock grazing during the nineteenth century and fire suppression during the twentieth century have changed tree species composition and increased the density of forest stands (Vankat and Major 1978). As early as the late 1800's, foresters in California were arguing for fire exclusion to protect timber resources for the future, and by the early twentieth century fire reduction was occurring (Skinner and Chang 1996). Suppression of low and moderate severity fires has caused conifer stands to become denser, especially in low- to mid-elevation forests, and shade tolerant, fire-sensitive tree species have become established. In turn, these vegetative changes have led to a profusion of wildfires that burn with greater intensity than in the past, with crown fires becoming more common (Skinner and Chang 1996). Further, the proportion of the annual acreage burned by the largest wildfire on National Forest land has trended upwards during the twentieth century (McKelvey and Busse 1996).

We hypothesize that a positive trend may be identified in the probability of observing large wildfires in the SSNM which may reflect these long-term changes in forest composition. A trend variable, $time_i$, was created by setting $time_i = 1$ for the first year of the data record, $time_i = 2$ for the second year, and so forth up to the final year of the data record.

4.1.2 Fire suppression technology

The use of air tankers for fighting wildfires began in California. The first air drop was made on the Mendenhall Fire in the Mendocino National Forest in 1955 in a modified agricultural biplane. These early aircraft had roughly a 100 gallon capacity and dropped about 124,000 gallons of water and fire suppressants during that year. By 1959, heavier air tankers with as much as a 3,000 gallon capacity were in operation and dropped nearly 3.5 million gallons in 1959 (Anon. 1960). Aircraft are now commonly used in fire suppression and their expense is a major component of suppression costs on large wildfires (Mangan 2001).

Although historical aircraft fire suppression cost data are not available for the SSNM, an aircraft variable was specified for use in our large wildfire probability model by creating a dummy variable, *air_dummy*, to approximate the effective use of air tankers for fire suppression in California. In particular, we set *air_dummy* = 0 for years prior to 1960 and *air_dummy* = 1 for subsequent years.

4.1.3 Climate

The moisture available in fuels is a critical factor in wildfire spread and intensity. Climatic effects are specified in our model using PDSI, which is an index of combined precipitation, evapotranspiration, and soil moisture conditions. PDSI has been used successfully in previous studies of climate-fire relationships (Balling et al. 1992, Swetnam and Betancourt 1998, Mitchner and Parker 2005, Westerling et al. 2003, Westerling, chapter 6). The index is negative when inferred soil moisture is below average for a location, and positive when it is above average. In this chapter, we investigate the relationship between climate and large fire sizes using observations on July values for PDSI for California region 7 (fig. 4.1). PDSI values from the U.S. Climate Division Data set were obtained from NOAA for 1895-2003. July PDSI calculated from monthly climate division temperature and precipitation is used here as an indicator of inter-annual variability in summer drought.

We note a pronounced trend toward drier summer conditions over the entire period of analysis, with a highly significant trend in July PDSI (fig. 4.4). This tendency toward drier summers is probably a function of both lower precipitation and higher temperatures. There has been a trend toward lower precipitation throughout the entire Sierra Nevada Mountain range over the period of analysis,

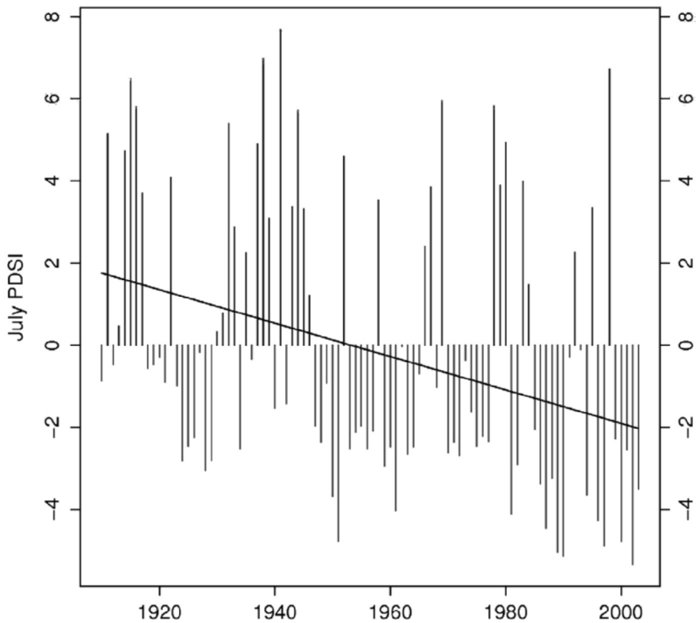

Figure 4.4. July PDSI index for California Climate Division number 7, 1910–2003. Diagonal line is ordinary least squares regression fit to a time trend.

while spring and summer temperatures have been much warmer since the early 1980s. Warmer springs in particular, combined with less precipitation, result in an earlier snow melt at mid and higher elevations, which in turn implies a longer, more intense summer dry season and fire season (Westerling et al. 2006).

4.2 Empirical Models

Two Generalized Pareto models were estimated that fit historical fire size data for Sequoia National Forest: (1) a basic model with a constant scale parameter, and (2) a covariate model that specified the scale parameter as a linear function of a time trend ($time_i$), an air tanker dummy variable (*air_dummy*), and climate effects (*PDSI*). The models were estimated using the Integrated Matrix Language (IML) programming code in the SAS statistical software.

Prior to estimating either model, it was necessary to choose the threshold fire size u above which large or “extreme” fires would be modeled. Mean excess plots were created to identify the location of the fire size threshold. As explained in Coles (2001), the Generalized Pareto distribution will be a valid representation of the distribution of exceedances above u if the plot is linear past that point. Visual inspection of the mean excess plot indicated that a threshold of 500 acres would be appropriate as the plot became generally linear beyond the $u = 500$ acre fire size. Reference to figure 4.3 also suggests that the rather long tail of the fire size distribution may be initiated at a threshold of 500 acres.

Although a relatively small proportion (30 percent) of the total number of fires in SQF exceeded 500 acres, they accounted for nearly all (94 percent) of the total area burned during the fire record (fig. 4.5). After eliminating observations with total burned area of 500 acres and less, 181 observations remained for estimation of the SQF large fire distribution. Summary statistics for the 181 fires, and the set of covariates included in the model, are given in table 4.1.

Maximum likelihood techniques were used to estimate the parameters in equation (4.6), where the scale parameter was specified using covariates as shown in equation (4.8). That is, the scale parameter was specified as: $\sigma_i = \beta_0 + \beta_1 time_i + \beta_2 air\_dummy_i + \beta_3 PDSI_i$. Standard errors for the parameter estimates were derived from the inverse of the observed information matrix, and allowed us to test whether the parameter estimates were significantly different than zero.

4.3 Results

In the simple model with no covariates, both the shape and scale parameter estimates were significantly different than zero at the 1 percent level. Since the parameter estimate for the shape parameter ξ is greater than 0, the distribution has a heavy-tail (Fréchet). This result is consistent with the studies in the literature reviewed above. Further, since $\xi > 1$, the distribution has an infinite mean and variance, which is consistent with the findings reported by Alvorado and others (1998). Although forest extent is finite and, therefore, average wildfire size must be finite, the finding of an infinite (divergent) mean and variance for

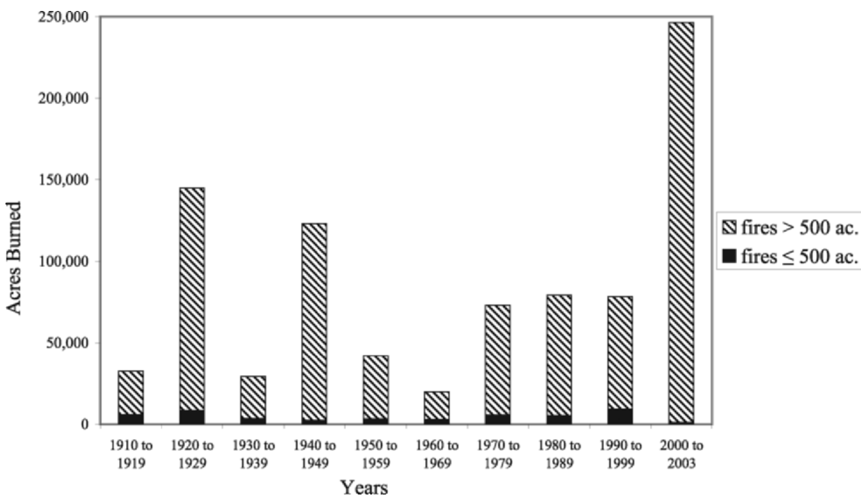

Figure 4.5. Acres burned by fire size class, Sequoia National Forest, 1910-2003

Table 4.1. Descriptive statistics for large fires and covariates included in the model.

	Mean	Std. Dev.	Min.	Max.
fire size (ac.)	4,533.49	13,365.25	504.00	149,470.00
time (trend)	47.96	29.75	1	94
air_dummy	0.47	0.49	0	1
PDSI	-1.00	3.13	-5.35	7.70

Table 4.2. Parameter estimates of the basic Generalized Pareto extreme value model.

Parameter	Value	Std. Error	t-statistic
shape ξ	1.02	0.15	6.84
scale σ	780.23	115.68	6.75
N = 181			
log likelihood	-1,570.74		

Table 4.3. Parameter estimates of the Generalized Pareto extreme value model with covariates.

Parameter	Value	Std. error	t-statistic
shape ξ	0.91	0.14	6.49
scale σ			
constant	593.47	199.12	2.98
time	15.19	7.88	1.93
air_dummy	-983.44	420.01	-2.34
PDSI	-62.70	21.52	-2.91
N = 181			
log likelihood	-1,564.25		

fires exceeding the 500 acre threshold implies that fires greatly exceeding fire sizes included in the historical record are possible. This finding is important because it indicates that large wildfire production is extremely variable despite the constraints imposed by physical conditions. In turn, extreme variability in the production of large wildfires makes fire program planning and budgeting difficult, especially if the variables driving the stochastic fire generation process cannot be identified or reliably forecast.

The parameter estimate on ξ in the Generalized Pareto model with covariates indicates that large fires in SQF have a heavy-tail, or Fréchet distribution, with infinite variance, which is similar to the basic model. However, because the parameter estimate $\xi < 1$ in the covariate model, the mean of the distribution is finite (in contrast with the basic model). This result is important, as it allows us to estimate the size of the average fire for wildfires exceeding the threshold (500 acres), and evaluate how changes in management inputs influence the average size of large wildfires.

Including covariates in the Generalized Pareto model suggests that the distribution of large fires in SQF has been non-stationary over the recorded fire history. The positive parameter estimate on the time trend is significant at the 10 percent level and indicates that the probability of observing a large wildfire has increased over the 94 year fire record. This result should be viewed as illustrative, not definitive, and a more fully specified model (considering, for example, non-linear effects and other covariates) may alter this finding. None-the-less, this result is consistent with the idea that land use history and fire suppression have contributed to altered tree species composition and density which, in turn, have contributed to forest conditions with greater flammability. We note further that this effect may be confounded to some degree by the increased development of roads and trails in SQF over the 94 year period, and the concomitant increase in the number of people visiting the forest may have contributed to the increasing trend in large wildfires.

Consistent with our *a priori* hypothesis, drier fuel conditions (as measured using PDSI) were found to be related with larger fires. The parameter estimate on PDSI was negative and significant at the 1 percent level. Recall that negative values of PDSI correspond with the driest conditions, while positive values correspond with wet conditions. Consequently, the model results indicate that very dry conditions are associated with an increased probability of large wildfires.

The parameter estimate on the air tanker dummy variable is negative and significant at the 5 percent level and suggests that the deployment of air tankers since 1960 has decreased the probability of observing large wildfires. This result is consistent with the finding reported by Moritz (1997) who concluded that air tankers have aided the containment of large wildfires in California's Los Padres National Forest. Again we note that these results are provisional and might change with improved model specifications.

Given the parameter estimates, various scenarios can be constructed to demonstrate the effect of the covariates on the expected large fire size and to evaluate the impact of management interventions (table 4.4). For example, the scenarios shown in table 4.4 indicate that the expected large fire size is quite sensitive to the use of air tankers for fire suppression. Under average drought conditions in the year 2002, the use of air tankers reduces the expected large fire size from 24,700 acres to 13,690 acres, a reduction of about 45 percent. Given data on fire suppression costs, this relationship could be used to estimate the expected benefits (reductions in cost) due to the use of air tankers.

Because they represent averages, expected large fire sizes may not be sensitive to extreme conditions experienced during a single fire year. The fire year 2002 provides an instructive example, as the July PDSI for that year was the driest

Table 4.4. Scenarios depicting the expected size of large fires (thousand acres) under alternative conditions.

Scenario	Year 2002	Year 2010	Year 2025	Year 2050
1. Average drought w/ tankers	13.69	15.05	17.60	21.85
2. Average drought w/o tankers	24.70	26.05	28.6	32.86
3. Extreme drought w/ tankers	16.74	18.10	20.66	24.91
4. Extreme drought w/o tankers	27.76	29.12	31.67	35.92

since 1910. The expected large wildfire size for SQF for 2002, incorporating the time trend and the effect of PDSI, was computed to be roughly 15,000 acres if air tankers were used in suppression and roughly 26,000 acres if air tankers were not used for fire suppression. During the summer of 2002, the largest fire for SQF since 1910 was recorded (the McNally fire) which burned nearly 150,000 acres. Our estimate of average fire size for that year is much too low and suggests either that our model has omitted some important variables (such as wind speed) or that other unobserved factors create the extraordinary variance observed in large wildfire regimes.

Although an annual estimate of expected large wildfire size may be inaccurate under extreme climatic conditions, averaging expected large wildfire sizes over time improves the precision of expected values. For example, the average large fire in SQF between 1994 and 2003 was 9,625 acres. Using the parameters in our Generalized Pareto covariate model, we estimated that the average large fire would be 12,247 acres, which is within 1 standard deviation of the sample average. Therefore, temporal averaging can smooth out the estimate of expected large wildfire size even during periods of extreme climatic conditions. In turn, this suggests that estimates of the benefits of large wildfire management interventions should likewise be temporally averaged and that confidence intervals should be reported.

5. SUMMARY AND CONCLUSIONS

Although extreme value statistical models are not widely used in wildfire modeling, the literature review, results, and analysis reported in this chapter suggest that further development of these models is warranted for four principal reasons:

- Wildfire production often does not follow a “light-tail” distribution such as a normal or log-normal distribution. Rather, fire size distributions reported for several regions around the globe have heavy-tails characterized by infinite moments.
- Standard statistical techniques, such as ordinary least squares regression, may produce very misleading parameter estimates under conditions of infinite variance (second moment).

- Extreme value models focus attention on the tail of the distribution which, in fire modeling, is where most of the ecological and economic impacts occur. These statistical models are stable under conditions of infinite moments and allow probabilities of catastrophic events to be rigorously estimated.
- A set of covariates can be included in extreme value models providing the ability to test hypotheses regarding variables that influence the production of large wildfires. Parameter estimates on covariates can be used to evaluate the impacts of management interventions on the production of large wildfires.

A major conclusion of this chapter is that large wildfires are intrinsic to fire-adapted ecosystems and that memorable events such as the Yellowstone fires of 1988 (Romme and Despain 1989) and the McNally fire of 2002 in SQF cannot be simply dismissed as catastrophic outliers or anomalies. Rather, the underlying fire generation process operates in a fashion such that wildfires greatly exceeding those represented in local or regional fire histories may occur sometime in the future. Infinite variance in wildfire production, or wild randomness, greatly complicates planning operations for large fires. For example, moving average models of acres burned in large fires likely provide poor forecasts of the size of future large fires because the first moment converges very slowly to its true value in a wildly random state. The development of decision-making strategies for resources exposed to the state of wild randomness remains a challenge for risk managers in the finance and insurance sectors as well as for wildfire managers.

The second major conclusion of this chapter is that the ability to include covariates in a model of large wildfires characterized by infinite variance provides a robust method for evaluating the impact of management interventions. For example, the impact of deploying air tankers in fire suppression (captured using a dummy variable) was illustrated. The parameter estimate on the air tanker dummy variable was shown to have a substantial effect on the expected size of large fires. Given the size of this effect, data on large wildfire suppression cost could be used to estimate the expected benefits (cost savings) attributable to air suppression. In turn, the expected benefits of air suppression could be compared with air suppression costs. This modeling approach can be more generally used to evaluate the costs and expected benefits of other management interventions on large wildfires. Future research will be directed at identifying and testing alternative extreme value covariate models on an array of large wildfire regimes and management interventions with the goal of understanding how large fire costs might be better managed.

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CHAPTER 5

THE PRODUCTION OF LARGE AND SMALL WILDFIRES

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1. INTRODUCTION

Natural large (catastrophic) disturbances are important because of their potential long-lasting impact on their surroundings, but underlying differences between frequent small and less common large disturbances are not well understood (Turner and Dale 1997, Romme et al. 1998, Turner and Dale 1998, chapter 4 of this book). Smaller disturbances may be better understood given their relative abundance, which lends itself more readily for study, but it is, perhaps, more useful to understand the forces driving damaging, catastrophic events. Wildland fires represent a perfect example. Nationwide, over 130,000 wildfires burn more than 4 million acres annually (1960-2002), these fires costing Federal agencies in excess of $768 million a year (1994-2002) in suppression alone (National Interagency Fire Center, http://www.nifc.gov/stats/index.html). Average wildfire size was 31 acres, with a suppression cost of $4800 per fire. The average wildfire does not appear a catastrophic threat, however this ignores the spatial distribution of these fires in relation to values at risk (an averaged size fire in a heavily populated area poses a different risk than a similarly size fire far removed from people and items of value). Catastrophic fire events, while relatively infrequent, do occur with some regularity—the 2000 Cerro Grande fire in New Mexico devastated 47,650 acres, two fire complexes in California in fall of 1999 each burned for three months and consumed a total of 227,647 acres, and during the 1998 Florida summer wildfire season, two fire complexes accounted for 205,786 acres or 9 percent of all wildfire acres, nationwide, in that year (National Interagency Fire Center, http://www.nifc.gov/stats/index.html).

Do the largest fires account for a disproportional amount of the area burned and damage? Is it possible that the largest 1 percent of fires account for 99 percent of the area, as Strauss et al. (1989) explored? For the state of Florida, the largest 1 percent accounted for 67 percent of total area burned with an average fire size of 2,641 acres versus 13 acres for the smallest 99 percent (1981-2001). Understanding the differences between small and large wildfires, including the exogenous factors influencing each, may provide decision-makers with better tools to mitigate future large-scale fire events. It is not necessarily true that large

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disturbances will respond to the same controls that smaller disturbances do (Romme et al. 1998), thus wildfires should be modeled in a way that is flexible to potential differences.

Our objectives in this paper are to examine the wildfires that occurred in the St. Johns River Water Management District (SJRWMD) in Florida between 1996 and 2001. We explore four main questions: (1) Do small and large wildfires behave differently? (2) What are the implications for wildland management decisions? (3) Does spatial information enhance wildfire modeling? (4) Does spatio-temporal scale matter? We are interested in differences exhibited by large and small wildfire regimes—differences in fire damages (area), causes (arson, lightning, and accidents), ignition sources (dominant fuel source), climatic and weather influences, land use and wildland management (fuels management) strategies, landscape characteristics, and spatio-temporal factors (including fire and fuels management on neighboring areas) and their relevance for future mitigation. We use a scale fine enough to allow spatio-temporal effects to be observed, yet at the same time, a scale broad enough to be policy relevant to decision-makers interested in minimizing the damaging effects of wildfire.

We model large infrequent wildland fires separately, those in excess of 1,000 acres, to ascertain whether these potentially catastrophically large fires are fundamentally different, and hence whether they respond differently to various mitigation approaches, than their smaller counterparts. We conclude that there are differences between the two fire regimes and examine factors correlated with the probability that a small wildfire will become large.

2. FIRE MODELING REVIEW

Previous empirical findings show wildfire behavior (whether meaning frequency, occurrence, size, or severity) to be related to four general sets of factors: wildfire specific characteristics, climate and weather, wildland/wildfire management and mitigation, and landscape attributes (including both landuse/landcover and socioeconomic characteristics). We review some of the results below, but first note the rarity of studies that include a full suite of factors from each set (chapter 3).

2.1 Wildfire Characteristics

Wildfire characteristics include factors to explain the *when*, *where*, and *why* of the fire occurrence. This includes factors such as the time of ignition (e.g., year, month, day, hour, or season), some set of locational factors (e.g., latitude and longitude or county), and fire cause (e.g., lightning, arson, or accidental). For instance, year and day variables, perhaps capturing seasonal and daily fluctuations, were found to be related to wildfire (Prestemon et al. 2002, Preisler et al. 2004). Location is important, whether meaning latitude and/or longitude (Donoghue and Main 1985, Preisler et al. 2004) or the geopolitical unit in which the fire ignition occurred (Garcia et al. 1995), which may signal the possibility that wildfires are spatially autocorrelated (Chou et al. 1993). Ignition cause also matters. Prestemon et al. (2002) found evidence that wildfires of different causes (lightning, arson, and accidents) were correlated with different exogenous factors.

2.2 Climate and Weather

Climate has been shown to influence wildfire size and severity in Florida (Barnett and Brenner 1992, Harrison and Meindl 2001, Prestemon et al. 2002, Beckage et al. 2003). The La Niña phase (colder than normal deviations in Pacific sea surface temperatures) of the El Niño Southern Oscillation (ENSO) has been associated with warmer, drier weather, but with more lightning strikes and more wildfire than the El Niño phase (Beckage et al. 2003).

The Keetch-Byram Drought Index (KBDI) provides a measure of organic fuels flammability and is calculated using maximum temperature and precipitation of the previous seven days (Keetch and Byram 1968). The KBDI provides an indicator (predictor) of fire danger (Butry et al. 2002, Goodrick 2002, Janis et al. 2002). Others have found that precipitation (Donoghue and Main 1985), temperature (Chou et al. 1993, Preisler et al. 2004) and humidity (Preisler et al. 2004) are each related to wildfire, with precipitation and humidity being negatively related and temperature positively. Preisler et al. (2004) included KBDI along with temperature into their models and found only temperature to be significant.

2.3 Management

Two dominant ways wildfire management may influence wildfire behavior are through fuels management (i.e., prescribed burning) and suppression. The relationship between prescribed fire and wildfire (either probability of ignition, fire size, or fire severity) has been shown to be negative at very fine scales (Brose and Wade 2002, Outcalt and Wade 2004) and even at very coarse scales (Davis and Cooper 1963, Gill et al. 1987, Prestemon et al. 2002). While prescribed fire has been found to be useful in reducing wildfire it does present users with several challenges, namely conducting prescribed fire on ideal weather days, as to prevent escapes and to limit its negative impacts (e.g., air quality) on local residents (Haines et al. 2001).

Much of the previous fire suppression literature has focused on understanding initial attack and fireline production (Fried and Fried 1996, Hirsch et al. 1998, Hirsch et al. 2004) or using simulations or other techniques to understand initial attack and containment (Donovan and Rideout 2003, Fried and Fried 1996). We know of no empirical research that quantifies the effectiveness of suppression, however defined, on wildfire behavior, at any scale, but especially at a relatively fine scale.

2.4 Landscape (Fuel & Socioeconomic) Characteristics

Landscape characteristics such as measures of landscape composition (e.g., fuel load, forest types, landcover, and landuse) and socioeconomic factors (e.g., population) are related to wildfire. Fuels buildup (Garcia et al. 1995), fuels moisture and susceptibility to burning (Preisler et al. 2004) have been found to be related to wildfire, where fuels buildup and susceptibility to burning were positively related to wildfire and fuel moisture negatively related. The fire spread index (a measure of fire spread potential) and the burn index (a function of potential fire spread and energy release) (Preisler et al. 2004), have both been found to be positively associated with fire probability (Garcia et al. 1995, Preisler et al. 2004).

Softwood and mixed (hardwood and softwood) forest were found to be positively correlated with wildfire occurrence (Zhai et al. 2003), with amount of forest cover (closed forests) to be negatively associated with high severity fires (Odion et al. 2004).

Previous wildfire has been shown to provide a protective effect on future wildfire (Chou et al. 1993, Prestemon et al. 2002), although nearby wildfire has been found to be positively correlated with fire probability (Chou et al. 1993).

Socioeconomic factors, such as population (Donoghue and Main 1985), distance to city (Zhai et al. 2003), and land ownership (Zhai et al. 2003) were found to be related to wildfire.

3. DATA SOURCES AND DESCRIPTION

This analysis focuses on the St. Johns River Water Management District (SJRWMD) located in northeast Florida, which includes portions of 18 counties (fig. 5.1). The SJRWMD was chosen primarily due to its abundance of wildfire, both large and small, within the wildland-urban interface (WUI) and availability of data. Wildfire presence within the WUI creates potentially large values at risk.

Wildfire data used in this analysis are divided into the four general categories outlined above (wildfire characteristics, climate and weather, wildland/wildfire management and mitigation, and landscape attributes).

3.1 Wildfire Characteristics

Data on individual wildfire occurrences were obtained from the Florida Division of Forestry (FDOF). FDOF's wildfire data contains detailed information of fires found on private and state-owned lands including, but not limited to, the date and time of ignition, location (Public Land Survey township, range, and cadastral section), size (acres), and cause (arson, campfires, cigarettes, children, debris burning, equipment, lightning, miscellaneous, railroad, and unknown) from 1981-2001. Fires on federal lands are excluded.

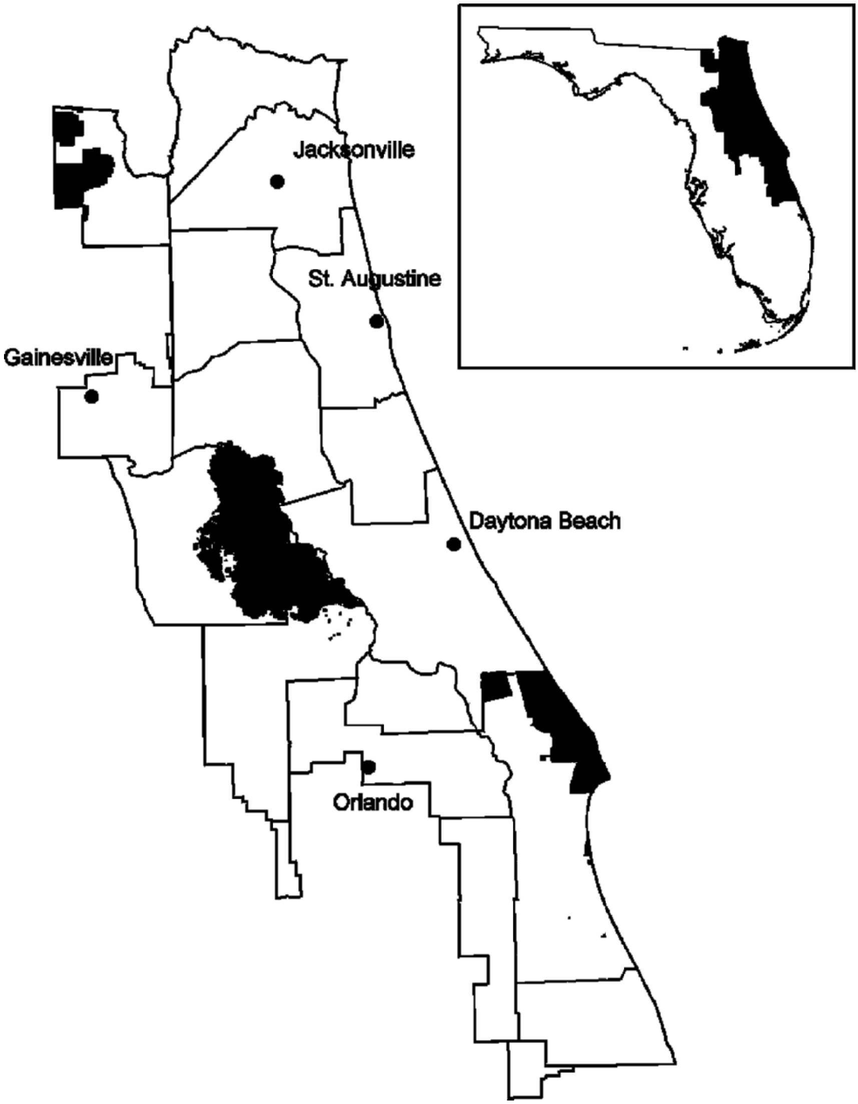

Figure 5.1. The St. Johns Water Management District, Florida. The black shading represents federal lands excluded from the analysis.

3.2 Climate/Weather

The ENSO measure used in this analysis is the Niño3 sea-surface temperature (SST) anomaly, which was obtained from the National Oceanic and Atmospheric Administration (National Oceanic and Atmospheric Administration, ftp://ftp.ncep.noaa.gov/pub/cpc/wd52dg/data/indices/sstoi.indices). The Niño

3 SST anomaly is measured as the positive (El Niño) or negative (La Niña) deviation from a moving average, in Celcius, of the Pacific sea surface temperature (at a specific location). KBDI was calculated for two weather stations in the SJRWMD region using daily data collected by the National Climate Data Center and provided by EarthInfo (2002). Each wildfire record was matched with a daily KBDI value based on its proximity to one of the two weather stations. The two weather stations reside in Volusia County (Deland) and in Duval County (Jacksonville).

The FDOF wildfire database also provides information on the humidity, wind speed, and dominant wind direction (calm, variable, east, north, west, south, northeast, northwest, southeast, and southwest) associated with each individual fire.

3.3 Management/Mitigation

The FDOF provided a second dataset that details all prescribed fire activities within the state (in order to conduct a prescribed burn in Florida, a permit must be obtained from the FDOF). Permit data includes information on the location (located by the township, range, and cadastral section), reason/type (hazard reduction, prior to seeding, site preparation, disease control, wildlife, ecological, or other), and total size (in acres). The dataset includes permits issued between 1989 and 2001.

The FDOF wildfire database also provides information on whether a fire is a "limited action wildfire" (whether a wildfire was allowed to burn). In addition, we use each wildfire start time and fire crew arrival time, from the FDOF database, to create a measure of initial attack/suppression (response time).

3.4 Public Land Survey Township/Range/Cadastral Section (Landscape) Characteristics

Section-level road and census data (population, income, and education) were created from U.S. Census Bureau TIGER/Line GIS data. Fire department location (Florida Department of Emergency Management, http://www.dca.state.fl.us/fdem/) was used to calculate the distance between each section and the closest fire department (straight line distance was used).

National Land Cover Data, based on the Multi-Resolution Land Characteristics (MRLC) Consortium's land cover map (30-meter resolution grid) was used to determine landcover composition within and surrounding each section. Five landcover classes were assembled—grass (grassland/herbaceous), upland forest (deciduous, evergreen, and mixed forest), urban (low intensity residential, high intensity residential, and commercial/industrial/transportation), water (open water), and wetland (woody wetland).

The FDOF database also provided an indicator for the fire district where each wildfire began (fig. 5.1 also depicts fire district boundaries), ignition fuel type (grass, hardwoods, muck, palmetto-gallberry, pine, swamp, and other), and information on fuels moisture (buildup index) and the potential that conditions may have on fire spread (fire spread index).

3.4.1 Descriptive statistics

Table 5.1 provides descriptive statistics of wildfire attributes, climate and weather, management and mitigation, and landscape/section characteristics, as defined above, for large and small wildfires. This table provides statistics based on wildfires occurring in 1996-2001, the period of analysis.

We examine 7,302 wildfires that occurred between 1996 and 2001 in the SJRWMD. These wildfires ranged in size from 0.1 acres to 61,500 acres. Of these 7,302 ignitions, only 53 were greater than 1,000 acres and the majority of large fires (32) occurred during the summer of 1998. Although large wildfires accounted for a mere 0.7 percent of all ignitions, they were a whopping 74 percent of the area burned!

The leading cause of large wildfires was lightning (55 percent), followed by accidents (unintentional human-caused fire—campfires, cigarettes, children, debris burning, equipment, miscellaneous, railroad, and unknown; 28 percent) and arson (17 percent). The leading cause of small wildfires was accidents (43 percent), followed by lightning (32 percent), then arson (25 percent). Roughly, the same percentage of large and small fire ignitions occurred in palmetto-gallberry fuel types (53 percent and 51 percent, respectively) and in pine (15 percent and 12 percent, respectively). A greater percentage of small fires occurred in grasslands (19 percent versus 9 percent) and hardwoods (5 percent versus 0 percent) than large fires. Of the remaining fuel type (swamp/muck/other), a larger percentage of large fires (23 percent) occurred there than small (13 percent).

Comparing large fires to small fires, we find several statistical differences (at the 5 percent level) between the estimated means of several of their attributes. Large wildfires appear to correspond with dry, hot days (larger mean KDBI values) with lower humidity, larger negative Niño3 SST anomaly values (negative values correspond with the La Niña phase), in areas with a greater accumulations of flammable fuels (fuels buildup), a greater propensity to spread (fire spread index), and in areas with fewer roads and fewer, but wealthier, people. It appears that large and small wildfires occurred in areas with similar landscapes, the exception being urban areas and areas under water. Larger fires occurred in areas with less urbanization and more water. Statistically, smaller fires were associated with hazard reducing prescribed burning during the previous year, burning three years lagged, and in neighboring areas during the current year.

3.4.2 Exploratory spatial data analysis

Next, we examine and compare the spatial distribution of the large and small wildfires. In particular, we were interested whether or not large or small fires demonstrate spatial clustering—do fires, either large or small, reside proximately to other fires? Alternatively, do large/small fires seem to occur in the

Table 5.1. Select descriptive statistics—for each variable the mean is shown with the exception of the categorical variables (mode) and the 0/1 variables (frequency).

		Wildfire			
Variable	Units	Large		Small	
		Statistic	*SE*	*Statistic*	*SE*
Area Burned	Acres	6240.1	1522.52	16.1	0.80
Fire Characteristics (X^F)					
1998	Count	32		1399	
Fire Cause					
Arson	Count	9		1834	
Accident	Count	15		3119	
Lightning	Count	29		2296	
Climate/Weather (X^C)					
Niño3 SST Anomaly	Celsius	-0.14	0.02	-0.03	0.01
KBDI	Index, 0-800	558.2	20.07	424.1	2.04
Humidity	Percent	44.5	1.13	48.8	0.16
Wind Speed	MPH	10.0	0.74	9.0	0.06
Management/Mitigation (XM)					
Response Time	Hours	7.7	2.16	3.5	0.11
Let Burn	Count	7		219	
Prescribed Burn (Hazard Reduction)					
Own Section—Current Year	Acres	8.6	6.06	3.9	0.59
Own Section—Lag 1 Year	Acres	0	0.00	3.6	0.52
Own Section—Lag 2 Year	Acres	8.5	5.00	5.6	0.96
Own Section—Lag 3 Year	Acres	1.5	1.03	4.7	1.11
Neighbor Sections—Current Year	Acres	8.9	4.66	18.8	1.40
Neighbor Sections—Lag 1 Year	Acres	68.5	31.07	34.6	2.04
Neighbor Sections—Lag 2 Year	Acres	122.9	74.20	34.8	2.53
Neighbor Sections—Lag 3 Year	Acres	46.3	20.26	39.5	2.73
Section Characteristics (X^S)					
Population Density	People/KM^2	15.4	2.58	93.5	2.52
Income	Dollars	31199.5	1238.28	28053.4	104.99
College	Percent	39.2	2.35	35.2	0.18
Roads	Kilometers	4.0	0.76	7.4	0.09
Distance to Fire Dept.	Kilometers	13.5	1.20	14.7	0.14
Buildup	Index, 0-250	68.1	5.67	47.0	0.40
Spread Index	Index, 0-100	27.5	2.24	20.4	0.16
Fuel Type					
Palmetto-Gallberry	Count	28		3682	
Grass	Count	5		1382	
Pine	Count	8		888	
Hardwood (Leafy)	Count	0		353	
Swamp/Muck/Other	Count	12		944	

(continued)

Table 5.1. Select descriptive statistics—for each variable the mean is shown with the exception of the categorical variables (mode) and the 0/1 variables (frequency). *(continued)*

		Wildfire			
Variable	Units	Large		Small	
		Statistic	*SE*	*Statistic*	*SE*
Own Section Landcover					
Grass	Percent	8.4	1.56	8.1	0.12
Upland Forest	Percent	37.0	4.10	35.9	0.29
Urban	Percent	3.7	1.30	15.4	0.26
Water	Percent	1.6	0.48	1.3	0.04
Wetland	Percent	22.8	2.53	17.3	0.18
Neighboring Sections Landcover					
Grass	Percent	8.6	1.09	8.1	0.10
Upland Forest	Percent	37.7	3.33	34.6	0.23
Urban	Percent	3.9	0.85	14.4	0.21
Water	Percent	1.7	0.38	1.6	0.03
Wetland	0-100 %	22.4	1.67	18.6	
	N	53		7249	

same area, year after year? Genton et al. (2006) analyzed the spatio-temporal distribution of the wildfire ignitions (using the same FDOF wildfire data), as a spatial-point process, and found that the degree of spatial clustering varied by year and by cause. They did not examine, however, differences in the spatial structure between small and large wildfires, meaning they did not examine how the spatial clustering was different between small and large fires. Figure 5.2 depicts each fire's location by cause (accident, arson, and lightning) and size for 53 large wildfires that occurred between 1996 and 2001. The majority of large fires were clustered along the coastline, where lightning fires appear to dominate. There were fewer large fires, regardless of cause, farther inland.

The spatial distribution of small wildfires is presented in figure 5.3, which depicts the location and cause of more than 7,000 small wildfires in our analysis. Small lightning fires were clustered along the coastline, similar to their larger counterparts, whereas accidental fires appeared mostly in the interior of the SJRWMD. Although not explicit in the figure, arson ignitions appeared to follow major roadways (especially the I-95, I-10, and I-4 corridors). Unlike large wildfires, small fires were fairly well distributed across the SJRWMD landscape, with a couple of notable exceptions. Areas without wildfires include the St. Johns River, which runs from the Jacksonville area southward to Lake George and that borders another notable void in the figure, the Ocala National Forest (federal data not included in the FDOF dataset), found in the middle of the SJRWMD area. Also, note the Intracoastal Waterway edging the coastline.

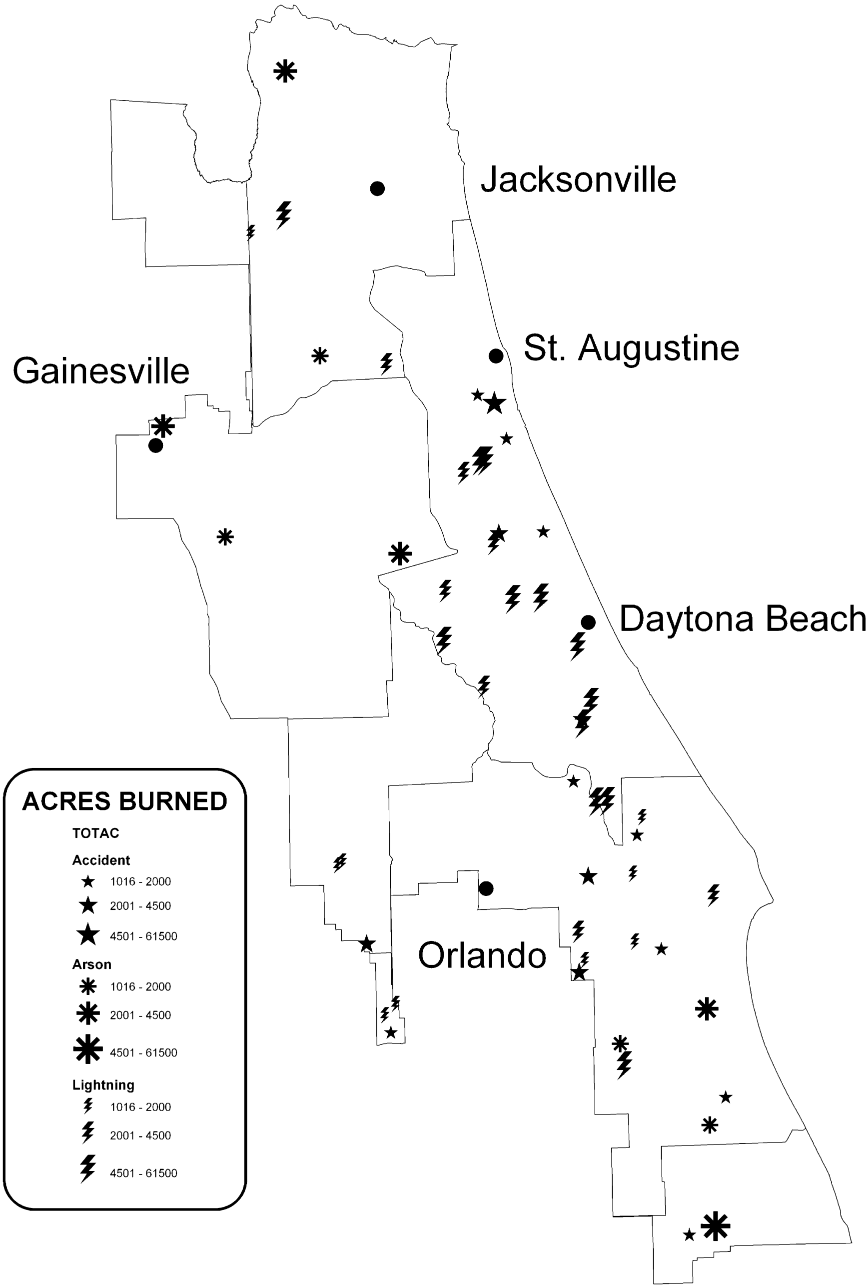

Figure 5.2. Spatial distribution of large wildfires (those fires greater than 1,000 acres) by cause by size from 1996-2001.

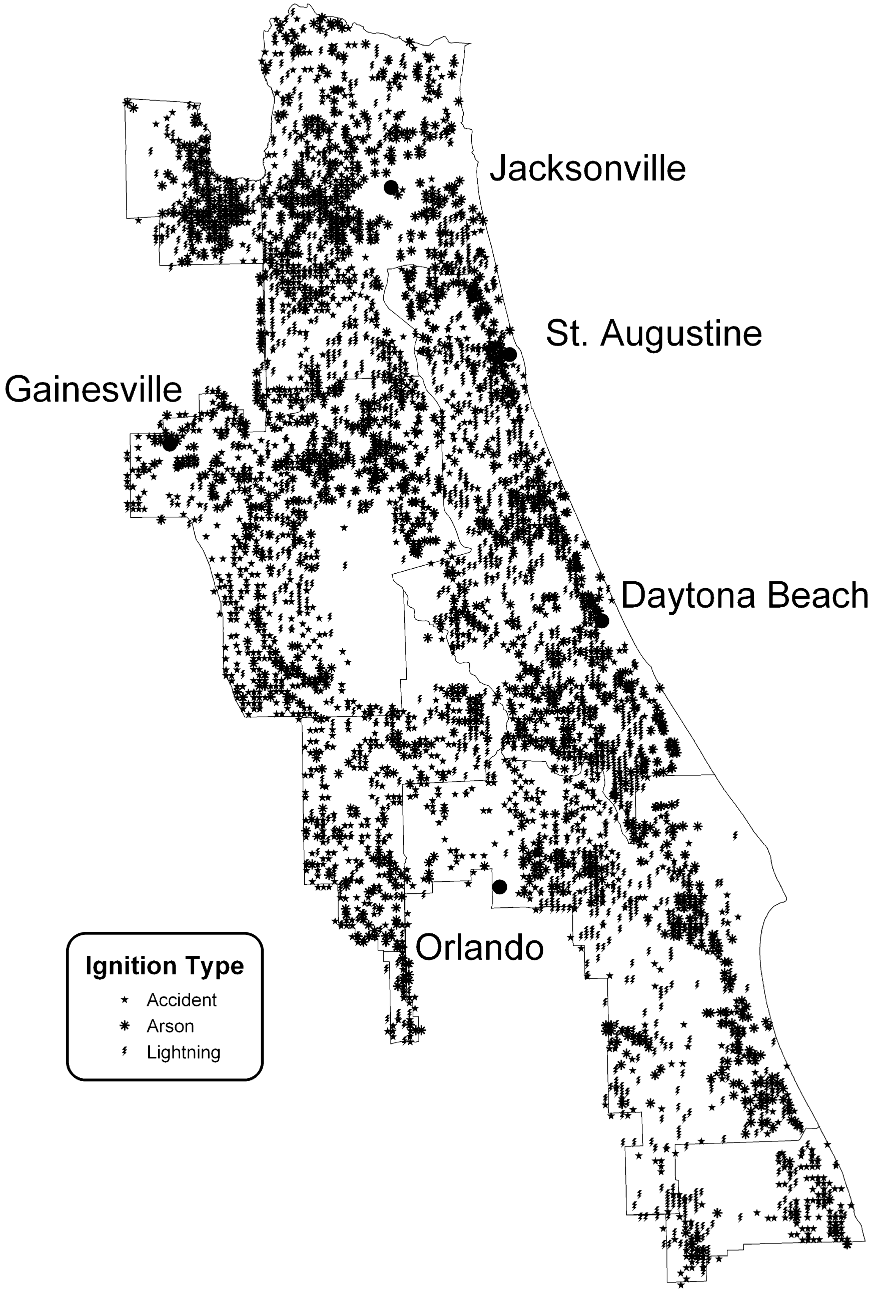

Figure 5.3. Spatial distribution of small wildfires (those less than or equal to 1,000 acres) by cause from 1996-2001.

4. MODELS

Three empirical models are estimated—two estimating the wildfire final size and one estimating the probability that a small wildfire (a wildfire less than or equal to a thousand acres) will become large (a wildfire greater than a thousand acres). Two wildfire size models are used to assess statistical differences between small fires and larger, more catastrophic fires. If there are differences, this implies that large, catastrophic wildfires are not simply big, small fires. Rather, differences might imply that large wildfires respond to different factors (and mitigation strategies) than smaller fires.

4.1 Wildfire Size Models

Wildfire size is modeled as a semi-log function specified as:

$$w = \alpha + \boldsymbol{X}^F\beta^F + \boldsymbol{X}^C\beta^C + \boldsymbol{X}^M\beta^M + \boldsymbol{X}^S\beta^S + \boldsymbol{Z}\gamma + \varepsilon \tag{5.1}$$

where w is a (Nx1) vector of the natural log of wildfire size, α is a constant term, $\boldsymbol{X}^F$ is a (Nxk_1) matrix of k_1 wildfire characteristics, β^F is a (k_1x1) vector of parameters for the wildfire characteristics, $\boldsymbol{X}^C$ is a (Nxk_2) matrix of k_2 climate and weather variables, β^C is a (k_2x1) vector of parameters for the climate and weather variables, $\boldsymbol{X}^M$ is a (Nxk_3) matrix of management variables, β^M is a (k_3x1) vector of parameters for the management variables, $\boldsymbol{X}^S$ is a (Nxk_4) matrix of section attributes, β^S is a (k_4x1) vector of parameters for the section attributes, $\boldsymbol{Z}$ is a (Nxk_5) matrix of variables specifying amount of previous wildfire in the same section or a neighboring section, γ is a (K_5x1) vector of parameters for the previous wildfire, and ε is a (Nx1) i.i.d. error vector. There are k parameters to be estimated ($k = k_1 + k_2 + k_3 + k_4 + k_5 + 1$).

The complete menu of exogenous variables includes:

Fire Characteristics ($\boldsymbol{X}^F$): start time (morning, afternoon, evening, overnight), start year, and cause (arson, accident, and lightning).

Climate/Weather ($\boldsymbol{X}^C$): Niño3 SST anomaly (La Niña and El Niño phase), KBDI, humidity, wind speed, wind direction, KBDI interactions (with kilometers of road, wind speed, buildup, La Niña, El Niño, response time, amount of upland forest, wetland forest, water, grass, and urban in the section, and all prescribed burning variables), and wind speed-buildup interaction. Second-order effects allowed for wind speed.

Management/Mitigation ($\boldsymbol{X}^M$): Response time, limited action fires (let burn), prescribed fire in section and neighboring sections including lags, response time interactions with all prescribed burning variables. Second-order effects allowed for response time.

Section Characteristics ($\boldsymbol{X}^S$): Population density, income, percent of population who have attended college, amount of road, distance to nearest fire department, percent of landscape and neighboring landscape in grass, upland forest, urban, water, and wetland forest, ignition fuel type (palmetto-gallberry, grass, pine, hardwood, swamp/muck/other), latitude, longitude, buildup, spread index, fire district, amount of previous wildfire in section and neighboring sections including lags, whether the section resided within a GIS "hole"[1], GIS "hole" and 1998 year interaction, GIS "hole" and wetland forest interaction, and GIS "hole" and water interaction. Second-order effects allowed for amount of road, distance to fire department, latitude, longitude, population density, income, and percent of population who have attended college.

The Niño3 SST anomaly variable is separated into two variables, La Niña and El Niño. The La Niña variable equals the Niño3 SST anomaly when it is negative (zero otherwise). Conversely, the El Niño variable equals the Niño3 SST anomaly when it is positive (zero otherwise). This allows us to examine the relationship between these two phases with wildfire size separately. We include the location of the fire (the latitude and longitude of the Public Land Survey section centroid), thereby allowing for spatial variation in wildfire size across the landscape not controlled by the other variables included in the model, as well as year dummy variables (1996 is included in the intercept) and start time (morning, afternoon, evening dummy variables; overnight is included in the intercept). In addition, we use the natural log of population density and income.

Because previous values of wildfire and prescribed burning appear to influence the wildfire size (Prestemon et al. 2002), we include total wildfire acres burned for the previous 12 years (we also include previous wildfire occurring in the same year as the current wildfire, but before the ignition date). For Florida, prescribed burn treatments are thought effective for around three years (Brose and Wade 2002, Outcalt and Wade 2004). Because prescribed burning is performed for several reasons and not all pertain to wildfire reduction, we include two different measures: hazard reduction and other (all non-hazard reducing prescribed burning)

[1] Originally, a Public Land Survey section (PLSS) GIS was obtained from FDOF and spatially matched with wildfire records to various explanatory variables. However, upon further inspection of the GIS it was revealed that there were several wildfires that did not have a match on the GIS (there was not a PLSS id with the same id). While only a relatively small number of wildfires could not be matched, these wildfires accounted for 37 percent of all wildfire acres burned. A new GIS was assembled (North Carolina State University Center for Earth Observation 2002) that is able to locate 98 percent of the ignitions and acres burned. In modeling wildfire size, we include as an explanatory variable a dummy variable that identifies those wildfires that did not have a match in the original GIS. The majority of these wildfires resided in section that are surrounded by or adjacent to water, thus we believe that perhaps these sections may be periodically inundated with water. In Mercer et al. (2000) it was found that many of the large wildfires of 1998 occurred in cypress swamps, areas normally surrounded by water (potentially limiting fire spread), however in 1998, severe drought conditions removed many of the normal wet areas. Thus, we hypothesize fires beginning in one of these "holes" (missing in the original GIS) will become large due to lack of constraints.

prescribed burning. We use two measures of hazard reducing prescribed burning—hazard reducing prescribed burning acres in the current year of the fire (but before ignition) and hazard reducing prescribed burning acres from the previous three years—that are calculated for the same section as the wildfire and for the neighboring section. One measure of other prescribed burning is used—all non-hazard reducing prescribed burning acres from the previous three years including the current year—for the section of the wildfire and the neighboring areas.

The model is made spatially explicit by incorporating latitude, longitude, and neighborhood-level information, including previous wildfire and previous prescribed burning by type in the neighboring cadastral sections. Neighboring sections are defined as those with a centroid distance no more than 2.8 kilometers from the section of reference. Each cadastral section is approximately one square mile with the layout of sections in a fairly regular lattice, so a neighborhood was defined as the eight surrounding sections (contiguous neighbors). Because the lattice is not exactly regular, sections are defined to be neighbors whose centers are no more than 2.8 kilometers apart (roughly 1.7 miles) to ensure that all contiguous neighbors are included.

The wildfire data includes records spanning back to 1981, the prescribed burning data does not exhibit complete (statewide) reporting until 1993 (only a few counties reported prescribed fire permits from 1989 through 1992), so because we include three years of lagged prescribed burning in the model, the analysis includes only those wildfires that occurred between 1996-2001. Two different wildfire size models are estimated based on equation (5.1)—one for small (≤1000 acres) wildfires and another for large (>1000 acres) wildfires.

4.2 Large Wildfire Probability Model

We estimate the probability that a fire will become a large wildfire once an ignition has occurred (conditional large fire probability) using logistic regression. The model is

$$\Pr[Y_i = 1] = \frac{e^{X_i\beta}}{1 + e^{X_i\beta}}, \tag{5.2}$$

where $Y_i = 1$ if the fire is large, $Y_i = 0$ if the fire remains small, $\boldsymbol{X}_i = [1, \boldsymbol{X}_i^F, \boldsymbol{X}_i^C,$ ${}^M_i, \boldsymbol{X}_i^S, \boldsymbol{Z}_i]$, $\beta = [\alpha, \beta^{Ft}, \beta^{Ct}, \beta^{Mt}, \beta^{St}, \gamma^t]^t$, and i indexes wildfire (the unit of observation). The variables contained in $\boldsymbol{X}_{\mathrm{i}}$ have been previously described.

The small wildfire size model and conditional large wildfire probability model are estimated using backward hierarchical selection, in which terms are dropped from the model if their significance level fails to reach 0.10. Potentially 100 explanatory variables can be included in the models, so the selection significance was set at 0.10 rather than a more customary 0.15 to keep the models parsimonious. The estimation starts by dropping the variable with the largest p-value. The model is re-run until all variables left achieve the 0.10 p-value level, the exception being those terms involved in a higher-order or interaction term. If the interaction term $A \times B$ is significant, then terms A and B must be included in the model regardless of their individual significances. Given the small number of observations in the large wildfire size model, backward selection cannot be used, so instead forward selection is used.

5. RESULTS

5.1 Wildfire Size Models

5.1.1 Small wildfire size model

We find statistically significant links between wildfire size and several exogenous variables (table 5.2). For continuous explanatory variables a positive coefficient indicates that the larger the exogenous variable, the larger the expected wildfire size; for qualitative factors a positive coefficient indicates that the category is associated with a larger wildfire size than a specified reference category. The following variables had significant positive coefficients (10 percent level): fire spread index, limited-action fires (those fires allowed to burn), palmetto-gallberry, grass, and pine fuel types (as opposed to swamp/muck), arson ignitions (as opposed to lightning ignitions), afternoon ignitions (as opposed to overnight ignitions), amount of wildfire in the neighboring sections lagged 1-12 years, and the amount of same section non-hazard-reducing prescribed burning lagged up to 3 years.

We would expect the fire spread index, limited-action fires, and fuel types (as opposed to swamp/muck) to be positively related to wildfire size. We had no prior expectation as to the sign of arson, amount of previous wildfire earlier (reduces fuel, yet proxies a higher probability of ignition), and non-hazard-reducing prescribed burning. Should arson fires be bigger than or smaller than lightning fires? It seems possible that lightning fires are more likely to occur in forested areas far removed from populated regions, thus they have the potential to grow before they are detected. However, lightning strikes are not targeted like arson ignitions are—the arsonist chooses the ignition point (chapter 7, Wildland Arson Management). It seems reasonable that an intentional fire setter would choose areas with a high probability of a successful ignition and for the ignition to become a larger fire. Analysis of the FDOF dataset reveals that the average size of arson fires is smaller than lightning fires; however, the partial effect of arson ignition is larger than that of lightning ignition if we adjust to common values of all other exogenous variables.

The area burned by previous wildfires (1-12 years previous) in the same section perhaps proxies for relative probability of ignition in that section that year. Non-hazard reducing prescribed burning is also correlated with increased wildfire size. While one might surmise that any prescribed burning might reduce the probability of ignition (because fuels material is removed), we find the opposite result.

La Niña, humidity, fire district 10, 14, and 16 (as opposed to district 6), years 1999-2001 (as opposed to 1996), amount of current year hazard reducing prescribed burning in the section, percent of water and wetland in the section,

Table 5.2. Small and large wildfire area model estimates. *Calm wind is the base case for the small wildfire model; however, there are no occurrences of calm winds and large wildfires, so variable wind becomes the base case for the large wildfire model.

	Small Wildfires			Large Wildfires		
Variable	Coefficient	S.E.	P-Value	Coefficient	S.E.	P-Value
Intercept (α)	-1.6879	13.8388	0.9029	0.9994	3.2785	0.7622
Fire Characteristics (β^F)						
Ignition Year (base=1996)						
1997	-0.0086	0.0926	0.9264			
1998	-0.1151	0.0869	0.1850	0.4986	0.2144	0.0256
1999	-0.3028	0.0881	0.0006			
2000	-0.5222	0.0807	<0.0001			
2001	-0.4821	0.0897	<0.0001			
Ignition Time of Day (base=night)						
Morning	0.0924	0.1317	0.4830			
Afternoon	0.2688	0.1193	0.0243			
Evening	-0.1156	0.1250	0.3551			
Fire Cause (base=lightning)						
Arson	0.1955	0.0665	0.0033	-0.6600	0.2487	0.0116
Accident	-0.0758	0.0624	0.2243	-0.5859	0.2183	0.0108
Latitude	-0.0383	0.0168	0.0222			
(Latitude)^2	2.9E-5	1.4E-5	0.0394			
Longitude	0.0538	0.0409	0.1883			
(Longitude)^2	-4.0E-5	3.2E-5	0.1600			
Climate/Weather (β^C)						
KBDI	-0.0026	0.0005	<0.0001			
KBDI*Roads	2.3E-4	1.2E-4	0.0458			
KBDI*Wind Speed	4.3E-4	1.5E-4	0.0036			
KBDI*Upland Forest	1.8E-5	6.3E-6	0.0044			
KBDI*Grass	2.9E-5	1.2E-5	0.0196			

(continued)

Table 5.2. Small and large wildfire area model estimates. *Calm wind is the base case for the small wildfire model; however, there are no occurrences of calm winds and large wildfires, so variable wind becomes the base case for the large wildfire model. *(continued)*

	Small Wildfires			Large Wildfires		
Variable	Coefficient	S.E.	P-Value	Coefficient	S.E.	P-Value
KBDI*Urban	1.4E-5	7.5E-6	0.0585			
KBDI*LN(Neigh. Haz. PB Lags 1-3)	-7.0E-5	3.8E-5	0.0706			
La Niña	-0.2942	0.0975	0.0026			
Humidity	-0.0069	0.0018	<0.0001			
LN(Wind Speed)	-0.0949	0.0710	0.1817			
Management/Mitigation (β^M)						
LN(Response Time)	0.5845	0.0383	<0.0001			
$(LN(Response Time))^2$	-0.1050	0.0130	<0.0001			
Let Burn	0.7875	0.1277	<0.0001			
Own Section PB						
LN(Hazard Lag 0)	-0.0504	0.0216	0.0200			
LN(Other Lags 0-3)	0.0221	0.0130	0.0902			
Neighboring Sections PB						
LN(Hazard Lags 1-3)	0.0181	0.0177	0.3066			
Section Characteristics (β^S)						
LN(Population Density)	-0.1495	0.0445	0.0008	0.1834	0.0755	0.0201
$(LN(Population Density))^2$	0.0175	0.0069	0.0108			
LN(Income)				0.5421	0.3202	0.0989
Roads	-0.3003	0.0555	<0.0001			
GIS 'Hole'				-0.0779	0.5066	0.8786
GIS 'Hole'*Own Water				0.3407	0.0948	0.0009
Spread Index	0.0088	0.0022	<0.0001	0.0269	0.0069	0.0004
Previous Wildfire						
Own Wildfire Lag 0						
LN(Own Wildfire Lag 0)	0.0297	0.0132	0.0251	0.0847	0.0331	0.0147

(continued)

Table 5.2. Small and large wildfire area model estimates. *Calm wind is the base case for the small wildfire model; however, there are no occurrences of calm winds and large wildfires, so variable wind becomes the base case for the large wildfire model. *(continued)*

	Small Wildfires			Large Wildfires		
Variable	Coefficient	S.E.	P-Value	Coefficient	S.E.	P-Value
Fire District (base=District 6)						
District 7	-0.1514	0.1850	0.4129			
District 8	0.2283	0.2224	0.3048			
District 10	-0.4080	0.2425	0.0925			
District 11	-0.2918	0.2792	0.2959			
District 12	0.0184	0.2940	0.9502	-0.6088	0.2213	0.0091
District 14	-0.8253	0.4173	0.0480			
District 16	-1.0766	0.4090	0.0085			
Fuel Type (base=swamp/muck)						
Palmetto-Gallberry	0.7339	0.7060	<0.0001	0.0783	0.2300	0.7356
Grass	0.4927	0.0796	<0.0001	-0.7734	0.3868	0.0530
Pine	0.9362	0.0868	<0.0001	-0.5475	0.3351	0.1107
Hardwood (Leafy)	0.1137	0.1182	0.3363			
Own Section Landcover						
Grass	-0.0120	0.0061	0.0492			
Upland Forest	-0.0105	0.0032	0.0012			
Urban	-0.0177	0.0036	<0.0001			
Water	-0.0152	0.0069	0.0277	0.0255	0.0316	0.4249
Wetland	-0.0037	0.0019	0.0582	0.0236	0.0054	<0.0001
Neighboring Sections Landcover						
Grass	-0.0126	0.0046	0.0063			
Upland Forest	-0.0057	0.0023	0.0137			
F Value	15.26		0.0003	6.81		<0.0001
R^2	0.16			0.73		
Number of Observations	7249			53		

and percent of grass and upland forest in the neighboring sections are all significantly negatively (10 percent level) related to wildfire area. We would expect humidity, La Niña, hazard reducing prescribed burning, and percent water in the section to be negatively related to fire size. As mentioned earlier, La Niña has been found to be positively correlated with fire in previous studies (Prestemon et al. 2002), which is what we find here. Hazard-reducing prescribed burning is targeted to areas for the express reason to reduce wildfire probability. The more a section is composed of water, the less burnable material is present. We have no prior expectations for the effects of fire districts or years on wildfire size.

A number of variables exhibit non-linear relationships with natural log of wildfire area. Response time, latitude, longitude, and population density all exhibit second-order effects. Increases in response time correspond with increases in wildfire size, up to 16 hours, where then it corresponds with decreases. Approximately 93 percent of all small fires are responded to within 16 hours. Increases in population density correspond with decreases in wildfire size, up to 71 people per square kilometer, where then it corresponds with increases in fire size. About three-quarters of all small wildfires occur in areas with population density less than 71. Population has at least two (opposite) influences on wildfires, one as an ignition source (arson and accidental ignitions), and two, as a source of fire detection. Also related to the second, with larger population we would expect greater fire fighting resources and capability. Wildfire size decreases going north, all else being equal, up to latitude 4 kilometers north of St. Augustine, beyond which it increases. Wildfire size increases going east, all else being equal, up to a longitude 13 kilometers west of Daytona Beach, beyond which it decreases.

Several statistically significant interactions exist between KBDI and other variables: roads, wind speed, percent of the section that is upland forest, grassland, and urban and hazard reducing prescribed burning acres from the previous three years in neighboring sections. Evaluating these variables at their means, we find that increases in KBDI reduce the expected size of wildfire. We expected KBDI to exhibit a positive relationship with wildfire size, which it does not at the means of the other interaction terms. However, a positive relationship does exist between KBDI and wildfire size for different combinations of the interaction terms. For example, if wind speed is set somewhere above its mean (with everything else held at its mean), then wildfire size increases with KBDI. For wind speeds at or above 14 mph, KBDI and wildfire size are positively related.

With KBDI set at its observed mean, wildfire size increases as wind speed and percent of the section in grass increases, whereas increased amounts of road, either upland forest or urban area in a section, and amount of neighboring hazard-reducing prescribed burning from the previous three years are negatively related with wildfire size. Figure 5.4 shows how the marginal effect of the natural log of prescribed burning on wildfire size changes for different levels of KBDI. Under medium-to-high drought conditions (KBDI>259), previous prescribed fire in the neighbors is correlated with smaller wildfire size, and the magnitude of this relationship increases as the drought index increases.

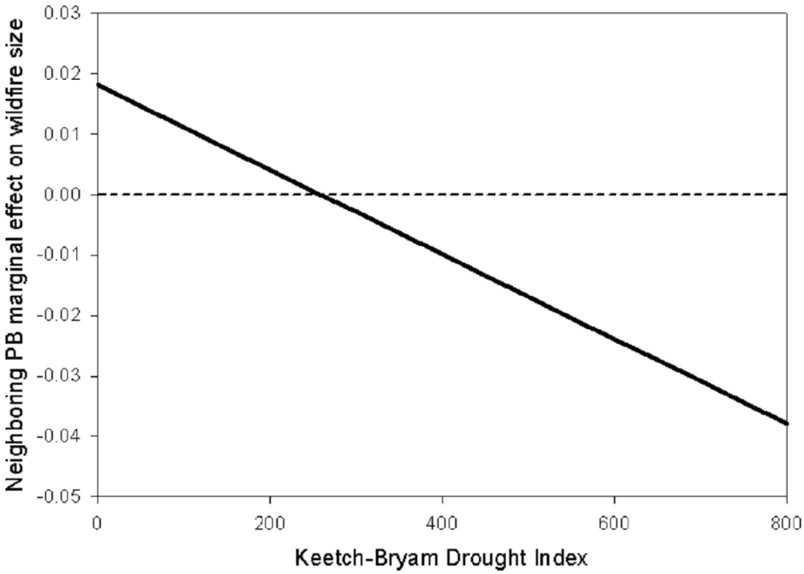

Figure 5.4. Marginal effect of small wildfire acre from an acre change in neighboring section prescribed burning, lagged 1-3 years, conditioned on KBDI.

Although we have found many significant predictor variables, a large proportion of the total variance in wildfire size remains unexplained. The model accounts for only 16 percent of wildfire size variation (table 5.2). We examine the residuals for the presence of spatial dependence using both regular and robust semivariograms of the residuals, for all years combined and then by year. When all years are included in one semivariogram, spatial dependency appears to exist. Pairs of neighbors that are, on average, no farther than 10 kilometers apart (lag distances are in meters) appear to be correlated. When the residuals are examined by year, it appears that all years, except 1999-2001, exhibit a white noise process, or no spatial dependence.

5.1.2 Large wildfire area model

We estimate the model of equation (5.1), but restrict the data to only large wildfires (those greater than 1,000 acres). The model explains 73 percent of the variation in large wildfire size and is highly significant (table 5.2). We find that population, spread index, 1998, percent of section in wetland forest, and amount of previous wildfire (1-12 years prior) have a significantly positive (to the 10 percent level) correlation with the size of large wildfire.

The fire spread index is a measure of potential fire spread, thus it makes sense that it should be positively correlated with larger fires. The 1998 wildfire season was quite notable for many large fires occurring during a six-week period in June and July, with 32 of the 53 large fires in our analysis beginning in 1998.

Grassland, fire district 12, and arson and accidental ignitions are found to have a significantly negative (to the 10 percent level) effect on the size of large wildfires. In addition, there are no instances of large wildfires beginning in hardwoods or occurring with calm winds. Grassland, fire district 12, and arson and accidental ignitions entered the model all as dummy variables. For grassland, the base case is swamp/muck/other fuel types. The base case for fire district 12 is all other fire districts in the SJRWMD, and the base case for arson and accidental ignitions is lightning ignitions.

There is an interaction between GIS "hole" and amount of water in the section that is statistically significant. Mercer et al. (2000) contends that 1998 was such a catastrophic year because areas that are usually under some standing water were arid due to the hot and dry conditions, thus increasing the potential wildfire connectivity and intensity across the landscape. Hence, the GIS "hole" would no longer be wet, and would no longer act as a natural firebreak. This coupled with the high fuel loads in these areas imply that GIS "holes" should be positively related to fire size.

A semivariogram analysis, like that discussed in the previous section, showed no spatial correlation among the residuals after fitting the regression model. All years were combined for the semivariogram analysis because there were only 53 large wildfires in all years combined.

5.1.3 Conditional large wildfire probability model

We use a backwards hierarchical logistic model (again, using a significance level of 0.10) to estimate the probability that a small wildfire will become large. The model explains 32 percent (pseudo R^2 from SAS Proc Logistic) of the variation of large versus small fire and is highly significant (table 5.3).

We find that the La Niña, natural log of income, fuels buildup, limited action fires, wind speed, percent of neighboring section in upland forest, and 1998 are significantly (10 percent level) positively related to the probability of a wildfire becoming large. We expect that increases in La Niña (linked to fire weather), fuels buildup, limited action fires (fires are allowed to burn), wind speed (quicker boundary spread), and 1998 increase the likelihood of a large wildfire. We have no prior expectations for income or upland forest in neighboring sections.

El Niño, latitude, and percent of urban areas in neighboring sections are statistically significantly (10 percent level) negatively associated with large wildfire probability. We also find an interaction between KBDI and percent of upland forest in the section, where KBDI exhibits a positive relationship with the probability of a large wildfire for percent of upland forest values in excess of 24 percent (note: the mean is 36 percent). Thus, areas with at least $1/4^{th}$ of the land cover in upland forest experience higher probabilities of large, catastrophic fire probability, given an ignition, when KBDI rises. About half of all sections in the SJRWMD are covered by at least 24 percent upland forest. We also find an interaction between fire crew response time and current year hazard mitigating

Table 5.3. Conditional large wildfire probability model estimates. *Standardized coefficients are calculated in SAS as $\hat{\beta}_i / (s / s_i)$ where $\hat{\beta}_i$ is the estimated coefficient of the ith explanatory variable, s_i is the ith explanatory variable's sample standard deviation, and s is $\pi / \sqrt{3}$ when computing the standardized coefficient for a logistic regression. They are not computed for the intercept orfor the interaction terms.

Variable	Coefficient	S.E.	P-Value	Odds Ratio	Standardized Coefficients*
Intercept	-12.1708	5.4839	0.0265		
Fire Characteristics					
1998	1.4617	0.3824	0.0001	4.3130	0.3199
Latitude	-0.0086	0.0025	0.0006	0.9910	-0.4066
Climate/Weather					
KBDI	-0.0032	0.0015	0.0298		
KBDI*Upland Forest	1.36E-4	4.6E-5	0.0031		
La Niña	5.7364	1.5054	0.0001	309.9360	0.9071
El Niño	-6.7036	2.9025	0.0209	0.0010	-2.7508
Wind Speed	0.0644	0.0181	0.0004	1.0660	0.1748
Management/Mitigation					
LN(Response Time)	0.3870	0.2532	0.1264		0.2288
LN(Response Time) *LN(Own Haz. PB)	0.2165	0.1124	0.0540		
LN(Response Time) *LN(Neigh. Haz. PB)	-0.1686	0.0680	0.0132		
Let Burn	1.9797	0.4784	<0.0001	7.2400	0.1890
Own Sections PB					
LN(Hazard Reduction Lag 0)	-0.1227	0.2700	0.6495		-0.0681
Neighboring Section PB					
LN(Hazard Reduction Lag 0)	0.0437	0.0763	0.5668		0.0512
Section Characteristics					
LN(Income)	1.2835	0.5026	0.0107	3.6090	0.7859
Buildup	0.0092	0.0035	0.0092	1.0090	0.1716
Own Section Landcover					
Upland Forest	-0.0958	0.0307	0.0018		-1.2892
Neighboring Sections Landcover					
Upland Forest	0.0316	0.0148	0.0331	1.0320	0.3493
Urban	-0.0746	0.0225	0.0009	0.9280	-0.7427
Likelihood Ratio (Chi-Square)	194.0171				
Pseudo R-Square	0.3183				

prescribed burning (in the same section and neighboring sections). Holding the two prescribed burning variables at their means, we find that the probability of fire becoming large increases with firecrew response time (fig. 5.5). If the response time is short, prescribed burning is negatively correlated with probability of a large fire, but if the response time is longer than about an hour and 20 minutes, prescribed burning in the section of ignition has no effect (fig. 5.6). On the other hand, if the response time is long then the probability of a large fire is negatively correlated with the amount of prescribed burning in sections adjoining the section of ignition (fig. 5.7).

We also report the odds ratio and the standardized coefficients (beta weights). The standardized coefficients imply that a one-standard deviation change in an exogenous variable is associated with a one-standard deviation change in the log-odds of the response variable multiplied by the standardized coefficient. The odds ratio describes the effect of a one-unit change in the odds of a large fire. For instance, if La Niña decreases by one-unit, then the expected odds of a large wildfire, given an ignition, increases by 310. Note that while the change in odds from a one-unit change in La Niña is large, the range of La Niña in the data is 0 to -1.61 with the mean being -0.29.

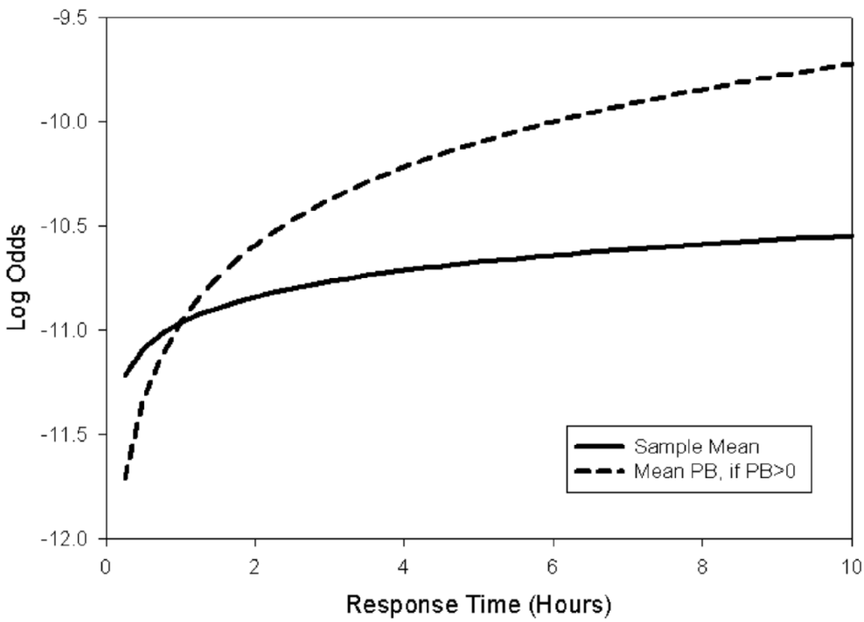

Figure 5.5. Response time versus log odds of catastrophic wildfire (from conditional large wildfire probability model) with (1) all explanatory variables set to their means, then (2) all explanatory variables set to their means except their own and neighboring section hazard mitigating prescribed burning, which is set to their means when there has been a prescribed burn (i.e., conditioned on PB>0).

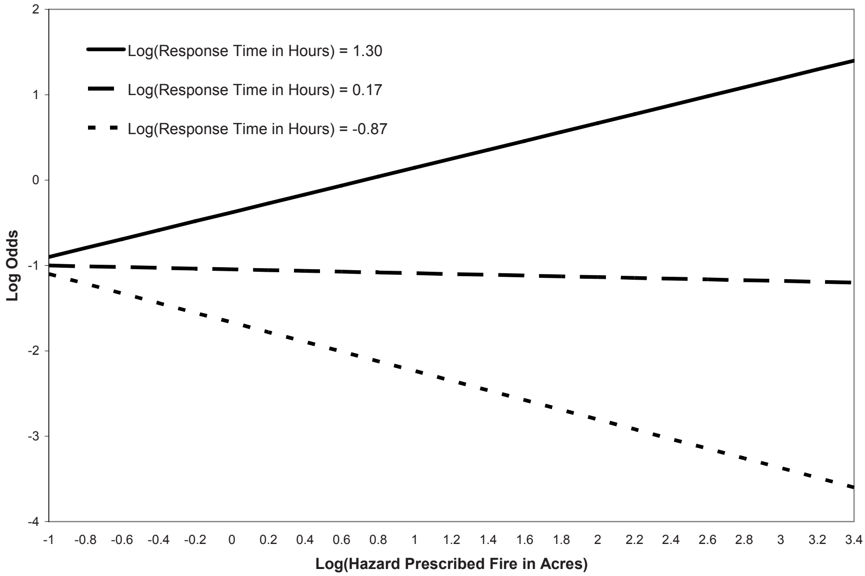

Figure 5.6. Predicted log odds of a large fire versus hazard reducing prescribed fire, varying firecrew response time.

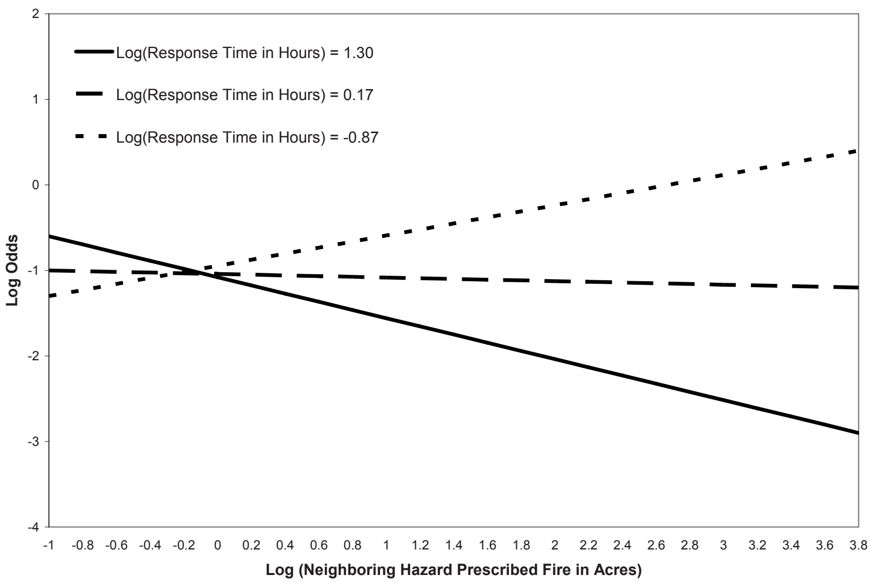

Figure 5.7. Predicted log odds of a large fire versus hazard reducing prescribed fire in the neighborhood, varying firecrew response time.

6. DISCUSSION

6.1 Does Spatial Information Enhance Modeling?

Incorporating spatial information into the wildfire models adds significant information and increases the amount of explained variation in wildfire size. For instance, removing some of the spatial variables (latitude, longitude, fire district, and neighborhood measures) reduces the amount of the explained variation of small wildfire size by 21 percent; removing the spatial variables (GIS "hole" and its interaction term) reduces the explained variation of large wildfire size by 11 percent.

6.2 What Does Fine Scale Modeling Add?

We use wildfire occurrence as the unit of observation, rather than an aggregated measure of wildfire (e.g., annual county or state level), which allows a finer exploration of the relationships between wildfire and others variables than at a coarser aggregated level. At a fine spatio-temporal scale, we find evidence that a wide range of factors matter, including fire specific characteristics, climate and weather conditions, management decisions, and landscape composition. We find strong empirical support for hazard reducing prescribed burning as mitigating wildfire size that occurs in the same section as the wildfire, at least when fires stay small, whereas at broader scales the evidence was shown to be weak (Prestemon et al. 2002).

6.3 Do Small and Large Wildfire Differ?

Our models suggest that small and large wildfires are truly different processes, related to a different set of factors. Interestingly, the two models have very few significant variables in common. If we regress small wildfire size on the set of exogenous factors found significant in the large model, they explain less than 1 percent of the variation in small wildfire size. It does not appear that large wildfires are simply small wildfires, only bigger, but something fundamentally different. This suggests that techniques used to mitigate small wildfires may not be appropriate for large wildfires.

6.4 What are Possible Management Implications?

Wildland management (as defined in this analysis) appears to have the greatest effect on reducing the likelihood that fires will become large (1,000 acres or more), and for those fires that stay small, management has the effect of mitigating final fire size (in acres). When fire crews cannot respond fast enough, perhaps when there are multiple fires, prescribed fire in surrounding areas limit ultimate fire size, thus retarding the probability that a fire will become large. In addition, prescribed fire was found to mitigate the effects of drought conditions on the probability of large fires. Keeping fires manageable is important, and unfortunately, we find no evidence that large wildfires respond to wildland management (again, as defined in this analysis). Instead, large fires appear sensitive only to weather and landscape conditions.

Ultimately, society may care less about fire size than fire-related damages. If acres burned by wildfire are closely related to wildfire-caused damages, then the above analysis provides insight into damage minimization and the role for fire management. However, if acres burned by wildfire are only loosely related to wildfire-caused damages, then the above analysis may underestimate the true effect wildfire management has on wildfire-caused damage.

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CHAPTER 6

CLIMATOLOGY FOR WILDFIRE MANAGEMENT

Anthony L. Westerling

1. INTRODUCTION

Forest and wildfire managers in the western United States are very familiar with weather information and forecasts provided by various public and private sources. Even very sophisticated users of these products, however, may be less familiar with climate information and forecasts and their applications. Partly this is because the scientific community has made rapid progress in climate, and particularly climate forecasting, as a field of applied study in recent decades. Integrating these new research findings into management systems is a difficult and time-consuming process. Sophisticated users of weather information may also be less comfortable using climate information because, conceptually, the two are so different: weather is something we all experience every day, while climate is an abstraction.

As an abstraction, however, climate provides powerful tools for understanding recent developments in forest wildfire in the western United States. Climate and wildfire research, and the practical experience of many forest managers, show that summer drought is a very important driver of interannual variability in forest wildfire (Balling et al. 1992, Swetnam and Betancourt 1998, Kipfmueller and Swetnam 2000, Veblen et al. 2000, Donnegan et al. 2001, Heyerdahl, Brubaker and Agee 2002, Westerling et al. 2003b). In turn, the duration and severity of summer drought in western forests is highly sensitive to variability in spring and summer temperature at higher latitudes and elevations and its effect on snow (Westerling et al. 2006). Trends in temperature and the timing of the spring snowmelt explain much of the dramatic increase in large forest wildfire frequency in the West in recent decades (Westerling et al. 2006), and these trends in wildfire are probably driving most of a similar increase in fire suppression costs.

Recent research has demonstrated the feasibility of producing seasonal forecasts of temperature, drought, and wildfire activity in the western United States (Alfaro et al. 2005a, Alfaro et al. 2005b, Westerling et al. 2002, Westerling et al. 2003a). Based on surveys of fire managers in California and the Southwest (conducted by Corringham, Westerling and Morehouse, in press), these forecasts may be useful for planning wildfire suppression budgets, allocating resources within the annual cycle of fire seasons in different parts of the United States, and prioritizing fuels management projects. Obstacles remain however, from

T. P. Holmes et al. (eds.), The Economics of Forest Disturbances: Wildfires, Storms, and Invasive Species, 107–122.

mismatches between the timing of decision-making and the time horizon of skillful forecasts, to institutional constraints on managers' ability to use climate information and forecasts for wildfire and fuels management.

This chapter has three goals. First, to define what climate, as opposed to weather, is, and to explain what this implies for climate versus weather forecasts. Second, to describe the scientific community's current understanding of the relationships between climate variability and forest wildfire in the western United States. And finally, to demonstrate a forecast application that exploits these relationships, and their potential applications to the business of forest wildfire management.

2. CLIMATE VERSUS WEATHER

The best way to think about climate and weather is that climate is a process, and weather is an outcome of that process. That is, the Earth's climate is a system of interactions between the Sun's energy and the Earth's oceans, atmosphere and biosphere. Weather is the observed precipitation, temperature, wind, etc., that result from those interactions. Understanding the difference between climate and weather is important because it gives us insight into the limitations of what a description of climate (i.e., a *climatology*) or a climate forecast can provide for fire and forest management applications.

A climatology is often expressed as a statistical description of the average or *normal* outcomes (weather) of the climate system, such as the average rainfall or temperature to be expected in a given place and time of year, and the typical variability that has been observed around that average. This is different from a description of all the possible outcomes. We might have an idea about the range of possible outcomes based on a physical model or inferences drawn from observations, but a typical climatology is a statistical description of past weather observations available for a particular region. An example is the empirical history of standardized August temperature anomalies (i.e., deviations from the mean for that month) for 110 western U.S. climate divisions collected from 1895 to 2004 (fig. 6.1, vertical grey bars) (Karl and Knight 1985, NCDC 1994). These temperature anomalies are normally distributed, such that the observed probability of any given mean August temperature can be approximated by a smooth curve calculated from the mean and variance of the historical temperature anomalies (the black line in fig. 6.1).

In conceptual terms, a climate forecast is a forecast of what kind of climate process will be operating at some point in the future, and a weather forecast is a forecast of what outcomes should be expected from the climate processes operating at some point in the future. In practical terms, the greatest distinctions between weather and climate forecasts are their lead times, duration and their degree of uncertainty. A weather forecast makes a prediction about rainfall, precipitation, wind, etc., over relatively short time horizons: the next hour, day, or week. A climate forecast makes a prediction about these parameters over

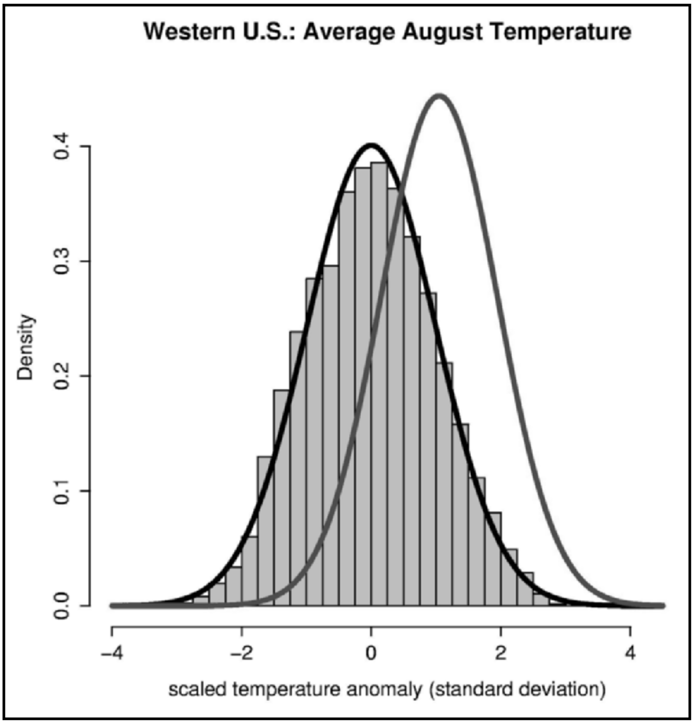

Figure 6.1. Scaled western U.S. August temperature anomalies (i.e., deviations from the mean for that month) for 110 climate divisions from 1895-2004 are used to calculate an empirical probability density (vertical grey bars) that is closely approximated by a normal distribution with mean = 0 and variance = 1 (left curve). The subset of August mean monthly temperatures following a July with mean temperature > 2 standard deviations above the mean for all July's is also normally distributed, with mean = 1 and variance = 0.8 (right curve). The grey line represents a forecast for August temperature conditional on observed July temperature being > 2 standard deviations above the July mean.

longer time horizons: over the next month, over a season starting a year from now, etc. Because a weather forecast deals with the near future, the range of likely outcomes for a weather forecast is narrower than for a climate forecast. This is because the conditions immediately preceding tomorrow's forecast (that is, today's observations) are well known, while the conditions immediately preceding next month's or next season's forecast are not yet observed. Since the climate is such a complex system, the longer the time between observations and prediction, the harder it is to confidently predict what will happen at any point in time. As a result, a weather forecast might specify the chance of rain tomorrow afternoon, while a climate forecast would more likely specify the chance that the cumulative total rainfall over the next month or season will be above or below normal.

A climate forecast usually provides a description of expected outcomes conditional on observations in the recent past. As an extreme case, a climatology can be thought of as a forecast of the future climate system based on all available past observations. Such a forecast might change gradually from year to year, as additional observations are incorporated into parameters like the mean and variance that describe some aspect of the climate system.

A more useful forecast would be one that uses past associations between observations and subsequent outcomes. For example, scientists have observed that after an El Niño develops—signified by warmer than average sea surface temperatures in the eastern equatorial Pacific—above average rainfall and temperatures have often subsequently been experienced in the U.S. Southwest (Dettinger et al. 1998, Gershunov et al. 1999). A climate forecast might describe the likelihood of these outcomes conditional on an El Niño signal having been observed (or not) in the Pacific. Going back to the western United States. August temperature example (fig. 6.1), a climate forecast might take advantage of persistence in temperature trends; warmer than average Julys tend to be followed by warmer than average Augusts. Looking at August temperature anomalies following very warm July temperatures (when the July temperatures were more than 2 standard deviations above the mean, i.e., in the top 2.3 percent of observed July monthly mean temperatures), mean August temperatures are warmer than average (1 standard deviation above the mean, i.e., in the top 15.9 percent of observed August monthly mean temperatures). The temperature forecast for August, conditional on the temperatures observed in July, can be represented by a curve that uses the mean and variance of the subset of August temperature anomalies that followed a very warm July to calculate the conditional probability of experiencing any given August temperature (fig. 6.1, grey line).

In terms of practical applications for fire management, weather forecasts are appropriate for supporting operational decisions during a fire season, since they deal directly with variables of interest to fire managers with short time horizons suitable to managing a fire. Climate forecasts are more appropriate for activities that precede a fire season or take place over a longer period, like budgeting and pre-positioning resources for fire suppression, undertaking pre-suppression activities to reduce the risk of wildfire ignition and spread, and prioritizing resources for fuels management projects.

3. CLIMATE AND FOREST WILDFIRE

3.1 Moisture, Fuel Availability, and Fuel Flammability

Climate affects wildfire risks primarily through its effects on moisture availability. Wet conditions during the growing season promote fuel—especially fine fuel—production via the growth of vegetation, while dry conditions during and prior to the fire season increase the flammability of the live and dead vegetation that fuels wildfires (Swetnam and Betancourt 1990, 1998, Veblen et al. 1999, 2000, Donnegan 2001, Westerling et al. 2003b). Moisture availability is a function of both cumulative precipitation and temperature. Warmer temperatures can reduce moisture availability via an increased potential for evapo-transpiration[1], a reduced snowpack, and an earlier snowmelt. In much of the West, three quarters or more of annual water year (i.e., October to September) precipitation occurs by the end of May (Westerling et al. 2003a). Snowpack at higher elevations is an important means of making part of winter precipitation available as runoff in late spring and early summer (Sheffield et al. 2004), and a reduced snowpack and earlier snowmelt consequently lead to a longer, drier summer fire season in many mountain forests (Westerling 2006).

For wildfire risks in most western forests, interannual variability in precipitation and temperature appear to have a greater effect on forest wildfire via their short-term effects on fuel flammability, as opposed to their longer-term affects on fuel production. By way of illustration, we show average Palmer Drought Severity Index (PDSI, NCDC 1994) values for the month of ignition and also for the March prior to the growing season one year earlier for 1166 large forest wildfires on USDA Forest Service (USFS) and USDI National Park Service (NPS) lands in the western United States, by elevation (fig. 6.2). PDSI is a meteorological drought

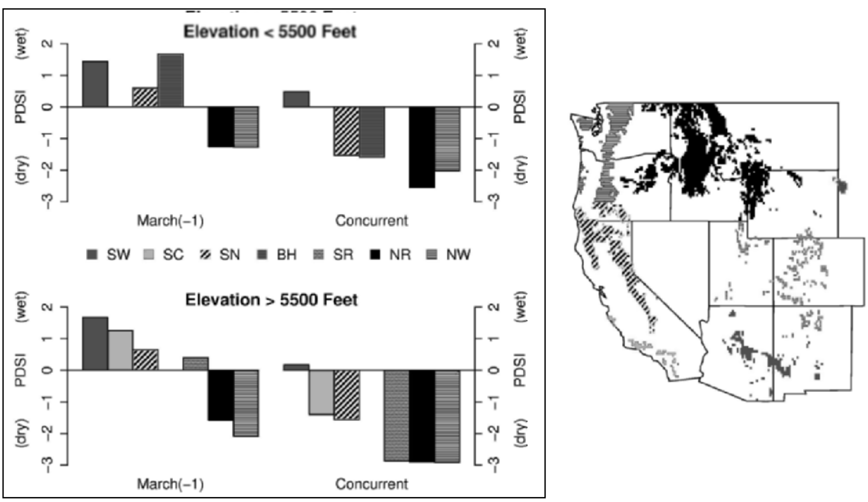

Figure 6.2. (*left*) Average Palmer Drought Severity Index values for the month a large forest wildfire started, and for the March a year or more before, by elevation for seven forest regions (*regions shown at right*). Regions are roughly ordered by mean latitude from south to north. BH had too few fires above 5500 feet, and SR too few below 5500 feet, to calculate representative mean PDSI values.

[1] I.e., evaporation from soils and surface water, and from vegetation.

index that uses precipitation and temperature anomalies to estimate the duration and severity of long-term drought (Alley 1984 and 1985). Positive values of the index represent wet conditions, and negative values represent dry conditions. We use it here as an indicator of the moisture available for the growth and wetting of fuels.

This analysis included all fires over 1000 acres that have burned since 1970 in land management units that have been reporting consistently since 1970 for the two agencies, and account for the majority of large forest wildfires west of 101°W Longitude in the contiguous United States (Westerling et al. 2006, online supplement) for a detailed description of this data set). The fires have been aggregated into seven regions: mountain ranges of Arizona and New Mexico excluding the Southern Rockies (SW), the mountains of coastal southern and central California (SC), the Sierra Nevada and southern Cascades and Coast Ranges (SN), the Black Hills of South Dakota and Wyoming (BH), the Central and Southern Rockies between 35.3 and 42° N (SR), the Northern and Central Rockies between 42 and 49° N Latitude (NR), and the Cascades and Coast Ranges above 43.1° N (NW) (fig. 6.2).

For all but SW, conditions were drier than average when large forest wildfires burned, with the magnitude of the drought increasing roughly with latitude and elevation (fig. 6.2). In the SW, the frequency of large wildfires peaks in June, ignited by monsoonal lightning strikes before the monsoon rains wet the fuels (Swetnam and Betancourt 1998). Since the lightning ignitions are associated with subsequent precipitation, it is possible that the monthly drought index may tend to appear to be somewhat wetter than conditions were at the time of ignition.

In the two northernmost regions—NR and NW—conditions also tended to be drier than normal in the preceding year: extended drought increased the risk of large forest wildfires in these wetter northern forests. In SW, SC, SN and SR, the preceding year is wetter than average. For fires above 5500 feet in elevation, the importance of surplus moisture in the preceding year was greatest for the southernmost regions. Swetnam and Betancourt (1998) found that moisture availability in antecedent growing seasons was important for fire risks in open pine forests in the SW where fine fuels play an important role in providing a continuous fuel cover for spreading wildfires, but not in mixed conifer forests. Looking at the western United States more broadly, the moisture necessary to support denser forest cover tends to increase with latitude and elevation. Consequently, the shift in figure 6.2 from wet to dry growing season conditions a year before a large fire as one moves from the forests of the SW to those of the NW is broadly consistent with a decreasing importance of fine fuel availability—and an increasing importance of fuel flammability—as limiting factors for wildfire as moisture availability increases on average.

3.2 Forest Wildfire and the Timing of Spring

There has been a dramatic increase in the incidence of large forest wildfire in the western United States since the early 1980s, with the number of fires increasing over 500 percent and area burned increasing nearly 760 percent (fig. 6.3, table 6.1). While the incidence of large forest wildfires has increased everywhere in the West, the change in the Northern Rockies has been extraordinary: an 1100 percent increase in large fires, and a 3500 percent increase in area burned (table 6.1). As a result, the NR accounts for a rising share of western forest wildfires—from under 28 percent before 1985 to over 55 percent subsequently—as well as a rise from 14 percent to 61 percent of area burned in western forests (fig. 6.3, table 6.1). With the NR accounting for 60 percent of the increase in western wildfires and 67 percent of the increase in area burned, understanding the factors behind the increase in NR forest wildfire activity is key to understanding the recent trends and interannual variability in western forest wildfire.

Westerling et al. (2006) note that the NR has a large forested area between about 5500 and 8500 feet in elevation where the length of the average season completely free of snow cover is relatively short (two to three months) and highly sensitive to variability in regional temperature, increasing roughly 30 percent in the earliest third of snowmelt years over the season length in the latest third of snowmelt years. They observed that in years with an early spring snowmelt, spring and early summer temperatures were higher than average, winter precipitation was below average, the dry soil moistures typical of summer in the region came sooner and were more intense, and vegetation was drier.

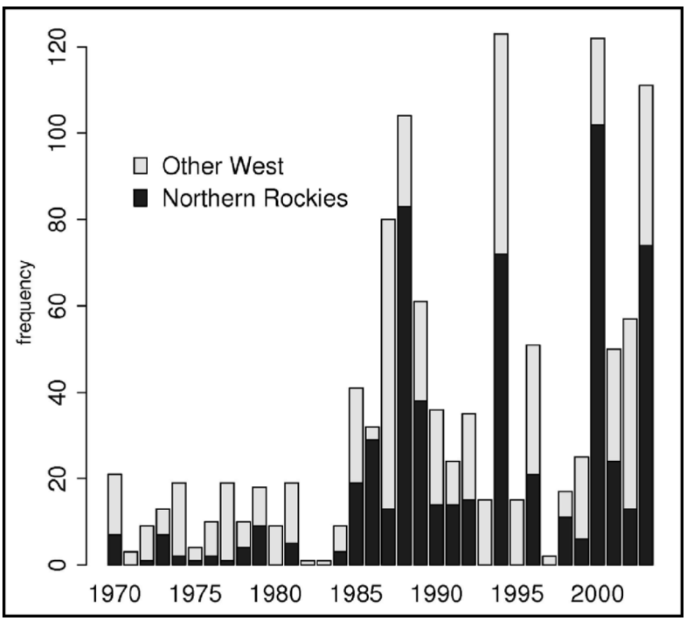

Figure 6.3. Annual number of forest fires greater than 1000 acres (total column height). Black area indicates the annual number of large fires in the Northern Rockies.

Table 6.1. Change in Large Fire Frequency and Area Burned, 1970–1984 versus 1985–2003.

	Change in Frequency	1970–1984 share of total	1985–2003 share of total	Change in Area Burned	1970–1984 share of total	1985–2003 share of total
NW	256% 1.84 (0.08)	10%	6%	558% 1.83 (0.08)	8%	6%
NR	**1100%** **3.64 (0.00)**	28%	55%	**3523%** **2.43 (0.03)**	14%	61%
BH	250% 1.76 (0.09)	4%	2%	42% 0.12 (0.90)	9%	2%
SN	**343%** **2.52 (0.02)**	24%	18%	**671%** **2.24 (0.04)**	22%	19%
SR	104% 1.30 (0.21)	12%	5%	464% 1.18 (0.25)	7%	5%
SC*	71% 0.65 (0.52)	4%	1%	-74% -1.23 (0.24)	27%	1%
SW	**354%** **3.52 (0.00)**	16%	12%	**371%** **2.70 (0.01)**	11%	6%
WEST	**507%** **4.68 (0.00)**	100%	100%	**759%** **3.22 (0.00)**	100%	100%

The statistics here are for only those wildfires greater than 1000 acres that burned primarily in forests, of which there were only 19 in SC since 1970. SC has experienced a number of large wildfires that ignited and primarily burned in chaparral, but spread to and burned substantial forested area, such as the Cedar fire in October 2003, that are not included here and might significantly affect the results for SC.

The consequences of an early spring for the NR fire season are profound. Comparing fire seasons for the earliest versus the latest third of years by snowmelt date, the length of the NR wildfire season (defined here as the time between the first report of a large fire ignition and last report of a large fire controlled) was 45 days (71 percent) longer for the earliest third than for the latest third. Sixty-six percent of large fires in NR occur in early snowmelt years, while only nine percent occur in late snowmelt years. Large NR wildfires in early snowmelt years, on average, burn 25 days (124 percent) longer than in late snowmelt years.

As a consequence, both the incidence of large fires and the costs of suppressing them in the NR are highly sensitive to spring and summer temperatures (figs. 6.4A and B). Both large fire frequency and suppression expenditure appear to increase with spring and summer average temperature in a highly non-linear fashion. Suppression expenditure in particular appears to undergo a shift near 15°C. Temperatures taken separately above and below that threshold are not

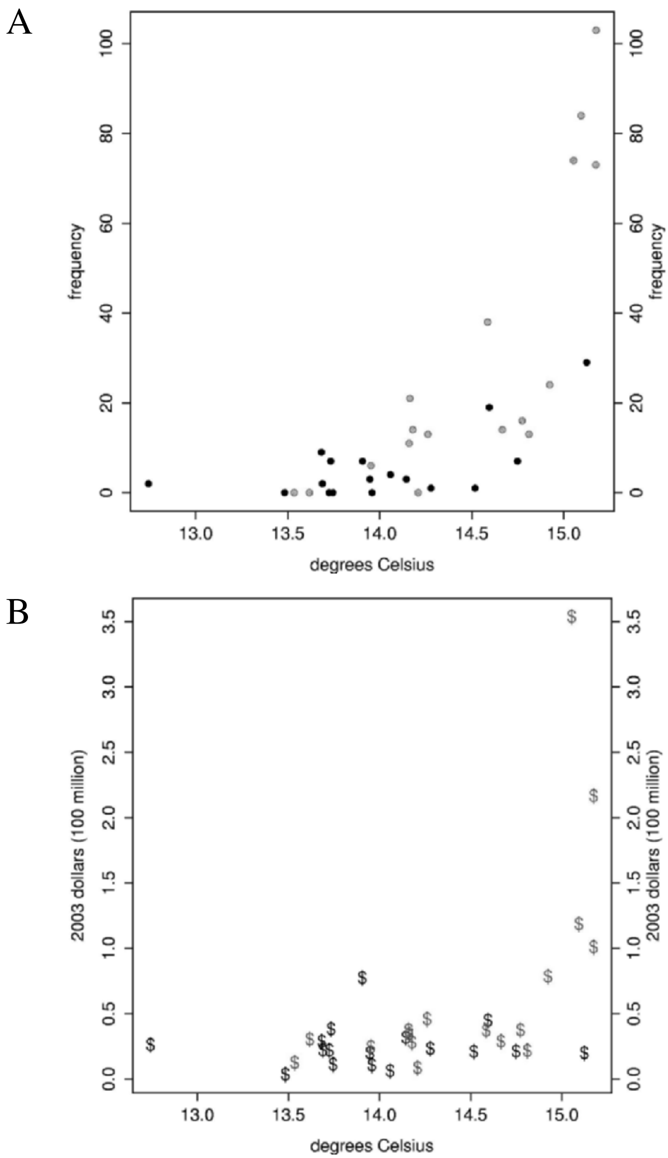

Figure 6.4. A) The annual number of large forest fires in the Northern Rockies versus average March–August temperature for western United States climate divisions. B) Annual inflation-adjusted suppression expenditures for USFS in the Northern Rockies versus average March–August temperature for western United States climate divisions. Light grey symbols indicate observations in the later half of the sample (1987-2003), black indicates observations in the first half (1970-1986). The preponderance of light grey symbols for the highest observed temperatures is indicative of the trend in recent decades toward higher spring temperatures.

significantly correlated with expenditures, but the mean and variance of expenditures increase dramatically above it.

4. SEASONAL FORECASTS FOR WILDFIRE IN THE NORTHERN ROCKIES

Given that the Northern Rockies has played such a dominant role in wildfire in western forests in recent decades, it makes a good candidate for an example forecast. Given its sensitivity to temperature, the key to forecasting wildfire activity in the Northern Rockies is whether spring and summer temperature can be forecast with any skill in the western United States. Alfaro et al. (2005, 2006) report significant skill forecasting maximum summer temperature on time horizons of a month to a season in advance. Westerling (2005) has applied the same methodology to forecast average March through August temperature as of April 1 for use in wildfire forecasting applications.

North Pacific sea surface temperatures (SSTs) and PDSI for western United States climate divisions, both observed in March, are used to forecast average spring and summer temperatures by climate division using Canonical Correlation Analysis (CCA, Barnett and Priesendorfer 1987, Westerling et al. 2002, Alfaro et al. 2005, for a detailed description of the forecast methodology). The CCA methodology matches spatial patterns in North Pacific SST anomalies and spatial patterns in western United States. PDSI observed in March to spatial patterns in March–August average temperatures for western U.S. Climate Divisions (fig. 6.5). SSTs are useful here because the oceans store heat, and the pattern of surface temperature anomalies in the oceans influences subsequent weather patterns, providing some predictive skill on seasonal time scales. The El Niño/La Niña cycle is an example of a well-known index describing patterns in the spatial distribution of SST anomalies in the Pacific that is associated with variability in climate in the western United States. PDSI, as a proxy for soil moisture, is useful as a predictor because "particularly for non-arid inland areas, a wet soil tends to depress the concurrent and subsequent monthly mean temperature, while a drier-than-normal soil is favorable for higher-than-expected monthly means…" (Durre et al. 2000).

A cross-validated regression model using forecast temperature and observed PDSI as predictors (fig. 6.6) explains 47 percent of the variability in the log-transformed NR fiscal-year area burned from 1977-2003 (adjusted R^2 of 0.47), and 42 percent of the variability in NR log-transformed fiscal-year suppression costs (adjusted R^2 of 0.42).[2] Cross-validation in this case means that the coefficients for

[2] The model specification is $\ln(Y) = T \times PDSI + \varepsilon$, where $\ln(Y)$ is the natural logarithm of either fiscal-year area burned or fiscal year suppression costs for fires on Forest Service lands in the NR, T is the regional temperature forecast, PDSI is the local climate division PDSI value for March, and ε are the errors. Fiscal year area burned data were provided by Krista Gebbert, USDA Forest Service Rocky Mountain Research Station.

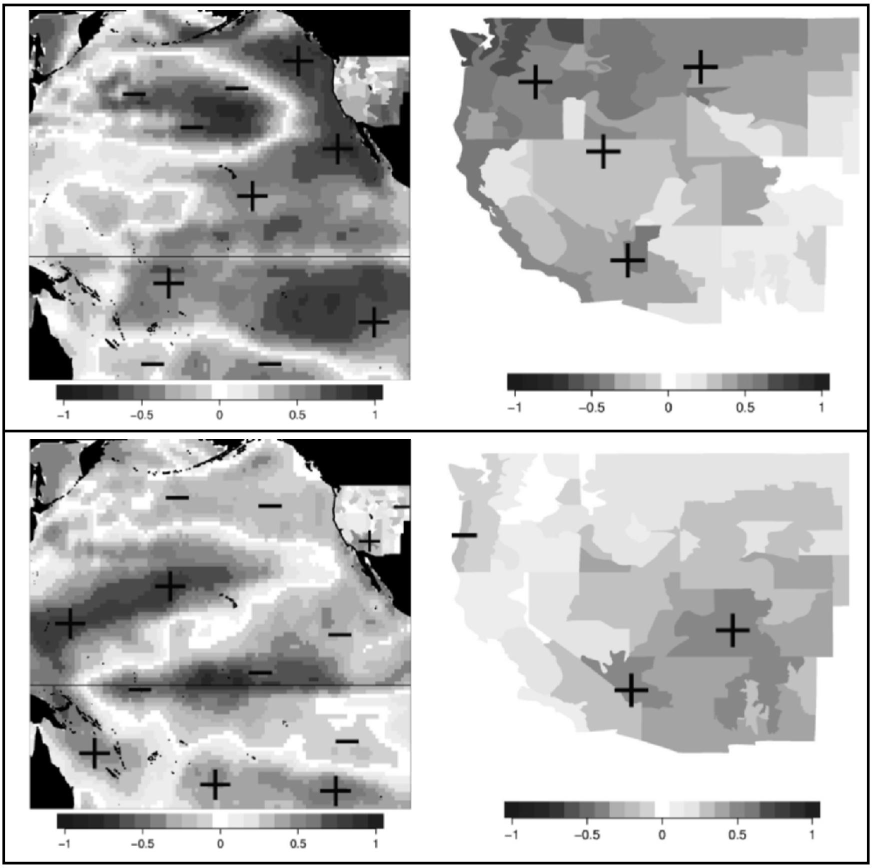

Figure 6.5. Spatial patterns in March Pacific sea surface temperatures and drought (PDSI) associated with spatial patterns in spring and summer (March to August) average temperature. Upper left: correlations between the 1st Canonical Correlation predictor and both gridded Pacific SSTs and negative climate division PDSI. Negative PDSI is used so that a positive correlation with PDSI indicates a positive correlation with dry conditions, and a positive correlation with SSTs indicates a correlation with warm anomalies in the Pacific. Upper right: correlations with the 1st Canonical Correlation predictor and MAMJJA average temperature by climate division. Lower left and lower right: as above, for 2nd Canonical Correlation predictor. A linear combination of these two patterns allows us to predict average temperature.

both the temperature forecast model and the log-area burned forecast model were recalculated for each year, while withholding information about the year being retrospectively forecast when calculating the model coefficients used to make that year's forecast. Consequently, the results appear less skillful than they would for

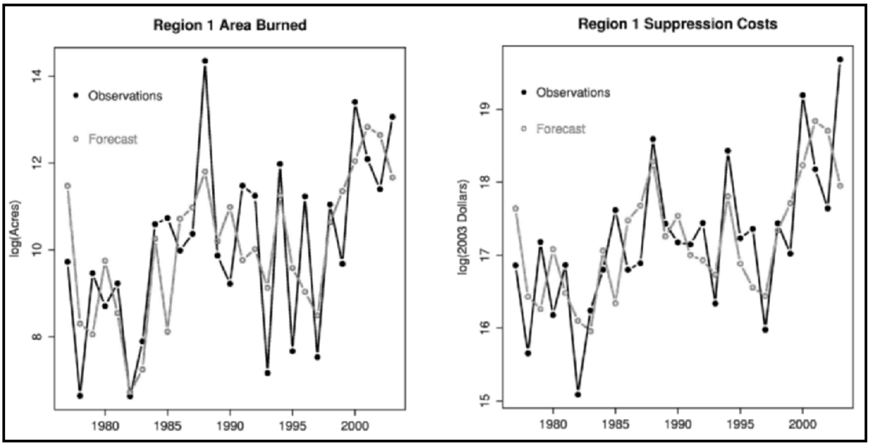

Figure 6.6. (*Left*) Region 1 log area burned forecast and (*Right*) Region 1 log suppression expenditures forecast models. The forecasts are leave-one-out cross-validated regressions on cross-validated CCA MAMJJA Temperature forecasts and the interaction between the March Palmer Drought Severity Index and the MAMJJA Temperature forecast.

the same model without cross-validation, but they are a better indication of the true forecast skill[3] to be expected on average.

While a cross-validated R^2 of 0.47 or 0.42 is actually considered to be quite good for a seasonal forecast, it is important to keep in mind the limitations of models like this. The quantities being forecast—area burned or suppression costs—are highly variable from year to year, are not conveniently normally distributed like the temperature anomalies in figure 6.1, and much of that variability is driven by factors other than seasonal temperature and precipitation. The log-transformation used in modeling area or cost (as in fig. 6.6) tends to obscure the very large absolute forecasting errors observed in high-temperature (and therefore high-fire-risk) years (fig. 6.7). For example, for the nine years with the highest temperatures (one third of the sample), the absolute level of the forecast errors averaged about $70 million (adjusting for inflation to 2003 constant dollars), or about 75 percent of the actual expenditures for each year.

While there is a high level of uncertainty associated with these forecasts using PDSI and Pacific SSTs, they are very good at telling us whether we are about to experience one of the infrequent but very active fire seasons that account for the majority of suppression costs (fig. 6.7). In most years there are relatively few large fires and suppression costs are low, while a handful of extreme years

[3] "Forecast skill" refers to the accuracy with which the forecast anticipates reality. There are many ways to assess this accuracy, including measures such as correlation, percentage of variance explained, and probability of surprise. These three particular measures are reported for the example forecast model presented here.

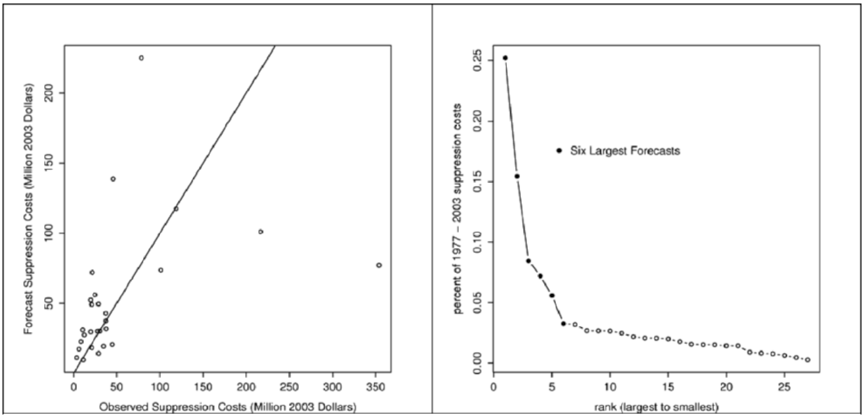

Figure 6.7. (*Left*) Northern Rockies observed versus forecast suppression cost 1977–2003. Diagonal line is the 45° line where observations would equal forecast values. Forecast model distinguishes between high and low suppression cost years. (*Right*) Ranked fiscal-year observed suppression costs for USDA Forest Service in the Northern Rockies (Region 1) as percent of total for 1977–2003. The six largest suppression cost years account for 65% of 1977–2003 USFS suppression expenditures in Region 1, and are also the six years with the largest forecast suppression costs.

accounts for the majority of the impacts from wildfire. In NR, the five largest years (all greater than $65 million) account for 62 percent of the Forest Service's suppression expenditures there since 1977 (and 83 percent of total area burned there since 1977). Cross-validated retrospective forecasts correctly estimated that 21 of the 27 years in the sample would cost below $65 million (table 6.2). Of the six remaining years—all forecast to be above $65 million—five were observed above $65 million. The sixth was still among the years with the six largest observed suppression costs.

Drought indices and North Pacific sea surface temperatures observed in March are sufficient to make a skillful categorical forecast that can distinguish between "mild" and "extreme" fire years in the Northern Rockies. In terms of the climate

Table 6.2. Northern Rockies Contingency Table: Observations versus Forecasts of Extreme Fire Years' Suppression Costs

	Forecast	
Observed	< $65 Million	> $65 Million
< $65 Million	21	1
> $65 Million	0	5

forecast framework introduced in the first section, this is equivalent to forecasting a season in advance which of two probability distributions for wildfire will be pertinent. While we do not have enough realizations (years) to describe these probability distributions empirically (as in fig. 6.1), it may be possible to parameterize them sufficiently to describe shifts in the probability of extremes (Holmes et al., this volume).

Forecasts of area burned or suppression costs made on April 1 come at least three months prior to the NR fire season, which is usually concentrated in July and August. While the Forest Service's fiscal year begins in October, an April forecast is still potentially useful for reallocating suppression resources across regions within the over-all allocation for suppression. In addition, with advance notice of a potentially active fire season, the agency may re-allocate funds from other activities to suppression. Forecasts at this lead-time may also be used to support seasonal hiring decisions, and decisions regarding the use of prescribed fire to meet vegetation management objectives.

5. CONCLUSION

While concerned with many of the same variables—such as precipitation and temperature—climate forecasts are different from weather forecasts. Climate forecasts are made at much longer lead times than weather forecasts (months to seasons rather than days to weeks in advance), but they are not simply long-lead weather forecasts. Climate forecasts are less precise than weather forecasts: they describe changes in underlying processes that are often expressed in terms of changes in average or cumulative conditions over longer time frames than a weather forecast.

Forest wildfire in the western United States is strongly influenced by spring and summer temperatures and by cumulative precipitation. The effect of temperature on wildfire risks is related to the timing of spring, and increases with latitude and elevation. The greatest effects of higher temperatures on forest wildfire in recent decades have been seen in the Northern Rockies, and a handful of fire seasons account for the majority of large forest wildfires and of total suppression expenditures in that location. A seasonal climate forecast for spring and summer temperatures would thus be of value in anticipating the severity and expense of the forest wildfire season in much of the western United States, and would be of particular value in the Northern Rockies.

In an example application of a climate forecast for the Northern Rockies, seasonal temperature forecasts using Pacific sea surface temperatures and proxies for soil moisture (PDSI) allow managers to anticipate extreme fire seasons in the Northern Rockies with a high degree of reliability. As is often the case with climate forecasts however, forecasts for the Northern Rockies do not provide a large degree of precision: while they can indicate whether a mild or active wildfire season is likely, they cannot provide a precise estimate of the level of area burned or suppression expenditures given a mild or extreme forecast.

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CHAPTER 7

WILDLAND ARSON MANAGEMENT

Jeffrey P. Prestemon and David T. Butry

1. INTRODUCTION

Wildland arson has received scant attention in the resource economics literature, yet is the cause of many large and damaging wildland fires. Research into wildfire management and policy in the United States has been principally concerned with wildfire suppression, fuels treatments, fire science (behavior), and economic efficiency questions. This is unfortunate, because wildland arson in some parts of the United States comprises well over a quarter of all fire starts and is the third most common type of arson behind arson fires in residential and educational structures (Icove and Estepp 1987). From an ecological and wildland management standpoint, wildland arson fires comprise an important piece of the overall wildfire production process. (See chapter 3 for a more detailed discussion of fire production processes). Its response to wildland management may differ from other sources of fire, such as lightning and other human caused ignitions, so special measures may need to be taken to address it. Additionally, the damages resulting from arson wildfires may differ from those ignited by other sources. Wildland arson fires are ignited close to high values at risk (Butry et al. 2002)—structures, especially, and so they are a threat to public safety that could be higher than are other kinds of fires. In fact, wildland fires in general threaten more homes than do fires that originate within structures (Cohen 2000). Recent high-profile, expensive, large arson wildfires such as the Hayman fire near Denver in 2002 (Kent 2003) and the Rodeo-Chediski fire in Arizona that same year illustrate the importance of understanding and focusing attention on potential costs and losses from arson. (See chapters 8 and 11 for in depth analyses of how to value the damages from wildfires and other disturbances.)

The dictionary definition of wildland arson is as a fire set intentionally and for malicious purposes. However, the popular usage of the term could be broader (Hall 2005). In history, fires were very often set intentionally, although not always with malice as a component of the intent (Doolittle and Lightsey 1979). As Doolittle and Lightsey (1979) found in their extensive surveys of firesetting in the South, a more inclusive term for unauthorized or even technically illegal setting of fires is "firesetting" and those setting the fires as "woods-burners." Indeed, a small but significant share of wildfires recorded, with some discretion,

T. P. Holmes et al. (eds.), The Economics of Forest Disturbances: Wildfires, Storms, and Invasive Species, 123–147.

by government agencies are those started by children, and these are identified as distinct from those by older individuals, which are classified as "incendiary." The implication is that children as firesetters are assumed to not possess the malice required of a classification of the fire as "incendiary." In this chapter, we use the word "arson" when we refer to the nominal classification of a fire cause by law enforcement or wildland managers. We use the term "firesetting" when we want to encompass all deliberately set wildfires, whether or not they are set with malicious intent. We are not able to discern the shares of fires started with and without malice using aggregate wildfire data, as all fires determined to have not been set by children are classified by law enforcement and wildland managers as "arson" or "incendiary."

Arsonists (structure and wildland) commit their crime with a wide array of motives. The Australian Institute of Justice (2005, p. 1), summarizing research by several analysts, including research on serial arsonists by Sapp et al. (undated) and Icove and Estepp (1987) (among others), indicates that wildland arsonists are driven by two classes of motives—vandalism and excitement. Vandalism is an umbrella term used to describe "wilful, mischievous, wanton destruction… [and is] often [the] result of boredom or frustration" (Australian Institute of Justice 2005, p. 1). Similarly, the umbrella term "excitement" captures a set of motives that include "thrills, attention, or recognition." Willis (2004) provides a summary of these potential motives. While a recurrence throughout the literature is that the typical arsonist is a white male, poorly educated, of lower intelligence, from a dysfunctional family, and a loner, there appears to be some variation by motivation (Inciardi 1970). For instance, Inciardi finds that arson for financial gain (e.g., as insurance fraud) is more likely to be done by older (approaching 30), middle class individuals with above-average intelligence, whereas vandalistic arson tends to be performed by teenagers. Nonetheless, Doolittle and Lightsey (1979) claim that a significant share of wildland arson fires are set as acts of retaliation and revenge. In the following pages, we describe how fires classified as arson have been explained by analysts and how and why they may have changed over time. The relevance of motive arises when seeking to understand the importance of hypothesized drivers of changes in wildland arson rates over time and differences in those rates across space.

The objectives of this chapter are to (1) review the importance of arson using recent data, (2) explain how wildland arson has been described and examined in various fields of study, (3) report new research that has characterized wildland arson as a complex, spatio-temporal process whose broad patterns may be familiar to wildland fire managers as well as to criminologists, and (4) synthesize the available research to make recommendations to wildland managers and law enforcement on potential methods of reducing aggregate arson rates. To accomplish these objectives, we provide data on wildland arson on national forests in the United States and in Florida statewide. After this background, we briefly discuss how sociologists and criminologists have or may offer explanations for observed arson wildfire. We then describe recent findings on wildland arson in

Florida. We follow this with a report on a small empirical analysis of how wildland arson compares with other crimes in Florida. The conclusion lays out our findings and describes future research directions that would advance our understanding of this phenomenon.

2. ARSON RATES IN THE UNITED STATES AND FLORIDA

Before advancing explanations for how to explain arson fire patterns, it is useful to provide some background on what has been observed in the United States. Figure 7.1 shows an index of area burned (index for area burned in 2002 = 100) by arson-ignited wildfires and the count of arson ignitions (index for number ignitions in 2002 = 100) on all national forest-managed lands of the United States. Over the period 1970-1985, arsonists successfully ignited an average of about 1,700 wildfires, while from 1986-2002, roughly 1,260 such fires were set. The dotted linear trend line in the figure illustrates that the rate of wildland arson ignitions has declined. Over that same period, the area burned by arson-ignited wildfires increased from about 57,000 acres per year to 88,000 acres per year, with peaks showing that arsonists had ignited so many large fires during some years that they burned over 200,000 acres.

More information, at finer spatial resolution, can be obtained by examining such trends and variations in plausible explanatory variables for arson. Figure 7.2

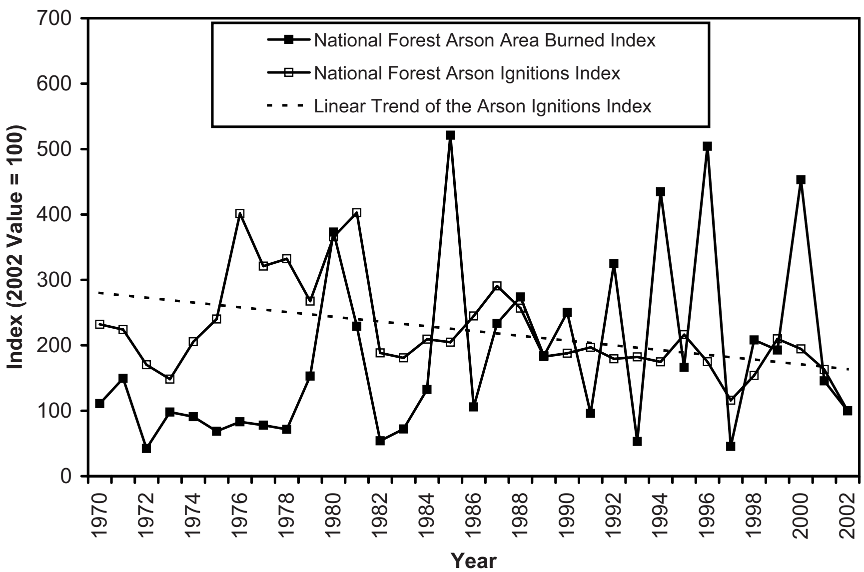

Figure 7.1. Indices of wildland arson area burned and ignitions on all U.S. National Forest System acres, 1970-2002.

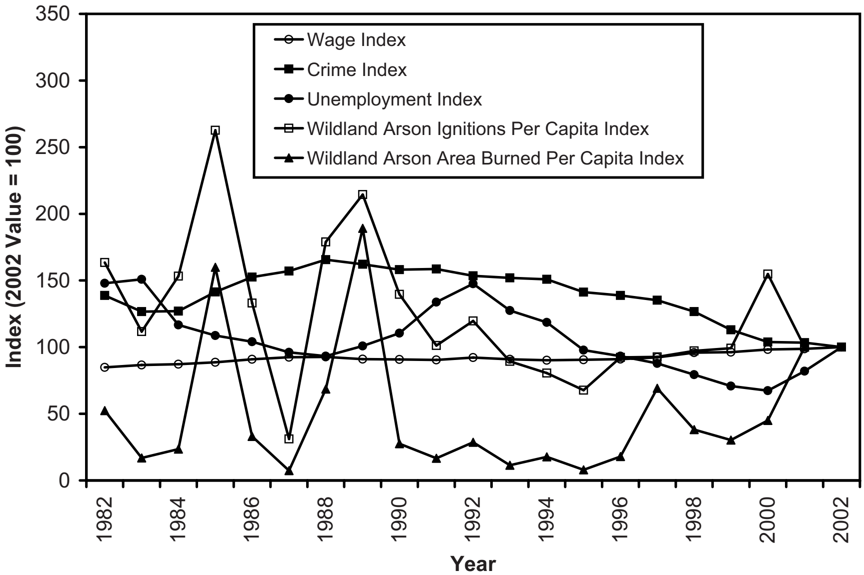

Figure 7.2. Indices of wildland arson ignitions per capita and area burned, index crimes, unemployment, and real retail wage in Florida, 1982-2002.

plots the number of arson wildfires ignited per capita statewide in Florida as an index (2001 = 100), 1982-2002. It also plots the area burned by arson wildfires as an index (2001 = 100).

Arson-ignited wildfires still burn a significant share of wildland in Florida—over the years 1992 to 2002, they comprised 20 percent of all burned acres—and negative trends in the occurrence might not have netted a better situation for residents of the state. From 1982-1991, arson wildfires annually burned an average of about 62,000 acres, while from 1992-2002, the average was about 38,000. Concurrently, annual ignitions dropped from 1,695 to 1,131 per year between those two periods, implying that the average size of an arson wildfire stayed approximately constant, at 31 and 33 acres in the two periods, respectively. In short, arson fires appear to have declined in both frequency and aggregate extent. But during this same time frame, the number of people and other values threatened by arson wildfires in Florida have grown. Between 1982 and 2002, Florida's population grew by 57 percent and total income grew by 114 percent. We recognize that research is lacking about whether actual values at risk from arson wildfires have changed. However, if the spatial distribution of values at risk were uniform and changed in magnitude directly with real income earned in Florida, then we could conclude that reductions in arson wildfires have been about sufficient to keep up with rising values at risk of arson wildfire loss. In other words, arson wildfires appear to be just as threatening today as they were over twenty years ago.

3. ARSON AND SOCIOLOGY

Sociological explanations for wildland firesetting center on normative (socially acceptable) and retaliatory behavior (Kuhlken 1999), implying that firesetting may or may not be incendiary, i.e., set with malicious intent. It is the diversity of intents that Doolittle and Lightsey (1979) describe in their typology of southern firesetters. These authors identified three firesetting archetypes among Southern United States "woods-burners." First, most active woods-burners were young (early to mid-20s), white males, of lower educational achievement, who were under-employed and of lower or lower-middle class economic status. Members of this group were loners whose favorite past-time was hunting. Second were individuals who were less active in firesetting than the first group, consisting of white males averaging 46 years of age, with higher living standards. This group perhaps was numerically larger than the first group. A third group was identified as low economic and social status individuals, often with criminal backgrounds, looked upon unfavorably by the community; they were the stereotypical, criminal woods-burners (arsonists), from the perspective of land management agencies and law enforcement.

Doolittle and Lightsey (1979) outlined how some firesetting can be classified as a normative activity, and evidence suggests its historical prevalence as such in the U.S. South (Kuhlken 1999). In general, firesetters belonging to the first two groups identified by Doolittle and Lightsey (1979) were not regarded negatively by the rest of the community because they usually were partaking in a normative activity. As a normative activity, firesetting could at least be classified from the community's perspective as a crime defined by a bad law, as it might have been classed by Cesare Beccaria (1738-1794), the "father" of modern criminology. Woods-burners looked upon fire as a necessary component of forest management: to clear undergrowth, eliminate pests, improve wildlife and livestock forage opportunities, and for site preparation following timber harvesting. Firesetting by some members of the first and second groups can be seen as carrying on long-standing traditions of active land management, in spite of nominal legal prohibitions. This assessment by Doolittle and Lightsey (1979) validates later descriptions by Pyne (1995), who documents such firesetting for management purposes as spanning many cultures worldwide. This normative behavior among firesetters also helps to explain Doolittle's (1978) finding that prescribed fire use can lead to reductions in incendiarism. Prestemon et al. (2002), Mercer and Prestemon (2005), and Prestemon and Butry (2005) provide empirical support for the negative relationship between incendiary fires and prescribed fire in Florida, as well. The last three studies based their results on data from the mid-1990s and later, not on fires set during the time of Doolittle and Lightsey's analyses. But aside from the obvious explanation that rising use of prescribed fire in some parts of Florida might have reduced fuels and incendiary firesetting success by arsonists, it is plausible that, at least historically in Florida, higher rates of arson observed there in early 1980s might have included normative firesetting activity that has abated as prescribed fire has expanded. Furthermore, it is possible that normative firesetting, while classified by government agencies and law enforcement as "incendiary," has changed over time as a result of changes in the culture that supported it.

Fire has also long been employed as a tool of antisocial behavior and political violence, at scales large and small (Goudsblom 1992). Doolittle and Lightsey (1979) indicate that fire has been used as a weapon of retaliation in the South, apparently by members of any of the three groups of firesetters that they identified. When used in this way, fire is employed to punish other landowners for restricting forest access for previously allowed public activities (e.g., hunting); the greater the degree of restriction, the greater the rate of illegal firesetting in retaliation. In their study, they found that such acts were frequently directed at corporate landowners. U.S. Forest Service lands (national forests) were also set afire when local residents faced similar access problems (through road closures, etc.) or when federal land managers imposed new regulations of public lands grazing, prosecuted illegal dumping, and created opportunities for developed recreation by non-local residents. Doolittle and Lightsey (1979) conclude that incendiarism is one way that landowners can retaliate against a more powerful neighbor or land manager. These findings document a rural, southern United States version of a phenomenon, described by Gamst (1974, p. 48), as being present for centuries in human societies in many cultures. Molina (1997), who documented the same phenomenon in northwestern Spain, Gamst (1974), and Doolittle and Lightsey (1979) have shown that fire is frequently used as tool of social protest and revenge in response to use-restrictions of the land by the politically and economically powerful.

4. ARSON IN CRIMINOLOGY

Criminology is the study of laws, their violation, and how society responds to violations. Some criminologists attempt to explain spatial and temporal crime patterns: how crimes of various kinds, once defined, vary over space and over time, typically as a function of social, environmental, and economic variables. Variables used to explain crime typically derive from theories about the causes of crime, and many theories are available from criminology that could help us to understand spatial and temporal patterns of wildland arson. Crime pattern modeling or crime mapping can be descriptive and it can be quantitative. Crime pattern modeling of wildland arson might reveal how wildland arson is both an environmental and a sociological phenomenon.

Cohen and Felson (1979) depict a routine activity approach in order to understand crime patterns. While their routine activity approach is focused on "direct contact predatory violations" (robbery, homicide, assault, etc.), in many ways it can be applied to arson crimes. For instance, Cohen and Felson (1979) discuss how, for a crime to be committed, several necessary factors must exist: an offender, a target, and a lack of "capable guardians" (e.g., police or neighborhood watch groups). The routine activities approach explains that crime varies across space and over time according to how everyday human activities vary over space and time—for example, among neighborhoods, over the course of a year, perhaps in response to seasonal or economic differences and changes. These variations modify the convergence among the offender, the target, and the lack of guardians. For instance, in many places, spring and summer times drive people out from their homes to parks, city streets, and vacation spots. Being away from one's home leaves one vulnerable to being personally targeted by criminals (who might assault or rob) or it leaves one's home vulnerable to thieves (who would burglarize it).

A large literature exists on mapping crime and predicting the places and times of future crime based on statistical models of crime patterns, often referred to as "hotspotting" (Townsley et al. 2003). These models have been developed especially for aiding in understanding and dealing with serial crime in urban settings, and the relevant literature dates back several decades (Shaw and McKay 1931, 1942, Lottier 1938, Boggs 1966, Harries 1980). More recently, the science of crime pattern understanding and prediction has gained traction because of advances in statistical modeling techniques, geographic information systems, and computing power (Corcoran et al. 2003, Deadman 2003, Bowers and Johnson 2004, Johnson and Bowers 2004). In wildland arson, hotspotting models are in their infancy. One reason for slow development has been lack of data or modeling constructs.

These theories of crime requiring an opportunity would seem to require elaboration to encompass wildland arson. For example, Prestemon et al. (2002), Mercer and Prestemon (2005), Butry and Prestemon (2005), and Prestemon and Butry (2005) have shown how arson wildfire area burned appears to react strongly to fuel conditions and weather—the same factors influencing other kinds of wildfire. Figures 7.3 and 7.4 highlight the seasonal arson peaks in Florida in late-winter to early-spring further and daily (hourly) peaks in ignitions in the mid-day, both of which are times of increased non-arson wildfire probability. But the broader crime literature does, indeed, include weather or seasonality in models explaining crime. For example, Rotton (1985) and Cohn (1990) found a positive correlation between certain kinds of crime and air temperature (among other atmospheric factors). Broadly across the United States, aggregate crime rates are positively correlated with average annual temperature, and the correlation seems stronger for violent crime.[1] Perhaps confirming the link between routine activities and weather, some kinds of crime follow systematic (seasonal) patterns (Farrell and Pease 1994, Felson and Poulson 2003). Other patterns are even finer scale and likely relate to routine activities associated with societies.

[1] The correlation between average annual temperature and the major crime index is 0.38, while that for violent crime is 0.43. These correlations are based on data from 1972-2004, where each pair of observation is a state's 1972-2004 average crime index or violent crime index and the state's 1970-2004 average annual temperature (Florida Department of Law Enforcement 2005c, National Climatic Data Center 2006).

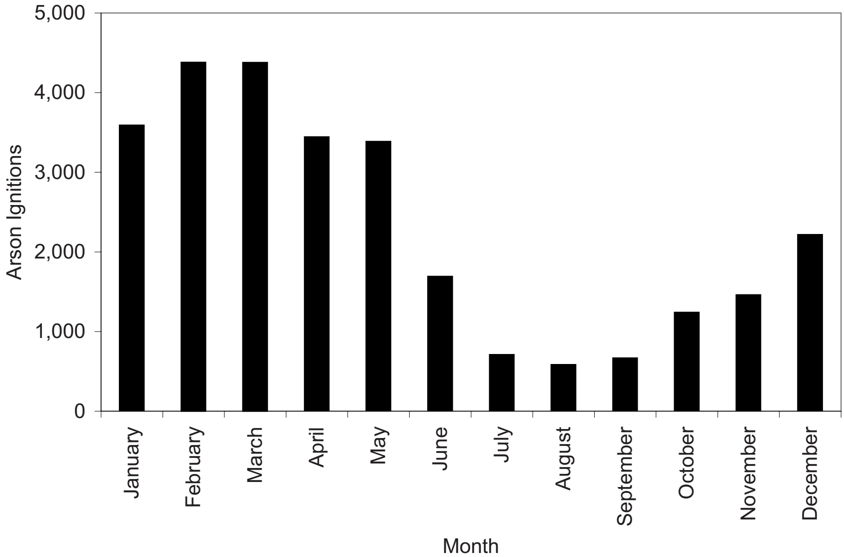

Figure 7.3. Arson ignitions by month in Florida, 1982-2002.

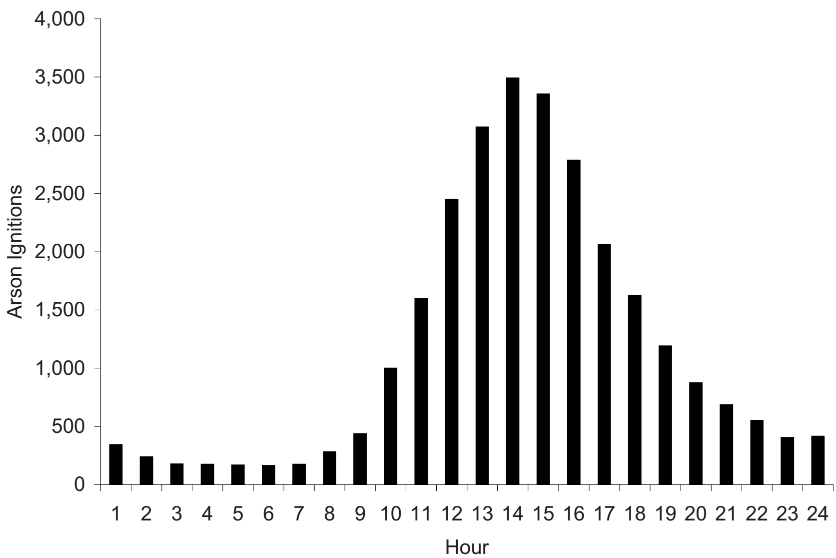

Figure 7.4. Arson ignitions by hour of the day in Florida, 1982-2002.

DiTella and Schargrodsky (2004) showed how car thefts in Argentina also had systematic components linked to days of the week. Also, many kinds of crimes vary diurnally, which has been attributed to the advantage provided to criminals by the cover of darkness and hence community oversight and policing efficacy, and to diurnal patterns of human activities related to work, leisure, and substance abuse timing (Cohen and Felson 1979, Cohn 1993, Hipp et al. 2004). While arson ignitions in Florida appear to follow seasonal and daily patterns, perhaps indicating that weather and climatic conditions exist for successful ignition. Figure 7.5 shows that other factors influence arsonist behavior, such as opportunities to burn. Figure 7.5 clearly illustrates a weekend effect, which is not likely to be driven by weather and climate conditions, but rather by other socioeconomic factors, such as leisure time.

The Federal Bureau of Investigation classifies wildland arson as a property crime, although we know of no research that has sought to characterize its relationships to other major categories of crime. Data show that the overall statistical relationship between wildland arson and aggregate crime measures are weak. Although some violent and non-violent crimes differ in their responses to labor market conditions and aggregate wealth (Grogger 1998, Gould et al. 2002, Burdett et al. 2003), law enforcement, and poverty (Hannon 2002), we know of no published study that has sought to evaluate whether wildland arson is more similar to certain types of crimes than to others. Perhaps a reason for the lack of identified statistical relationships is that wildland arson is a less frequent crime, implying that there is insufficient information to clearly establish statistical relationships. We might hypothesize that, because arson involves an effort to attack a target, it could be classified as a violent act (Crowe 2000) and so should share underlying causes with violent crimes. In contrast, Rider (1980) found that convicted arsonists are more likely to have committed prior property

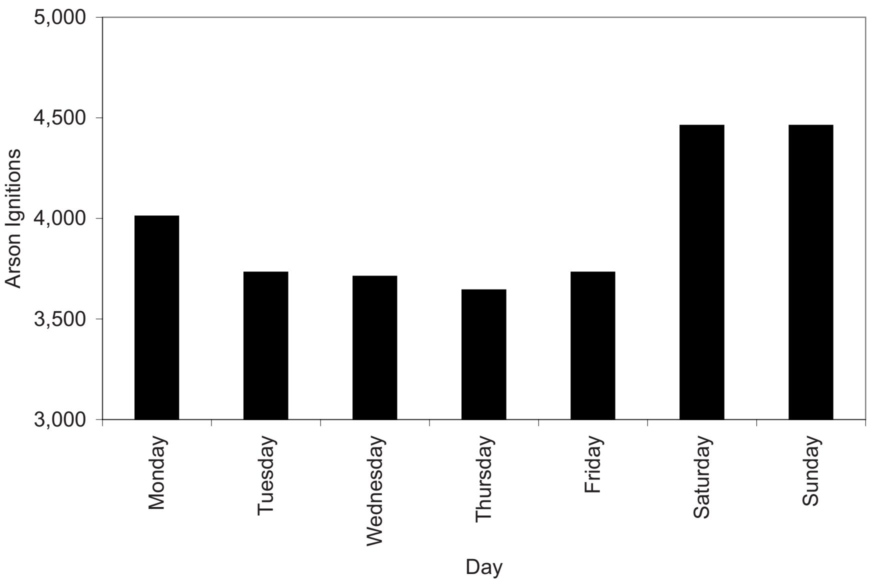

Figure 7.5. Arson ignitions by day of the week in Florida, 1982-2002.

crimes and that convicted non-arsonists are more likely to have committed violent crimes. Indeed, evidence suggests (Cabe 1996) that some arsonists ignite fires in order to obtain employment or even salvable timber (Molina 1997). Therefore, wildland arson incidence rates should have characteristics of both categories.

There is statistical evidence that wildland arson should be related to other kinds of crimes, even while correlation analyses could fail to turn up connections. For example, arson wildfire ignitions in Florida counties relate to the same set of socioeconomic variables that affect other kinds of crimes (Prestemon and Butry 2005), even while simple correlation statistics do not appear to support this contention. Data on wildland arson and general categories of crime in Florida (Florida Division of Forestry 2002, Florida Department of Law Enforcement 2005b) show that wildland arson ignitions per capita, statewide, over the period 1982-2001 were slightly negatively correlated with violent crime per capita (murder, forcible sex offenses, aggravated assault, and robbery) ($\rho = -0.06$) and slightly positively correlated with property crimes per capita (burglary, larceny, motor vehicle theft) ($\rho = +0.13$), while violent and property crime indices were very positively correlated ($\rho = +0.86$). Similarly, pooled county-level data for the period 1989-2001 in Florida show that county arson ignitions and a county index of aggregate violent and non-violent crime were negatively correlated ($\rho = -0.19$) (Prestemon and Butry 2005). As well, departures from 1989-2001 county average wildland arson ignitions per capita and county average crime indices (crimes per capita) had a population-weighted correlation of –0.08. What we show in the empirical application section of this chapter is that controlling for weather variables in conjunction with socioeconomic and law enforcement variables can enlighten hypotheses about whether wildland arson is similar to major crime categories.

5. ARSON IN ECONOMICS

We can synthesize the findings of criminological and sociological studies of wildland arson into an economic model of firesetting. Criminological research continues to define individual decision making as deriving from economic models of crime (Becker 1968), where crime is deterred when the overall expected costs (its opportunity costs) of committing a crime are high relative to its perceived expected benefits. This description of criminal decision-making dates to Jeremy Bentham's (1748-1832) utilitarian view of criminal activity. Bentham posited that criminals commit crimes when the expected rewards from commission exceed the expected losses, so that preventing or reducing crime could be achieved by adjusting the expected losses. Indeed, analysts have found that economic models of crime, while arguably a narrow view of criminal behavior, often fit crime data fairly well. In the case of wildland arson, such an "economic model" would appear to need to index the prevalence of normative behaviors supporting firesetting in the target population. Greater prevalence of normative beliefs in favor of firesetting would translate into higher perceived benefits from the firesetting. Places or populations with greater prevalence of normative firesetting would have higher arson rates, *ceteris paribus*.[2]

Abstracting from the issue of normative firesetting, and following Becker's (1968) approach, we formalize the prospective arsonist's expected utility from successfully starting a wildland arson fire as:

$$E[U_i(O_i)] = \pi_i U_i(g_i - c_i - f_i(W_i, w_i)) + (1 - \pi_i)U_i(g_i - c_i) \qquad (7.1)$$

where E is the expectations operator, U_i is the prospective arsonist's utility, O_i is the number of offenses committed, π_i is the probability of being caught and convicted, g_i are the arsonist's psychic and income benefits from illegal firesetting, c_i is the production cost for firesetting, and f_i is the wealth loss experienced by the criminal if caught and convicted. Wealth loss is a function of wages (w_i) and employment status (W_i) (Grogger 1998, Gould et al. 2002, Burdett et al. 2003, Jacob and Lefgren 2003).

Expected utility theory implies that an arsonist will continue to ignite additional arson fires until the expected marginal gain in psychic or income benefits from the last ignition is exceeded by the sum of the expected loss in wealth from being caught and the production cost of the firesetting. Empirical findings in crime research also lends weight to the hypothesis that psychic and income benefits would be higher if there is more available wealth in a community, encouraging crime (Gould et al. 2002). For wildland arson, costs would be affected by opportunity costs of time (i.e., the net benefits accruing to the prospective arsonist of using time in an alternative way), weather affecting firesetting success, and the availability of information regarding potential firesetting success. The opportunity cost of time could be captured by wages, employment status, and the availability of leisure time. Greater wages or the opportunity to work and earn wages tends to raise opportunity costs; these costs could also be higher on non-leisure days. In the aggregate, wage rates of the target arsonist population and the unemployment rate would index such costs.[3] On fine time

[2] We contend that steady changes in such normative behavior over time can be captured in a statistical sense through time trends within populations and by cross-sectional dummies and time trends in panel data analyses. Average differences across space would be captured by the cross-sectional dummies in panel studies. Differences in the amount of normative behavior across populations in purely cross-sectional analyses would be more difficult to capture but could be indexed by measures of rural, lower economic status populations.

[3] Grogger (1995) shows how past arrests and convictions affect wages subsequently earned, but that the effect is limited. Grogger (1998) then elaborates a model wherein wages for youth are endogenous to those individuals' past crime committed and other variables, and some of his findings support this hypothesis. Further, wages were found to be negatively related to youth crime. Burdett et al. (2003) describe a labor market equilibrium model where employment levels, the wage rate, and the crime rate are jointly determined, but without formal tests of the theory.

scales, an indicator of weekends and holidays may control for the apparent lower opportunity cost of time on days of leisure.

Firesetting success, as measured by the time spent attempting to ignite a fire, would tend to be higher on dry, warm days and would depend on the availability of flammable fuels (Gill et al. 1987, Vega Garcia et al. 1995, Prestemon et al. 2002). In other words, firesetting costs, as captured by the opportunity cost of time spent attempting to ignite a fire, are higher when fuels are limited and the weather is not amenable to successful ignition (e.g., in wet conditions). A potentially cheap way to monitor firesetting success would be to observe success. The prospective arsonist would do this by observing the firesetting activities of another arsonist or by observing the success of his own firesetting attempts. Expectations of high success rates would translate into expectations of low opportunity costs of time in firesetting. Prestemon and Butry (2005) and Butry and Prestemon (2005), in their daily models of wildland arson ignitions, statistically controlled for this kind of short-run information-gathering by arsonists by lagging ignitions in both time and space. Their findings identified significant spatial and spatio-temporal clustering, which, in addition to validating casual evidence of spatial clustering of arson fires (Doolittle 1978), provide statistical support for the contention that arsonists observe and use firesetting success of themselves or others to facilitate copycat and serial behavior.

To the extent that so-called "copycat" firesetting derives from more complex psycho-social pathologies, the "firesetting production cost" approach may be an overly simplistic representation of observed human activities. Surrette (2002) describes juvenile copycat criminal behavior as facilitated through media. His assessment is that initial stimuli may differ across crime categories and social groupings. As well, copycat crimes emanate from established criminals who view crimes of others as a learning experience, practice the crime, and then commit it under amenable conditions. Copycat and serial firesetting behavior is a topic meriting additional exploration. To confirm spatio-temporal or temporal firesetting, analysts would need to establish serial behavior using forensic evidence. However, we believe that the economic model of crime elaborated above can be captured statistically by including measures of recent and (or) nearby ignition activity.

Research that relates the numbers of wildland arson ignitions to hypothesized drivers dates back at least two decades. This literature is smaller than the also limited literature modeling human-ignited wildfires (Gill et al. 1987, Vega Garcia et al. 1995), although the daily models that were estimated by those authors clearly have lessons on how to model intentional firesetting. Below, we describe a few of these statistical studies of wildland arson.

Donoghue and Main (1985) develop a state-level time series cross-sectional model that relates the annual total wildland arson ignitions to police levels and state-level dummy variables. They show how law enforcement might play a role in wildland arson rates in the eastern United States, suggesting an economic model that would minimize the sum of arson wildfire losses and law enforcement costs. The general idea of including law enforcement or even fires of specific types has not been adopted in a fuller specification of the problem of optimal wildfire management (Rideout and Omi 1990, Donovan and Rideout 2003) or in models of optimal timber management under catastrophic risk (Martell 1980).

Prestemon and Butry (2005) model arson ignitions as a daily autoregressive process (prior ignitions influence future ignitions) that is also sensitive to law enforcement, socioeconomic variables (poverty, unemployment, retail wage), fuels management, wildfire history, and fire weather (Keetch-Byram Drought Index, El Niño) for several high arson multi-county regions in Florida. There are three notable findings from this research: (1) law enforcement is negatively correlated with arson rates; (2) fuels management is negatively correlated with arson rates; (3) arson ignitions are clustered in time in episodes that can last up to 11 days, which is consistent with serial and copycat arson. This last result is consistent with Surrette (2002), and it suggests copycat behavior. With arson wildfire, it validates a claim by Dennett (1980) and Crowe (2000) that media reports of wildfire lead to future instances of arson, as would-be arsonists are notified of the favorable ignition conditions or spur fantasies of heroism.

Butry and Prestemon (2005) adapt their purely temporally autoregressive arson ignition model to examine arson outbreaks at a finer spatial resolution (Census tracts and individual counties) in a way that would allow for detection of spatio-temporal clustering of firesetting in Florida. These findings include strong evidence that arson ignitions are clustered in both time and space, implying an arson outbreak in one area should be a signal to law enforcement that there is temporarily higher likelihood of future arson fires in surrounding areas. The implication here is that increased arson targeting by law enforcement could be effective at deterring fires in surrounding areas. This study also confirms initial analyses conducted by Doolittle (1978), which found that incendiary (arson) fires occur in clusters.

Brantingham and Brantingham (1981) describe a method by which the spatial pattern of serial crimes can be used to aid in apprehension of criminals. Canter and Larkin (1993) show how using an offense map to circumscribe the physical space containing all linked serial criminal acts can provide a zone within which law enforcement can search for a criminal base location (e.g., the criminal's home). Canter et al. (2000) and Ratcliffe (2004) provide examples of how to develop and use crime maps to improve law enforcement efficiency, leveraging information regarding the physical and temporal domain of the criminal activity. It would be possible, then, to use this spatio-temporal clustering of fires to focus law enforcement's efforts in deterrence of future arson fires. The multidimensional concentration of firesetting would appear to make it amenable to tools such as crime maps.

6. WILDLAND ARSON AND CRIME IN FLORIDA: AN EMPIRICAL ANALYSIS

Our empirical analysis seeks to clarify whether wildland arson is similar in its response to hypothesized causal factors as other kinds of crimes. We focus on wildland arson and crime in Florida. We specify our model based on research by statistical criminologists (Gould et al. 2002) and the recent research into wildland arson. The hypotheses that we test are whether arson wildfires respond to the same factors shown to have an influence over the rates of major categories of crime. As well, we seek to evaluate whether the degree and direction of the response to these factors is similar across crime types and thereby draw conclusions that could be helpful for managers, law enforcement, and analysts.

6.1 Data and Empirical Models

Our statistical approach is to estimate eight individual equations relating crime to the hypothesized causal or driving variables. These crime categories are the ones routinely reported to the Federal Bureau of Investigation and are reported as an index, in terms of the number of crimes committed per 100,000 residents of the state. As such, they are called index crimes: the violent crimes of murder (all types), rape, robbery, and aggravated assault; and the property crimes of burglary, larceny, motor vehicle theft; a fourth property crime sometimes included in federal data is arson (all types included). Here, our fourth property crime and eighth crime index is simply wildland arson. The wildland arson equation relates the number of arson ignitions (Florida Division of Forestry 2005) per capita statewide in each year to: the statewide count of full-time equivalent sworn police officers per capita (Florida Department of Law Enforcement 2005a), which indexes the arrest rate and deterrent effect of law enforcement; the statewide average length of sentence imposed on convicted nonviolent criminal offenders (Florida Department of Corrections 2005), lagged one year because these apply to averages from July of the previous year to June of the nominal year, which also accounts for one source of the opportunity cost of the crime, and should be negatively related to arson; the statewide retail wage rate (Bureau of Labor Statistics 2005), which has a negative expected effect on arson; the statewide unemployment rate (Bureau of Labor Statistics 2005), another measure of the opportunity costs of crime, which should be positively related to arson; the statewide per capita income (Census Bureau 2005), which should be positively related to arson, as it would capture the relative economic inequality between the prospective arson population and the broader population (Ousey 2000), although; the statewide poverty rate of all persons (Census Bureau 2005), which has long been suspected of being linked to crime (Ousey 2000); two variables that index environmental factors that would affect the success rate (hence some of the cost) of firesetting, including an El Niño-Southern Oscillation (ENSO) measure called the Niño-3 sea surface temperature (SST) anomaly

(National Oceanic and Atmospheric Administration 2005) and a dummy variable that controls for the extreme ENSO cycle of 1997-1998 ("D1998") (Prestemon et al. 2002, Prestemon and Butry 2005); and a time trend, which could capture changes in policing practices (including, in the case of wildland arson, efforts to contact potential firefighter arsonists [Cabe 1996] or normative firesetters [Doolittle 1978]), technology, and other influential demographic variables not directly modeled. Equations for the seven major index crimes are related to all of the same variables as in the wildland arson model, except that each has its own applicable statewide average sentence length and that these seven equations exclude the environmental variables.

Equations are estimated using three-stage least squares methods[4], using instruments to control for the simultaneous determination of police levels and crime rates on an annual basis and estimating all eight equations simultaneously. All variables, including the wildland arson and crime index variables we attempt to explain, are expressed in natural logarithms. Temporal autocorrelation is abated by temporally "lagging" dependent variables (i.e., having last year's crime index value help to explain this year's crime index value) and (or) by (in four cases) first-differencing (i.e., subtracting the previous year's value from the current year's value) the dependent (crime index) variable and the regressors (except for the time index variable, "year").

6.2 Results

Table 7.1, shows that wildland arson statewide in Florida behaves mainly according to our expectations from theory—it is negatively affected by police levels and wages and positively affected by unemployment and per capita income. However, in contrast to findings by Prestemon and Butry (2005) and in contrast to descriptive research by Doolittle and Lightsey (1979), it is not significantly related to the poverty rate. The former study had more detailed data on poverty, allowing a tighter match between locations and temporal variations of arson fires and locations temporal variations of poverty. The effect of poverty found here therefore might have been erased by aggregation bias, which tends to attenuate parameter estimates. The measures used to control for the success rate of firesetting, the Niño-3 SST anomaly and the 1998 dummy variable, relate to wildland arson as expected from previous analyses and theory: Drier weather associated with the cold phase of the ENSO cycle (negative values of the Niño-3 SST anomaly) leads to greater wildland arson rates; the severe 1997-1998 ENSO cycle explains a higher rate in 1998 with weak statistical significance (a probability level of 0.18).

The explanatory powers of the other seven crime models are comparable to, or higher than, our equation for wildland arson, but broad similarities exist between

[4] Due to an insufficiently long time series, we did not attempt to estimate this as a cointegrated system. This is an area worthy of additional analysis, however.

Table 7.1. Three-stage least squares estimates of statistical relationships between index crimes and wildland arson and hypothesized explanatory variables, statewide in Florida, 1982-2001.

	Parameter Estimate	Standard Error	t-statistic	Probability Level
Wildland Arson				
Constant	3563.63	1369.01	2.60	0.01
Police Per Capita	-10.26	3.65	-2.81	0.01
Sentence_{t-1}	1.31	0.58	2.25	0.03
Real Retail Wages	-13.56	6.52	-2.08	0.04
Unemployment Rate	1.69	0.66	2.57	0.01
Real Per Capita Income	23.01	6.86	3.35	0.00
Poverty Rate	0.98	1.25	0.79	0.43
1998 Dummy	0.40	0.29	1.34	0.18
Niño-3 SST Anomaly	-0.29	0.08	-3.44	0.00
Year	-483.64	184.70	-2.62	0.01
Murder				
Constant	55.34	10.08	5.49	0.00
Police Per Capita	-0.63	0.18	-3.47	0.00
Sentence_{t-1}	-0.56	0.07	-8.11	0.00
Real Retail Wages	-0.74	0.49	-1.50	0.14
Unemployment Rate	0.10	0.05	2.04	0.04
Real Per Capita Income	0.23	0.41	0.56	0.58
Poverty Rate	-0.11	0.10	-1.12	0.27
Year	-0.023	0.006	-4.21	0.00
Lagged Dep. Var.	0.09	0.09	0.96	0.34
Rape				
Constant	-8.90	11.95	-0.74	0.46
Police Per Capita	-0.10	0.26	-0.37	0.71
Sentence_{t-1}	-0.14	0.06	-2.21	0.03
Real Retail Wages	-1.51	0.62	-2.42	0.02
Unemployment Rate	0.03	0.06	0.53	0.60
Real Per Capita Income	0.28	0.49	0.57	0.57
Poverty Rate	0.09	0.11	0.83	0.41
Year	0.0081	0.0069	1.17	0.24
Lagged Dep. Var.	0.63	0.13	4.95	0.00
Assault				
Constant	22.99	3.84	5.99	0.00
Police Per Capita	-1.61	0.36	-4.48	0.00

(continued)

Table 7.1. Three-stage least squares estimates of statistical relationships between index crimes and wildland arson and hypothesized explanatory variables, statewide in Florida, 1982-2001. ***(continued)***

	Parameter Estimate	Standard Error	t-statistic	Probability Level
Assault *(cont.)*				
Sentence$_{t-1}$	0.00	0.05	-0.02	0.98
Real Retail Wages	-0.70	0.57	-1.24	0.22
Unemployment Rate	0.17	0.07	2.49	0.01
Real Per Capita Income	-0.39	0.47	-0.84	0.40
Poverty Rate	0.26	0.08	3.12	0.00
Year	-0.012	0.002	-5.99	0.00
Lagged Dep. Var.	-0.44	0.11	-4.00	0.00
Robbery				
Constant	9.20	11.87	0.77	0.44
Police Per Capita	-0.76	0.25	-2.97	0.00
Sentence$_{t-1}$	-0.30	0.05	-6.45	0.00
Real Retail Wages	-1.66	0.73	-2.27	0.03
Unemployment Rate	0.24	0.07	3.21	0.00
Real Per Capita Income	1.79	0.57	3.11	0.00
Poverty Rate	0.13	0.12	1.06	0.29
Year	-0.0025	0.0072	-0.35	0.73
Lagged Dep. Var.	0.61	0.06	9.81	0.00
Burglary				
Constant	6.44	3.74	1.73	0.09
Police Per Capita	-1.29	0.37	-3.51	0.00
Sentence$_{t-1}$	-0.18	0.06	-2.93	0.00
Real Retail Wages	-0.88	0.65	-1.34	0.18
Unemployment Rate	0.15	0.07	2.04	0.04
Real Per Capita Income	1.07	0.58	1.85	0.07
Poverty Rate	0.09	0.08	1.02	0.31
Year	-0.0032	0.0019	-1.73	0.09
Lagged Dep. Var.	0.28	0.10	2.74	0.01
Larceny				
Constant	9.07	3.40	2.67	0.01
Police Per Capita	-0.68	0.33	-2.07	0.04
Sentence$_{t-1}$	-0.06	0.06	-0.93	0.35
Real Retail Wages	-0.08	0.58	-0.14	0.89
Unemployment Rate	-0.03	0.07	-0.47	0.64

(continued)

Table 7.1. Three-stage least squares estimates of statistical relationships between index crimes and wildland arson and hypothesized explanatory variables, statewide in Florida, 1982-2001. ***(continued)***

	Parameter Estimate	Standard Error	t-statistic	Probability Level
Larceny *(cont.)*				
Real Per Capita Income	-0.86	0.53	-1.63	0.11
Poverty Rate	0.01	0.08	0.08	0.94
Year	-0.0045	0.0017	-2.66	0.01
Lagged Dep. Var.	0.38	0.13	2.88	0.00
Motor Vehicle Theft				
Constant	21.79	5.69	3.83	0.00
Police Per Capita	-1.72	0.52	-3.32	0.00
Sentence_{t-1}	-0.06	0.09	-0.70	0.48
Real Retail Wages	-0.30	0.94	-0.31	0.75
Unemployment Rate	0.11	0.12	0.97	0.33
Real Per Capita Income	0.09	0.81	0.11	0.91
Poverty Rate	0.20	0.12	1.64	0.10
Year	-0.011	0.003	-3.83	0.00
Lagged Dep. Var.	0.28	0.12	2.37	0.02

Equation Statistics	Obs.	R^2	Adj. R^2	Durbin-Watson
Wildland Arson	20	0.61	0.26	1.88
Murder	22	0.99	0.98	2.62
Rape	22	0.83	0.72	2.22
Assault				
First-Difference Model	21	0.60	0.33	2.39
Robbery	22	0.98	0.96	2.45
Burglary				
First-Difference Model	21	0.68	0.47	1.56
Larceny				
First-Difference Model	21	0.56	0.26	1.87
Motor Vehicle Theft				
First-Difference Model	21	0.65	0.41	1.78
Whole System	170			

Source: National Interagency Fire Center (www.nifc.gov/stats/suppression_costs.html) arson and these other crimes in how variables relate to the modeled crime. In all other crime categories except rape, the most commonly significant explanatory variable for crime is police per capita, where it is negative and significant at a probability level of 0.05 or smaller for seven out of eight crime equation estimates. The negative relationship between crime and wages is observed for all four violent crimes (murder and assault with very weak statistical significance, probabilities of 0.14 and 0.22, respective; and rape and robbery with stronger statistical significance, probabilities of 0.02 and 0.03); a weak statistical effect (probability of 0.18) is also shown for burglary. After controlling for other factors, unemployment strongly (probability smaller than 0.05) and positively relates to murder, assault, robbery, and burglary. Besides wildland arson, real per capita income relates positively and significantly (probability of 0.07 or smaller) to robbery and burglary, which, with the exception of larceny, is as expected: criminals who take others' wealth steal more often when greater wealth exists.[5] Poverty is significant and positively related to only assault and motor vehicle theft, although with weak statistical significance. Measures of trends in these crime rates are typically negative and highly statistically significant (probability smaller than 0.01) for wildland arson, murder, assault, burglary, larceny, and motor vehicle theft.

These statistical results allow us to make several observations about the similarities and differences between wildland arson on the one hand and other index crimes on the other. We find that wildland arson behaves similarly to other crimes in response to variables capturing socioeconomic conditions. The primary contrast between wildland arson and other crimes is in the size of statistical relationships that crimes have with many of the explanatory variables. The absolute sizes of parameter estimates associated with each explanatory variable in the models shown in table 7.1 are measures of the statistical sensitivity of the crime index to changes in the explanatory variable. Wildland arson responds more sensitively to police force levels, wages, unemployment, per capita income, and unspecified other variables captured in the time trend, compared to other crimes[6]. The finding of the strong sensitivity of wildland arson to wages is consistent with what is

[5] Note that the retail wage rate, an indicator of the kind of work available to the low-skill criminal, is used as one measure of the opportunity cost to the criminal of participating in crime. Per capita income indexes theft opportunities. An alternative explanation for the positive effect of per capita income is that greater average wealth is correlated positively with greater crime reporting due to higher insurance coverage (Pudney et al. 2000). For wildland arson, the relationship to per capita income is more difficult to explain, but it could be that greater per capita income, for a given retail wage rate, indexes a greater aggregate income inequality and hence greater rates of social injustice and use of arson as a means of retaliation by the relatively less powerful (Doolittle and Lightsey 1979). Alternatively, as suggested by one reviewer of this chapter, greater wealth in a location or in time might be linked to less prescribed fire, which is unmodeled in our regression but whose relationships are documented by Butry et al. (2002).

[6] Part of this difference is that, perhaps, successful wildland arson crimes are reported, because they are so visible. With other crimes, under-reporting is likely to be serious (Pudney et al. 2000).

expected from a crime dominated by youth (Grogger 1998). That is, potential young criminals are highly sensitive to labor market opportunities as a result of their marginal status in such markets. Another explanation for the stronger measured sensitivity of wildland arson is that wildland arson is a narrowly defined crime type, while other crimes (e.g., larceny) are actually collections of crimes of a wide range of differences. Statistically, the result can be that the effects of individual variables are attenuated by errors in variables bias (Greene 1990). As well, wildland arson (like motor vehicle theft) appears to show no deterrent effect from stiffer sentencing. Aside from this, our results for arson and other crimes parallel findings from Gould et al. (2002) regarding labor market conditions: criminal activity responds to both unemployment and wages.

In summary, (1) wildland arson responds in the same way to many of the same variables as other crimes, and (2) wildland arson appears more responsive than other crimes to these variables. It is with robbery that it apparently shares the most common statistically significant relationships with explanatory variables, although similarity was not statistically tested. Still, greater confidence in our results for these similarities and differences could be obtained from more spatially detailed analyses of these crimes—perhaps at the county level, and for other places—and we believe that this would be a fruitful topic for additional study.

Data have shown that wildland arson has trended downward in Florida since the early 1990s, and the research reported here has identified some of the apparent socioeconomic underpinnings to these trends. Greater certainty about why arson has trended downward in Florida, after accounting for socioeconomic variables, might require more detailed analyses. These analyses would control for fuels levels, which might have been altered through fuels management programs in the state. Are these reductions statewide a function of wildland changes, fuels management, or in fact explained by the arrests and convictions of key individuals? If most arson fires are serial or copycat, catching the serial arsonist or the first arsonist at the beginning of a potential copycat string would have a relatively large impact on arson rates. If there has been success in this arena in Florida, it could have been enabled by new policing tactics and technologies that aid in locating and catching criminals. The negative trends found for other crimes lend weight to this hypothesis, we believe.

7. CONCLUSIONS AND MANAGEMENT IMPLICATIONS

The survey of research presented in the first five sections of this chapter show that wildland arson behaves in patterns of spatio-temporal clustering and exhibits temporal regularity on daily and intra-annual scales that are similar to patterns found for other kinds of crimes. Data presented on national forest arson rates and those for Florida show that long run trends in wildland arson are similar to long run crime trends, as well. The empirical research conducted for this chapter shows that wildland arson in Florida appears to have undergone long run changes that can be explained by the same factors that explain such changes for other major crime categories but that it has responded more sensitively to these factors than have other crimes. Hence, wildland managers and law enforcement should expect this crime to respond strongly in the future to changes in the variables expected to affect other major crimes. In other words, after accounting for weather and fuels, managers and law enforcement should expect that rising (falling) rates of other crimes would correspond with (or even be predictive of) rising (falling) rates of wildland arson.

Given the available research and the empirical results of this chapter, we conclude that wildland management and law enforcement actions are not the only variables explaining wildland arson rates. Arson ignitions also are responsive to weather and climate, and labor market variables. Changes in law enforcement and labor markets can explain much of the underlying trends in observed arson. Taking the change in the retail wage rate only (holding other variables constant) as an example, using the parameter estimates shown in table 7.1, and assuming that the statistical relationships found reflect causality, if these wages in 2001 were at the (lower) level experienced in 1982, then wildland arson ignitions per capita would have been 93 percent higher in 2001 than they actually were. Similarly, if the unemployment rate in 2001 were at the same level as in 1982, ignitions per capita would have been 45 percent higher.

Labor market, law enforcement, weather and climate patterns, and other socioeconomic variables do not tell the whole story of the long run changes in wildland arson. Remaining negative trends in wildland arson (and other crimes), after accounting for those other factors, lead us to believe that other factors should also be credited with reduced rates of wildland arson in Florida, and potentially for wildland arson on national forests in aggregate in the United States. Possible among these are rising rates of fuels management, efforts to reduce volunteer firefighter arson rates through special programs, and rising arrest rates of key individuals. Again, if most arson fires are serial or copycat, then catching the serial arsonist or the first arsonist at the beginning of a potentially copycat string would have a potentially large impact on wildland arson rates. If true, perhaps the negative residual trend in wildland arson seen in Florida, at least, is due to improving policing tactics and technology.

Based on the existing research, it appears that there are several possible avenues of attack against the problem of wildland arson. These include: (1) catching arsonists early and often by identifying arson hotspots in space and time, moving police into hotspots and into areas with higher overall wildland arson rates during those hours of the day, days of the week, and months of the year when wildland arson is most likely; (2) increasing police aggregate levels, which could enhance deterrence and raise the arrest rate; (3) reducing hazardous fuels levels; (4) monitoring underlying socioeconomic drivers of wildland arson and the rates of other crimes, which can be used as predictors of wildland arson rates in advance of a coming fire season; (5) monitoring underlying weather and climate drivers affecting wildland arson success, which can sometimes be predicted in advance; following Cabe (1996), (6) working with fire departments to reduce volunteer firefighter-set fires; and, for policy makers, (7) seeking means of expanding labor market opportunities in rural, fire prone areas. Based on our statistical analyses and the literature, changes in variables affecting wildland arson also should affect rates of other crimes, indicating complementarities and trade-offs among crime types and wildland arson. Incumbent upon wildland arson researchers is to demonstrate the effectiveness of each approach in the context of the larger picture of crime.

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SECTION III

VALUING THE ECONOMIC IMPACTS OF FOREST DISTURBANCES

CHAPTER 8

DESIGNING ECONOMIC IMPACT ASSESSMENTS FOR USFS WILDFIRE PROGRAMS

Karen L. Abt, Robert J. Huggett, Jr., and Thomas P. Holmes

1. INTRODUCTION

As often happens in the wake of a series of extreme fire seasons, such as those in 2000, 2002 and 2003, federal wildfire policy is being scrutinized and recommendations regarding changes both large and small are prevalent (Stephens and Ruth 2005, Busenberg 2004, Dellasalla et al. 2004, Dombeck et al. 2004). It is common practice for increases in acres burned and in suppression costs to be cited as evidence that existing policy is a failure and that changes must be made. For example, Busenberg (2004) argues that "the wildfire crisis in America was created by a longstanding policy failure" which "greatly increased the risk of wildfire damages."(p. 145). However, there is scant empirical evidence regarding the magnitude of total economic damages (much less, the benefits) resulting from wildfire, and empirical evidence that would permit an overall evaluation of wildfire programs is limited.

Stephens and Ruth (2005) make suggestions for reducing the trend in wildfire acres burned, which begs the question of whether the objective of federal wildfire policy is to minimize acres burned, economic impacts, or some other measure. Although the 2001 Federal Fire Policy has as its primary tenets the protection of life, property and resources, it does this through 9 guiding principles, 28 findings, 17 policy statements and 19 implementation actions (United States Interagency Federal Wildland Fire Policy Review Working Group 2001). Thus, despite a clear statement about protecting life, property and resources, the firefighting agencies are often faced with determining priorities in the face of multiple guiding statements that may imply contradictory objectives. One interpretation of this policy could be an objective of minimizing acres burned, although this results in treating all acres as equal in value, whether they are endangered species habitat, wildland-urban interface, or some other designation.

In the past, prior to the current epoch of increasing fuel loads and the expansion of the wildland urban interface into fire-prone areas, and with easier success in suppression, a goal of minimizing acres burned may have been synonymous with minimizing damages. Over the last 100 years, however, changes in suppression

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success in conjunction with increases in the values at risk have likely led to a divergence between economic damages and acres burned. Certainly, it is apparent that at least the largest and most well known fires are damaging (Kent et al. 2004, Butry et al. 2001, Franke 2000), but data are insufficient to identify trends in local, regional, or national impacts.

An analysis of the costs and losses associated with any natural disaster will be influenced by the inclusiveness and scope of the cost and loss categories used to conduct an assessment. In particular, an economic assessment will be sensitive to the spatial scale (geographic area to be assessed), temporal scale (time span used to assess impacts), and sectoral scale (economic sectors included). Further, programmatic scale issues derive from differences in evaluating the costs and losses of an *event* as compared to the costs and losses of a *program*. Finally, economic costs of individual wildfire events and wildfire programs are important not only because of the magnitude of the costs and losses, but also because these events and programs will have distributional consequences, influencing who gains and who loses from each event and program (chapter 9 of this book). This chapter discusses design of economic impact assessments for natural disasters, describes a feasible design for wildfire programs, and suggests immediate improvements to data collection that could enhance the ability of the U.S. Forest Service to evaluate trade offs for private property owners and public land managers.

2. ECONOMIC IMPACT ASSESSMENTS OF NATURAL DISASTERS

2.1 Defining Economic Impacts: Costs, Losses, Benefits, and Damages

Over the last decade, evaluations of empirical methods to assess the costs of disasters have been conducted by three organizations: (1) the Bureau of Transport Economics of Australia (BTE) (2001), (2) the Economic Commission on Latin America and the Caribbean (1999) (ECLAC), and (3) the National Research Council (NRC) (1999). Each of these evaluations promotes a slightly different method of tallying the costs, losses, impacts and damages of disasters, but the overall intent of the evaluations was to provide a consistent method for tallying disaster costs. We discuss the classifications of costs and losses recommended by these studies, noting where an evaluation of a wildfire program would be different from the evaluation of nationally or internationally designated disasters.

In the United States natural disasters are defined by either the insurance industry or by presidential proclamation. The Property Casualty Services unit of the Insurance Service Organization, an industry group, began collecting data on disasters, which they defined as an event with over $1million in insured losses, in 1949. The dollar limit increased several times to $5 million in 1983, and was most recently set at $25 million in 1997. The second determination, allowed under the Stafford Act (P.L-93-288) passed in 1988, is a presidential proclamation of disaster, which allows federal resources to be used for assistance and reimbursement of local, state, and uninsured damages and costs. Few wildfires, and no non-fire program activities, have been classified as disasters, and thus would not be tallied under a disaster evaluation program. However, although each event may be small, we still need to know cumulative impacts in order to address the trade-offs inherent in developing a wildfire program.

The terms costs, losses and damages are used in the BTE, ECLAC, and NRC evaluations of disaster costs similarly, and are consistent with the cost plus loss (least cost plus net value change) model traditionally used for assessing wildfire suppression (chapter 16 of this book). Economic impacts of a wildfire program will include both market (e.g., timber) and non-market (e.g., water quality and quantity) effects. One component of market effects is costs—expenditures made by agencies or individuals to directly influence the wildfire program or recover from a wildfire event. Costs include suppression expenditures, as well as disaster aid, rehabilitation expenditures and pre-fire treatments and activities.

Two types of damages, direct and indirect, are identified in the 3 listed reports, which can be either monetized (also referred to as losses) or nonmonetized (e.g., intangible losses). Direct damages are the physical assets destroyed by a catastrophic event and are typically measured in monetary terms. Indirect damages are the subsequent, or downstream, effects of the disaster on the rest of the economy. These downstream effects include losses in production and gains due to reconstruction and rehabilitation.

Damages to environmental assets may be of more importance in evaluating wildfires than in evaluating other natural disasters such as earthquakes or tsunamis. These can include damages to soil, water, cultural resources and wildlife habitat. Suppression efforts themselves have also been identified as a source of environmental damage (Backer et al. 2004) as have timber salvage activities (McIver and Starr 2001). The three studies disagree regarding whether losses to environmental assets are considered direct losses (loss of capital) or indirect losses (loss subsequent to the event).

In addition, while all three studies refer to intangible losses, and the potential significance of these losses, they acknowledge that there are no methods for computing either the values or quantities of these losses. These intangible losses include loss of memorabilia, sense of trauma or fear, and loss of sense of place. Indirect damages are also difficult to quantify, and there is some evidence that these downstream damages may be less important for wildfires because wildfires rarely destroy major economic infrastructures in the manner of disasters such as earthquakes and floods (NRC 1999). There are also potential positive impacts from wildfire that are rarely quantified, even though these effects are one reason that the behavioral model is now referred to as cost plus net value change rather than cost plus loss.

2.2 Scale

The disaster cost plus loss tallies such as those suggested by the NRC, BTE and ECLAC are specifically designed to address individual events, not a land management program which happens to include events that may end up classified as disasters. In the case of wildfire and other forest disturbances, damages and benefits will accrue from both the events themselves and from the mitigation and rehabilitation efforts, and will accrue each year, whether activities and events occur or not. Thus, tallies such as those recommended by NRC, BTE and ECLAC do not adequately address a program such as that used by federal agencies for all wildfire activities. Expanding these tallies so that data are recorded for all events in a program (including prevention, presuppression, suppression, and recovery and rehabilitation) would require substantial, and unavailable, investment by the land management agencies. Yet, without addressing programs as a whole, the usefulness of these tallies will be limited to addressing single questions rather than overall program goals. Agencies conducting tallies of detailed costs and losses for individual fire events will need to determine if the agency and public would be better served by a broader assessment of the economic impacts of a program, or if they will continue to place energy and funding to tallying details of only a select few events.

Program evaluation is complicated by the fact that it requires calculating the interactions and trade-offs between the various activities of the program. For example, the impacts of a wildfire on life and health are undeniably negative. This does not lead to the conclusion that wildfires are to be avoided, unless, of course, the consequences and costs of avoiding wildfires are also assessed. Evaluating the impacts of the wildfire *program* on life and health, however, will require assessing the health impacts of prescribed fire (which may be different than wildfire), mechanical fuel treatments (logging is still a dangerous occupation), and wildfire impacts under different suppression scenarios (e.g., full suppression, wildland-urban interface only, increased use of wildland fire use fires).

The geographic, temporal and sectoral scales of an assessment will affect the total measured outcome. It is possible that effects of a natural disaster may be close to zero if the measured part of the economy is large enough (NRC 1999). Similarly, impacts will differ if the geographic area of the analysis is small or if the time span of the analysis is short. If the area of impact is the nation or state, the effect of any single wildfire event or even the total program will be dwarfed by the size of the economy. Geographic trade-offs will occur in nearly all market sectors, where timber prices may influence adjacent markets, and tourism may be redirected to adjacent recreation areas, resulting in gains in areas otherwise unaffected by the wildfire. In this case, for a large geographic area, the only losses that may result are from additional costs incurred to travel to the new location.

If only the immediate effects of a wildfire program are measured, the assessment might easily exclude potential benefits from a fire or treatment (such as improved ecosystem health). Likewise, certain damages (such as later flooding or water quality degradation) might be omitted from a short-term assessment. Thus, a time scale appropriate for the type of each activity or event must be used to correctly evaluate overall impacts. The sectors to include in the analysis will also influence the outcome, especially as there are often gains in one sector or part of the market even as there are losses in another sector.

2.3 Distributional Impacts

Although tallies of costs and losses are important for current and future economic analyses, an optimally designed economic assessment would also include information that would allow the distributional impacts of wildfire program costs and benefits to be evaluated (Holmes et al. 2007). Due to the complex interactions of weather conditions, fuel loads, and topography that affect wildfire management decisions, it is unlikely that fire suppression decisions fully reflect the consequences of a fire event from the perspective of households with various income, age, ethnic, or racial characteristics. If some socio-economic groups are more likely to reside or work in locations with a high fire risk, then they would be more vulnerable to potential losses from a fire event. Likewise, if some groups have a lesser ability to recover economic losses from a catastrophic fire, either because they are uninsured or have a lesser ability to receive disaster assistance, they would have greater vulnerability to long-term economic losses.

Most research evaluating the linkages between demographic characteristics and the severity of impacts from natural disasters has been conducted in the context of low-income countries (Morduch 1994). Within the United States, Bolin and Bolton (1986) evaluated the role of race, religion, and ethnicity on the ability of households to recover from natural disasters in four different case studies. They concluded that poor families and large families have the greatest difficulty acquiring aid and recovering from a natural disaster. They note that, because members of ethnic minorities, particularly Hispanics and blacks, are more likely to belong to such families, these ethnic groups are more vulnerable to natural disasters. This conclusion is echoed in the sociological review conducted by Fothergill and Peek (2004) who found that, within the United States, the poor are more vulnerable to losses from natural catastrophes because of their location decisions, poorer quality housing, less frequent purchase of insurance, and lesser ability to travel the bureaucratic pathways necessary to claim disaster assistance.

We are unaware of any studies that have specifically evaluated the relation between demographic groupings and the economic impacts of wildfire related damages. However, rapid population growth in fire-prone regions of the wildland urban interface, combined with the structure of the local economies in these areas, suggest that such studies may be warranted. Johnson and Beale (1994) reported that, during the 1990s, the fastest growing counties in the United States were non-metropolitan counties that were destinations for retirement-age migrants or were outdoor recreation centers. Because service industry jobs in the outdoor recreation and tourism sector generally provide lower levels of income than other sectors of the economy, the impact of income inequality on the ability to recover from wildfire damages may be an emerging issue in some fire prone communities.

A second distributional concern is that the provision of disaster relief by the federal government creates what economists call a “moral hazard”. By offering financial assistance to insured and uninsured households and businesses in the wake of a natural disaster, disaster relief lowers the recovery costs faced by people who voluntarily choose to locate in high hazard areas. This moral hazard creates an economic incentive to locate in hazard prone areas (Shughart II 2006). Further, it has been argued that both presidential and congressional politics affect the rate of disaster declaration and allocation of recovery expenditures (Garrett and Sobel 2003). These findings raise questions as to whether federal disaster recovery funds are reaching the people in greatest need of assistance.

3. FEASIBLE ECONOMIC IMPACT ASSESSMENT FOR WILDFIRE PROGRAMS

Without adequate information, landowners and land managers can not make the best decisions. Risk analyses, optimization models, and program assessments of varying degrees of detail have the potential to provide better information for both land management agencies and for homeowners, reducing economic losses associated with both property owner response to wildfire risk (often presumed to be inadequate) and land management agency response, variously assumed to be excessive (if one is paying the bills) or inadequate (if one’s home was destroyed by wildfire).

A complete model of an economically optimal wildfire program maximizes net social welfare summed across all participants and over time. Such a model would include values for all market and non-market products, services and attributes; incorporate ecological tradeoffs between wildfire, prescribed fire, fuel treatments, logging and grazing; recognize how suppression influences fires and affects forests; and incorporate climate and weather linkages to fire, suppression and forest regrowth. Developing data sufficient for this type of model across all ownership types, temporal and spatial scales, and wildfire programs is overwhelming and likely prevents realistic optimization in the near future.

It is feasible, however, to develop assessments of economic impacts (including damage estimates) that address policy issues, even if the data are not sufficient to develop a fully specified cost+loss model. These assessments can help land management agencies determine the appropriate level of suppression as compared to fuel treatments, prevention, prescribed fire and other land management. Further, they can help landowners determine the appropriate level of insurance and averting behavior. In section 4 we suggest an immediate economic impact assessment that could be implemented within the current data structure with few changes. In the remainder of this section we describe a more fully-specified feasible economic impact assessment.

Four analyses of large, recent wildfires are used to illustrate the fire-only components of a feasible economic impact assessment, and to illustrate where additional research might be needed before components are suitable for inclusion. These wildfires are Florida 1998, Hayman 2002, California 2003 (selected fires), and Northern Rockies 2000. Table 8.1 summarizes the values derived from these assessments. The tallies are inconsistent due to the fact that different attributes were significantly affected in each of the fires and different methods were used to estimate the various impacts. This table shows the total economic impact and the percentage of the valued total that was attributable to each loss category. These totals and percentages, combined with our understanding of the time involved in evaluating some of these losses, contributed to the feasible assessment design.

Certain losses caused by wildfire, such as those from watershed impacts, tourism and recreation impacts, health impacts, and the damage and destruction of insured property need additional research to ensure consistent and reliable estimation of each impact's value. These damages and losses will take significant time to determine even after accepted methods are developed. However, delaying development

Table 8.1. Economic impact assessments of four recent wildfires.

	1998 Florida	2000 Northern Rockies	2002 Hayman Colorado	2003 Old, Grand Prix and Padua
Damages				
Size of fire(s) (acres)	500,000	3,104,000	138,000	161,175
Structures destroyed #				
Residential	340	135	132	1,130
Commercial	33	5	1	11
Outbuildings		325	466	60
Human losses				
Deaths #		4	5	6
Injuries #		14	3	

Costs + Losses	mm$	% of total	mm$	% of total	mm$	% of total	mm$	% of total
Loss of structures and contents	12	2%			39	25%	576	50%
Loss of timber	480	64%			0.036	0%		
Suppression costs	100	13%	378	100%	43	28%	61	5%
Disaster relief costs	22	3%			6	4%	45	4%
Watershed costs and losses					66	43%	478	41%
Health costs	0.52	0%						
Tourism costs	138	18%						
Total costs plus losses	753		378		154		1,160	

of an assessment program until these issues are resolved could postpone an evaluation of damages and trends from wildfire programs for many years.

Note that many of the entries in table 8.1 are left blank. These values or numbers were not found in the studies we used. This blank entry could represent a 0, or perhaps it was not possible to estimate this value, or the value may have been estimated by others and thus not included in the economic analysis. This illustrates the difficulty in deriving total wildfire impacts, let alone total wildfire program impacts, by using estimates from the few fires that were deemed worthy of additional analysis. These elements are discussed further below.

3.1 Impacts Included in the Feasible Design

3.1.1 Agency expenditures (all activities including suppression)

Although there is substantial discussion and importance placed on suppression expenditures (chapters 13, 15, 16, and 17 of this book, for example) these expenditures averaged only 21 percent of the total cost-plus-loss for the 4 assessed fires (table 8.1). They are, however, of critical importance to the agencies faced with limited budgets and increased pressure to reduce costs. Accurate tallies of these expenditures, both for suppression and other wildfire programs, are also critical for determining trade offs between different activities such as prescribed fire and wildfire, or mechanical treatments and prescribed fire. While there are significant issues associated with this data it is relatively easy to collect, consistent and reliable.

3.1.2 Natural resource impacts (excluding timber, all activities)

Damages from suppression and from activities such as prescribed fire and fuel treatments are necessary for evaluation of wildfire programs, and some estimate of these losses and benefits may be attainable. These tallies, however, could always be presented in physical terms, with values in dollar terms provided where available. The development of valuation estimates for natural resource damages is difficult and time consuming, and is unlikely to be available for all wildfire program activities, but could be presented where available. None of the four studies presented in table 8.1 show these impacts.

3.1.3 Timber (all activities)

Earlier versions of the USFS Wildfire reports (FS 5100-9) included an estimate of timber value destroyed. Timber values destroyed and damaged could be included for areas where commercial timber harvest is still a viable economic activity. Butry et al. (2001) provide a welfare theoretic method for assessing these values in detail, but for most fires a simple estimate of volumes destroyed and volumes damaged but salvageable could be included. Timber comprised an average of 20 percent of all costs+losses recorded for the four sample fires, but variations from fire to fire are extreme. In addition, the methodology used varied depending on this level of importance. For example, for the Florida fires, the calculation included losses and gains to both consumers and producers in all sectors, while the Hayman estimate represents only the total loss to the USFS from timber sales (primarily firewood and Christmas tree sales). There was an additional estimate of total timber value destroyed on the Hayman of $34 million which appears to be based on projected volume destroyed times average price, but is not related to actual or projected timber harvested on the affected area.

3.1.4 Human life and injury (all activities)

Although human life and injury is number one on the list of federal fire policy objectives, the USFS makes a limited effort to tally the effects of wildfire, and especially the effects of a wildfire program including treatments, on human life and injury. OSHA maintains records by job category, but the detail needed to link these to presuppression, initial attack, wildfire, wildland use fire, or prescribed fire are not available. For the other program activities, it is similarly difficult to determine if fatalities and injuries result from traditional logging or fuels treatments. This information is crucial to developing reliable economic impacts, especially in view of the importance given to this objective in federal wildfire policy. Human fatalities and injuries are, however, generally available for large and damaging fires and these numbers are displayed in table 8.1 where available.

3.1.5 Threatened and evacuated structures (wildfire and escaped prescribed fire only)

Calculating the number of threatened and evacuated structures may be difficult, but is important for determining the negative effect of wildfires on communities, and for determining the positive effect of suppression on reducing damages. Knowledge of the potential size and damages of a fire without suppression is unattainable, but the threats to development will provide some information on these potential damages. Evacuations are ordered by neighborhood or street, and local governments may have accurate numbers of dwellings in a neighborhood. Commercial evacuations may also be available from local governments. The number of threatened structures is a core element of a post-fire assessment of values at risk. Currently, there is little guidance regarding what constitutes a ‘threatened’ structure. Evacuated structures can be classified as threatened, but additional research and discussion are needed to develop a more precise measure of ‘threatened’ areas, be they acres or structures.

3.1.6 Infrastructure destroyed or damaged (wildfire and escaped prescribed fire only)

Damages to major infrastructure, such as highways, communications facilities, recreational areas and electric power lines, could be recorded. These damages are usually less than structural damages, but could be critical to recovery and rehabilitation efforts. These could be recorded as dollar values whenever possible.

3.1.7 Structures destroyed or damaged (wildfire and escaped prescribed fire only)

Standards could be developed and used to determine whether a structure is destroyed or damaged, and levels of damage could also be included based on the percentage of total value destroyed. It is critical to make distinctions between types of structures, because the loss of an outbuilding is not likely as important as the loss of home or business. The preliminary and final reports for the 2000 Northern Rockies fires both report that 465 structures were damaged, but only the preliminary report provides the detail that 135 homes and 5 businesses were destroyed, the remainder were outbuildings (table 8.1). A protocol could be developed to clarify the use of terms representing the type of structures destroyed rather than continuing to refer to the all-encompassing 'structures lost' which can be misleading.

3.2 Impacts Requiring Additional Research

3.2.1 Watershed impacts (all activities)

One particular impact of wildfire is on municipal watersheds—leading to two distinct outcomes. First, is the change in the quality of water produced for municipal use from increased sediment, nutrients, and salts. Second is the change in the quantity of water, leading to flooding and mudslides. Municipal water managers must address both of these, and there may be substantial costs associated with both the quantity and quality changes resulting from the fire. As of yet, however, the data are not available to consistently estimate the costs of fire on watersheds.

Few assessments attempt to value watershed impacts of fire. Dunn (2005) included an estimate from the 2003 fires in the San Bernardino Mountains in Southern California. Estimates from the Santa Ana Watershed Project Authority, Natural Resources Conservation Service and others amounted to $478 million, nearly 8 times the estimate for suppression expenditures and 83 percent of the estimate for structural losses (table 8.1). Making programmatic decisions based on these impact estimates could lead to the conclusion that only slightly more of our suppression effort should be directed at structural protection than at watershed protection. However, including these damage and restoration estimates as stated is questionable due to the unknown methodologies and assumptions used in their construction. In addition, a full programmatic assessment would require estimates of the impacts on water quantity and quality from other program events, such as prescribed fire, mechanical treatments, and wildland fire use. We recommend that additional research on the costs and values of the impacts of the wildfire program on municipal watersheds be conducted before these estimates are included in wildfire program tallies of costs and benefits.

3.2.2 Tourism and recreation impacts (all activities)

Locally, wildfires and prescribed fires may have significant effects on immediate (fire-year) recreation and associated tourism expenditures. Documentation of declines in tourism expenditures (Butry et al. 2001), outfitter and guide trips (USDA Forest Service 2001), and national forest visits (Kent et al. 2003) indicates that for some market participants the effects could be significant. These effects, however, may be mitigated in the larger economy by the substitution of other recreation sites for the fire-affected sites (Kent et al. 2003). Medium-term (1-5 years) effects are also uncertain, with some studies suggesting losses and others finding increases subsequent to the fire season, presumably by curiosity-seekers (Franke 2000, chapter 10 of this book). In situations where fires dramatically alter ecosystem attributes, the dynamics of forest regeneration and recovery may continue to induce long-term (spanning decades) declines in visits to affected areas (chapter 10 of this book). Substitution patterns over space and time appear to be rather complex, suggesting the need for future research.

The issue of substitutability between recreation sites and activities can be seen in the varying results from the four fires evaluated in table 8.1. The large negative values from the Florida fires (Butry et al. 2001) assumed that all tourism was lost, and no substitutes were available. In contrast, Kent at al. (2003) assumed that substitutes were available and used, resulting in a much lower loss estimate. Direct effects (losses occurring from closures and/or destruction of property) could be separated from indirect effects (losses occurring later because of publicity or effects on the resource that attracted the tourism in the first place). We recommend that additional research be conducted on these issues regarding recreation and tourism impacts of a wildfire program before efforts are made to include these data in economic impact tallies.

3.2.3 Insurance values and losses (wildfire and escaped prescribed fire only)

Currently, tallies of total insured losses are available only for select wildfires, usually the largest and/or most damaging. The Insurance Service Organization (ISO) gathers data from all insurance companies, but this information is not available free of charge. In addition, the records do not always distinguish between wildfire and structural fire as the cause, unless the fire is considered a disaster (exceeding $25 million in losses). It may be possible to work with insurance organizations to develop reporting that would be useful to both the ISO and to the agency. Once insured losses are known, a simple conversion is usually used to derive total losses, including uninsured, deductibles and underinsured costs.

One additional issue remains with collecting and utilizing insurance losses for use in an assessment. Because the access to insurance differs across economic, social, and demographic strata, reliance on this aggregate level of values information alone may mask differential equity effects. While a complete tally of costs and benefits would measure the values at risk in order to compare these values with the costs of protecting these values, the inequities inherent in these value-based analyses must also be addressed. In some respects, the number of dwellings and commercial buildings destroyed, damaged and threatened may be equally appropriate as a measure of economic impact. Tallies of types of structures damaged/destroyed must always accompany any structural dollar loss totals.

3.2.4 Other health impacts (all activities)

At this time, data are not readily available for estimating total health impacts from wildfire programs. The Butry et al. (2001) analysis of the Florida wildfires included a monetized assessment of the costs of smoke from the wildfire. More recently, Rittmaster and others (2006) present a method for estimating the health impacts of elevated particulate matter associated with a wildfire in Alberta, Canada. They report that the economic impacts are substantial and only second to the impacts on timber. We recommend that additional research be conducted that would allow estimation of these impacts for all wildfire program activities.

4. IMMEDIATE USFS ECONOMIC IMPACT ASSESSMENT FOR WILDFIRE PROGRAMS

Within the USFS, and through other federal agencies, we have various systems to record data on fires, but these are primarily oriented toward tallying suppression efforts and suppression resources used, and to documenting the path and course of the fire itself. And many of these data are collected only for large fires (greater than 100 acres). In addition, while the databases often allow for entry of information on specific suppression activities or on threatened structures, these entries are not required. Prudent data entry personnel would not likely allocate time for optional entries, particularly when there is inadequate time for the required entries. Even so, this information on damages is necessary to understand trade-offs between the various wildfire program elements, over space, and through time. The USFS could begin acquiring the necessary information by requiring the collection of the following information:

1. Require that all wildfire program events (fuel reduction treatments and prescribed fire) be recorded, including all information possible, similar to the recording of wildfire events done currently including at least location, acres, costs, fuel model, and start and end dates. Additional fields to record the type of treatment could be added.
2. Require that firefighter and non-firefighter (including civilian) deaths and injuries be recorded for all wildfire program activities.
3. Require that evacuations and threatened, damaged and destroyed residential and commercial structures be recorded for wildfires and escaped prescribed

fires. Develop precise and understandable criteria for determining what constitutes a threatened, damaged or destroyed structure and how to measure evacuations.

4. Require corrected agency expenditures, accounting code and acres affected (for wildfires—acres burned by a predetermined classification of severity or intensity; for other activities—total acres only).

5. Require a list of affected communities, perhaps by zip code, name or census tract. Population, income and other demographic variables in destroyed, damaged and threatened areas can be determined subsequent to a fire provided the spatial extent of the affected areas is recorded.

We believe that a credible and useful immediate impact assessment for wildfire programs could be developed if these 5 suggestions are immediately implemented.

5. CONCLUSIONS AND SUGGESTED RESEARCH

Over the last 15 years, trends in wildfire acres burned and suppression costs have increased and have become increasingly volatile. While intuition, common sense and anecdotal evidence indicate that damages are also increasing, data are insufficient to develop trends for economic impacts and damages. Many post-fire reports and analyses have been produced, each addressing the issues important to that fire or season. These reports are produced by different groups or agencies, and there currently is no single location where data on economic impacts from wildfire are archived. Evaluations of damages are typically conducted when some unusual event occurs, such as an escaped prescribed fire (Cerro Grande in 2000), higher than average suppression costs (Biscuit 2002), large numbers of homes destroyed (California 1991, 1993, 2003), or widespread fire seasons (USFS Northern Region 2000, Yellowstone 1988, Florida 1998). Many other large fires, some equally damaging, have received little attention, and small fires, even if they result in loss of life or structures, receive no economic impact analyses at all. In addition, the other interrelated components of a wildfire program, including fuel treatments and prevention, are not recorded in the same manner. Thus, we have inconsistent data for the fires collected, inconsistency in the reporting, and inconsistency in data accessibility. And, while Emerson (1841) eschews “a foolish consistency” as the “hobgoblin of little minds”, scientific analyses and decision making at both the property owner and governmental level wisely require consistent and available data.

While data on numbers of ignitions and acres burned are of crucial importance to land managers in preparing for upcoming fire seasons, without similar values for structural and other damages, neither private landowners nor public land managers will have the necessary information to develop optimal responses to the risk of wildfire damages. Accurate, well-defined entries for the number of destroyed, damaged and threatened structures will be an important step in developing an adequate economic impact summary for wildfires.

Developing data sufficient to model economic optimization across ownership types, over time and for overall wildfire programs requires data on wildfires as well as on other land management activities with direct links to wildfire occurrence and severity, including pre-suppression (initial attack), fuel treatments (both mechanical and prescribed fire), fire prevention programs, and changes in external hazards such as the wildland urban interface and climate. Costs that could be assessed include financial costs to agencies, businesses and individuals, and all losses to capital including buildings, other infrastructure, human life and injury, and ecosystems. Losses to ecosystems from suppression and the positive effects of wildfires and other program components would also be included. In addition, data on external influences, such as insurance, population and demographic variables would be needed to fully evaluate trends in economic impacts.

The cost, damage and benefit data for all wildfires would not be confined to large fires (>100 acres) or disasters (more than $25 million in damages). Cumulative impacts and damages from small fires could be considerable, and if a wildfire program is successful, less damaging individual fires may become the norm. Second, structures damaged, destroyed and threatened, as well as structures evacuated would be collected in conjunction with other existing fire records. Third, lives lost and serious injuries must be recorded for all fires. Fourth, acres burned by a predetermined classification by severity will assist in developing loss and damage estimates for non-market or non-quantified attributes. Fifth, other program elements would have the same degree of detail as included for wildfires, perhaps by using the fire records database to include prescribed fire and other fuel treatments data.

The USFS could implement a basic improvement to their data collection that would substantially improve our ability to assess economic impacts and damages from fires. This would begin the process of developing data necessary for understanding trends in damages and impacts. Without this information, we run the risk of making changes to a program that could be worse than continuing with existing programs.

Beyond these changes, additional research needs to be conducted before some costs and benefits of wildfire events can be included in numerical tallies. Suggestions for further research includes (1) evaluate losses to recreation and tourism resulting from wildfire programs, specifically addressing substitution, and considering the endemic nature of fire in ecosystems, (2) evaluate costs, damages and benefits to watersheds resulting from wildfire programs, (3) evaluate health and fatality impacts resulting from wildfire programs and (4) evaluate the effect of wildfire programs on insurance and distribution of wealth, and the effects of wealth and insurance on wildfire programs. For each of these, it is imperative that the analyses be conducted to include wildfire, fuel treatments and prescribed fire and that a multi-year approach be taken.

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CHAPTER 9

TIMBER SALVAGE ECONOMICS

Jeffrey P. Prestemon and Thomas P. Holmes

1. INTRODUCTION

It could be argued that perhaps the most dismal sub-discipline within the dismal science of economics is salvage economics. In the wake of catastrophic events such as pest epidemics, storms, or fires, forest managers make complex and often controversial decisions about scavenging value from wounded, dead, or dying trees. For profit-maximizing landowners, salvage decisions must balance the cost of harvesting operations in difficult conditions with the revenue obtainable from damaged timber. On public forest lands, salvage decision-making is further complicated by the fact that managers need to consider trade-offs between the net value of timber extracted and the impact of salvage activities on a suite of ecosystem services that are valued by people.

Prior research has shown that, in aggregate, salvage provides short-run benefits to timber market participants (Holmes 1991, Prestemon and Holmes 2004) and helps to mitigate long-term timber value losses (Prestemon et al. 2006). Some have argued that substantial timber market benefits can be obtained while incurring only minor impacts on non-timber values (Sessions et al. 2003). Others would argue that catastrophic events are intrinsic to the normal functioning of natural systems and that salvage activity can be detrimental to biogeochemical processes and other ecosystem functions that occur after a natural disturbance (Foster and Orwig 2006, Lindenmayer and Noss 2006). Timber salvage has the potential to alter natural post-disturbance plant associations, introduce invasive species, decrease the available habitat for certain bird species, increase erosion, and reduce water quality (McIver and Starr 2001, McIver and McNeil 2006). The view that the net timber market benefits of salvage on public lands are outweighed by these and other non-timber value losses may induce organized resistance by stakeholder groups.

Previous research regarding the economics of timber salvage has occurred at two scales—the firm level and the aggregate market level. Beginning with Martell (1980) and Reed (1984), firm level models describe the salvage decision from the perspective of individual landowners in a Faustmann-type framework and address the question of optimal timber management in even-age stands subject to the risk of catastrophic loss. This modeling framework has been extended

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to include optimal rotation decisions in the presence of fire risk on multiple use forests (Englin et al. 2000) and the impact of intermediate fuel treatments and initial planting densities on salvage values if a fire occurs (Amacher et al. 2005). Timber salvage market models describe the economic impacts of aggregate, large-scale salvage operations on prices and the economic welfare of timber market participants. Beginning with Holmes (1991) these models use time series analysis and economic welfare theory to identify market impacts and transfers in economic welfare. Short-run price impacts have been identified for southern pine beetle epidemics (Holmes 1991) and hurricanes (Prestemon and Holmes 2000, Yin and Newman 1999). In addition, Prestemon and Holmes (2000, 2004) identified long-run price and welfare impacts due to substantial changes in timber inventories. Market-level analysis is aimed at governmental decision-makers whose salvage programs can affect market prices and quantities.

The goal of this chapter is to provide the reader with an overview and working knowledge of the main topics in timber salvage economics. The following section of this chapter describes how large scale natural disturbances affect timber markets and timber market participants. This is followed by a discussion of the role of timber salvage in private and public landowner decision models. To provide a concrete example of the methods described in this chapter, we include a case study of the timber market effects of a recent, large disturbance—the Biscuit Fire of 2002.

2. TIMBER MARKET IMPACTS OF SALVAGE

In the wake of a large scale forest disturbance, timber markets demonstrate a discernable price decline due to a pulse of salvaged timber entering the market and may also manifest longer run effects if timber inventory losses are large. The salvage price effect occurs immediately after the disturbance, as affected landowners rush to harvest as much damaged timber as possible in order to avoid additional decay-related losses in quality and volume (Holmes 1991, Prestemon and Holmes 2000, 2004). In contrast, the price and quantity impacts due to losses in timber inventory can last much longer than the salvage period and depend upon the growth rate of the subsequent inventory.

The effects of the salvage and inventory losses can be illustrated with a supply-demand graph. Figure 9.1 shows a demand curve (D) and three supply curves that represent the three main epochs of timber market conditions following an inventory-destroying large scale disturbance. Pre-disturbance equilibrium price (P_0) and quantity (Q_0) is located at point *a*. Consumer surplus is defined as the area bounded by the demand curve from above and the price line below. Producer surplus is defined as the area bounded by the price line from above and the supply curve S_0 from below. During the salvage period two phenomena occur. First, the "green" supply curve shifts back to $S(I_1)$ due to a smaller inventory $I_1 < I_0$ available for harvest. Second, a salvage supply curve, V_1, is introduced in the days,

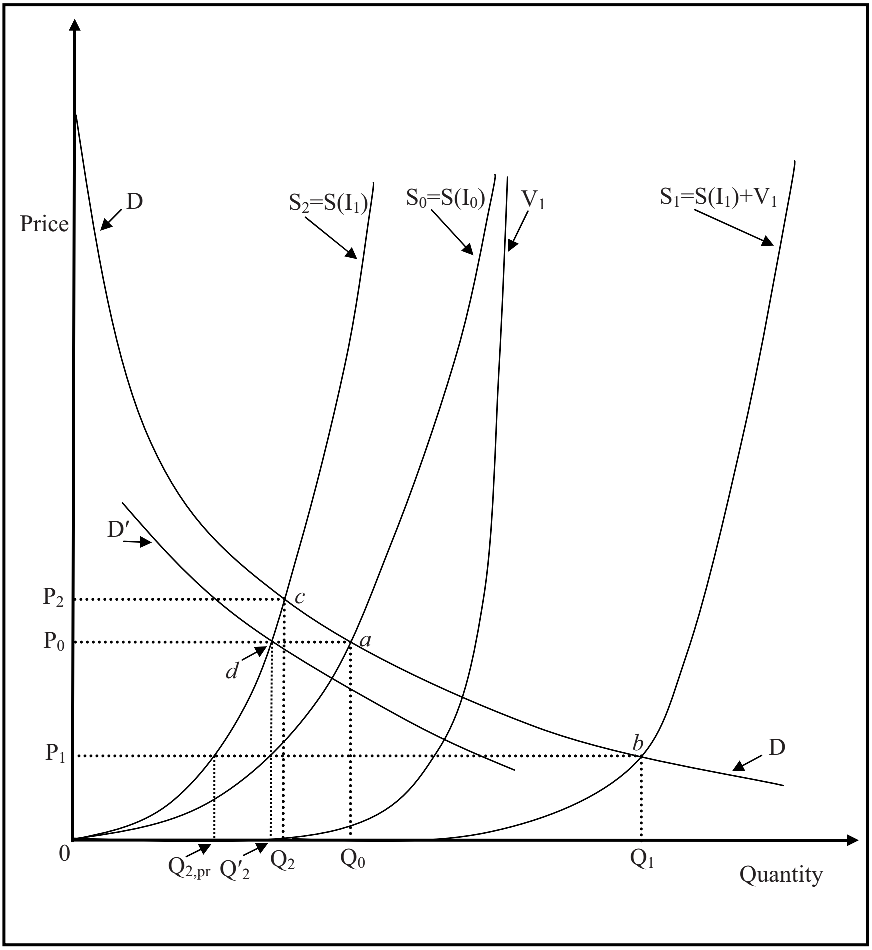

Figure 9.1. Market supply and demand shifts following a large-scale natural disturbance, including a price enhancement due to inventory loss. Point a marks the pre-disturbance supply and demand equilibrium, b marks the salvage period equilibrium, and c marks the post-salvage equilibrium.

months, or even years after the disturbance. This curve is drawn to be highly inelastic or nearly vertical throughout most of its range. Because the timber is no longer growing, due to severe damage or tree mortality, owners of salvage will take almost any stumpage price above zero (recall that the stumpage price is the delivered mill price of the logs obtained from the stand minus the cost of removing the timber and transporting it to the mill). Note that V_1 is the quality-adjusted volume of timber; the volume shown is adjusted downward due to defect (Holmes 1991). Added together, $S_1 = S(I_1) + V_1$ intersects with D to define the salvage epoch price, P_1 ($< P_0$), and quantity, Q_1 ($> Q_0$), at equilibrium at point *b*.

Over time, salvage is exhausted and the salvage supply curve shifts back toward the vertical axis and eventually disappears. This second epoch lasts 6–12 months in the case of hurricanes in warm and humid regions and may last several years in the case of beetle or fire killed timber in cold and dry locations.

During the third epoch, the price and quantity equilibrium is defined by the intersection of the inventory-adjusted supply curve, $S_2 = S(I_1)$, and demand, D. The equilibrium price is higher than the pre-disturbance price ($P_2 > P_0$), and the equilibrium quantity is lower than the pre-disturbance quantity ($Q_2 < Q_0$). This third epoch lasts as long as it takes timber inventories to return to pre-disturbance levels, and so will generally be shorter in high productivity locations. In the case of Hurricane Hugo in South Carolina, Prestemon and Holmes (2000) found a price enhancement of about 15 percent for southern pine timber due to inventory reductions, and Prestemon and Holmes (2004) concluded that this epoch will last 23 years for southern pine sawtimber.

The spatial extent of the timber price dynamics described above depends on the scale of the disturbance and on the costs of material transport between affected and unaffected regions. In the case of Hurricane Hugo, the use of cointegration and intervention analyses demonstrated that the salvage induced price depression was not evidenced beyond the boundaries of South Carolina, where the hurricane struck (Prestemon and Holmes 2000). This result is explained by the fact that spatial arbitrage—the equilibration of prices across space due to product movement—does not operate across great distances if the costs of product movement are high or if commodity prices are low.

The timber demand impacts following a large scale disturbance are not well understood from existing studies, and it seems as though demand may shift in either direction. On one hand, market timber price increases induced by the loss of timber inventories may force some marginally profitable sawmills out of business, thereby dampening demand to D′ in figure 9.1 (Prestemon and Holmes 2004). On the other hand, hurricanes which destroy or damage large numbers of homes and other structures work to increase demand for construction inputs such as lumber and panels. For example, Hurricane Katrina—the most damaging hurricane in recorded U.S. history—is projected to require the reconstruction of over 100,000 houses. This translates into a net increase in lumber consumption of 2–3 percent in 2006 and 2007 (Spelter 2005). Such outward shifts in demand in output markets translate into outward shifts in timber demand, serving to prop up timber prices. Timber price increases resulting from increases in demand for building products, however, would naturally be smaller than the effects of salvage and timber loss caused by the hurricanes. The effects of building product price increases on timber prices are likely to be dampened through spatial arbitrage in building product markets, although this is an empirical question not yet evaluated, as far as we know.

In addition to timber price impacts induced by catastrophic disturbances, transfers in economic welfare among timber market participants can be identified as well (Holmes 1991, Prestemon and Holmes 2004), and can be understood using

Figure 9.1. The supply curve S_2 represents green timber supply from producers holding undamaged timber stocks after a catastrophic event. Due to the price depressing effect of salvage (P_0–P_1), these producers reduce their harvest volume from Q_2 to $Q_{2,pr}$ during the salvage period. Consequently, they suffer a loss of economic welfare even though their stands are undamaged. After the supply of timber salvage is exhausted, supply from undamaged stands expands to Q_2 as price increases to P_2. If price P_2 exceeds the pre-disturbance equilibrium price, owners of undamaged timber enjoy a windfall. The net welfare impact on producers holding undamaged timber depends on the magnitude of these two effects and the levels of supply and demand elasticities.

During the salvage period, consumer surplus increases due to the drop in price (P_0–P_1) and higher quantity consumed (Q_1–Q_0). After the supply of salvaged timber is exhausted, consumers lose surplus as prices increase. The post-salvage price may be as high as P_2 and consumption as low as Q_2. If wood products capacity shrinks enough to drop demand back to Q'_2, then consumers lose even more surplus.

3. ALTERNATIVE SALVAGE DECISION FRAMEWORKS

Timber salvage decisions depend on the degree to which landowners or land management agencies value multiple outputs provided by post-disturbance forests. In what follows, we assume that private landowners make decisions to maximize profit or land value and public managers make decisions to maximize the value of timber and non-timber outputs. For the interested reader, private landowner decision-making is further elaborated in a technical Appendix.

3.1 Private Landowner Decision-Making

For a landowner interested in recovering timber value from a damaged stand, the value of the post-disturbance stand must exceed the cost of logging and transport to market. The decision on whether to salvage requires a comparison of the expected value of salvage versus the expected value of no salvage. If the salvage option has the greater expected value, then timber recovery should proceed.

The timber salvage problem can be embedded in a model of optimal capital management subject to risk of a catastrophic loss. The models of Martell (1980) and Reed (1984) focused attention on the optimal rotation age for timber stands prior to the onset of a catastrophic event. Either the stand attains its optimal rotation age or it is destroyed by a catastrophic event, with salvage of damaged timber a special case. These models assume that timber prices are unaffected by the catastrophic event which, as noted above, is not the case for large-scale disturbances. In order to address this gap in knowledge, we include a technical Appendix that demonstrates how governmental salvage programs (provision of subsidies for land clearing or long-term log storage, public and private road clearing to improve access to stands, enactment of temporary rules that reduce costs of log transport, etc.) can affect optimal forest management decisions. Nonetheless, these prior studies demonstrated that the risk of catastrophic loss, even when mitigated by salvage activity, shortens the optimal rotation age relative to stands facing zero risk of catastrophic loss.

Optimal timber management decision-making by timberland owners following a catastrophic event requires an accurate estimate of the reduction in timber price due to a loss in timber quality. We refer to this change in price as the salvage discount. The salvage discount depends on many factors, including species, climate, pre-disturbance timber quality, the degree and nature of timber damage, the effect of disturbance on harvest and transport costs, and time. Following hurricanes, for example, internal damage to wood may be extensive, whereas with fire or beetle-killed trees, damage can be limited to an outer ring of wood. In warm, humid climates, the price discount increases rapidly over time due to fungal staining and decay (Forest Products Laboratory 1999, p. 13-2). Examples of degrade losses are reported in the literature: (1) de Steiguer and others (1987) found that southern pine beetle mortality caused a 25–75 percent reduction in timber value, (2) Lowell et al. (1992) and Lowell and Cahill (1996) found a 10 percent reduction in timber value following wildfire induced mortality, and Prestemon and Holmes (2004) found a reduction value due to degrade of 80–90 percent following Hurricane Hugo. Figure 9.2 provides a general schematic of the change in timber quality degrade over time.

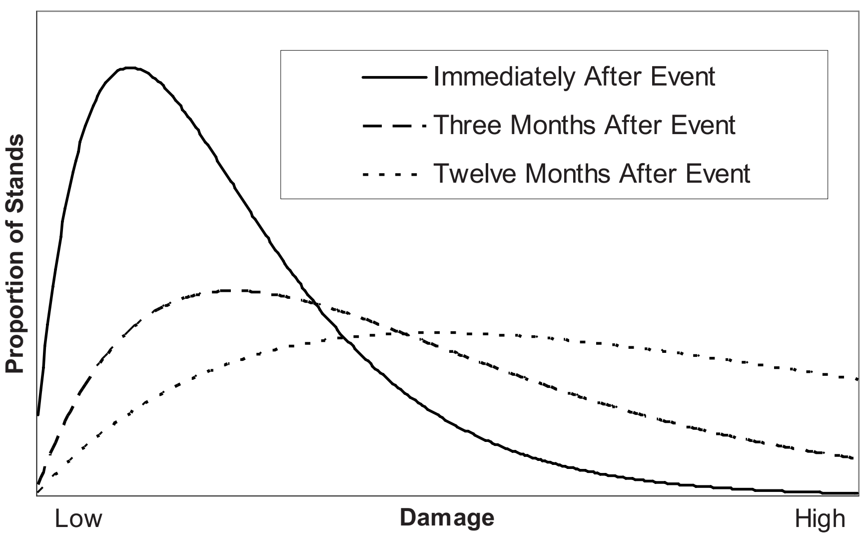

Figure 9.2. Hypothetical damage progression of stands in damage zone following a catastrophic disturbance over time.

From the perspective of profit-maximizing landowners holding stands of even-aged timber, the salvage decision depends on whether salvaging results in a decrease or an increase in profitability over the long run. In turn, this decision depends on the price of timber, the salvage discount, and the age of the stand when the damage occurred. Haight et al. (1995) evaluated the impact of varying rates of damage in young stands on the decision of whether to clear-cut and replant or to let the stand grow. They discovered that, for low disturbance frequencies (3 percent or less), stands with disturbance-caused stocking reductions that are less than 25 percent would optimally be left to grow until a rotation age is reached that is very similar to the no disturbance case. Higher disturbance frequencies call for an immediate clearing of young stands or earlier entry for a commercial thinning to remove injured trees and concentrate growth on larger ones. Because this study did not address the issue of the market effects of a large-scale disturbance on the price of timber, we can surmise that a market price drop due to aggregate salvage activities would decrease the attractiveness of timber salvage and increase the probability that a damaged stand should be left untouched following a disturbance.

The market price dynamics of a widespread disturbance such as a hurricane, catastrophic wildfire, or a large pest outbreak provide both opportunities and risks for affected and unaffected landowners. Owners of damaged timber should understand that salvage prices might be lower than the quality reductions would imply. Owners of undamaged timber may receive higher prices for several years following the exhaustion of salvage supply and may benefit from delaying their harvests. As well, owners of lightly damaged (but live) timber might do well to wait until timber is mature rather than go forward with an immediate harvest. Dunham and Bourgeois (1996), however, caution against letting leaning trees grow to maturity, as these can develop significant timber quality problems related to reaction wood as they age.

3.2 Public Landowner Decision-Making

Government land management agencies typically manage public forests to provide multiple goods and services. If the harvest of timber from public lands is a profitable activity, timber salvage on those lands following a natural disturbance can help mitigate the overall negative economic impact of the disturbance. The economic efficiency of any salvage effort is a logical objective so long as efficiency efforts do not reduce non-timber values produced from the disturbed landscape. Unfortunately, studies that precisely quantify the non-timber consequences of timber salvage are very scarce, thereby impeding a fully specified description of the trade-offs between timber and non-timber values resulting from salvage operations. Greater understanding of these effects is worthy of new research.

One approach would be to model the public decision using a modified Hartman (1976) model in which non-timber benefits flow from intact forests and where salvage potentially reduces those benefits. Generally, the Hartman decision framework reveals that non-timber values can lead to an optimal rate of timber harvesting that is lower than that deriving from a pure timber profit-maximizing decision structure. Depending upon the value of non-timber goods and services provided by disturbed forests, it is possible that salvage on public lands rarely produces positive net benefits from a social welfare perspective. Alternatively, social welfare optimizing solutions may restrict salvage to a subset of the stands that would have been economically salvaged under a timber-only (Faustmann) model. In this case, decisions on which individual stands to salvage would depend on non-timber impacts of salvage, distances to markets (affecting stumpage values), the expenses of cutting, the species mix, timber quality, and the nature of the damage. Because all of these factors vary across disturbance events, the decision calculus could vary dramatically from case to case.

Even within the narrow perspective of timber revenue maximization, several factors work together to reduce the volume of timber salvaged on public lands. First, public timber salvage can reduce market timber prices during the salvage period. Price reductions negatively affect prices received by private landowners holding undamaged timber, reducing their profits and (or) forcing them to delay harvest. Price declines also have a negative impact on private landowners holding damaged timber who are seeking to salvage some of their timber, driving down their salvage revenues. Further, such price reductions can reduce revenues obtained from regular public harvests that proceed during governmental salvage operations. To the extent that public agencies consider the impacts on other market participants from salvage-induced market price reductions, public salvage will be constrained. Although we have not seen evidence indicating that concern for these impacts explicitly influences agency decision making, it would be possible to evaluate the question empirically.

A second factor that can reduce government salvage is the perception by some members of the public that timber salvage is a subsidy to private sector mills who buy the damaged wood. Although such windfalls might be passed on to consumers of forest products in the form of lower prices of building and paper products, competition in the forest products industry suggests that this is unlikely and that mills receive most of the short-run gains from lower timber prices. Although we have no evidence that wood product prices do not reflect lower input prices, public perception that salvage is a giveaway to industry is evident in public communication by the environmental community. Governmental decision-makers may, in response to these kinds of communication campaigns, reduce the amount of timber offered in salvage sales or replace their green timber harvests with salvage harvests (Prestemon et al. 2006).

A third factor is that government efforts to salvage timber are limited by the potential costs of litigation by interested outside parties who oppose salvage sales and by their own institutional capacity. In the latter case, governments might have personnel capacity limitations for managing an increase in overall timber sale activity on affected public forests.

Fourth, and in recognition of the last two factors, mandated decision frameworks can lead to a reduction in salvage. As has been discussed, timber salvage can harm the provision of non-timber values such as ecosystem functions (McIver and Starr 2001), which governments are charged with protecting (Fedkiw 1997).

Fifth and finally, the decision to salvage timber from public lands must not only consider the effects of salvaging on non-timber values but also the consequences of moving economic resources within an agency to carry out a timber salvage sale. In the United States, laws require planning and public hearings, including the preparation of an environmental impact statement. The time required for planning and hearings typically will delay a timber sale, reducing the quantity and quality of salvable timber through decay. Research by Prestemon et al. (2006) quantifies how these kinds of delays may have resulted in real economic welfare losses in the timber market. These losses accrue as timber decays, reducing the net value of standing timber and hence the viability of proposed timber sales on government lands.

4. CASE STUDY: TIMBER SALVAGE FOLLOWING THE BISCUIT FIRE

In order to provide an illustrative example of the economic consequences of a public salvage project, we provide a model that describes the potential market impacts induced by the salvage of fire-killed timber in southwest Oregon. This example highlights some aspects of the decisions facing public land managers. Additionally, this example utilizes the concept of spatial equilibrium in the transport of salvaged timber to regional markets.

4.1 The Biscuit Fire

Between July 13 and November 9, 2002, the catastrophic Biscuit fire burned 499,965 acres, mainly on the Siskiyou National Forest, but also included some Bureau of Land Management land in southwestern Oregon, the Six Rivers National Forest in northern California, and some private land. Most of the Kalmiopsis Wilderness Area, contained within the boundaries of the Siskiyou National Forest, experienced the fire. Burn intensities varied greatly across the area affected, with generally low intensity within the Kalmiopsis Wilderness Area and higher intensities in other zones. Suppression expenditures for the fire exceeded $150 million (USDA Forest Service 2003a).

Four main categories of National Forest lands were burned. These were: Congressionally Reserved (CR, 152,900 acres burned), Administratively Withdrawn (AW, 64,100 acres), Late Successional Reserves (LSR, 133,700 acres), and Matrix land (33,000 acres). The CR land includes the entire Kalmiopsis Wilderness Area. By virtue of legislation and administrative rule, the CR and AW lands are off limits for salvage harvesting and do not contain inventory available to the timber market.

4.2 Model Assumptions

Our analysis provides estimates of the net impacts of the fire under a no-salvage alternative and under alternative rates of salvage, up to 1,500 million board feet (MMBF), which we model as being carried out over two years (2004, 2005). Economic impacts are disaggregated by market participants: owners of damaged timber, owners of undamaged timber, and consumers. The model structure and some underlying assumptions are based on research conducted by the authors, including Holmes (1991), Prestemon and Holmes (2000, 2004), Butry et al. (2001), and Prestemon et al. (2003, 2006), as well as methods pioneered by Samuelson (1952) and Takayama and Judge (1964).

Critical inputs to the analysis include: the pre-fire regional inventory volume and the amount of timber inventory volume killed by the fires (mentioned above), the current price of softwood (green) stumpage, the rate of degrade for fire-killed timber over time, the size of market within which salvage volumes would flow, discount rates, market price sensitivities (measured as the price elasticities of supply and demand), and the starting date of the salvage. These are discussed in turn below.

4.2.1 Timber volumes

We take as given by Sessions et al. (2003) that 40 percent of timber in the burned area was killed. Within LSR and Matrix lands, this amounts to 1,951 MMBF killed (Sessions et al. 2003, p. 22), 83 percent of which was softwood. Therefore, the total volume of softwood killed is assumed to be 0.83 x 1,951 = 1,619 MMBF.

4.2.2 Timber markets

Two markets are identified for analysis: "fire-zone" and "outside fire zone." The fire-zone market constitutes the local area within which much of the salvage volume would likely be consumed and consists of five southwest Oregon counties (Coos, Curry, Douglas, Jackson, and Josephine). In 2003, the sawtimber harvest level in these counties was estimated by the Forest Service to be about 1,441 MMBF. The regional "outside fire zone" market constitutes a larger region around the fire zone, which would be available to absorb additional volume. These counties include seven counties in California (Del Norte, Humboldt, Lassen, Modoc, Shasta, Siskiyou, and Trinity), and one in Oregon (Lane). These eight counties processed approximately 1,825 MMBF of timber in 2003, the base level used in our analysis.

Salvage is assumed to be consumed within the fire zone until the price of the salvage is low enough that it is economically optimal to ship logs from the burn area to the eight counties outside the zone. Inside the fire zone, maximum capacity of sawmills is assumed to be 50 percent above current production. Outside the fire zone, capacity constraints are never reached, because so little salvage exits the fire zone and since timber price reductions are modest, in aggregate. It is possible for some capacity to go unused even while some material exits the fire zone, due to the spatial arbitrage occurring through transport. The two sets of counties therefore have separate and differential impacts from the salvage activity. When timber is moved outside of the five fire zone counties, an additional \$60/MBF is deducted from the defect-adjusted salvage stumpage value, to cover an average of 60 miles additional transport distance to a non-fire zone mill (i.e., per unit transport costs are set at \$1/MBF/mile).

4.2.3 Standing inventory

The Biscuit fire killed approximately 0.5 percent of non-reserved standing softwood timber in the two market areas. Note that hardwood timber impacts are not addressed by this analysis. Inventory re-growth rates are obtained from tables 30 and 34 in Smith et al. (2004). Dividing the Pacific Northwest softwood inventory net growth volume by the region's softwood inventory volume yields a growth rate of 2.1 percent.

4.2.4 Timber prices

Initial equilibrium timber prices for softwood stumpage are taken as the approximate average sale price for stumpage in majority Douglas-fir sales made on National Forests in southwest Oregon in 2002. This price is \$333/MBF (USDA Forest Service 2003b).

4.2.5 Timber supply and demand elasticities and discount rate

Base estimates of the timber market supply elasticity with respect to price are obtained from Adams and Haynes (1996, p. 23), and represent the average of industry and private nonindustrial softwood timber (0.43). The elasticity of supply with respect to inventory volume is set at 1.0, consistent with Adams and Haynes (1996) and with economic theory. The elasticity of demand with respect to price is taken from Abt and Ahn (2003) (-0.5). The base discount rate is set at 4 percent.

A sensitivity analysis is conducted to evaluate the effect of changes in these model parameters on economic welfare estimates. In our simulations, the elasticity of supply and demand with respect to price were halved and doubled. The effect of the discount rate on all economic measures and prices were evaluated using alternative values of 2 percent and 7 percent.

4.2.6 Degrade factors

Salvage volume deterioration rates are weighted averages of expected annual rates by species. The rates of deterioration for one year after the fire (2003), two (2004), and three (2005) are obtained from Lowell and Cahill (1996). The rates of deterioration for years 4 (2006) and 5 (2007) are based on an analysis of the Bitterroot fires of 2000 (Prestemon et al. 2006), which show that the available volume from fire-killed timber of similar species will be about zero by year 5 (fig. 9.3). The volume proportions of fire-killed species were obtained from Sessions et al. (2003, p. 18). The weighted average net volume discount factors used in this analysis are 0.99 after 1 year, 0.89 after two years, 0.58 after three years, 0.22 after four years, and zero after five years (and later).

4.2.7 Demand sector

Salvage logs are assumed to be processed only by sawmills. Some residues from production of dimension products from these mills naturally can be diverted to other processors. The additional defect associated with salvage implies that each cubic foot of salvage sawlog produces both less lumber and less marketable residue than the typical green log of the same dimensions. In this analysis, we ignore the impacts of salvage logging on the residue-using sector. However, as demonstrated by Thurman and Easley (1992), the economic effects on the residue-using sector should be fully accounted for by examining the primary sawlog sector.

4.3 Modeling Approach

Our analysis evaluates the timber market effects—including price changes—attributable to various levels of timber salvage harvesting. The value of the salvage removed is strictly in terms of the net volume of the salvage removed times the

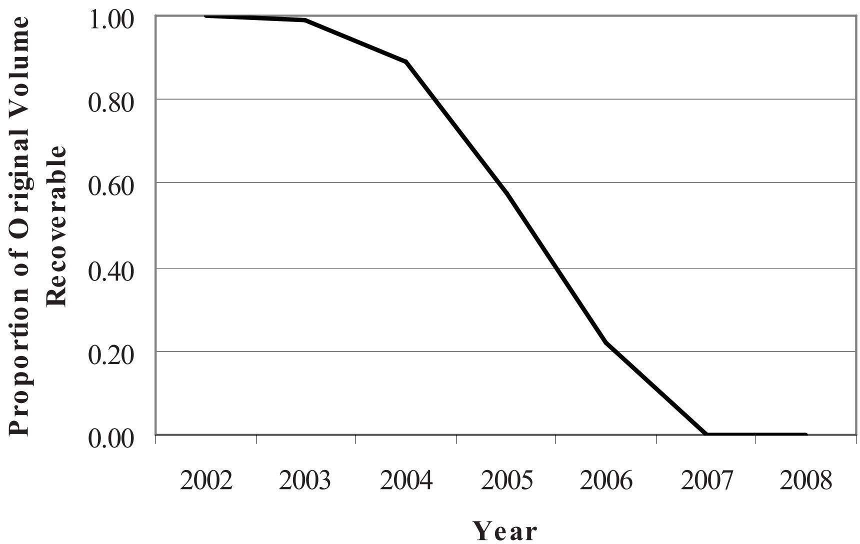

Figure 9.3. Proportion of volume recoverable from fire-killed timber, weighted by species groups (Douglas-fir, Ponderosa Pine, Sugar Pine, true firs), 2002-2008.

market-clearing price of the timber, appropriately adjusted for the fire-related defect shown in figure 9.3. The value of the timber lost from the fire and the effects of the fire on consumers and owners of undamaged timber are reported in economic surplus: consumer surplus and producer surplus. Consumer surplus is defined as the quantity times what consumers would be willing to pay for the timber, minus what they actually paid for the quantity. This can be visualized graphically in a supply-demand graph as the area above the market-clearing price (where supply intersects demand) and below the demand curve. The producer surplus is defined as the net revenues generated from timber production—in effect, the price received minus the cost of producing it. This is visualized graphically as the area above the supply curve and below the market-clearing price line. Methods applied are described in Just et al. (1982) and validated by Thurman and Easley (1992) for one resource-based market.

For non-salvage years (i.e., 2003, 2006, and later), market equilibrium prices and quantities in the two markets are determined separately. For 2004 and 2005, salvage volumes, prices, and green volumes in each subregion are jointly determined jointly using spatial equilibrium methods outlined by Takayama and Judge (1964). The joint solution is found by maximizing the sum of total net economic welfare across the two markets. That is, the combination of salvage volume in each market, green production in each subregion, and market-clearing prices in each market are determined by maximizing the two regions' sum of producer and consumer surplus minus the additional transport cost associated with moving some of salvage from the subregion in the fire zone to the subregion outside the fire zone. Hence, the price differential between the two subregions never differs by more than the cost of transport between the two regions. Net welfare impacts from salvage reported in the tables of this chapter account for the transport costs and are reported as sums across the two subregions.

4.4 Results

4.4.1 "No salvage" economic welfare estimates

The results of our market simulation show that, under a "no salvage" scenario, base case elasticities and a 4 percent discount rate, the Biscuit fire caused producer surplus to decrease for producers of damaged timber (the National Forests and Bureau of Land Management lands) on LSR and Matrix lands by \$51.5 million (table 9.1). The effect of the fire on consumers (mills) is to reduce long-run consumer surplus by \$79.7 million, due to the lower volumes produced on the burned-over area during the ensuing decades (inventory effect), and the higher long-run equilibrium price due to inventory loss. For owners of undamaged timber, slightly higher market prices due to the inventory reduction in the 13-county market area lead to a net benefit from the fire, amounting to about \$79.2 million. Thus, a roughly equivalent economic value would be transferred from mills to producers holding undamaged timber in the long-run, under this scenario. When these three economic welfare impacts are summed, the total

Table 9.1. No Salvage Scenario, $ million changes in market welfare by group, alternative discount rates and market elasticities, for Late Successional Reserve and Matrix lands, Biscuit Fire burned area, 2002 dollars.

	Discounted Consumer Surplus Change	Discounted Value of Timber Lost	Effects on Undamaged Producers	Total Discounted Surplus
Base Case Values	-79.7	-51.5	79.2	-52.1
Low Discount Rate	-101.2	-65.4	100.5	-66.1
High Discount Rate	-59.5	-38.5	59.1	-38.8
Low Supply Elasticity	-84.8	-49.5	84.2	-50.1
High Supply Elasticity	-44.5	-32.3	44.2	-32.6
Low Demand Elasticity	-89.3	-42.1	88.7	-42.7
High Demand Elasticity	-42.3	-42.1	42.0	-42.3

Note: the Total Discounted Surplus column does always exactly equal the sum across the other three columns, due to rounding error.

market impact of the fires is a loss of $52.1 million. Using alternative market parameters, total impacts on economic welfare ranged from a $32.6 million loss under a high supply elasticity, to a $66.1 million loss when a low discount rate (2 percent) and base case values for other parameters are assumed.

4.4.2 Price impacts

Timber salvage reduces average market prices for both salvage and green timber, particularly in the fire zone (fig. 9.4). As stated above, a small price increase would occur in the fire zone if no timber is salvaged (about 1 percent in 2004 and 2005). From this initial equilibrium point, the price depressing effects of timber salvage would strengthen monotonically along with salvage volume. For the maximum salvage volume we considered, 1,500 MMBF out of the estimated 1,619 MMBF loss of softwood inventory, timber prices within the fire zone would decrease by 28.7 percent in 2004 and 22.3 percent in 2005. In the regional market outside the fire zone, the stumpage price reductions are 10.7 and 4.3 percent for 2004 and 2005, respectively for the maximum salvage volume. Outside the fire zone, detectable price effects are not registered until salvage volumes reach or exceed 700 MMBF. The effect in 2004 would be larger than the effect in 2005, other variables held constant, due to the decay and greater quality discount applied to salvage timber as time progresses.

4.4.3 Economic impacts of timber salvage

Salvage revenues range from about $24 million for 100 MMBF of salvage up to $265.2 million for maximum salvage effort (table 9.2). Mills are positively affected by salvage because market prices drop and they purchase greater timber volumes (salvage as well as non-salvage) at lower prices during the two years

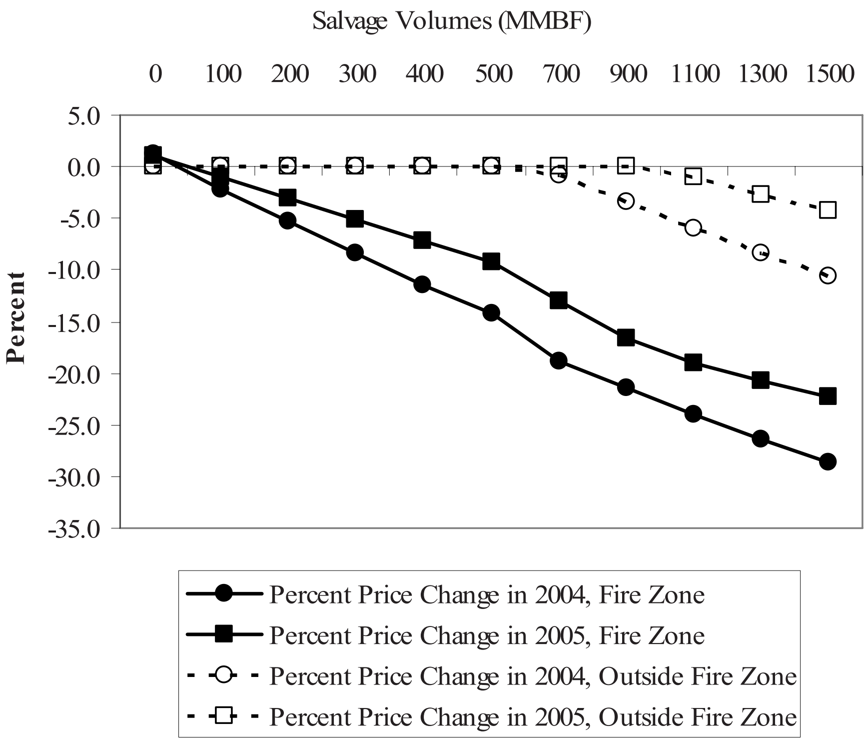

Figure 9.4. Price impacts (percent price changes) in the five-county region of southwest Oregon ("Fire Zone") and an eight-county region outside of that ("Outside Fire Zone"), from alternative salvage volumes removed from some parts of Late Successional Reserve and Matrix lands in the Biscuit Fire burn area, under base case assumptions of the discount rate and market sensitivities to prices and inventory.

of salvage activity. At a salvage volume of just over 300 MMBF, the benefits to mills from salvaging operations are approximately equal to the loss of consumer surplus due to the long-run inventory effect. At greater salvage amounts, mills would be better off in the long run than they would have been if the Biscuit fire had not occurred.

Timber salvage reduces producer surplus for forest owners holding undamaged timber, net of the windfall benefits they enjoy as a result of the elimination of inventory from the regional market (the $79.2 million benefit, mentioned above). At low salvage volumes, these producers still receive net benefits from the fire. If salvage volume exceeds about 330 MMBF, however, net windfall benefits become net losses in producer surplus. These owners are worse off with salvage because they harvest less timber during the salvage period and because they receive a lower price for their timber. Relative to the "no salvage" scenario, producers lose $24.4 million for 100 MMBF of salvage, and $308.4 million if

Table 9.2. Changes in welfare effects ($ million) resulting from alternative salvage plans, by producer and consumer group. The no salvage scenario is the point of comparison.

Volume Salvaged, MMBF, Total over 2004 & 2005	Discounted Consumer Surplus Change	Effects on Undamaged Producers	Value of Salvage Removed	Total Discounted Surplus	Percent Price Change in 2004, Fire Zone	Percent Price Change in 2005, Fire Zone	Percent Price Change in 2004, Outside Fire Zone	Percent Price Change in 2005, Outside Fire Zone
0	0.0	0.0	0.0	0.0	1.2	1.1	0.0	0.0
100	24.7	-24.4	24.0	24.2	-2.2	-1.0	0.0	0.0
200	49.1	-47.8	46.6	47.7	-5.4	-3.1	0.0	0.0
300	73.0	-70.1	67.9	70.5	-8.5	-5.2	0.0	0.0
400	96.5	-91.5	88.0	92.6	-11.4	-7.2	0.0	0.0
500	119.6	-112.0	107.0	114.1	-14.3	-9.2	0.0	0.0
700	165.2	-151.8	142.4	155.1	-18.9	-13.0	-0.9	0.0
900	210.9	-192.0	176.2	194.2	-21.5	-16.6	-3.4	0.0
1100	256.2	-231.6	207.7	231.2	-24.0	-19.0	-5.9	-1.0
1300	301.3	-270.8	237.4	266.6	-26.4	-20.7	-8.4	-2.7
1500	345.6	-308.4	265.2	300.9	-28.7	-22.3	-10.7	-4.3

Note: the Total Discounted Surplus column does always exactly equal the sum across the other three columns, due to rounding error.

1,500 MMBF are salvaged over the two years from the Biscuit Fire burn area. The larger the salvage program, the larger the negative impact on these producers.

These ideas are further illustrated in figure 9.5, which shows the production volume during the year of the fire and in subsequent years, including salvage years, in the five counties of the fire zone. The figure shows the production volume of owners of undamaged timber as well as the volume of salvage removed, under a 400 MMBF salvage program. Owners of undamaged timber would produce about 6 percent less in 2004 and 4 percent less in 2005, due to the lower prices. The entire five county fire zone market, however, would produce about 7 percent more timber than usual in 2004 and 4 percent more than usual in 2005, adding together the volumes of green and defect-adjusted salvage.

Economic analysis of this type can help decision-makers evaluate the economic impacts of alternative timber salvage programs. For example, at maximum timber salvage effort, net economic welfare in the market would increase by about $300 million, compared to not salvaging at all. A salvage program of 1,000 MMBF would approximately compensate for the combined timber market surplus losses and the fire suppression expenditures on the Biscuit fire. A timber salvage program of roughly 300 MMBF would reduce the non-timber impacts of salvage relative to the maximum salvage program, maintain timber consumer (mills)

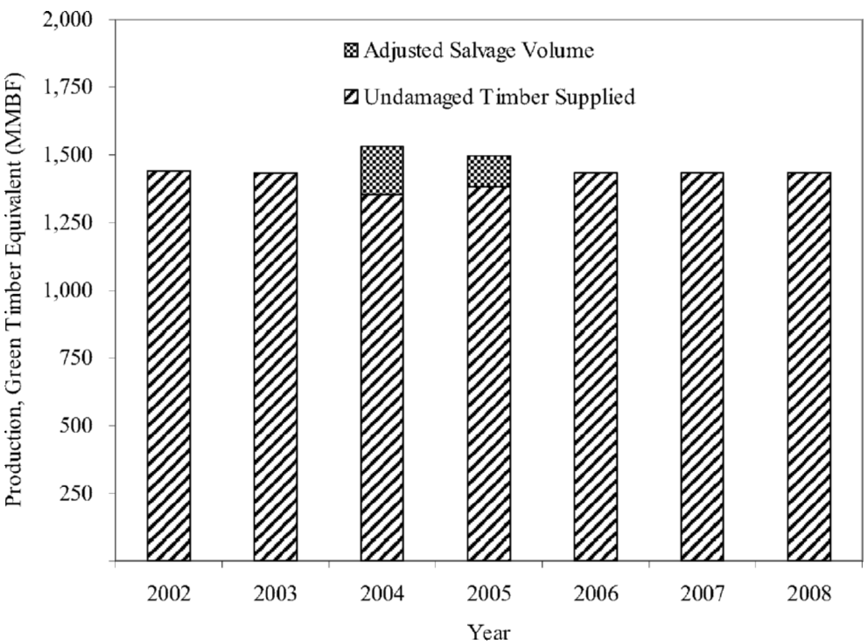

Figure 9.5. Market volume of timber produced, 2002-2008, including 200 MMBF of salvage (adjusted for degrade) occurring in each of 2004 and 2005, in the five counties of Southwest Oregon contained in the Fire Zone.

and undamaged producer surplus at roughly the pre-disturbance level, and yield timber market revenues of nearly $68 million for the suppliers of salvaged timber (the government).

5. CONCLUSIONS

Timber salvage, like other kinds of salvage, provides benefits that can help to mitigate the overall economic impacts arising from a catastrophic event. Salvaging timber, like the natural disturbance preceding it, often induces a rearrangement of economic wealth. Private forest owners holding damaged and undamaged timber need to understand the implications of price changes that occur during the aftermath of a catastrophic forest disturbance and alter their timber harvest plans accordingly. Public decision-makers need to be aware that governmental programs supporting large-scale salvage operations can accentuate timber price and welfare impacts.

Perhaps the greatest challenge facing public decision-makers is managing the suite of trade-offs between timber and non-timber economic benefits deriving from a salvage program. Unfortunately, very little is known about the value of post-disturbance ecosystem goods and services that are impacted by salvage operations. However, if public sentiment regarding governmental salvage activities is an accurate barometer of these values, we would suggest that their omission from a full economic analysis may provide biased policies. Incorporating public values in timber salvage analysis presents an urgent challenge for forest economists.

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APPENDIX

Salvage decisions faced by private landowners can be modeled within the framework of optimal capital management. In the presence of the risk of a catastrophic loss, the profit maximizing forest landowner seeks the optimal rotation age T that maximizes $L(T)$, the land rent (Reed 1984). In this Appendix, we show how the optimal capital management model can be modified to account for a short-run market price decline during the salvage period, due to a pulse of salvaged timber, as reported by Holmes (1991), Yin and Newman (1999), and Prestemon and Holmes (2000). Then we discuss how governmental interventions during the salvage period might affect landowner decisions.

We begin by modifying equation (20) in Reed (1984) by the addition of two new variables: (1) g, the relative price decline for *green* timber due to a supply pulse of salvaged timber, and (2) $\tilde{\kappa}(t)$, the volume-weighted salvage price discount ratio which reflects the price decline for *salvaged* timber, relative to green timber, due to a loss in quality. The value of an infinite series of rotations in an even-aged stand potentially subject to salvage can then be written:

$$L(T) = \frac{[h+r]\{[V(T)-c_1]e^{-(h+r)T} + \Phi(T)\}}{r(1-e^{-(h+r)T})} - \frac{h}{r}c_2 - c_1,$$
$$\Phi(T) = \int_0^T hgV(t)\tilde{\kappa}(t)e^{-(h+r)t}dt \tag{9.1}$$

where $L(T)$ is the land expectation value at the optimal rotation age $t = T$; $V(t)$ is the value of the stand at age t; h is the constant annual probability of a damage-inducing forest disturbance (described here as an independent probability, implying that the landowner's decisions have no effect on the probability); r is the discount rate; c_1 is the reforestation plus site preparation costs without a pre-harvest disturbance; c_2 is the reforestation plus site preparation costs with a pre-harvest disturbance; e is the exponential function; $\Phi(T)$ is the expected present value of salvage in the presence of a constant annual disturbance risk. The volume-weighted salvage price discount ratio obtained in year t, $\tilde{\kappa}(t) = \sum_{j=1}^{J} s_j(t)[\kappa_j(t) - d_j(\kappa_j)]$ where $s_j(t)$ is the value share at green prices of the timber in the stand year t with damage level j; $\kappa_j(t) \sim [0, 1]$ is a measure of the timber quality discount (i.e., the ratio of the value of timber with damage level j to undamaged timber); $d_j(\kappa_j) \sim [0, 1]$ measures the extra removal and transport cost (proportional loss of stumpage value) for trees with a κ_j level of damage following a disturbance.

The economic significance and variability of the timber quality discount is indicated by noting that de Steiguer et al. (1987) determined $\kappa(t)$ ranged from 0.50 to 0.75 for southern pine-beetle damaged wood. Lowell et al. (1992) and

Lowell and Cahill (1996) found $\kappa(t)$ to be at or above 0.9 in fire-killed timber for two years following tree mortality in Oregon. Prestemon and Holmes (2004) showed that the volume-weighted salvage price discount ratio $\widetilde{\kappa}(t)$ aver-averaged 0.22 for sawtimber and 0.11 for pulpwood in the year following Hurricane Hugo.

Optimal decisions regarding salvage in stands damaged by a disturbance where $L(T) \leq 0$ are somewhat simpler. Here, a landowner must ask whether the following inequality holds (ignoring discounting, as salvage usually proceeds within a year of the disturbance):

$$gV(t)\widetilde{\kappa}(t) - (c_2 - c_1) > 0 \tag{9.2}$$

where $c_2–c_1$ is the extra cost of harvesting the stand following a disturbance compared to a harvest of an undamaged stand. If (9.2) holds, then salvage can take place profitably for non-timber-managing forestland owners.

The above decision framework applies to stands with older trees, where salvage can yield damage mitigating revenues. Following Haight et al. (1995) and Reed (1984), the probability of a catastrophic event can be defined by a Poisson process, whereby the cumulative probability that a disturbance will have occurred by year t is given by $\Pr[X(t)] = 1–e^{-ht}$, where X is the time between successive stand damaging events or clearcuts. The cumulative distribution is then $\Pr[X<t] = 1-e^{-ht}$ if $t < T$ equal to 1 if $t=T$. Hence, the probability that either the rotation age (T) or the disturbance has occurred by year t is given by $1–e^{-ht}$ if $t < T$ and 1 if $t \geq T$. The expected present value of managing a young stand without salvable wood but with the ongoing risk of stand damage is modeled as in Reed (1984, equation (3)):

$$J = [V(T) + L]e^{-(h+r)T} + \frac{h(L-c)[1-e^{-(h+r)T}]}{h+r} \tag{9.3}$$

where c is the simple land clearing cost following a disturbance.

When disturbances are widespread, post-disturbance market prices will be different from the pre-disturbance market prices, even after accounting for the reduced quality of some timber entering the market. Holmes (1991), Yin and Newman (1999), and Prestemon and Holmes (2000) all noted substantial market price declines. The studies of Hugo found green-timber price reductions averaged about 30 percent during the salvage period, or a value of g in equation (9.2) equal to 0.7 for both southern pine pulpwood and southern pine sawtimber. Holmes' (1991) study of southern pine beetle found green-timber price reductions that averaged about 20-30 percent (g = 0.7 to 0.8). A market price drop would tend to decrease the attractiveness of timber salvage while simultaneously increasing the probability that a damaged stand should be left untouched following a disturbance.

Evaluation of the equations (9.1), (9.2), and (9.3) offers the opportunity to identify ways in which government interventions could change the optimal decision-making calculus for private landowners, and they offer insights into potentially optimal strategies for landowners generally. For example, the effect of post-disturbance subsidies of land clearing or planting costs on land value could be assessed by evaluating how incremental changes in subsidies affect long run profits from the land use. At a 6 percent discount rate and an annual disturbance probability of 3 percent, each dollar of subsidy to clearing plus replanting cost following disturbance would yield about $0.50 in land value increase. More generally, given a clearing plus planting cost of $400 per acre and a land value of $1,000 per acre, each 1 percent increase in the land value subsidy would increase land value by 0.2 percent. In other words, if governments want to encourage timber growing, maintenance of land in forest, and timber salvage in the event of a disturbance, then provision of a post-disturbance subsidy to affected landowners can help.

Part of the salvage discount is the extra cost of removal and transport of wood from disturbance-damaged stands. Therefore, another way for government to intervene is in the facilitation of transport. In South Carolina following Hugo, for example, the State relaxed weight limits on roads temporarily to allow for larger log loads. The State also invested generally in road clearing, which likely aided the salvage effort. Unfortunately, it is unclear how much harvest costs and transport costs are directly affected by disturbance events.

Another consideration is how governments can act to increase the level of market demand, which will enhance prices offered to all landowners in the months or years immediately following a catastrophe. To the extent that demand capacity can be expanded, the ratio of damaged to undamaged timber prices will be higher, which will encourage salvage, increase land values, and raise the economic incentive to reestablish stands. For example, assume that the volume of pre-event timber on a stand is 1,700 ft^3, the value is $976/acre, the discount rate is 6 percent, the annual rate of disturbance is 3 percent, and the optimal harvest age is 25 years with a salvage value per unit volume that is 30 percent lower than for an undamaged stand. Now, assume government intervention to encourage a demand expansion following the disturbance event such that there is only a 20 percent loss in stand value immediately following the storm, then land values would increase by about 7 percent. The value of salvage would rise by exactly the proportional rise in the salvage period price, about 14 percent in this case. Although these numbers are somewhat arbitrary, they are reasonable and therefore informative. If government efforts to facilitate log storage achieve this kind of dampening of the salvage price glut, then widespread benefits could be experienced by affected landowners by enhancing their salvage revenues and land rents.

In this last case, where government could intervene by subsidizing temporary log storage capacity expansion begs an important question: Would government spending on private capacity be economically efficient? If market decision makers possess all of the same, correct information about probabilities of timber-damaging natural disturbances as the government, then we might expect an optimal distribution of production inputs devoted to storage capacity to exist in the market. In this context, government provision of a storage subsidy would be inefficient. Presumably, mills make decisions about log storage capacity by balancing the cost of the last unit of capacity with the additional expected long run stream of extra revenues gained by creating it. The capacity decision should therefore incorporate the probability that disturbances will occasionally offer gluts of raw materials. One possibility justifying such subsidies would be that some firms have under-invested in capacity because of capital (credit) constraints or because poor decisions were made (e.g., the firm underestimated the actual frequency of such disturbances). Alternatively, the sector might have under-invested in capital in anticipation of government intervention; if such intervention does not happen, then capital would have been misallocated.

CHAPTER 10

WILDFIRE AND THE ECONOMIC VALUE OF WILDERNESS RECREATION

Jeffrey Englin, Thomas P. Holmes, and Janet Lutz

1. INTRODUCTION

The idea that wildfires play an integral role in maintaining healthy forests has begun to change the ways that scientists, managers, and the general public view fire policy and programs. New approaches to forest management that seek to integrate natural disturbances with the provision of goods and services valued by people impose a greater need for a full accounting of the economic effects of wildfire (as well as other disturbances). In addition to the effects that forest fires have on commodities and assets that are traded in markets, such as timber and residential structures, fires also affect the condition and value of public goods that are not traded in markets, such as outdoor recreational sites. Understanding the economic consequences of wildfires on the provision and value of public goods requires the use of non-market valuation methods (Champ et al. 2003). The goal of this chapter is to demonstrate how wildfires affect the demand for, and value of, Wilderness recreational sites, which is illustrated using the travel cost method.[1]

Wilderness areas provide the public with a special opportunity to observe the effects of wildfires on natural processes in fire-adapted ecosystems. Lightning-caused fires are sometimes allowed to burn in Wilderness areas (a prescribed natural fire) when conditions are deemed suitable. Management ignited prescribed fires are also used to reduce fuel loads and mimic natural processes (Geary and Stokes 1999). Although fire suppression activities are permitted in Wilderness areas, management of forest regeneration and succession after a wildfire (including timber salvage and tree planting) is not permitted. Consequently, Wilderness areas provide a natural laboratory where visitors can experience first-hand the ecological dynamics following the occurrence of wildfire.

Since the passage of the Wilderness Act in 1964, more than 100 million acres of wild lands have been included in the National Wilderness Preservation System. Recent estimates suggest that roughly 15 million annual visits were made to

[1] The focus of this paper is on Wilderness areas located within the National Wilderness Preservation System as designated by Congress. To maintain this distinction from other land uses, we capitalize the word Wilderness.

T. P. Holmes et al. (eds.), The Economics of Forest Disturbances: Wildfires, Storms, and Invasive Species, 191–208.

Wilderness areas during the mid-1990s, up from roughly 5 million visits in 1970 (Loomis et al. 1999). Projections made using data from the National Survey on Recreation and the Environment indicate that the number of people participating in Wilderness recreation will increase by roughly 26 percent between 2002 and 2050 and total nearly 20 million visits by the mid-twenty-first century (Bowker et al. 2006). Wilderness use data, where they are maintained, provide researchers with an excellent opportunity to observe the recreational choices made by outdoor enthusiasts. Because wildfires alter the condition of forest ecosystems, and set into motion a dynamic process of fire succession, we hypothesize that concomitant shifts in recreation demand will occur. In this chapter, we illustrate how the travel cost model can be used to identify linkages between fire succession and shifts in recreation demand that span several decades.

The next section of this chapter describes several conceptual issues faced by researchers who seek to evaluate the impact of wildfires on forest recreation, and how these issues have been treated in the literature. This is followed by a brief, but technical, presentation of the theoretical and econometric models used in our subsequent empirical analysis. The methods used to collect and organize a large-scale data set, spanning more than 2.5 million acres of Wilderness, 15 years of Wilderness use, and 60 years of fire history, are then described. This is followed by a presentation and discussion of the empirical results. The chapter concludes with some remarks about the limitations and potential extensions of the analysis, and a discussion of how recreation demand modeling can help land managers make more informed decisions.

2. ASSESSING THE IMPACT OF FIRE ON FOREST RECREATION DEMAND

The economic effects of wildfires on the demand for outdoor recreation have been evaluated from two broad perspectives. The first approach focuses attention on the economic sectors of local economies that are impacted during and following a wildfire, primarily (1) tourist expenditures, and (2) employment and wages in tourism related sectors (Butry et al. 2001, Kent et al. 2003). It is generally recognized that the influx of fire fighters and other personnel during the period of fire suppression and restoration activities confounds the identification of economic impacts due solely to changes in recreation demand. The second approach focuses attention directly on the behavior of people participating in outdoor recreation activities and evaluates the impacts of wildfires on recreation demand and the value of recreation sites (Boxall et al. 1996, Englin et al. 2001, Loomis et al. 2001, Hesseln et al. 2003, Hesseln et al. 2004). Although the emphasis of this chapter is on the latter perspective, there are several conceptual challenges that are common to both approaches.

The first challenge in evaluating wildfire impacts on forest recreation is identifying a control or a counterfactual basis for comparison. Even in situations where *ex ante* and *ex post* data exist on visits to an area burned by wildfire, it is difficult to know with certainty what level of visitation would have occurred in the absence of wildfire. To provide a proxy for without fire data, some sort of model is typically imposed to estimate a counterfactual rate of visitation. A simple solution was provided by Franke (2000) who compared changes in visitation to Yellowstone National Park subsequent to the 1988 wildfires to visitation trends in Montana as a whole. Visitation dropped during the year of the fires, due to Park closures. However, Park records showed that visitation increased each of four years after the 1988 fires and by 1992 visitation had increased 41 percent above the 1985 pre-fire level. Some observers might conclude that the increased rate of visitation could be attributed to a surge in visits from people who were curious to see how the Yellowstone landscape had been altered by the wildfires. However, Franke (2000) notes that tourism in Montana rose about 54 percent during that same period. If the general rate of tourism increase in Montana during this period is taken as the true counterfactual data for rates of change that would have occurred within the Park with no fire, then it could be concluded that the wildfires of 1988 caused a decrease in the subsequent rate of visitation.

Another approach to constructing a counterfactual scenario is to use a statistical model. Butry and others (2001) used a simple statistical model to test the hypothesis that the 1998 wildfires in Florida caused a loss in tourism revenue during the summer in which the fires occurred. To estimate counterfactual without fire scenarios, they computed the 95 percent confidence interval around the average annual percentage change in tourism revenues for each county in the wildfire impact area for ten years prior to the 1998 wildfires. Then, they tested whether or not the actual tourism revenues for June, July and August of 1998 fell inside the confidence intervals. Using this approach, they identified a statistically significant loss in tourism revenues for the month of August (only) for each of the counties in the impact area during the year of the fires.

Kent and others (2003) also used a statistical approach to evaluate the economic impacts of the Hayman fire in Colorado during the summer of 2002. Counterfactual without fire scenarios were estimated for the months of June and July for each of 5 counties in the primary impact area using statistical models for wages, employment, and retail sales in the eating and drinking, lodging, and recreation sectors of the economy. Although the analysis was able to identify some statistically significant changes in some sectors during specific months, the overall pattern of changes in economic activity was mixed and it was not possible to arrive at a definitive conclusion regarding the economic impacts of the Hayman fire on local economies.

A second issue when attempting to evaluate the impact of wildfires on tourism and/or recreation is the possibility of contemporaneous (or same season) substitution. People planning outdoor recreation trips have options regarding where to visit, and the temporary closing of destinations such as Yellowstone Park might induce people to alter their plans and visit an alternative destination rather than simply canceling their trip. Contemporaneous substitution is important to recognize if the goal of economic analysis is to understand the impact of a natural disturbance on the general economic system. If alternative recreational destinations are available, the economic loss from closing a single site will overestimate the total economic impact to the system because some economic value is transferred as an economic gain to the alternative sites visited.

A third issue to consider when evaluating the impact of wildfires on tourism or recreation is the possibility that fire succession induces inter-temporal (time-dependent) substitution. Although recreation sites are often closed during the wildfire burn period in order to protect public safety, people interested in viewing the aftermath of wildfires might substitute some other trip for a post-fire visit to the site that burned. Further, the number and value of visits to recreational sites that have burned might be anticipated to change over time as the quality of the site changes due to ecological succession. We would expect that patterns of inter-temporal demand will vary across specific forest ecosystems due to different patterns of regeneration and recovery after a wildfire.

Data that portray actual ecological conditions in a recreational area before and after a wildfire, and data representing actual recreational use of that area pre- and post-fire, are scarce. In lieu of such data, Vaux and others (1984) recommended using photographs to illustrate typical processes of fire succession, which will vary across forest ecosystems. Then, by asking people to respond to questions regarding how their use of the recreational area would change in response to the illustrated changes in conditions, contingent behavior data can be obtained and analyzed.

The contingent behavior approach to data collection has been employed by several researchers using micro-econometric travel cost models (Boxall et al. 1996, Englin et al. 2001, Hesseln et al. 2004, Hesseln et al. 2003, Loomis et al. 2001). A typical approach is to conduct intercept interviews at recreation sites that have recently burned as well as sites that have not recently experienced wildfire. Cross-sectional data provide a counter-factual no-fire control that can be compared with data collected at sites that have burned. Contemporaneous substitution across recreation sites is implicitly addressed in the micro-econometric studies by including site quality variables in the econometric specification. The micro-econometric studies also ask survey participants to respond to several contingent behavior questions which are included to increase the number of observations related to post-fire trail conditions. Two themes have emerged from this literature: (1) demand shifts over time in response to wildfires can be identified, and (2) the economic impact of wildfires on the demand for outdoor recreation differs by activity (e.g., hiking or mountain biking).

The analysis presented in this chapter tries to surmount some of the limitations faced by previous micro-econometric studies by using data spanning nearly two decades of Wilderness use across 7 Wilderness areas in the mountains of California. We argue that there are some substantial advantages to working at large temporal and spatial scales. First, it is reasonable to assume that, by and large, much substitution behavior through time and across space is captured in these data. Second, the pattern of fires used in the analysis is the natural pattern of fires across the landscape, rather than a simulated pattern of fires imposed by the research team. As a result the economic welfare measures reflect actual ecological dynamics and behavioral responses. Third, working at large temporal and spatial scales provides very large data sets that make robust estimation possible.

3. THEORETIC AND ECONOMETRIC MODELS

Harold Hotelling is usually credited with the insight that the price of access to outdoor recreation sites can be inferred from information on travel costs. This idea was subsequently developed by Marion Clawson and Jack Knetsch in a general work on the economics of outdoor recreation (Clawson and Knetsch 1966). The basic Hotelling-Clawson-Knetsch approach to estimating the demand for outdoor recreation is to statistically regress the number of trips taken to a recreation site on the round-trip cost of travel between trip origins and the site. A set of demand shift variables are typically included in the regression model to control for socio-economic characteristics of visitors, site characteristics, and costs of visiting alternative sites. Once a demand curve is estimated, the consumer surplus associated with a recreational site is computed by integrating the area under the demand curve and above the travel cost associated with accessing the site.

The ordinary least squares regression model was used in early estimates of travel cost demand models. However, since the seminal work of Shaw (1988) it has become popular to apply count data models to recreation demand. A review of count data models in estimating forest recreation demand is provided by Englin and others (2003). Count data models emphasize the non-negative, integer nature of trip visitation data, and are most useful when the number of counts is small (Hellerstein 1991).

A functional form that guarantees that trip counts will be non-negative is the linear exponential (semi-log) demand function. The linear exponential functional form of site demand is linked with a count data distribution by setting the expression for demand equal to the count data parameter for the mean (equation 10.1):

$$\lambda_{ij} = E\left[Trips_{ij}\right] = e^{(\beta' X_{ij})} \qquad \text{i} = 1,2,\ldots..\text{N} \qquad (10.1)$$

where λ_{ij} is the mean number of trips demanded by person i for site j; $E[Trips_{ij}]$ is the expected number trips by visitor i to site j; X_{ij} includes the travel cost to site j by the i^{th} person, socio-economic characteristics for individual i, and the fire characteristics for site j; and β is a vector of parameters to be estimated. This approach pools all of the data on visitation to j sites to estimate a single travel cost demand function.

For the analysis presented in this chapter, Negative Binomial count data regression models were used. The Negative Binomial is attractive because it does not constrain the mean to equal the variance, allows the model to be over-dispersed relative to the Poisson model, and can be corrected for truncation and endogenous stratification (Englin and Shonkwiler 1995). The likelihood for the Negative Binomial distribution is:

$$\Pr ob(Trips_i = q_i) = \frac{\Gamma(q_i + \frac{1}{\alpha})}{\Gamma(q_i + 1)\Gamma(\frac{1}{\alpha})}(\alpha\lambda_i)^{q_i}[1+\alpha\lambda_i]^{-(q_i+1/\alpha)} \qquad (10.2)$$

where α is the over-dispersion parameter. Notice that this likelihood collapses to the Poisson if α equals zero. The log likelihood (L) for the Negative Binomial is:

$$L = \sum_{i=1}^{N}[\ln\Gamma(q_i + 1/\alpha) - \ln\Gamma(q_i) - \ln\Gamma(1/\alpha) + q_i \ln\alpha + q_i X_i\beta - (q_i + 1/\alpha)\ln(1+\alpha e^{\beta X_i})] \qquad (10.3)$$

where $\exp(\beta' X_i)$ replaces λ in equation (10.2).

The data used in this chapter are panel data rather than a single cross-section. As discussed in the data section (below), trip origins are described by zip codes, which provide the basic unit of observation. Wilderness trips originating in specific zip codes appear in our data for multiple years of analysis, and each zip code is treated as a group. Because households that reside in some origins may demand more or fewer trips relative to the average household, panel data models can be employed to capture these unobserved effects. The random effects model treats these effects as being randomly distributed across the groups and independent of any of the explanatory variables in the demand model. In contrast, the fixed effects model allows correlation between the unobserved effects and the explanatory variables. In particular, we suspect that the unobserved fixed effects may be correlated with the travel cost variable.

Wilderness demand parameter estimates are obtained for random effects and fixed effects Negative Binomial models using the modeling approach described in the seminal paper by Hausman, Hall and Griliches (1984). The random effects count data model is:

$$\log\lambda_{it} = \beta' X_{it} + \delta_{it} + u_i \qquad (10.4)$$

where δ_{it} is the dispersion parameter, which is allowed to vary randomly across groups yielding the random effect u_i. It is assumed that the inverse of the dispersion is Beta distributed with parameters a and b. The model is estimated by integrating out the random effect and estimating the parameters using maximum likelihood. The likelihood function for the random effects model is:

$$\Pr[q_{i1},\ldots,q_{iT}] = \left(\prod_t \frac{\Gamma(\lambda_{it}+q_{it})}{\Gamma(\lambda_{it})\Gamma(q_{it}+1)}\right)\times \frac{\Gamma(a+b)\Gamma(a+\sum_t \lambda_{it})\Gamma(b+\sum_t q_{it})}{\Gamma(a)\Gamma(b)\Gamma(a+b+\sum_t \lambda_{it}+\sum_t q_{it})} \tag{10.5}$$

where $\Gamma(\cdot)$ is the gamma function. Because this model adds a heterogeneity term to a model that already contains a heterogeneity term (the over-dispersion parameter), Greene (2002, p. E20-120) warns that the random effects Negative Binomial model might be over-parameterized and convergence might not be attained. However, as shown in the Results section below, random effects were successfully estimated using this model on the permit data.

The fixed effects count data model is:

$$\log\lambda_{it} = \alpha_i + \beta' X_{it} + \delta_{it} \tag{10.6}$$

where α_i is the fixed effect. For the fixed effects model, the joint probability of the counts for each group is conditioned on the sum of the counts for the group (which solves the incidental parameters problem), and is estimated using conditional maximum likelihood. The conditional likelihood of the fixed effects negative binomial is:

$$\Pr[q_{i1},\ldots,q_{iT} \mid \sum_{t=1}^{T} q_{it}] = \frac{\Gamma(\sum_t \lambda_{it})\Gamma(\sum_t q_{it}+1)}{\Gamma(\sum_t \lambda_{it}+\sum_t q_{it})}\prod_t \frac{\Gamma(\lambda_{it}+q_{it})}{\Gamma(\lambda_{it})\Gamma(q_{it}+1)}. \tag{10.7}$$

Notice that the conditional likelihood function eliminates the fixed effect and the probability is a function of β alone.

Once the parameters of the count data models are estimated, it is straight forward to estimate consumer surplus by integrating the area under the demand curve. Because the estimator used to obtain parameter estimates is nonlinear, total consumer surplus is found by simulating the demand equation using observed values for the explanatory variables. For the linear exponential demand function, total consumer surplus is estimated as:

$$\textit{Consumer surplus} = \int_{p0}^{p1} \lambda dp = (-1)\bullet\frac{\lambda(\beta' X)}{\beta_{tc}} \tag{10.8}$$

where p^0 is the actual travel cost, p^1 is the choke price, and β_{tc} is the parameter estimate on the travel cost variable. Marshallian consumer surplus per trip ($-1/\beta_{tc}$) is computed by dividing the total consumer surplus by the number of trips (λ).

Because wildfires are included as an additive term in the vector of explanatory variable (X) in the regression model, wildfires affect the number of trips taken to Wilderness areas, but not the value per trip.[2] The change in consumer surplus induced by wildfires occurring during various periods antecedent to the time of a visit is estimated by computing λ(β'X) with and without fires of specific vintages:

$$Wildfire\,\Delta\ in\ Consumer\ surplus = \frac{\lambda(\beta' X^{obs}) - \lambda(\beta' X^{vintage})}{\beta_{tc}} \tag{10.9}$$

where X^{obs} is the vector of explanatory variables set at their observed level for the simulation, and $X^{vintage}$ substitutes counterfactual area burned for fires of specific vintages. Note that equation (10.9) allows us to estimate the total change in trips resulting from wildfires of different vintages and sizes, but it does not allow us to determine whether specific groups are changing their recreational behavior (such as new entrants).

4. DATA

The analysis presented in this chapter required merging data assembled from three sources: (1) Wilderness permit data, (2) socio-economic data, and (3) wildfire data (fig. 10.1). An explanation of these data and how they were merged is presented below.

4.1 Wilderness Permit Data

Visitors to National Forest and National Park Wilderness areas are required to obtain a permit before entry. For the purpose of recreation economic research, the key information provided by a Wilderness permit is the location of the visitor's place of residence (zip code), which can be used to estimate travel distance from the place of residence to the Wilderness area.

In an attempt to collect as much permit data as possible from Wilderness areas throughout the mountainous regions of California, our permit data search process began with phone calls to National Forest ranger stations and National Park offices. Of the offices that maintained permit data for 1 or more years, appointments were made to meet with the data managers. Prior research had yielded permit data for several California Wilderness areas for the years 1990-1992. The current research effort yielded new permit data and the complete data set included the following Wilderness areas (and years): Ansel Adams (1990-1992; 2001-2002), Golden Trout (1990-1992; 2001-2002), Hoover (1990-1992;

[2] As noted in the Conclusion section of this chapter, future research will be conducted to evaluate whether or not wildfires affect the value of a trip as well as the number of trips taken.

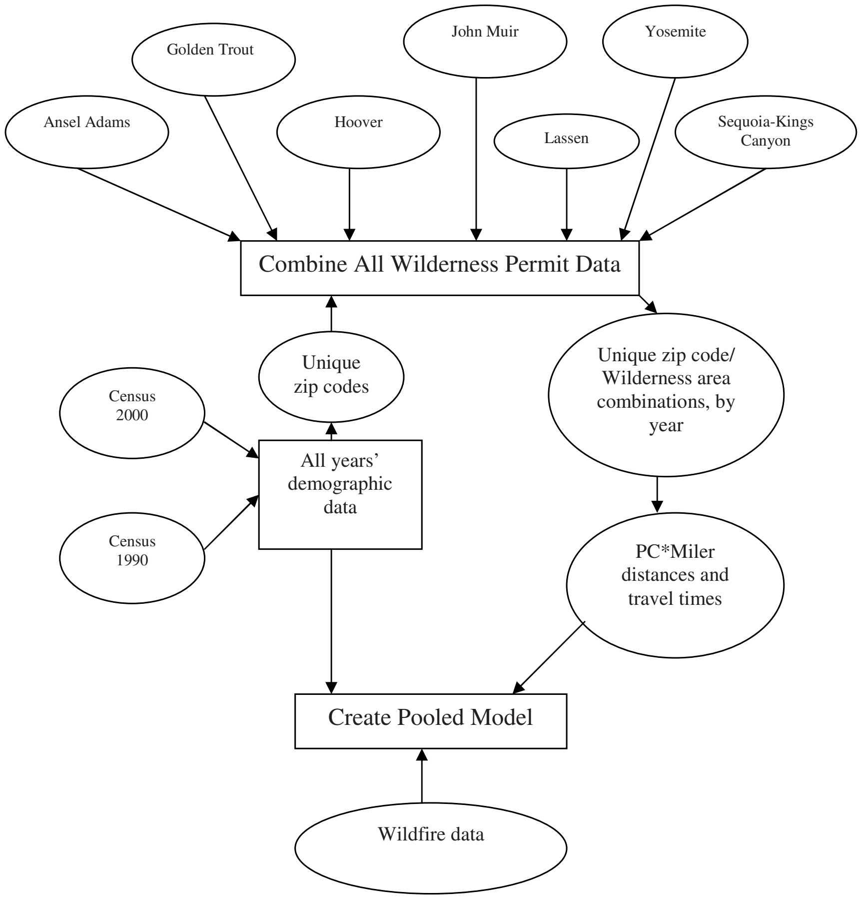

Figure 10.1. Flow chart of data set construction.

2001-2002), John Muir (1990-1991; 2001-2002), Lassen Volcanic National Park (1990-1992; 2001-2002), Yosemite National Park (1998-2004), and Sequoia-Kings Canyon National Park (1990-1992; 2001-2002). Except for Lassen Volcanic National Park, all of these Wilderness areas are located in the Sierra Nevada Mountains (fig. 10.2). These land management areas encompass about 2,670,082 acres and roughly 2,739 miles of Wilderness trails are located within their boundaries. In total, permits for 182,987 trips to these Wilderness areas were obtained for analysis.

Complete data records were imported, one land management unit at a time, into the STATA statistical software package. In order to pool all of the data into a single dataset, several steps were followed to create identical subsets of data for each Wilderness area. First, a variable identifying each Wilderness area was

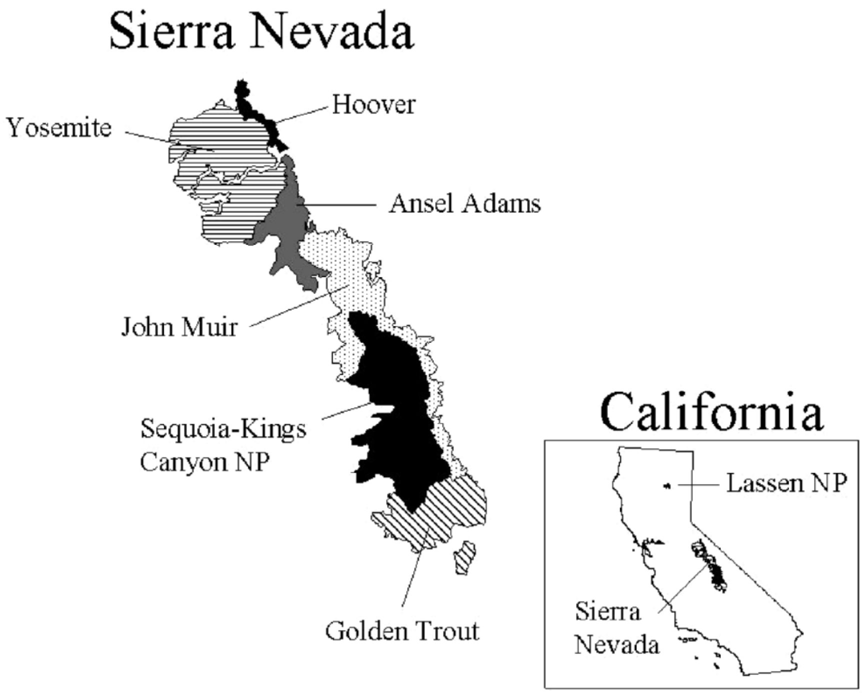

Figure 10.2. Map showing Wilderness areas used for analysis.

created to classify the destination for each trip. Second, a zip code variable was created, identifying the origin of the trip. Third, variables identifying the entry and exit locations for each trip were created. Fourth, variables were created for both entry and exit dates for the trip (where possible). Finally, if available, party size, activity, and fee variables were included. The activity variable provides the primary purpose of each trip. A large majority of the trips are hiking trips and were the focus of this study.

4.2 Merging Data Subsets

Data subsets for each Wilderness area were created using a consistent format that allowed data to be merged. The first step was to remove all permits having missing or clearly erroneous zip code origin data. Second, in order to focus on the demand for backcountry hiking, all trips associated with other primary activities were deleted. Finally, all trips that had invalid entry years (for example, 1900 or beyond 2005) were removed.

To reduce the possibility of including multiple destination trips in the analysis, it was decided to retain only those trips that originated in California or Nevada. All zip codes from these two states that are common to the 2000 and 1990 census were merged onto the trip data set. Zip codes within these two states and for which no Wilderness permits were recorded were retained in the data set. This procedure ultimately simplifies the econometric model of population demand by eliminating the need to control for truncation in the data.

The next step of the data combination process was to create a dataset that included a row for each unique combination of zip code origin, park destination, and entry year. This involved a two stage process. First, a dataset was created that identified each unique combination of zip code, Wilderness area and entry year for all possible entry years. Trip counts were then added for each unique combination (the number of trips per zip code ranged from 0 to 528). Then, all entry years were removed for those Wilderness areas where no permit data had been obtained. For example, permit data were obtained for trips into all Wilderness areas in the Inyo National Forest during 1990, but permits were not obtained for trips into Yosemite National Park during that year. Therefore, all zip code, Wilderness area, entry year combinations for 1990 excluded Yosemite National Park as a destination. The final data set included 38,907 observations on trip counts observed for each unique combination of zip code, Wilderness area and entry year (or zero if no trips were observed).

Next, distances were computed for each zip code/Wilderness area combination using the PC*Miler software. PC*Miler can calculate driving distance and estimated travel time from a zip code to a specific latitude-longitude. A USGS website provided precise latitude and longitude data points either for the center of the area of interest or near a major highway or road that all visitors to the area would most likely use. No zip codes in the data submitted to PC*Miler were invalid and therefore both the distance traveled as well as travel time were added to the data.

Finally, demographic information by zip code was added to the data set using data from the 2000 and 1990 census. Data were included for household income, population, average age, percent white, average household size, and years of education. The 2000 census data were obtained from the census web site. The 1990 census data were obtained from a library CD-ROM. Demographic variables were interpolated to unique values for each year in between the census years by assuming a linear relationship.

With both the demographic and distances data added, travel cost was calculated using the following equation:

$$\text{Travel cost} = (\text{cost per mile} * \text{round trip miles}) + \text{opportunity cost of time.} \tag{10.10}$$

Cost per mile was computed for each year based on the IRS allowance for business mileage.[3] The opportunity cost of time was calculated as follows:

$$\text{Opportunity cost} = (\text{round trip hours traveled})*(1/3)*(\text{income}/2040). \qquad (10.11)$$

Fire data were obtained from CALFIRE01_3 GIS data files.[4] These data include information on fire size, fire perimeter, location and year (spanning the period 1908 through 2001.)[5] The data were available for all Wilderness areas in California and therefore could be merged onto the permit data. Once the wildfire data were added, variables were created to capture the total area burned within each of the Wilderness areas for various vintages based on each entry year in the permit data.[6] For example, the Ackerson fire occurred in 1996 in Yosemite National Park and burned 55,960 acres. For an individual entering Yosemite in 1997, this would be a one year old fire, whereas it would be a 4 year old fire for an individual entering in 2000. To simplify the model specification, vintages were then aggregated into age classes: 1 to 3 years old; 4 to 9 years old; 10 to 19 years old; 20 to 29 years old; 30 to 39 years old; 40 to 49 years old; and 50 to 59 years old. Aggregation was based on natural break-points in the data and not specifically on expected visual or aesthetic changes in vegetation due to fire succession.[7] Fire data were sparse for vintages beyond 59 years, and were not used in the model specifications.

Some Wilderness areas in California are more prone to wildfires than others. At one extreme is the Hoover Wilderness which, due to its high elevation (8,000–12,000+ feet) and minimal forest area, reported almost no area burned. In contrast, large wildfires fires have been regular occurrences in Sequoia-Kings Canyon and Yosemite Wilderness Areas. Since 1908, wildfires have burned roughly 12 percent of Sequoia-Kings Canyon Wilderness and 14 percent of Yosemite Wilderness has burned since 1991.

Descriptive statistics are shown in table 10.1. Relatively few trips were taken per zip code on average (2.54) and the relatively high variance (73.44) suggests that the negative binomial distribution is a better choice than the Poisson (which restricts the mean to equal the variance). Using the formulas shown in equations

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[3] These values were obtained from IRS tax form 2106. For the years from 1990 forward, these forms are available online at http://www.irs.gov.

[4] Personal communication (A.L. Westerling; January 3, 2003).

[5] Although fire sizes greater than 0.1 ha are included in the data, it appears that smaller fires were not generally recorded during the early decades of the twentieth century.

[6] Fire area data were re-scaled for analysis. One fire area unit was equivalent to 23,393.07 acres.

[7] Future research will evaluate alternative specifications of the econometric model, including specifications based on anticipated changes in major fire succession visual characteristics.

Table 10.1. Descriptive statistics.

Variable	Mean	Std. Dev.	Maximum
Number of trips	2.54	8.57	528
Travel cost ($)	292.49	113.97	933.07
Income ($1,000)	39.76	17.84	169.06
Log of population (#)	8.97	1.84	11.64
Age (years)	35.94	5.72	79.33
Education (years)	12.42	1.60	18.18
Race (percent white)	0.74	0.22	1.23
Household size (#)	2.74	0.57	6
Entry year 1990-1992 (dummy)	0.58	0.49	1
Ansel Adams (dummy)	0.13	0.34	1
Golden Trout (dummy)	0.13	0.34	1
Hoover (dummy)	0.13	0.34	1
John Muir (dummy)	0.13	0.34	1
Lassen (dummy)	0.13	0.34	1
Sequoia-Kings Canyon (dummy)	0.13	0.34	1
Area burned 1-3 years prior (units)	0.31	0.64	2.42
Area burned 4-9 years prior (units)	0.71	1.05	3.20
Area burned 10-19 years prior (units)	1.10	1.49	4.24
Area burned 20-39 years prior (units)	0.13	0.20	1.36
Area burned 40-49 years prior (units)	0.12	0.19	0.62
Area burned 50-59 years prior (units)	0.07	0.12	0.62

Note: 1 fire area unit = 23,393.07 acres.

(10.10) and (10.11), the average travel cost was $292.49, which is consistent with a single-use trip. Because the data were balanced to reflect all possible combinations of zip code origins and Wilderness destinations, including zero trips from origins to destinations, mean values for the dummy variables on Wilderness areas are equal by design. More than half of the data (58 percent) represent trips taken during 1990-1992.

5. MODEL RESULTS

As indicated above, Wilderness recreation demand was estimated using a simple negative binomial count data model and specifications for random and fixed effects. Parameter estimates for all models are shown, with standard errors in parentheses (table 10.2). All variables are statistically significant at the 1 percent level. The over-dispersion parameter estimates (alpha) were highly significant, indicating that the negative binomial model was an appropriate specification. The chi-square (χ^2) goodness-of-fit statistics indicated that the random effects model performed better than the other two models, and parameter estimates from this model are used in the discussion that follows.

Table 10.2. California Wilderness area recreation demand parameter estimates.

Variable	Negative Binomial		Random Effects Negative Binomial		Fixed Effects Negative Binomial	
	Mean	Standard error	Mean	Standard error	Mean	Standard error
Travel cost	-0.007	0.000	-0.006	0.000	-0.006	0.000
Income	0.017	0.001	0.017	0.001	0.020	0.001
Log of population	0.718	0.008	0.670	0.013	0.521	0.019
Age	-0.047	0.018	0.113	0.027	0.072	0.031
Years of education	0.446	0.014	0.414	0.002	0.318	0.027
Race (percent white)	0.261	0.055	-0.267	0.097	-0.735	0.124
Household size	-0.653	0.037	-0.153	0.057	0.052	0.072
Age^2	0.0004	0.000	-0.001	0.000	-0.001	0.004
Ansel Adams	5.560	0.556	6.606	0.338	6.302	0.348
Golden Trout	4.268	0.553	3.570	0.324	3.391	0.334
Hoover	3.784	0.559	4.690	0.326	4.411	0.336
John Muir	5.652	0.493	6.833	0.301	6.556	0.310
Lassen	2.365	0.554	3.543	0.324	3.256	0.335
Sequoia-Kings Canyon	25.831	1.035	10.396	1.115	10.942	1.142
Entry year 90,91,92	1.594	0.051	1.768	0.061	1.795	0.062
Area burned 1-3 years prior	0.702	0.108	0.531	0.064	0.513	0.066
Area burned 4-9 years prior	8.769	0.185	7.104	0.153	7.080	0.156
Area burned 10-19 years prior	-0.499	0.102	-0.998	0.082	-0.986	0.084
Area burned 20-39 years prior	-0.355	0.099	-0.175	0.083	-0.217	0.084
Area burned 40-49 years prior	-45.882	1.767	-15.320	2.088	-16.691	2.137
Area burned 50-59 years prior	-34.801	0.771	-36.103	0.958	-35.496	0.966
Constant	-13.668	0.728	-18.785	0.775	-15.521	0.932
Ln alpha (ln _a)	-0.219	0.019	(1.579)	(0.051)	--	--
Alpha (ln_b)	0.803	0.015	(1.356)	(0.057)	--	--
a	--	--	4.851	0.248	--	--
b	--	--	3.880	0.220	--	--
Number of Observations	38,907		38,907		38,907	
χ^2 statistic	32,299.92		38,359.95		33,540.45	

Note: STATA estimates the dispersion parameter indirectly using natural logarithms, then undoes the transformation.

As expected, the parameter estimate on the travel cost variable had a negative sign, which is consistent with a downward sloping demand curve. Substituting the parameter estimate for travel cost (β_{tc}) into the formula for Marshallian consumer surplus ($-1/\beta_{tc}$), the economic value provided by Wilderness sites in our study area is $174.73 per hiking trip. Income, education, and population were also found to have a positive effect on the demand for Wilderness recreation.

Of primary interest in this chapter is how wildfires of different vintages alter the demand for, and value of, a Wilderness hiking trip. The parameter estimates shown in table 10.2 indicate that wildfires occurring 1-3 years prior to the observed trip had a modest impact on Wilderness demand. However, fires occurring 4-9 years prior to the observed trip were seen to stimulate demand, presumably for hikers who were curious to observe fire succession after the initial impact of the fires had been buffered. Forest recovery during this stage is often typified by the establishment of low vegetation such as grasses and flowers. Fire succession during the next three decades appeared to have little impact on Wilderness recreation demand, as trees in burned over areas began to regenerate. Somewhat surprisingly, burned-over areas with 40-60 year vintages appeared to have a strong negative impact on recreation demand. This may be due to increasing density of forest regeneration which would restrict views. Although these results are statistically robust, and are consistent with other research (Englin et al. 2001), alternative model specification need to be tested to further evaluate the linkages between fire succession and Wilderness demand.

The parameter estimates can be used to simulate the linkages between wildfire vintage and consumer surplus. For example, we simulated the loss in consumer surplus resulting from a 250 acre fire occurring at different points in time prior to observed trips. The observed pattern (fig. 10.3) is a result of changes in the number of trips and not with the value associated with a trip. Understanding the relationship between the value of a Wilderness trip and fire succession dynamics is a topic deserving future modeling effort.

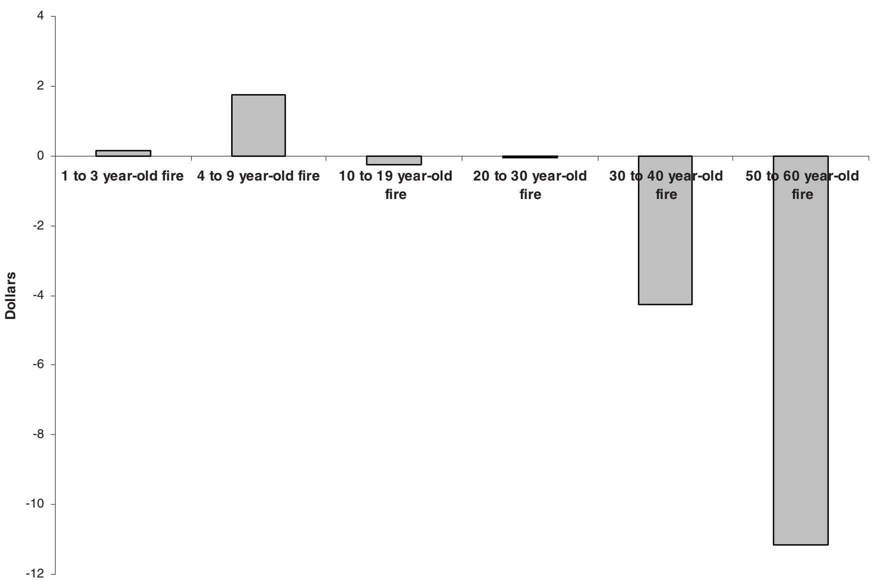

Figure 10.3. Simulated change in Wilderness Area consumer surplus for 250 acres burned at different vintages.

6. CONCLUSION

Although the analysis presented in this chapter is exploratory, and should not be viewed as definitive, it represents the first attempt to estimate the effects of forest fires on recreation demand that exclusively uses observations on actual behavior and does not rely on responses to contingent behavior questions. Relative to other studies that have modeled the impact of wildfires on outdoor recreation, the dataset presented here is enormous. The model estimates are based on nearly 183,000 observations of actual Wilderness trips taken under a variety of conditions. The panel data set spans a decade and a half of Wilderness recreation behavior and is linked to a wildfire data set that spans nearly 6 decades. As the data cover such a long period of time, as well as including several alternative Wilderness destinations, an exceptionally broad range of demand substitution patterns is captured.

Based on our exploratory analysis, the major conclusion of this chapter is that fire succession is linked to Wilderness recreation demand in a complex fashion. Wildfires of recent vintages appear to increase the number of trips to Wilderness areas, and wildfires of older vintages appear to decrease the number of trips. The robust statistical results we obtained strongly suggest that Wilderness managers need to be aware of a potential flux in recreation demand for several years following large wildfires. The outward shift in demand we observed is consistent with visitation shifts reported for the Yellowstone fires of 1988 (Franke), with the Shenandoah fire complex of 2000 (Morton et al. 2003), the Rat Creek–Hatchery Creek fire in Leavenworth, Washington (Hilger 1998), and various fires in the intermountain western United States (Englin et al. 2001). It appears that a significant number of hikers and other outdoor recreation enthusiasts desire to observe fire behavior and its impacts on forest succession. We suggest that these demand shifts provide a good opportunity for land managers to provide educational and scientific information about fire ecology to this segment of the population. Further, volunteers might be recruited from among this demand segment to collect information on fire succession, such as the location and abundance of plant species of interest. This pattern also suggests the need for sufficient resources to reduce potentially hazardous situations created by wildfires such as snag trees close to trails and campsites. Over the longer run, these results suggest that large areas burned by wildfires may begin to experience reductions in demand. Understanding these long-run demand shifts is important for trail and infrastructure planning in the impacted areas.

The data reported in this chapter are very rich and present analytical complexities on many levels. As such, we conclude this chapter by suggesting various avenues for future research. First, wildfire area was treated as a linear variable in the analysis. However, future research should consider the possibility of non-linear responses to wildfires, perhaps occurring at different spatial thresholds and degrees of fire intensity (such as crown fires vs. low-intensity ground fires). Second, a precise understanding of why visitors seem to prefer recent fires to fires of older vintages is lacking, and this demand behavior might be clarified through on-site surveys of Wilderness users. Third, partitioning the data and performing micro studies could be illuminating. For example, it would be useful to identify demand shifts among substitute Wilderness areas in response to large, recent fires such as the 55,957 acre Ackerson fire in the Yosemite Wilderness. Not only would such analyses permit validation testing of the hypotheses evaluated in this large-scale study, but such micro-analyses would likely provide greater detail about the patterns of cross-sectional and inter-temporal substitution by Wilderness travelers. Finally, it is plausible that, in addition to affecting the number of Wilderness trips, wildfires might affect the quality and value of trips taken. This hypothesis will be tested in future research and should help to further clarify the effects of wildfires on the economic value of outdoor recreation.

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CHAPTER 11

FOREST DISTURBANCE IMPACTS ON RESIDENTIAL PROPERTY VALUES

Robert J. Huggett, Jr., Elizabeth A. Murphy, and Thomas P. Holmes

1. INTRODUCTION

Natural environments and the amenities they offer have fueled much of the population growth in the rural United States (Deller et al. 2001, English et al. 2000). In fact, the fastest growing counties in the United States during the early 1990s were non-metropolitan counties that were destinations for retirees or that offered outdoor recreation opportunities (Johnson and Beale 1994). Migration to these rural and exurban areas from urban and suburban locations, along with growth in the United States population, has resulted in an increased mixing of humans, their artifacts, and natural environments. These expanding interface and intermix areas expose more lives and property not only to desirable natural amenities, but also to natural disturbances and disamenities.

Households choose the type and amount of natural amenities, along with other structural, neighborhood, and environmental characteristics, in their location decisions. These amenities, such as access to recreation, viewshed, and air and water quality, are capitalized by housing markets into prices. Wildfires, pest outbreaks and other natural disturbances can alter the quantity and quality of amenities available to the household. Damage or destruction of the property itself or any of the surrounding amenities by natural disturbances affects that property's value and if the impacts are widespread, the broader property market is impacted as well. Even in the absence of a disturbance event, property markets respond to the presence of disturbance risk alone since this risk represents the potential for future damages to property and natural amenities. In the context of this chapter, risk will refer to both the probability of a disturbance event and the probability of the loss associated with an event.

The primary theoretical framework for studying the relationship between a property's portfolio of characteristics and its price is based on the hedonic model of Rosen (1974). The application of this theory to property markets is known as the hedonic property model (HPM). The empirical use of the HPM in the literature is extensive as it is a popular method to explain the effect of trees, forests, and

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woodland on residential property markets. Morales (1980), Anderson and Cordell (1985), and Dombrow et al. (2000) examine how residential prices respond to the presence of trees. The relationship between urban forests and housing prices in Finland is treated by Tryväinen (1997) and Tryväinen and Miettinen (2000). Price response to woodland in Great Britain is the subject of work by Garrod and Willis (1992a). The effects of open space, a more general classification, on property prices are considered by Geoghegan et al. (1997), Acharya and Bennett (2001), Shultz and King (2001), and Geoghegan (2002).

Despite the depth of literature using the HPM to look at how forest and woodland amenities impact property prices, there are far fewer examples which examine the impacts of forest disturbances and the risks they represent. PricewaterhouseCoopers (2001) performed an analysis of how the Los Alamos, New Mexico real estate market responded to the 2000 Cerro Grande fire. The results report a temporary dip in prices of 3 percent to 11 percent following the fires. No insight is offered on the possible cause for this drop—a shock to the overall housing market, the loss of forest amenity, or an increased awareness of wildfire risk. Loomis (2004) estimates that house prices in Pine, Colorado decreased by approximately 15 percent following the Buffalo Creek fire due to updated risk perceptions and the loss of forest amenity. In a study of the Flagstaff, AZ property market, Wells (2001) reports that households place a higher value on medium canopy density vs. high canopy density. Lower risk of fire and increased viewshed afforded by medium canopy closure are offered as possible explanations. Donovan et al. (2007) find that the publication of a website that rated wildfire risk in the wildland-urban interface of Colorado Springs had an impact on housing price.

Payne et al. (1973) provide an accounting procedure for calculating property value losses from gypsy moth damage, which was based on a hedonic study of the contribution of trees to property value in Massachusetts (Payne and Strom 1975). Derived from the later published hedonic study, an equation was presented which describes the relationship between the number of trees on a lot and the dollar amount those trees contribute to property values. Using data on tree mortality from insect infestations, lost property value is calculated as the difference between pre-attack and post-attack valuations. However the model does not account for lost value from trees that are unsightly or unhealthy, nor does it consider the nuisance impact of gypsy moths.

Garrod and Willis (1992b) suggest that replacing mature conifers, which reduce price in their study when located within 1 km of a house, with other species would result in lower disamenties. However they offer no insight into the nature of the disamenities. Geoghegan et al. (1997), Tryväinen (1997), and Schultz and King (2001) report negative relationships between some natural amenity variables and housing prices but do not suggest the risk or realization of disturbances as a reason. The response of property prices to other natural hazards, such as earthquakes, volcanoes, and hurricanes, has received treatment by Brookshire et al. (1985), Bernknopf et al. (1990), Beron et al. (1997), and Bin and Polasky (2004).

This chapter seeks to provide a basic framework for modeling the effects of forest and other natural disturbances on property markets. The modeling section will begin by introducing the hedonic property model in a simple, accessible format. Several important modeling issues and aspects of forest disturbances that make them special in regard to describing their impact on property markets will be discussed next. These include the tension between risks and amenities embodied in a forest resource, the temporal dynamics of disturbance manifestation, and spatial dependence among observed outcomes present challenges to capturing the effects of disturbance shocks. Two case studies will follow, examining the price responses of residential housing to wildfire and an invasive species, the hemlock woolly adelgid. The chapter will conclude with a discussion of management and policy implications of disturbance shocks to property markets.

2. HEDONIC PRICE THEORY

The hedonic model simply states that a good's price is a function of the various qualities and characteristics that make up that good. The intuitive nature of the theory underlying the hedonic model, variation in characteristics embodied in a good creates variation in prices (Taylor 2003), is very appealing. In general the hedonic model estimates how the total price of a good changes at the margin—that is, when one of its characteristics changes and all others are held constant. Using the HPM to analyze the residential property purchasing decisions made by households, where houses with differing portfolios of characteristics and prices are bought and sold in a single market, allows the researcher to find the marginal willingness to pay (MWTP) for an additional unit of each characteristic.

Let $z_1, z_2, \ldots, z_m$ be the set of *m* characteristics of a property such as lot size, square footage, age, the quality of local schools, distance to a trailhead, etc. We can denote this set as the vector **Z**. The market for property is comprised of buyers on the demand side and sellers on the supply side and is assumed to be in equilibrium. Each buyer's willingness to pay for vector **Z** while one characteristic z_i is changed and all others are kept constant is described by a bid curve. For each seller, the willingness to accept a price for vector **Z** while one characteristic z_i varies and others are held constant is represented by an offer curve. The hedonic price function, P(**Z**), an equilibrium relationship between buyers and sellers, is an envelope of tangencies of buyer bid curves and seller offer curves (Taylor 2003 for an extended discussion of bid and offer curves). The first derivative of P(**Z**) with respect to characteristic i, $\partial P / \partial z_i$, yields that characteristic's implicit price, also called the hedonic price. The implicit price is the MWTP for an additional unit of that characteristic.

Using statistical techniques, such as linear regression or maximum likelihood estimation, P(**Z**) can be estimated and implicit prices for the various characteristics inferred from the results. A variety of functional forms are available for use in empirical applications, including linear, log-linear, semi-log, quadratic, and Box-Cox. However little guidance from economic theory is available for the selection of the proper form. It has been demonstrated that the linear and semi-logarithmic forms were among those that performed best when unobserved variables were proxied by others or are not included in the hedonic function at all (Cropper et al. 1993). In some cases the dependent variables or the error terms of different locations may be correlated. Spatial hedonic property models can account for both spatial dependence in the dependent variable and the error structure.

Estimating P(**Z**) and the implicit prices for each z_i is known as first stage analysis. Using first stage results, demands for characteristics of interest can be estimated in the second stage analysis. Because the implicit price represents only one point on the buyer's bid curve, identifying demands can be difficult. Estimating demands requires information beyond that required in the first stage, such as demand shifters and in some cases a second set of implicit prices from another market. Despite the difficulty, second stage analysis is useful because demands can be used to estimate welfare changes that result from changing the quantity of a characteristic. The two applications presented in this chapter will focus only on first stage estimation.

3. EMPIRICAL ISSUES IN MODELING DISTURBANCE IMPACTS

The data used in an empirical application of the HPM must be extensive enough in geographic coverage to capture the disturbance shock, but not so large that the single market requirement of hedonic theory is violated. Defining the extent of the area to be studied is the first step in the broader task of identifying variables of interest for capturing the impact of natural disturbances using the HPM. Natural disturbances possess several unique aspects, including the interaction between risk and amenities, and temporal and spatial dynamics, that have consequences for measuring their influence on the variable of interest. Depending on these unique factors, the price response in property markets to a natural disturbance can be subtle and therefore difficult to detect, or robust and easy to identify. The choice of variables that will relate the disturbance impacts to observable outcomes in the market, as well as the econometric techniques to be employed, requires thoughtful consideration.

3.1 Risk and Amenities

Many natural areas present some risk of disamenity in addition to the amenities they provide. The same measures that are chosen to capture the positive spillovers from a resource in a HPM may also represent a source of risk to the household. For example, while decreasing the household's distance to a forest boundary may increase scenic woodland views, the risk of property damage due to a fire may go up as well. There is a danger that the use of a single distance or neighborhood variable to capture the price shock from a disturbance can result in the "netting" of amenity and risk components in the results. For example, Portney (1981) cautions that the estimated value of risk reduction from improvements in air quality can be conflated with the amenity values of cleaner air. In some circumstances it may be possible to include variables which represent both the amenity and risk components of a resource in an attempt to account for this tension. Donovan et al. (2007) consider a novel approach to show that positive amenity values overcome the negative impacts of risk. Modeling the price impacts of risk and amenity attributes requires very careful selection in the variables which will convey the impacts in the model.

Common variable choices for measuring disturbance risk and changes in natural amenities from a disturbance event include the distance to at-risk or impacted areas and the share of land in a neighborhood surrounding the house that is at-risk or impacted. Very precise variables, such as the number of trees within a 100-meter radius that are infected with an invasive species, are also possible but require significant time and effort in data preparation. In choosing to use neighborhood measures, the extent of impact around each data point must be considered. This involves identifying how a disturbance shock to price decays as distance from the impacted area increases. For example, a 50-meter neighborhood around a house would not be sufficient to model the price impacts of a wildfire that damaged a trailhead one mile away. However such a localized neighborhood might suffice for an invasive species study where house values capitalize dead and damaged trees near the property.

The actual, or objective, probability that a household will experience a disturbance may be quite low. Vectors for invasive species may be relatively rare such that the likelihood that a household has one within its parcel boundary or experiences spillovers is small. Likewise, the chance of wildfire burning any one acre and hence affecting a household is very small. Measuring the true, objective probability level for these very infrequent events is difficult both for the household and the modeler. While distance and neighborhood variables can be used to proxy for risk, variables that provide information on the risk-averting behavior of the household can be useful. Homeowners take precautionary steps, such as installing fire-resistant roofs or treating trees to thwart insects and diseases, to protect their property from disturbances. They may also participate in collaborative efforts with other households to reduce risk in their broader neighborhood. These self-protection and community efforts can help to reveal the household's perceived, or subjective, assessment of risk, the value of which can be inferred from the HPM.

3.2 Temporal Dynamics

The speed with which a disturbance occurs and spreads across the landscape can vary dramatically. The damage from hurricanes may occur in a matter of hours, whereas wildfire impacts may occur over days to weeks and insect outbreaks may last for years. The shock to the market also has its own profile across time that may differ from that of the disturbance. Natural disturbances that operate at slow speeds may confound attempts to identify "before" and "after" time periods necessary for choosing the temporal window from which to select data and measure market shocks.

For disturbances that manifest at fast time scales, such as hurricanes or wildfires, variables that indicate the date of sale can identify the impact of the shock to the overall housing market. Interacting these time variables with variables that identify spatial variation in the shock, such as risk or amenity proxies, produces measures of a disturbance shock at a fine combination of temporal and spatial resolution. To better understand this technique, called "difference-in-differences", consider two different locations, one exposed to a natural disturbance and one not exposed. Let P^{td} be the price of a house at time period t (t=1 during or after the disaster, t=0 before) in location d (d=1 for the affected location, d=0 for the unaffected) and $\mathbf{Z}$ be a vector of housing characteristics. If $E[.]$ is the expectations operator, then the conditional difference-in-differences estimator,

$$\left\{E[\mathbf{P}^{11} \mid \mathbf{Z}] - E[\mathbf{P}^{10} \mid \mathbf{Z}]\right\} - \left\{E[\mathbf{P}^{01} \mid \mathbf{Z}] - E[\mathbf{P}^{00} \mid \mathbf{Z}]\right\} \tag{11.1}$$

accounts for the differences in price across locations as well as changes in price due to time that are not attributable to the disturbance. The first term in brackets is the difference in prices between the locations after the disturbance while the second bracketed term is the difference in prices between the locations before the disturbance. By subtracting out the difference in price that prevailed *ex-ante* from the difference *ex-post*, only the effect of the disturbance remains.

The difference-in-differences technique may not appropriate for modeling impacts from disturbances that do not occur on fast time scales with distinct start and stop dates. The slow, continuous spread of a disturbance at fine spatial scales complicates the identification of time t when the impact occurs. For instance, the relatively slow spread of an invasive species through small patches in the landscape blurs both the "before" and "after" necessary to identify the time of infection. A further complication arises when insects or diseases take several years to cause mortality in susceptible hosts. In contrast, a natural disturbance that rapidly spreads across a large area is well-suited to this technique since t can be easily identified.

3.3 Spatial Dependence

A final issue that has implications for the empirical estimation of natural disturbance impacts on housing markets is the identification and control of spatial dependence in the data. Spatial dependence is expected when the relative locations of sample observations matters (Bell and Bockstael 2000). Said differently, spatial dependence refers to a spatial association between values observed at different locations. Two potential sources of spatial dependence are of concern:

structural or spatial lag dependencies across observations on the dependent variable and spatial dependence across error terms. In the context of hedonic property value modeling, structural dependence arises, for example, when the sales value of one property is systematically influenced by the sales value of nearby properties. Spatial dependence among the errors is generally due to omitted variables, which are themselves spatially correlated but could also be due to errors in measurement that are systematically related to location. Property characteristics omitted from the hedonic property value model that are spatially correlated would result in spatially autocorrelated or dependent errors.

Spatial dependence has implications for the validity of OLS parameter estimates and variance-covariance estimates and therefore for the validity of hypothesis tests based on such results. If spatial lag dependence is present and ignored in the analysis, OLS will give biased and inconsistent parameter estimates. If spatial error dependence is present and ignored, OLS will produce unbiased parameter estimates but the standard errors associated with these estimates will be biased (inefficient). Spatial lag and error models can be used to correct for spatial dependence problems in the data. Refer to Anselin (1988) for a comprehensive discussion of spatial dependence.

The econometric modeling of spatial effects in housing price studies is at an early stage of development, and little is known about the spatial impact of natural disturbances on housing markets. However, we suspect that spatial econometric methods may be well-suited for identifying the property value impacts of localized disturbances, such as invasive species that operate at small patchy spatial scales and where value spillovers from infected to non-infected properties occur (Holmes et al. 2006). In contrast, spatial econometric methods may prove to be less useful for modeling disturbance impacts which are uniformly distributed across a housing market.

The forgoing discussion emphasizes that the temporal and spatial scope of the data, the list of variables of interest, and the specification of econometric models all need to be evaluated to account for the special nature of disturbances. Two case studies utilizing the HPM will now be presented to illustrate how the specific characteristics of a natural disturbance influence modeling decisions. The first case study analyzes the impact of a large wildfire on housing prices using the difference-in-differences estimator. The second case study investigates how a decline in forest health induced by an exotic forest insect—the hemlock woolly adelgid—is capitalized into housing prices.

4. WILDFIRE IMPACTS ON RESIDENTIAL PROPERTY VALUES

Wildfire is a common natural hazard in eastern Oregon and Washington. Rapp (2002) explains that frequent, low-intensity fires dominated the historic fire regime of the ponderosa pine forests of this area. Vegetation on the forest floor and small diameter, less fire-resistant trees burned but larger trees survived. This regime resulted in an open forest with low fuel levels. However, Rapp (2002) further reports that the fire regime has changed to one of more lethal fires that occur more often. As a result of timber harvesting, grazing, the introduction of nonnative plant species, and wildfire management policies that stressed fire suppression and exclusion, these forests have experienced an increase in the probability of severe, stand replacement fires. The result is that many dry, east-side forests have missed between 7 to 10 fire-return intervals.

A set of three wildfires burned over 180,000 acres in the Wenatchee National Forest and Chelan County, on the east side of the Cascades in central Washington, during the late summer of 1994. Suppression expenditures were almost $70 million, and the economic impacts included losses of personal property, timber, and tourism revenue (Carroll et al. 2000). This empirical application of the HPM will examine the property market impacts from these fires.

4.1 Empirical Model

This model uses the difference in differences technique to capture the impact of the fires on the amenity value of the forest. With a log-linear functional form, the general ordinary least squares (OLS) difference-in-differences hedonic estimating equation is

$$\ln P^{td} = \sum_i \beta_i z_i + \alpha t + \varphi d + \delta(t \cdot d) \tag{11.2}$$

where the z_i are housing characteristics, d is a measure of forest amenity (d = 1 if high, d = 0 if low), and t is an indicator of whether the house was sold after a disturbance event (t = 1 if after, t = 0 otherwise). In a very general sense, d could describe the proximity to a trailhead (d = 1 if close, d = 0 if far) or the quality of a viewshed (d = 1 if good views, d = 0 if poor views). Assuming that d measures only the amenity role of the forest (and does not include any risk components), it should be the case that $\phi > 0$ so that price increases with the amenity level. The outcome of interest is the coefficient on the product of the time and location dummies, δ, which is the equivalent of the conditional difference-in-differences estimator in equation (11.1):

$$\begin{aligned}
&\left\{E[\mathbf{P}^{11} \mid \mathbf{Z}] - E[\mathbf{P}^{10} \mid \mathbf{Z}]\right\} - \left\{E[\mathbf{P}^{01} \mid \mathbf{Z}] - E[\mathbf{P}^{00} \mid \mathbf{Z}]\right\} \\
&= \left\{\ln P^{11} - \ln P^{10}\right\} - \left\{\ln P^{01} - \ln P^{00}\right\} \\
&= \left\{\left(\sum_i \beta_i z_i + \alpha + \varphi + \delta\right) - \left(\sum_i \beta_i z_i + \alpha\right)\right\} - \left\{\left(\sum_i \beta_i z_i + \varphi\right) - \sum_i \beta_i z_i\right\} \\
&= \delta .
\end{aligned} \tag{11.3}$$

If $\delta < 0$ then it is possible to claim that the disturbance had a negative impact on the market price of a house due to an impaired amenity level. In this application, d is not binary but semi-continuous to represent how the amenity level varies with distance to the household.

4.2 Data

Residential housing transactions for 1992 through 1996 were obtained from the Chelan County Assessor's Office. A review of federal fire records (Coarse Scale Spatial Data 1999) showed that large fires in the study area were unusual. With the exception of the three large fires in 1994, during the period 1992-1996 on the Wenatchee NF in Chelan County, fewer than twelve fires exceeded 100 acres, the largest just under 600 acres. This reveals that the three largest fires in the summer of 1994, when added together, comprised the largest fire event recorded over the period covered by the set of sales transactions.

In addition to sales price, this dataset included a variety of structural variables such as date of sale, living area, date of construction, type of roof, and whether the house included a fireplace, hot tub, garage, carport, patio, or basement. Lot size in acres was also included. The Assessor's office provided a parcel map for the county which was used to spatially reference the sales transactions with ArcView GIS. The centroid of each parcel was used as the location of each property. To account for differences in neighborhood or community characteristics, census tract data for median household income were obtained from the U.S. Census Bureau. Additionally, the kilometers of road in a 0.40-kilometer (0.25 mile) radius around the parcel centroid (from the 2000 TIGER road file for Chelan County) were computed to account for the differing levels of urban development in the data. Lake Chelan is a large lake located in the county which is a popular recreation area. The distance to the lake was included to capture its amenity.

The measures for the amenity role of the forest are the distance in kilometers from the parcel centroid to the closest fire boundary, *fire_dist*, and the distance in kilometers from the parcel centroid to the national forest boundary, *nat_for_dist*. Distance to the national forest embodies characteristics such as access to recreation and viewshed while distance to the burned area controls for the fire-induced change in forest condition.

NFPA 1144, Standard for Protection of Life and Property from Wildfire by the National Fire Protection Association, Inc. (NFPA 2002) and the *Urban-Wildland Interface Code 2000* by the International Fire Code Institute (IFCI 2000) include surrounding vegetation and slope in their systems for assigning risk levels to properties in the urban-wildland interface. The National Fire Danger Rating System (NFDRS: Deeming et al. 1977) breaks vegetative fuel into three broad classes (not including slash): shrub (*shrub*), grass (*grass*), and evergreen (*egreen*). The National Land Cover Data (NLCD) grid provided the link between the NFDRS and the vegetation surrounding each parcel for measuring vegetative risk. Vegetative risk was measured by the percent of land in a 190-meter neighborhood surrounding a parcel centroid that was in each of the three broad fuel classes of the NFDRS as shown by the NLCD grid. A mosaic of 7.5-minute digital elevation models (DEMs) with 10-meter resolution from the USGS was used to produce a countywide slope grid and measures of slope (*slope*) were developed for the 190-meter neighborhood. Together, the vegetation and slope variables proxied for the level of wildfire risk around each property.

Roofing type, which was included in the data received from the assessor's office, was chosen as the measure of structural fire resistance. Roofing class is measure of fire-resistance, with class-A being the highest level. The *roof* variable (*roof* = 1 for class-A, *roof* = 0 otherwise) will be used to represent the household's self-protecting or averting behavior and infer attitudes on risk.

To account for changes in the general price level in the Chelan residential property market, binary variables for the six month period the sale occurred were included and named *sd921*, *sd922*, *sd931*, etc. where *sd9xy* indicates a sale in year *199x* during six month period *y* (*y* = 1 for the first six months, *y* = 2 for the second six months).

The amenity variables *nat_for_dist* and *fire_dist*, the distances from the parcel centroids to the national forest and fire boundaries which proxy for the level of forest amenity, were applied to the difference-in-differences technique. To control for the possibility that the effect of the three fires was transient and would not be detected using a simple before and after measure, the post-fire indicator is decomposed into the five six-month periods during and after the fire. The corresponding five sales date dummy variables were multiplied by *nat_for_dist* and *fire_dist* and are named *nat_for_dist942*, *fire_dist951*, etc. These variables are the equivalent $t \cdot d$ of in equation (11.2). This technique was also applied to the roofing material dummy (*roof*) to examine how the valuation of self-protection evolves after the fires. Table 11.1 contains the summary statistics of the focus amenity and risk variables.

Table 11.1. Descriptive statistics for risk and amenity focus variables in Chelan County, WA.

Variable Name	Symbol	Mean	Std. Dev.
Class-A roof (Yes = 1, No = 0)	*roof*	0.822	0.383
Distance to national forest boundary (km)	*nat_for_dist*	3.638	2.328
Distance to fire boundary (km)	*fire_dist*	13.778	8.353
Share in grass (%)	*grass*	6.861	15.279
Share in shrub (%)	*shrub*	10.055	16.688
Share in evergreen (%)	*egreen*	8.505	21.387
Slope	*slope*	4.619	4.969

4.3 Results

Ordinary least squares regression results are presented in table 11.2. Only coefficient estimates for the variables of interest related to the wildfire impacts are presented here. Huggett (2003) includes the complete results with coefficients for the structural variables. All implicit prices are evaluated at the mean price of \$114,315[1]. The general price level falls by \$16,377 from the second half of 1994 to the first half of 1995 as evidenced by the coefficients on *sd942* and *sd951*. Comparing these results with previous work that uses pre- and post-fire indicator

Table 11.2. OLS results for Chelan County, WA. Dependent variable is the natural log of price.

Variable	Estimate	$p > \chi^2$
sd942	0.268	<0.0001 **
sd951	0.152	0.020 **
sd952	0.272	<0.0001 **
roof	-0.118	<0.0001 **
roof942	0.070	0.070 *
roof951	0.055	0.198
roof952	0.112	0.005 **
roof961	0.175	0.001 **
roof962	0.014	0.673
nat_for_dist	-0.006	0.314
nat_for_dist942	-0.006	0.447
nat_for_dist951	0.005	0.671
nat_for_dist952	-0.001	0.894
nat_for_dist961	0.015	0.082 *
nat_for_dist962	0.007	0.452
fire_dist	-0.006	0.000 **
fire_dist942	-1.93e-4	0.930
fire_dist951	0.006	0.048 **
fire_dist952	-1.04e-4	0.965
fire_dist961	-0.006	0.010 **
fire_dist962	-0.003	0.127
egreen	-0.001	0.008 **
shrub	-7.47e-5	0.866
grass	-3.55e-4	0.487
slope	0.002	0.193
R^2	0.61	
No. of obs.	4,720	

Note: ** denotes significance at 5%, * at 10%.

[1] With a semi-log functional the implicit price for non-binary variables is $B*P$ where B is the coefficient estimate and P is price. For binary variables, the implicit price is $\{\exp[B - .5 * V(B)] - 1\}*P$ where $V(B)$ is the variance of B (Kennedy 1981).

variables, this drop of 13 percent to 14 percent of mean price is between the upper bound of 11 percent in the PricewaterhouseCoopers report (2001) on the Cerro Grande fire and the 15 percent loss that Loomis (2004) found with the Buffalo Creek fire. Absent any other contemporaneous shocks this represents a broad fire-induced response in the Chelan County residential housing market in the first half of 1995. However this shock appears to be short-lived as the price level in the second half of 1995 (*sd952*) increases to the pre-fire level.

The coefficient estimate for *fire_dist* is negative and highly significant, indicating that prior to the fires households placed a premium on living near the area that would burn. An additional kilometer from the burned area prior to the fire discounts price by $676. The negative coefficient estimate of *fire_dist* reveals that the area that burned, in its unburned state before the second half of 1994, possesses some qualities that were unique from the rest of the national forest such as viewshed or opportunities for recreation. The coefficient on *fire_dist951* is positive and significant. For the first half of 1995 *fire* and *fire_dist951* combine to add $48 to price for each additional kilometer from the burned area. None of the distance-to-national-forest variables are significant for the 18 month period during and after the fires. These results indicate that while the fires had no impact on the overall value that households place on living near the national forest, the value for living near the burned area did fall in the first half of 1995 in response to the decreased amenity level. However this response was temporary and disappeared after the first six months of 1995.

For the neighborhood risk proxies, each percent increase in evergreen cover in a 190-meter neighborhood of the house decreases price by $165. This discount for higher evergreen density corresponds to the findings of Garrod and Willis (1992b) and Wells (2001). The coefficients of *slope*, *shrub*, and *grass* are not significant.

The signs on the coefficient estimates of the roof and sales date interaction variables indicate that having a fire-resistant roof detracted from the price of a house prior to the fire. A class-A roof lowers price by $12,742 from the beginning of 1992 through the beginning of 1994. The value of a fire-resistant roof increases by $8,190 in the second half of 1994, $13,452 in the second half of 1995, and $21,699 in the first half of 1996 over the pre-fire value. There are several explanations for the increase in the valuation of a fire-resistant roof, including a reassessment of the prevailing risk of wildfire in Chelan County in the presence of increased information (Kask and Maani 1992), increased post-fire demand for fire-resistant roofing, and a supply restriction due to fire-delayed plans to put property with class-A roofs on the market.

This example of the HPM has sought to empirically measure the relationship between the realization of a wildfire event and residential housing prices by accounting for both the spatial variability in fire risk and the change in amenity. The results reveal significant post-fire price impacts on the general price level, the valuation of forest amenity, and the valuation of self-protection.

5. EXOTIC FOREST INSECTS AND RESIDENTIAL PROPERTY VALUES

Our second example of the economic impacts of a forest disturbance on private property values considers the case of an invasive forest insect—the hemlock woolly adelgid. The hedonic property value method is used to evaluate both the timing and the magnitude of economic impacts resulting from a gradual decline in forest health.

The HWA was accidentally introduced into Virginian forests from Japan during the 1950's and causes mortality to eastern and Carolina hemlocks. During the past half-century, the HWA has spread to hemlock forests in the Northeast, the Mid-Atlantic region, and the South. Eastern and Carolina hemlocks have shown no resistance to HWA, and once trees are moderately or severely infested, there is little chance for recovery. Dramatic losses of hemlock forests throughout the eastern United States are likely unless successful control measures are found. Hemlocks are also widely used as ornamental trees in residential landscapes. During the early stages of an infestation, individual trees in residential landscapes, or specimens located close to roads, can be successfully treated using insecticidal methods. Severe infestations cause defoliation and a gradual loss of tree vigor, typically resulting in tree death as the extent of defoliation progresses over several years.

Northwestern New Jersey was chosen for the study site as the impact of HWA on hemlock forests in this area is well documented (Royle and Lathrop 1999). The 80 square mile township of West Milford is located in the Highlands region, and had a population of 26,410 according to the 2000 census. The area is characterized by farms, small villages and towns, lakes, forests and wetlands.

5.1 Model

This case study employs the hedonic property value model to examine the effects of hemlock decline on residential property values. To understand the model, first recall that a semi-log hedonic price function can be specified as:

$$\ln\mathbf{P} = \mathbf{Z}\boldsymbol{\beta} + \boldsymbol{\varepsilon} \tag{11.4}$$

where $\ln\mathbf{P}$ is an n x 1 vector of the natural log of price, $\mathbf{Z}$ is an n x m matrix containing explanatory variables, and ε is the n x 1 vector of errors which are distributed normally with zero mean and variance of σ^2. As the impacts of HWA on hemlock health are gradual, with symptoms of decline and finally death extending over several years, a pertinent issue is to identify the point in time at which hemlock decline registers an impact on property prices. We hypothesize that, as hemlock health declines, a threshold is crossed beyond which the presence of hemlocks on the landscape quantitatively shifts the property value function.

The impact of hemlock decline on property value is specified using two related variables. The first variable, *h_forest*, specifies the total area of hemlock trees on parcels sold throughout the period covered by the data record. A second hemlock variable, *h_threshold*, is specified to evaluate the point in time at which hemlock decline shifts the property value function. The model we estimate is specified as:

$$\ln\mathbf{P} = \alpha_1 h\_forest + \alpha_2 h\_threshold + \mathbf{Z}\boldsymbol{\beta} + \varepsilon \tag{11.5}$$

where

$$h\_threshold = dummy_t \cdot h\_forest \tag{11.6}$$

and $dummy_t = 1$ for year t and all subsequent years in the data record; $dummy^t = 0$ otherwise. The parameter α_1 provides an estimate of the percentage change in property value with respect to a one unit change in the area of hemlock trees on properties that sold prior to the threshold year. The sum of the parameter estimates ($\alpha_1 + \alpha_2$) provides an estimate of the percentage change in property value with respect to a one unit change in the area of hemlock trees on properties that sold after the threshold was crossed. Alternative threshold values are tested in the model specification in order to isolate the value of t at which a statistically significant impact on the property value function is identified. If more than one value of t is associated with a statistically significant impact, the year associated with the greatest level of statistical significance is reported.

5.2 Data

Housing data for 1992 through 2002 were obtained from the town clerk of West Milford, New Jersey. After cleaning the raw data there were 4,373 usable observations. Available in the data were sales prices and the date each residential property was sold. Structural housing characteristics included square footage of living area, number of bedrooms, number of bathrooms, the year the house was built, and whether the basement and/or attic had been finished. The data also included the size of the parcel in acres.

The average sale price in the sample was $177,752 (nominal dollars). Dummy variables were included in the model specification for the year of sale. The parameter estimates on these variables control for housing price inflation in this market.

Landsat satellite imagery, at a resolution of 30m^2, was used to construct land cover and land use variables for each individual parcel. At this degree of spatial resolution, observations on land cover variables represent groups of trees or other cover types and do not represent individual trees. Land cover variables were measured in acres. Variables used in the model specification include highly developed land, medium and low development, deciduous forest cover, hemlock forest cover, other (non-hemlock) coniferous forest cover, mixed (deciduous and other coniferous) forest cover, agricultural land, wetlands, and area covered by streams, ponds, and lakes. Roughly 8 percent of the total land area in West Milford was covered by hemlocks. Of the total number of observations in the cleaned data set, 329 observations were for parcels with the hemlock cover type present.

Hemlock health data were available for the years 1992-2002 (fig. 11.1). Four hemlock health classes were created from the remote sensing data: (1) a combination of healthy and lightly defoliated hemlocks (less than 25 percent defoliation), (2) moderately defoliated hemlocks (25-50 percent defoliation), (3) severely defoliated hemlocks (50-75 percent defoliation), and (4) dead hemlocks (greater than 75 percent defoliation). Although a mix of healthy and unhealthy (moderately defoliated, severely defoliated, and dead) hemlocks was identified on parcels sold throughout this period, it is apparent that hemlock health declined rapidly on parcels sold in 2000 and subsequent years.

Descriptive statistics for land cover variables at the parcel level are shown in table 11.3. The average parcel sold during the study period was approximately 0.6 acres in size. On average, the most common land cover on parcels sold was low and medium development, followed by deciduous forests. Stands of hemlocks occupied about 7 percent of the land area, on average, on sales parcels. However, parcels with hemlocks were somewhat larger than average (0.8 acres) and hemlock coverage was the dominant land cover on these parcels (0.5 acres).

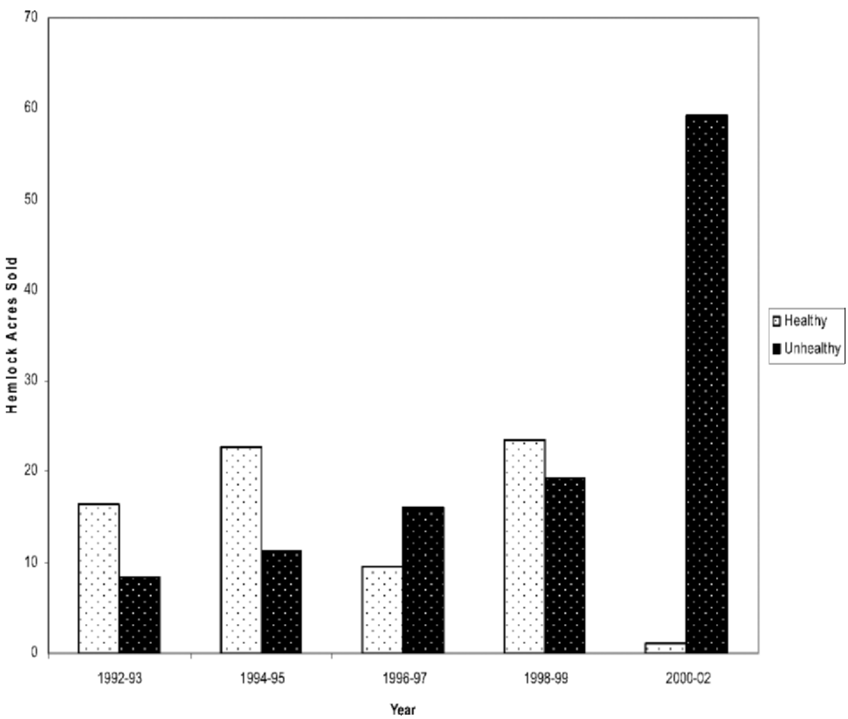

Figure 11.1. Area of healthy and unhealthy (moderately defoliated, severly defoliated, and dead) hemlocks on parcels sold, by time period for West Milford, NJ.

Table 11.3. Descriptive statistics for land cover variables in West Milford, NJ.

Variable Name	Symbol	Mean	Std. Dev.
Hemlock forest (ac.)	*h_forest*	0.039	0.197
Hemlock threshold (ac.)	*h_threshold*	0.012	0.107
Deciduous forest (ac.)	*d_forest*	0.159	0.503
Other coniferous forest (ac.)	*oc_forest*	0.001	0.017
Mixed forest (ac.)	*m_forest*	0.024	0.124
Wetland (ac.)	*wetland*	0.017	0.126
Other water (ac.)	*o_water*	0.003	0.024
Agriculture (ac.)	*ag*	0.002	0.019
Development: high (ac.)	*dev_high*	0.006	0.032
Development: low/med (ac.)	*dev_lm*	0.357	0.296
Grass (ac.)	*grass*	0.001	0.020

Similar to hemlock forest cover, other coniferous and mixed forest stands were, on average, relatively scarce across the entire sample. Wetlands, lakes/ponds, agricultural land, and grass held minor, but potentially important, positions in the distribution of land cover types represented in the sales records.

5.3 Results

Results of the OLS regression model are shown in table 11.4. Although not included in the table, all of the parameters for structural housing characteristics except "finished basement" were significant at the 1 percent level, with the expected signs. Additionally, all of the time dummy variables that were used to control for house price inflation were significant at the 1 percent level.

The model fits the data relatively well, with an R^2 value of 0.58, and the results indicate that several land cover variables are capitalized into property values. Parameter estimates for deciduous forest cover, mixed forest cover, water, agricultural land, grass, high development, and low and medium development were statistically significant at the 10 percent level or higher. The parameter estimate for hemlock forests during the period early on in the HWA outbreak was not significantly different than zero (indicating that hemlock forests during this period did not add to or subtract from property value). However, the parameter estimate on hemlock forests late in the epidemic was negative and significant at the 5 percent level. The best fitting model indicated that the decline in hemlock health crossed a threshold for sales occurring during and subsequent to the year 2000, which is consistent with the distribution of hemlock health classes shown in figure 11.1. In particular, the results indicate that a one acre increase in the area of hemlock decreases property value by 8.3 percent during this period. This loss in value is presumably due to the presence of severely defoliated and dead hemlocks which detract from the aesthetic quality of the landscape.

Table 11.4. OLS results for West Milford, NJ. Dependent variable is the natural log of price.

Variable	Estimate	$p > \chi^2$
h_forest	-0.036	0.184
h_threshold	-0.083	0.036 **
d_forest	0.018	0.100 *
oc_forest	-0.127	0.794
m_forest	0.072	0.041 **
wetland	-0.009	0.809
o_water	0.576	0.005 **
ag	0.409	0.013 **
dev_high	0.495	0.013 **
dev_lm	0.155	<0.0001 **
grass	-0.538	0.081 *
R^2	0.58	
No. of obs.	4,373	

Note: ** denotes significance at 5%, * at 10%.

6. MANAGEMENT AND POLICY IMPLICATIONS

The effect of a disturbance on property markets is one component of its overall economic impact. For example, Butry et al. (2001) estimate that the 1998 wildfires in northeastern Florida, which burned approximately 500,000 acres and which were concentrated in the St. John's Water Management District, resulted in $600 to $800 million in economic losses. This estimate includes $10 to $12 million in insured property losses but no realized losses from the sale of undamaged property. The results presented here from the wildfire and hemlock woolly adelgid case studies reveal statistically significant disturbance-induced impacts to housing prices beyond those related to the structural damage to the house. Although disturbance price impacts may be transient and are unrealized by the household until a sale, any realized losses (or gains) unrelated to insured damages warrant inclusion in economic analyses of disturbance events.

Aggregation of disturbance impacts across a property market must be done with care, as specific assumptions about the stability of the hedonic price schedule must be acknowledged. Bartik (1988) and Freeman (1993) suggest models for calculating the social welfare change from a change in amenity using HPMs. The transient nature of some disturbance impacts implies that there is a transfer of wealth from the seller to the buyer if a sale occurs before price returns to the pre-disturbance level. In the case of the Chelan fires, the 421 residential properties that sold in the first half of 1995 experienced a total decline in sales price of almost $6.9 million compared to a hypothetical sale date in the second half of 1994 assuming all else equal. This figure does not include impacts from decreased amenity values.

In the case of the HWA outbreak in West Milford, New Jersey, the 104 properties that sold after the threshold year 2000 experienced a total loss of about $1.2 million relative to sales of parcels with hemlocks before that period. This loss is presumably due to a loss in amenity value as well as pending costs associated with restoring the site to a desirable condition. As such, it is reasonable to propose that property owners with stands of hemlock that did not sell their property during this period likewise experienced a utility loss and other associated damages, although such losses were not capitalized into property values because a sale did not occur.

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CHAPTER 12

CONTINGENT VALUATION OF FUEL HAZARD REDUCTION TREATMENTS

John B. Loomis and Armando González-Cabán

1. INTRODUCTION

Increasing numbers of wildfires each summer has brought forward legislative and administrative proposals for expanding prescribed burning and mechanical fuel reduction programs. A policy of accelerating the amount of land to be mechanically thinned or prescribed burned is not without opposition. Prescribed burning can generate significant quantities of smoke that affects visibility and aggravates health problems for people with respiratory conditions. Prior initiatives to increase prescribed burning in states such as Florida and Washington have often been limited by citizen opposition due to smoke and health effects. The prescribed burning program is also expensive and costs as much as $250 per acre or more in some parts of the country. Thus, a policy relevant issue is whether the benefits of fuel reduction policies exceed the costs.

This chapter presents a stated preference technique for estimating the public benefits of reducing wildfires to residents of California, Florida, and Montana from two alternative fuel reduction programs: prescribed burning and mechanical fuels reduction. The two wildfire fuels reduction programs under study are quite relevant to people living in California, Florida and Montana because of these states' frequent wildfires[1]. The methodological approach demonstrated here has broad applicability to other fire prone areas of public land as well.

Wildfire on public land reduces the quality of forest recreation for some types of visitors and reduces the level of public goods arising from the forests such as water quality, and habitat for some wildlife species. Most of these resources adversely affected by wildland fire are not traded in markets, and thus, society does not have market prices as a guide to the economic values lost. Further, many of the goods and services lost from forests are public goods. The defining characteristics of public goods are that once the good is provided, no one can be excluded from consuming them, and that one person's consumption does not

[1] During the last several years these three states have experienced some of their worst fire season: California in 2003, Florida and Montana in 2000.

T. P. Holmes et al. (eds.), The Economics of Forest Disturbances: Wildfires, Storms, and Invasive Species, 229–243.

reduce the amount available to others. Protection from wildland fire and the associated risks is the public good under study in this paper. Households living at the wildland urban interface receive benefits from fuel reduction projects that reduce the intensity and extent of forest fires nearby where they live. These benefits reduce unhealthy levels of wildfire smoke, risk to their property, and loss of the aesthetics of surrounding forest landscapes. Others that enjoy forest recreation also benefit from protecting forests from catastrophic wildfire (Loomis et al. 2001).

There are several ways to estimate some of the effects of fire, such as expenditures made by households to avert some of the effects of wildfire on their private property. But this situation is different when public forests are involved, and the effects of wildfire affect non market public goods such as air quality, water quality, and habitat of non-game and endangered species. The public good values affected include those with direct human use values (e.g., air quality, water quality for drinking purposes), as well as passive use or existence values for protection of endangered species habitat. Stated preference methods such as the contingent valuation method (CVM) and conjoint/choice experiments are the primary methods capable of valuing both use and passive use values. Both of these techniques involve construction of a simulated market or simulated referenda to allow people to state how much they would pay for a particular level of one or more public goods.

In this chapter, the contingent valuation method is used to elicit how much households would pay for fuel reduction programs that reduce the number of acres of wildfire and number of houses lost. CVM has been applied to valuing a reduction in wildfire in approximately 3,000 acres of old growth forests that were habitat to threatened spotted owls (Loomis and González-Cabán 1998) and reducing risk of wildfire to property in rural Michigan (Winter and Fried 2001). The Winter and Fried study asked household how much of an increase in property taxes they would pay for a 50 percent reduction in probability of a wildfire. Their results averaged \$57 per year per household. Recently, Talberth et al. (2006) conducted a CVM study that elicited willingness-to-pay (WTP) of homeowners for private fire risk reduction actions (\$240), neighborhood fire risk reduction (\$95) and public land wildfire risk reduction (\$64). These annual WTP amounts are highest to protect one's own house, then neighborhood and then the public forests.

If forest managers simply wish to value an entire program or bundle of actions, CVM is often the easiest way to do it. However, if managers are interested in the individual values of different features of a fuel reduction program (e.g., smoke, probability of escape, etc), then choice experiments provide a method to estimate marginal values for each of the attributes of a fuel reduction program. In this chapter, we demonstrate the simpler contingent valuation approach. For more details on choice experiments and attribute based methods see Holmes and Adamowicz (2003).

1.1 WTP Model

In a contingent valuation survey we elicit an individual's WTP for the public program or public good. WTP is the conceptually correct measure of benefits for a new or expanded program. There are several ways in which WTP can be elicited from respondents. It can be asked as an open-ended question (e.g., what is the most you would pay), a payment card (e.g., please circle the maximum amount you would pay), or a closed-ended or dichotomous choice question format (e.g., would you pay a given monetary amount—yes or no?). As suggested by the National Oceanic Atmospheric Administration panel on contingent valuation, a closed-ended voter referendum WTP question format was used (Arrow et al. 1993). This casts the willingness-to-pay decision as voting for or against a given monetary amount. The magnitude of the monetary amount (call the bid amount) is varied across the sample.

Hanemann (1984), suggests how a respondent may answer a voter referendum or dichotomous choice CVM question. We assume that an individual's utility is a function of the public good bundle that represents the nonmarket benefits of reduced wildfires such as water quality, and endangered species habitat protection. This is represented by PG. The utility is also a function of the consumption of all private goods. Given the budget exhaustion by consumers, we can represent this composite commodity by their initial income (I) prior to paying for the public good. Therefore the utility function can be represented as:

$$U = f(\mathrm{PG}, I) \tag{12.1}$$

The utility derived from the combination of public and private goods is known to the individual but it is not directly observable by the researcher, because some part of the preferences are not solely determined by observable socio-economic variables. Thus, while a portion of the utility function can be treated as deterministic, the unobservable portion is treated as stochastic. Therefore, the resulting indirect utility function and a stochastic element, is:

$$U = f(PG, I) = v(PG, I) + e \tag{12.2}$$

where e represents an independent identically-distributed error term with a zero mean.

Under the dichotomous-choice approach, survey respondents are asked whether or not they would pay to maintain the public good if the costs to them were \$$X$. The respondent will answer Yes if her/his utility from the public good (with the associated loss of \$$X$ in income) is greater than or equal to her/his utility level with full income, but without the public good. Thus, a "YES" respondent intends to receive the public good (PG=1), and a reduction in income by \$X; while the "NO" respondent does not receive the increment in the public good, but retains their full income (PG=0). Therefore, the probability of a YES response is represented as follows:

$$P(\mathrm{YES}|\$X) = P[f(\mathrm{PG}{=}1, I\text{-}\$X) > f(\mathrm{PG}{=}0, I)] \tag{12.3}$$

Because the individual's utility function is not observable to the researcher, we introduce the stochastic element from the utility function in equation (12.2), which results in the following transformation of the probability function into equation (12.4):

$$P(\text{YES}|\$X) = P[v(\text{PG}=1, I\text{-}\$X) + e_1 > v(\text{PG}=0, I) + e_2] \qquad (12.4)$$

where e_1 and e_2 are error terms with means of zero (Hanemann 1984). If the utility difference with the public good is greater than the difference in the error terms then the respondent is presumed to answer Yes to paying \$X. If the difference in error terms is distributed logistically (Hanemann 1984, Loomis 1987) then the responses to the dichotomous-choice question are analyzed using a binary logit model to estimate the parameters, and to allow for calculation of WTP. The basic form of the logit equation is:

$$\ln\left(p_i / 1 - p_i\right) = \beta_0 + \beta_1\left(Bid\right) + \beta_2\left(X_2\right) + \beta_3\left(X_3\right) + \ldots + \beta_n\left(X_n\right) \qquad (12.5)$$

where p_i is the probability of a yes response; β's are coefficients to be estimated; Bid is the dollar amount the household is asked to pay; and X's are other demographic and taste variables. A probit model results if the utility difference is distributed normally. The distributions of the probit and logit models are fairly similar over much of their range.

From the estimated coefficients in the logit or probit model, net WTP per household per year can be calculated using the formula from Hanneman (1989). Equation (12.6a) is used to calculate mean WTP from the logit models when WTP is greater than or equal to zero.

$$MeanWTP = \ln\left(1 + \exp^{\alpha}\right) / \beta \qquad (12.6a)$$

In equation (12.6a) α is the product of the coefficient and mean values for all independent variables excluding the bid coefficient, plus the constant; and β is the absolute value of the bid coefficient. Equation (12.6b) presents the median WTP, which in the logit model is equivalent to allowing WTP to be positive or negative (i.e., if a fraction of households receive negative utility from the prescribed burning program due, for example, to smoke emissions):

$$MedianWTP = \alpha / \beta \qquad (12.6b)$$

Because forest fire prevention programs are public goods that increase the safety of all households living in the area influenced by the fuels reduction program, the value per household can be multiplied by all the households in the region that would benefit to arrive at a total annual value. The total annual benefits can be summed over the years that the prescribed burn or mechanical fuels reduction program are effective to yield a present value, which can be compared to the costs of the prescribed burning or mechanical fuels reduction program to determine the economic efficiency of the program.

1.2 Survey Design

The crux of any contingent valuation survey is an accurate and clear description of the resource to be valued; the consequences of paying and not paying; as well as specifying the means by which the respondent would pay for the program. In this example, a survey booklet was developed in conjunction with forestry professionals in California, Florida, and Montana to convey information on the extent of the problem, and two possible programs to reduce the problem (i.e., prescribed burning and mechanical fuels reduction).

Specifically, the survey booklet described the acreage that is burned by wildfires in an average year in each state, as well as the typical number of houses lost to wildfire each year in each state. The effect of wildfire on forests, houses and air quality was illustrated with a color drawing showing the flame height and rate of fire spread. This picture is shown in figure 12.1 and was designed to allow comparison to prescribed burning (fig. 12.2).

A program increasing the use of prescribed fire or controlled burning in California, Florida, or Montana was described and illustrated next (fig. 12.2). Specifically, respondents were told that the prescribed burning fuels reduction program would reduce potential wildfire fuels through periodic controlled burning. It was acknowledged that prescribed burning does create smoke, although far less than a wildfire. Then, the survey booklet provided additional information and drawings contrasting wildfire and prescribed fire. As can be seen in figure 12.2, prescribed fire is shown to have much lower flame height and slower fire spread.

The cost of financing this prescribed burning program was described as a cost-share program between their state government and the county the individual lived in.

WILDFIRE

Figure 12.1. Illustration of a wildfire effects on forests lands, houses, and air quality, with a fire spread rate of one-half to two miles per hour and a flame height of 30 to 60 feet.

PRESCRIBED BURNING

Figure 12.2. Illustration of prescribed burning effects on forest lands, houses and air quality, with a fire spread rate of 60 to 120 feet per hour and flame height of 4 to 8 feet.

The WTP elicitation wording for California was:

> *"California is considering using some state revenue as matching funds to help counties finance fire prevention programs. If a majority of residents vote to pay the county share of this program, the Expanded California Prescribed Burning program would be implemented in your county on federal, state, and private forest and rangelands. Funding the Program would require that all users of California's forest and rangelands pay the additional costs of this program. ...If the Program was undertaken it is expected to reduce the number of acres of wildfires from the current average of 362,000 acres each year to about 272,500 acres, for a 25 percent reduction. The number of houses destroyed by wildfires is expected to be reduced from an average of 30 a year to about 12. Your share of the Expanded California Prescribed Burning program would cost your household $__ a year. If the Expanded Prescribed Burning Program were on the next ballot would you vote:*
> *___ In favor ___ Against?"*

Identical wording was used in Florida and Montana, except the number of acres and numbers of houses burned were changed to correspond with particular state numbers. For example, in Florida, currently 200,000 acres burn and 43 houses are destroyed in an average year. With the proposed program this would be reduced to 150,000 acres and 25 houses. In Montana, currently 140,000 acres burn and 20 houses are destroyed. With the program, these would be reduced to 105,000 acres and 8 houses, respectively.

The mechanical fire fuels reduction program was defined in the booklet as the following: "*Another approach to reducing the buildup of fuels in the forest is to "mow" or mechanically chip the low- and medium-height trees and bushes* *into mulch. This is especially effective at lowering the height of the vegetation, which reduces the ability of fire to climb from the ground to the top or crown of the trees. In addition, mechanical "mowing" slows the growth of new vegetation with the layer of mulch acting as a barrier….Mowing or mulching …is more expensive…due to increased labor and equipment needs…However, unlike prescribed burning, mulching does not produce any fire smoke."*

For the mechanical fuel reduction program, the survey booklet stated the same wildfire acreage reduction as achieved with the prescribed burning program, and stated that only one of the two programs would be implemented. The mechanical fire fuels reduction dichotomous choice WTP question in California was stated as follows:

> *"If the Mechanical Fire Fuels Reduction Program was undertaken instead of the Expanded Prescribed Burning Program, it is expected to reduce the number of acres of wildfires from the current average of approximately 362,000 acres each year to about 272,500 acres, for a 25 percent reduction. The number of houses destroyed by wildfires is expected to be reduced from 30 a year to about 12. Your share of this Mechanical Fire Fuels Reduction Program would cost your household $X a year. If the Mechanical Fire Fuels Reduction program were the ONLY program on the next ballot would you vote:*
> ___ *In favor* ___ *Against?"*

Identical wording was used in Florida and Montana, except the number of acres burned and numbers of houses were changed to correspondent with particular state numbers discussed after the wording of the prescribed burning WTP question.

The funding of both of these fuels treatment programs was explained as being on a county-by-county basis, where if a majority of the county residents voted for the program, the state would match funds for the approved counties and everyone in the county would be required to pay the additional stated amount for their county. The bid amount, denoted by $X, varied across respondents and had the following values: $15, $25, $45, $65, $95, $125, $175, $260, $360, and $480. The bids were allocated equally across the sample. This range of values was based on prior surveys regarding Oregon and California respondents WTP for reducing fires in old growth forests to protect spotted owls (Loomis and González-Cabán 1998).

2. DATA COLLECTION AND SURVEY MODE

The surveys were conducted in Florida in 1999 and in 2001 in California and Montana through a phone-mail-phone process in all three states. To obtain a representative sample of households, random digit dialing of the households living in a sample of California, Florida, and Montana counties was performed. The counties were selected so there was a mix of counties that frequently experience wildfires, counties that occasionally experience wildfires, and counties that rarely experience wildfires. This variation had two useful features. First, it ensured variation in responses to questions like whether the respondent "had seen a wildfire". Second, targeting this sample aided in generalizing the results to all areas of the state.

Once initial contact was established, language was verified (except in Montana), along with elicitation of initial attitude and knowledge of wild and prescribed fire, followed by the scheduling of appointments with individuals for detailed follow-up interviews. During the interim time period, a color survey booklet was mailed to the household. These interviews were conducted with the aid of this color booklet. The booklet was sent in English to Caucasians and in Spanish to Hispanic households. The individuals were asked to read the survey booklet prior to the phone interview. Phone interviews were conducted in either English or Spanish depending on the language of the booklet received.

2.1 Survey Response Rate

Because the survey was conducted in two waves and with two ethnic groups in California and Florida, the response rates are compared from the initial random digit dial phone survey and the follow-up indepth interviews separately (table 12.1). While, the response rates to the initial phone calls were all over 40 percent, only in California there is a statistically significant difference between the groups in response to the initial phone call. The highest response rate (85 percent) is by

Table 12.1. Comparison of response rates in California, Florida, and Montana

	California		Florida		Montana
	Caucasians	Hispanics Spanish	Caucasians	Hispanics Spanish	Caucasians
First Wave – Screener Interview					
Total initial sample Contacted	794	620	840	652	602
Completed initial	328	468	714	553	406
1st Wave response rate (%)	41.3	75.5	85	85	67
Second Wave – In Depth Interview					
Net sample for 2nd wave	257	420	714	553	381
Total surveys completed	187	139	443	336	272
2nd wave response rate (%)	72.8	33.1	62	61	71

Hispanics phoned by a Spanish-speaking interviewer in Florida. The extra effort to contact people in their native language was definitely worthwhile in the initial interview.

Unfortunately, in the indepth interviews, after mailing a Spanish language booklet to Hispanic households, a relatively low response rate of 33 percent for California was obtained in this phase. The experience in Florida was different where the English and Spanish response rates to unit non-response, completed screener, and completion of the entire survey process are very similar. The participation rate for the initial screener was the same at 85 percent for both populations and near identical for the indepth interview at 62 percent for English and 61 percent for Hispanics (table 12.1, Second wave). A χ^2 statistic confirms no significant difference in response rate by Caucasians and Hispanics to the initial screener survey or the main interview in Florida.

In Montana, a total of 602 Caucasian households were contacted and 406 of them or 67.4 percent completed the initial interview. Of these 381 were available for the in-depth interview. The in-depth interview resulted in 272 completions, for a response rate of 71.4 percent on this phase (table 12.1, Second wave).

2.2 Specification of the Logit WTP Models

Because we are dealing with two programs, in three states and two ethnic groups, there could potentially be up to 12 separate logistic regression equations. For the purposes of this chapter, and to estimate a generalized WTP function for each program that could be applied to other states in a form of benefit function transfer, a relatively simple logistic regression model was estimated. To facilitate transfer to other states only simple demographics like household income and education levels were included, and survey variables were omitted since their values would not be known for other states. We did test whether a state specific intercept shifter dummy variable and a corresponding state specific bid interaction variable were statistically significant. As a general rule, we retained variables whose coefficients had t-statistics about one or higher.

The general form of the logistic regression model for the prescribed burning and mechanical fuel reduction for Caucasians interviewed in California, Florida, and Montana (where Montana is the base case, so no shift variable is needed):

$$\ln\left(p_i/1-p_i\right)=\beta_0+\beta_1\left(Bid\right)+\beta_2\left(CAState\right)+\beta_3\left(CABid\right)+ \beta_4\left(FLState\right)+\beta_5\left(FLBid\right)+\beta_6\left(Income\right)+\beta_7\left(Ed\right) \tag{12.7}$$

where CABid and FLBid are interaction terms of CAState or FLState and the Bid amount variables, and Ed is the household education levels in years.

The Hispanic logistic regression models (equation 12.8) are similar to equation (12.7), except that because interviews only took place in California and Florida, the state of Florida is used as the base case, so there are no intercept shifter variables or bid interaction variables for Florida.

$$\ln\left(p_i/1-p_i\right)=\beta_0+\beta_1(Bid)+\beta_2(CAState)+\beta_3(CABid)+\beta_4(Income)+\beta_5(Ed) \tag{12.8}$$

3. RESULTS

3.1 Descriptive Statistics by Fuel Reduction Program and Samples

The percent yes to the dichotomous choice CVM question for each state and each program, along with the key demographics were computed (table 12.2). As shown in table 12.2, the prescribed burning program consistently received 60 percent or higher Yes responses, ranging from a high of 84 percent among Hispanics in California to 60 percent among Caucasians in Montana. The mechanical fuel reduction program support was much lower among Caucasians, being only 34 percent to 50 percent, but 50 percent to 68 percent among Hispanics. Education levels and household income were highest in California and lowest in Montana for Caucasians. Hispanics education levels and income were higher in Florida than in California.

3.2 Results of Logit Regressions

In the logistic regression equations that follow, we started estimations with the full model specified in equation (12.7) (i.e., state intercepts and bid interaction terms, income and education) and then dropped any variable that was not significant at least at the 0.33 p value, correspondingly roughly to a t-statistic of one. This seemed a reasonable trade-off between avoiding omitted variable bias and minimizing variance due to inclusion of irrelevant variables.

The logit equation for Caucasian residents of the three states for the prescribed burning and mechanical fuel reduction programs are presented (table 12.3a). The

Table 12.2. Selected descriptive statistics of the sample by fuel reduction program

	California	Florida	Montana
Caucasians			
Income	$71,797	$53,078	$45,905
Years of Education	15	15	14
Yes Prescribed Burning (%)	75	73	60
Yes Mechanical (%)	50	45	34
Hispanics			
Income	$32,947	$37,982	
Years of Education	12	14	
Yes Prescribed Burning (%)	84	64	
Yes Mechanical (%)	68	50	

Table 12.3a. California, Florida, and Montana logistic regression for Caucasians WTP for Prescribed Burning and Mechanical Fuels reduction

	Prescribed Burning		Mechanical Fuel Reduction	
Variable	Coefficient	t-Statistic	Coefficient	t-Statistic
Constant	1.5986	2.376**	-0.3826	-1.850*
CASTATE	0.6782	2.683***	0.3694	1.227
CASTATEBID			0.0015	1.103
FLSTATE	0.3789	1.756*	0.3182	1.696*
INCOME	2.94E-06	1.033	4.11E-06	1.863
EDUC	-0.457	-0.976		
BID	-0.0042	-5.981***	-0.0032	-3.941***
Mean dependent var		0.6875		0.4238
McFadden R-squared		0.0708		0.0391
Log likelihood		-341.64		-451.15
Rest. Log likelihood		-367.68		-469.54
Likelihood Ratio statistic (5df)		52.067***		36.774***
Probability (LR stat)		5.23E-10		6.65E-07
Obs with Dep = 0	185			397
Obs with Dep = 1	407			292

*, **, and *** Indicates significance at the 0.1, 0.05, and 0.01 levels, respectively.

bid coefficients are negative and statistically significant (p<.01), indicating the higher the dollar amount respondents were asked to pay, the lower the chances they said they would pay. This shows internal validity to the CVM responses, i.e., respondents took the dollar amount they were asked to pay seriously, otherwise the bid coefficient would not be statistically significant and negative. In the prescribed burning program, the California (CAState) and Florida (FLState) intercept shifters were positive and statistically significant at the 1 percent and 10 percent level respectively, mirroring the higher percentage of Yes responses of these two states relative to Montana for the prescribed burning program.

In the mechanical fuel reduction program, the FL intercept shifter and income are positive and statistically significant at the 10 percent level. The California bid interaction variable (CABid) was also positive, which when added to the own price bid variable, indicates a more price inelastic response to the dollar bid amount than Florida and Montana.

A logit equation for Hispanic residents of California and Florida for the prescribed burning and mechanical fuel reduction programs (Florida is the base case) was estimated (table 12.3b). The bid coefficients are negative and statistically significant, indicating the higher the dollar amount respondents were asked to pay, the lower the chances they said they would pay. The California intercept shifter (CAState) and California bid interaction (CAStateBid) variables are

Table 12.3b. California and Florida logistic regression for Hispanics' WTP for Prescribed Burning and Mechanical Fuels reduction

	Prescribed Burning		Mechanical Fuel Reduction	
Variable	Coefficient	t-Statistic	Coefficient	t-Statistic
Constant	2.2285	2.700***	3.2653	5.509***
CASTATE	0.9883	3.947**		
CASTATEBID			0.0022	2.401**
INCOME	-4.94E-06	-1.102		
EDUC	-0.0796	-1.367	-0.2074	-4.829***
BID	-0.0026	-3.213***	-0.0022	-2.503**
Mean dependent var		0.7469		0.5885
McFadden R-squared		0.0711		0.0523
Log likelihood		-255.35		-380.65
Rest. Log likelihood		-274.93		-401.69
Likelihood Ratio statistic (5df)		39.141***		42.079***
Probability (LR stat)		6.51E-08		3.86E-09
Obs with Dep = 0 123				244
Obs with Dep = 1 363				349

*, **, and *** Indicates significance at the 0.1, 0.05, and 0.01 levels, respectively.

positive and statistically significant indicating that Hispanics in California are more likely to pay for these programs, and are less price sensitive. However, for the Mechanical fuel reduction program, the combined effect of the California bid interaction variable and bid coefficient is to essentially net each other out. Given this result, it is not possible to calculate WTP for California Hispanics for the Mechanical Fuel reduction program.

3.3 Willingness-to-Pay Results

Using equation (12.6a) Mean WTP was estimated for Caucasians and Hispanics (table 12.4). Mean (median) WTP of Caucasians for prescribed burning was \$460 (\$424) in California, \$392 (\$344) in Florida, and \$323 (\$254) in Montana. For the mechanical fuels reduction program the mean WTP of Caucasians in California was \$510, while it was much lower in Florida at \$239 and Montana at \$186. For the mechanical fuel reduction program the median WTP, which also allows for negative WTP of any respondent, is substantially less than the mean, with median WTP being about one-fifth in California (\$87) and Florida (\$48), and even slightly negative in Montana. These results are consistent with the lower percent yes in table 12.2, and suggest far less support for mechanical fuel reduction program than prescribed burning in these three states.

Table 12.4. WTP for Prescribed Burning and Mechanical Fuels reduction programs in California

	Prescribed Burning		Mechanical Fuel Reduction	
State	Caucasians	Hispanics	Caucasians	Hispanics
California				
Mean	$460	$838	$510	n/a
Median	424	794	87	n/a
Florida				
Mean	$392	$473	$239	$373
Median	344	344	48	124
Montana				
Mean	$323		$186	
Median	254			

Hispanics in Florida mean (median) WTP is $473 ($344) for the prescribed burning program, about half what Hispanics in California would pay $838 ($794). Hispanic's in Florida WTP for the mechanical fuel reduction program is $373 with a median WTP of $124.

The ranking of mean WTP per household in the states follows the magnitude of acres protected from fire and houses saved. While all the state programs represented an equivalent proportional reduction (25 percent reduction) in acres and houses burned, the absolute magnitude or amount of fewer acres that burned and number of houses saved did vary across states. In terms of the three states program, California would protect 90,000 acres and 18 houses. Florida would protect 50,000 acres and 18 houses, and Montana's 35,000 acres and 12 houses. The ranking of mean WTP per household is in the same order as the amount of acres prevented from burning and houses saved. In particular, mean WTP of California households are noticeably higher than Florida households, which are noticeably higher than Montana residents.

With mean willingness-to-pay of more than $400 per household, and more than 13 million households in California, the willingness-to-pay for either of these fuel reduction programs is about $5 billion. In Florida, with 7.6 million households this translates to about $3 billion for the prescribed burning and $2 billion for the mechanical fuels reduction program. In Montana, with only 366,000 households state level benefits would be close to $118 million for prescribed burning and $68 million for mechanical fuel reduction. Note, the survey explicitly indicated that only one of the programs would be implemented, so that it would be incorrect to add the values of these two fuel reduction programs together. However, these state level values reflect only what state residents would pay for the program, and Loomis and González-Cabán (1997) found that non-resident households often have a willingness-to-pay to prevent wildfires in ecologically important forests in other states.

4. CONCLUSIONS

This chapter demonstrated how the contingent valuation method (CVM) could be used to estimate willingness-to-pay for prescribed burning and mechanical fuels reduction programs among Caucasian and Hispanic households in California, Florida, and Montana. The simple format of the willingness-to-pay function including income, education, and a state intercept shifter may make the function suitable for benefit function transfer for calculating benefits of the two fuels reduction programs to other states.

The survey and statistical results suggest substantial willingness-to-pay of California, Florida, and Montana households for a prescribed burning or mechanical fuels reduction program that would decrease the number of acres burned by wildfires in their respective states by, at least, 25 percent. The range of California households' willingness-to-pay for the reductions in about 90,000 acres burned in the wildland urban interface where houses are at risk is \$400-\$500. These \$400-\$500 amounts are substantially larger than the \$75 per year Loomis and González-Cabán (1997) found for California household's willingness-to-pay to reduce fires in 3,000 acres of remote National Forest old-growth that was habitat for spotted owls. The relative magnitude of willingness-to-pay in the two studies are sensible, in that this current study involved a reduction of 90,000 acres, predominantly in the wildland urban interface, while the 3,000 acres was more remote public land forests without houses. Our values are also larger than Winter and Fried (2001) who used property taxes as a payment vehicle to elicit willingness-to-pay for a 50 percent reduction in risk. Their lower values of \$57 per household may in part be due to a large number of zero and protest responses to use of property taxes as the payment vehicle, and the fact this was a rural area in Michigan with relatively low house prices at risk compared to California.

The strong support in our study for the two fuel reduction programs is demonstrated by the high mean WTP for both the prescribed burning and the mechanical fuels reduction programs, and suggests these kinds of treatments to be economically feasible and efficient. However, unlike California where there is only a 10 percent difference in household's WTP for prescribed burning and mechanical fuel reduction, Florida and Montana resident's have much higher WTP for prescribed burning than for mechanical fuels reduction (by about one-third in Florida, and nearly double in Montana).

Information provided by fire managers in California indicates that prescribed burning is expensive, costing more than \$250 per acre in many locations. Although prescribed burning costs may be less in other part of the country, nonetheless, is an expensive proposition. However, when compared to the benefits estimated here attributable to prescribed burning programs, up to \$5 billion in Californian and $2-3 billion in Florida, the results indicate that many fuels reduction projects may be economically efficient as the benefits per acre are an order of magnitude greater than the costs in these two states.

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SECTION IV

DECISION MAKING IN RESPONSE TO FOREST DISTURBANCES

CHAPTER 13

ANALYZING TRADE-OFFS BETWEEN FUELS MANAGEMENT, SUPPRESSION, AND DAMAGES FROM WILDFIRE

D. Evan Mercer, Robert G. Haight, and Jeffrey P. Prestemon

1. INTRODUCTION

With expenditures to suppress wildfires in the United States increasing rapidly during the past couple of decades[1], fire managers, scientists, and policy makers have begun an intense effort to develop alternative approaches to managing wildfire. One alternative is "fuels management,"[2] which typically uses prescribed fire or mechanical methods (or both) to reduce fuel loads in dense, overstocked forests. Despite meeting strong resistance from many wildland policy makers and resource managers throughout much of the 20th century (Yoder et al. 2003), within the past decade prescribed fire has become one of the most frequently promoted approaches to reducing wildfire risk and intensity (Bell et al. 1995, Haines and Cleaves 1999, Hesseln 2000). For example, the Healthy Forests Restoration Act of 2003 called for dramatic increases in the use of fuel treatments to reduce hazardous fuel loads and the economic costs of wildfire, and one of the main objectives of the National Fire Plan (USDI/USDA 1995) is reducing fuels on 3 million acres annually. Graham et al. (2004) estimated that 100 million acres of forest lands historically burned by frequent surface fires in the western United States may benefit from surface fire restoration and 11 million acres need to be treated to protect communities (Graham et al. 2004), while Rummer et al. (2003) calculated that 66 million acres could benefit from fuels reduction. Progress has been slow, however. Obstacles include public resistance to smoke, planning and regulatory review difficulties, potential impacts on threatened and endangered species, budgetary limitations, risk of escaped fires, and lack of incentives (Stephens and Ruth 2005).

[1] Fire suppression expenditures by the USDA Forest Service rose from $160 million in 1977 to $760 million in 2005, when adjusted to 2003 dollars (Mercer et al. 2007).

[2] Fuels management is defined in the USDA Forest Service Manual as the "practice of controlling flammability and reducing resistance to control of wildland fuels through mechanical, chemical, biological or manual means, or by fire, in support of land management objectives." (USDA 1995).

T. P. Holmes et al. (eds.), The Economics of Forest Disturbances: Wildfires, Storms, and Invasive Species, 247–272.

The objectives of this chapter are to (1) characterize the overall problem of economically rational interventions into wildfire processes, (2) describe how economists and other analysts have evaluated the efficacy of fuel treatments, and (3) provide some empirical examples of how we have evaluated the trade-offs among fuel treatments, wildfire suppression, and wildfire damages. A goal of the chapter, however, is also to provide an overall characterization for the many complexities of the problem, due to its spatial and temporal dimensions and its need to account for the multiple impacts of both wildfire and proposed interventions.

2. LITERATURE REVIEW

Conventional wisdom suggests that reducing fuel loads may enhance wildfire management efficiency by reducing the resources needed for fire suppression, increasing fire fighter safety, and allowing more flexibility for suppression strategies. All too often, however, fuels management advocates promise an array of benefits yet to be validated by science. For example, fuels reduction has been promoted to restore forest structure and function, eliminate today's out of control wildfire behavior, and reduce suppression costs, acres burned, and economic and ecological damages (Finney 2003). Several recent studies, however, provide evidence that reducing fuel loads may have only a short-lived effect on wildfire spread rates (Fernandez and Botelho 2003).

The effectiveness of fuels management for reducing wildfire risk varies by ecosystem, fuel type, and weather. Although reducing fuel loads may facilitate wildfire management during most weather conditions, Graham et al. (2004) suggest that placing too much emphasis on fuels treatments for reducing the risk of catastrophic wildfire may underestimate the more important role played by weather. Under extreme weather conditions (low fuel moisture, low humidity, high winds), intense wildfires often burn through or breach most fuel treatments (Fernandez and Botelho 2003). For example, in the Southern Canadian Rockies, Graham et al. (2004) found surface fire intensity and crown fire initiation were affected more by weather rather than fuel loads. Crown spread, however, was slightly more dependent on fuels.

Piñol et al. (2005) examined the effectiveness of fuels treatments using simulation models and fire history data from Tarragona, Spain and Coimbra, Portugal. They found that the total amount and proportion of large fires decreased with increasing prescribed fire while the total area burned was not affected by fire suppression or prescribed fire. Suppression slightly enhanced dominance of large fires and prescribed fire reduced the importance of large fires. Finney (2003) concludes that changes in fire behavior associated with reduced fuel loads may enhance the effectiveness of fire suppression tactics, but it is impossible for fuel treatments alone to stop fires from burning or spreading. A more realistic objective for fuel treatments may be to reduce the risk of crown fires that tend to produce higher economic and ecological damages (Graham et al. 2004).

Research on the operational effectiveness of fuels management has been primarily based on anecdotal case studies, most of which only report on areas recently prescribed burned (i.e., within 4 years prior to the wildfire). However, since wildfires are produced from a combination of several random events (e.g., weather, ignition sources, ecological conditions) the usefulness of conclusions drawn from even the best of case studies is limited and needs to be validated with statistical analyses across a variety of spatial and temporal scales (Fernandez and Botelho 2003).

We know of only two studies (Prestemon et al. 2002, Mercer et al. 2007) that have rigorously subjected time-series data to statistical analyses of the impact of fuels management on wildfire risk. Both studies used data on wildfires and prescribed fire in Florida from 1994-1999 (Prestemon et al. 2002) and 1994-2001 (Mercer et al. 2007) and reported similar results. Mercer et al.'s more recent analysis showed that prescribed burning reduces wildfire risk for at least three years. Averaged over three years, each percentage increase in prescribed burned area in a county reduced wildfire area by 0.27 percent. In the short run (0-2 years), a 1 percent increase in prescribed burning acreage reduced the areal extent of wildfire by 0.65 percent and, when acres burned were weighted by fire intensity, by 0.71 percent (Mercer et al. 2007).

Scant research addresses the economic success of fuels management programs (Hesseln 2000). The focus of most economics research on fuels management has been on estimating per acre costs of prescribed burning or identifying factors that affect those costs (González-Cabán et al. 2004, González-Cabán and McKetta 1986, Rideout and Omi 1995). Following an in-depth review of the economics literature on prescribed burning, Hesseln (2000) concluded that existing economic research and methodology is insufficient for implementing cost-effective fire management programs based on sound economic principles. Two of the most important unanswered economic questions are whether the resources expended to reduce wildfire risk result in net economic gains and how to quantify the tradeoffs between increasing expenditures on suppression and fuels management.

Although some previous analyses have found that fuel treatments may produce positive short-term net benefits, most of the studies were site-specific (González-Cabán and McKetta 1986). Little work has evaluated whether this holds for larger geographic areas (e.g., a county) and over longer time frames (greater than two years); for example, how prescribed burning in a landscape affect subsequent wildfire patterns across the landscape (Prestemon et al. 2002). We need comprehensive risk research that focuses on stochastic processes, investment-return relationships, and changes in wildfire risk as a result of fire management activities (Hesseln 2000). This requires evaluating the effects of management activities on physical and financial outcomes over time.

Previous research, however, has tended to ignore the dynamic and spatial aspects of wildfire. Although Donoghue and Main (1985) evaluated wildfire on a broad scale, they did not consider the dynamic effects of presuppression activities that extend beyond the current time period or the immediate location of the activities. Since wildfires affect fuel levels by consuming and fragmenting flammable vegetation, the effects of wildfire and fuels management are expected to operate across a range of scales of space and time (Prestemon et al. 2002).

Although an increasing body of evidence supports the efficacy of using prescribed fire and other fuels management methods to reduce the extent and especially the intensity of wildfires (Brose and Wade 2002, Butry 2006, Davis and Cooper 1963, Hesseln 2000, Koehler 1992-93, Martin 1988, Stephens 1997, Wagle and Eakle 1979), economic analyses of the effectiveness of fuel treatment programs and of the tradeoffs between fuel treatments, wildfire suppression efforts, and economic impacts are rare (Kline 2004). The absence of trade-off analyses between fuels treatments and wildfire suppression has been attributed to problems specifying production functions for fuel treatments (Prestemon et al. 2002), lack of knowledge of the rates of technical substitution between treatment alternatives, and lack of fuel treatment data, which typically have not been collected or reported in formats that allow analysis of relative returns to treatments (Omi 2004).

One recent exception to the lack of analyses of economic efficacy of fuels management is a study by Butry (2006). Butry uses propensity score techniques to identify the individual effects of suppression and prescribed fire on wildfire activity in Florida. The analysis shows that a reduction in the suppression response time of firefighters to a reported wildfire has a large, negative impact on the resulting intensity-weighted acres burned—with an elasticity of about 0.40—implying that a 1 percent reduction in response time yields a 0.40 percent reduction in intensity-weighted acres burned. Similarly, prescribed fire in a section[3] and its neighboring sections has a significant negative impact on observed intensity-weighted acres burned, although the current-year elasticity, generally no larger than −0.05, is smaller than the long-run effect identified by Mercer et al. (2007). Nevertheless, Butry (2006) found that the benefit-cost ratio of damages averted per dollar spent in prescribed fire is about 1.5. Because little is known about the cost of reducing suppression response times, a similar ratio could not be found for wildfire suppression.

Next, we present two case studies for applying economic models to analyze the tradeoffs involved in fuels management for both strategic and tactical management applications. The first case study develops a dynamic stochastic programming and Monte Carlo simulation model to evaluate the tradeoffs between fuels management (prescribed fire) and resulting economic damages from wildfires. This approach is directed at strategic decision-making for wildfire management: how to allocate fuels management resources across regions in a way that maximizes societal welfare in the long-run. The second case study uses operations research methods (linear-integer optimization) to examine the tradeoffs between

[3] A "section" is a geographic area in U.S. land surveying. Sections are one mile square, containing 640 acres (2.6 km²). Thirty-six sections make up a survey township on a rectangular grid.

investments in fuels management and wildfire suppression resource deployment within a fire planning unit. The second analysis is based on a tactical decision model, and includes assumptions about how fuel treatments affect the ability of initial attack resources to contain fire ignitions. Scaling up the analysis through a set of assumptions about landscapes and costs and losses associated with fires could permit a strategic analysis to identify societal net benefit of spending on both fuels management and initial attack resource deployment.

The first case study shows, for one county in Florida, that prescribed fire does pay off for society, in terms of damages averted compared to the costs of prescribed fire. The second study shows that there is a trade-off between investing in initial attack resource deployment and fuels management and that some combination of the two should yield a globally optimal outcome.

3. CASE STUDIES

3.1. A Stochastic Programming Simulation of Fuel Treatment Effects on Wildfire in Florida[4]

Government agencies commonly intervene in wildfire processes through prescribed burning and other types of fuel treatments. In Florida, managers conduct and encourage landowners to reduce the risks of catastrophic wildfires through prescribed fire. Little is known about the overall efficacy of prescribed burning in reducing catastrophic wildfire damages, often because data are lacking and because wildfire processes are inherently spatial and intertemporal and proposed interventions have similar dimensional complexities. Because of this lack of information, decision makers find it difficult to evaluate how large scale programs of prescribed fire may result in net public benefits. Recently, several studies have quantified the net effects of both wildfire (Butry et al. 2001) and prescribed fire and other factors on wildfire in Florida (Prestemon et al. 2002, Butry 2006, Mercer et al. 2007). The following analysis summarizes the research of Mercer et al. (2007), who investigated how prescribed fire may affect wildfire activity and net economic benefits over the long run in Florida.

In general, determining the publicly optimal amount of prescribed burning requires solving a stochastic dynamic optimization problem. Therefore, to find the optimal levels of prescribed fire (or other vegetation management) inputs for wildfire risk reduction, we maximize the sum of expected current and future net present value of welfare[5]:

$$\begin{aligned}&\max_{x_t} A = E\left\{VW_t - \mathbf{v(x)'x_t} + \sum_{m=t+1}^{T} e^{-ri}(VW_i - \mathbf{v(x)'x_m}\right\},\\ &\text{subject to} \quad W_t = W(\mathbf{Z_t}, \mathbf{W_{t\text{-}j}}, \mathbf{x_{t\text{-}k}}) + \varepsilon_t, \mathbf{x_t} \geq 0(\forall t)\end{aligned} \tag{13.1}$$

[4] This section is derived from Mercer et al. (2007).

[5] This is a type of cost plus net value change model discussed in Chapter 16.

where A is the maximization criterion (a welfare measure), V is the net value change per unit area of wildfire, W_t is area (acres) burned by wildfire[6] in year t for the spatial unit of observation, $\mathbf{v}$ is a vector of the costs per unit area of suppression, pre-suppression, and vegetation management inputs[7], $\mathbf{x} = (\mathbf{x_t},\mathbf{x_{t+1}},\ldots,\mathbf{x_T})$ is a vector of the amount of suppression, pre-suppression, and vegetation management inputs for year t through T (the planning horizon), $\mathbf{x_{t\text{-}k}}$ is a vector of k lags of prescribed burn area, $\mathbf{Z}_t$ are exogenous inputs to wildfire production including stochastic climate variables, $\mathbf{W}_{t\text{-}j}$ is a vector of j lags of wildfire area, and r is the discount rate. Solving this optimization problem produces a Tx1 vector of optimal input quantities, **x,** and a Tx1 vector of wildfire quantities, W_t, over time. The uncertainty associated with random events (errors in prediction of weather, for example) means that $W(\cdot)$, is known only with error, complicating the solution process. In the presence of such error, simulation techniques may be used to identify, for example, the amounts of prescribed burning most likely to maximize the welfare criterion. Hadar and Russell (1969) describe how to evaluate these types of uncertain prospects.

Optimization models like equation (13.1) may involve as many choice variables as periods in the simulation[8], making them difficult to solve. Alternatively, the problem can be simplified to identifying the single optimal (stationary) policy from the set of possible policies that yields the highest expected net welfare benefits and which is consistent with any utility function that demonstrates non-increasing marginal utility.

3.1.1 The simulation model

Identifying the long-run expected impact of prescribed fire requires accounting for variable weather and the uncertainties associated with the "true" form of equation (13.1). While equation (13.1) was estimated using historical data on fire output and wildfire production inputs, observed wildfire output always differs from that predicted by an empirical model because of the random nature of the phenomenon and the imprecision of model specification. To identify the "best" level of prescribed fire to apply in a fire-prone landscape, Mercer et al. (2007) first estimated two versions of equation (13.1)—one expressing wildfire output in area burned and one in intensity-weighted area burned (tables 13.1 and 13.2).

[6] W_t could, alternatively, be expressed as a quantity measure of resources "saved" by applying resource inputs. In that case, V would be a positive number, reflecting positive values. As currently expressed in (1), V would be a negative value per unit, measuring damages per unit of wildfire realized.

[7] The "price" to the economy would be the net welfare change arising from the diversion of resources to vegetation management and away from other economically productive activities in the economy; in other words, this is the opportunity cost of foregone uses of these resources in the economy.

[8] The number of periods could be specified as infinite. Discounting would, of course, place a practical limit on the number of periods that need to be considered.

Research has shown that wildfire intensity is closely related to the resulting damages to forests (Kennard 2004). So, measuring how prescribed fire affects the intensity of wildfire output should provide a more accurate prediction of the impacts of prescribed fire on wildfire damages.

Next, the results from the empirical estimates of equation (13.1) were used to forecast the expected damages from wildfire under different prescribed fire scenarios for Volusia County, which is representative of the fire-prone landscape of Florida. Forecasts of annual wildfire activity were made for 100 years into the future. The 100-year realization of wildfire output was done by (1) selecting a fixed level of prescribed fire to apply every year; (2) randomly selecting the values of two climate variables found to influence wildfire in Florida (a measure of El Niño and a measure of the North Atlantic Oscillation); (3) randomly selecting a forecast error for wildfire area burned and wildfire intensity-weighted area burned from the historical distribution of weather factors and from prediction errors; and then (4) calculating the total annual expected wildfire damages and suppression costs and the annual cost of applying the fixed amount of prescribed fire to the county. In the final step, we varied the amount of prescribed fire chosen in step 1 and then repeated steps 2-4. This process was continued, starting from 5,000 acres prescribed burned per year, up to about 100,000 acres per year (out of 313,000 acres of forest in the county). After all of these simulations were completed, the total, long-run discounted cost plus losses associated with wildfire and prescribed fire were compared across all levels of prescribed fire to identify the level of prescribed fire where the costs and losses were smallest.

Data were obtained from the Florida Division of Forestry, the Florida Bureau of Economics and Business Research, the National Oceanic and Atmospheric Administration (NOAA), and federal agencies. The Florida fire data on state and private lands, 1981-2001, included daily records of the location and the features of the wildfire, sufficient information to construct a damage measure of fire intensity-weighted acres burned per year in each county. Data on wildfires on Federal lands were obtained from the U.S. Forest Service, U.S. Fish and Wildlife Service, and the U.S. Park Service. The prescribed fire data, 1994-2001, were derived from permits granted by the State of Florida for prescribed fire. The National Oceanic and Atmospheric Administration (2003a) provided data on the Niño-3 SST (sea surface temperature) anomaly, 1994-2001, a measure of the strength of El Nino (fires burn more in Florida when the Niño-3 SST anomaly is negative). NOAA (2003b) also provided the values of the North Atlantic Oscillation, 1994-2001, another ocean temperature measure linked to wildfire in Florida. The U.S. Forest Service provided information on the amount of forest in each county. The Florida Bureau of Economics and Business Research (2002) provided information on housing counts in each county, our instrument for measuring the impact of available wildfire suppression resources.[9]

[9] We assume that counties with more housing units will have larger fire departments than counties with fewer housing units.

The wildfire intensity-weighted risk variable was calculated from observations of the average flame length for each fire. We summed (for each county) the acres of fire for each flame length category and calculated the fireline intensity with Byram's (1959) equation, $FI = 259.833(L)^{2.174}$, where FI is fireline intensity (kW/m) and L is flame length in meters. The annual intensity-weighted risk was derived by summing for each county the product of the annual acres burned in each intensity class times the average intensity for that class divided by the county's total forest area.

Two county fixed-effects time series models[10] were estimated: (1) intensity-weighted area burned and (2) area burned. The dependent variables for the two models were: (1) intensity-weighted acres per acre of forest area in the county in the year and (2) total wildfire area burned per acre of forest area in the county.

The calculations of losses associated with wildfire were based on the 1998 wildfires (Butry et al. 2001). Two versions of losses were generated. One version assembled timber and housing losses and suppression expenditures in terms of market values—prices times quantities. The second version assembled losses in terms of social welfare—consumer plus producer surplus changes. Due to data limitations, suppression expenditures were not included in the social welfare analysis.

3.1.2 Results

The original statistical models, relating fire area burned and fire intensity-weighted area burned, show that prescribed burning at the county level has a large, statistically significant effect on both intensity-weighted area burned and on area burned in the county (tables 13.1 and 13.2). The elasticity of intensity-weighted area burned with respect to prescribed fire was −0.9 in the short-run (0 to 2 years) and −0.31 in the long-run (greater than 2 years). The elasticity of wildfire area burned with respect to prescribed fire was −0.72 in the short-run and −0.28 in the long-run.

We also estimated a model describing the supply of prescribed fire services[11] and found that prescribed fire services had a long-run elasticity of about 0.54. This indicates that the cost of prescribed fire per acre would increase twice as fast as the increase in the areal extent of prescribed fire. This extra cost associated with higher levels of prescribed fire was included in the cost plus loss simulations.

The simulations showed that the optimal levels of prescribed fire depend on whether wildfire is measured in area burned or in intensity-weighted acres. Figure 13.1 shows the impact of prescribed fire on both wildfire intensity-weighted acres

[10] A "fixed effects" time series regression model assumes that differences across units (counties in our case) can be captured in the constant term.

[11] Prescribed fire services refer to the human and capital inputs required for peforming prescribed burns.

Table 13.1. Model Parameter Estimates of Fully Specified and Parsimonious Forms of Intensity-Weighted Risk Functions (Source: Mercer et al. 2007).

Explanatory Variables	Full Model		Parsimonious Model	
	Parameter	Z value	Parameter	Z value
ln (*Prescribed Burn Area/ Forest Area*)	0.323***	-2.51	-0.388***	-3.29
ln (*Prescribed Burn Area$_{t-1}$/ Forest Area*)	-0.161	-0.096	----	----
ln (*Prescribed Burn Area$_{t-2}$/ Forest Area*)	-0.395***	-2.44	-0.513***	-3.13
ln (*Wildfire Area$_{t-1}$/Forest Area*)	-0.333***	-4.19	-0.314***	-4.64
ln (*Wildfire Area$_{t-2}$/Forest Area*)	-0.276***	-3.50	-0.308***	-4.53
ln (*Wildfire Area$_{t-3}$/Forest Area*)	-0.217***	-2.56	-0.292***	-3.95
ln (*Wildfire Area$_{t-4}$/Forest Area*)	-0.302***	-3.11	-0.318***	-3.95
ln (*Wildfire Area$_{t-5}$/Forest Area*)	-0.152*	-1.56	-0.171**	-2.05
ln (*Wildfire Area$_{t-6}$/Forest Area*)	-0.266***	-2.92	-0.309***	-4.11
ln (*Wildfire Area$_{t-7}$/Forest Area*)	0.816	0.84	----	----
ln (*Wildfire Area$_{t-8}$/Forest Area*)	0.174*	1.67	----	----
ln (*Wildfire Area$_{t-9}$/Forest Area*)	-0.081	-0.84	----	----
ln (*Wildfire Area$_{t-10}$/Forest Area*)	-0.239***	-2.70	-0.191***	-2.62
ln (*Wildfire Area$_{t-11}$/Forest Area*)	0.004	0.04	----	----
ln (*Wildfire Area$_{t-12}$/Forest Area*)	-0.001	-0.01	----	----
ln (*Pulpwood Harvest$_{t-1}$/ Forest Area*)	0.483**	1.81	----	----
ln (*Pulpwood Harvest$_{t-2}$/ Forest Area*)	0.075	0.27	----	----
ln (*Pulpwood Harvest$_{t-3}$/ Forest Area*)	-0.813***	-3.25	-0.932***	-5.65
ln (*Housing Density/Forest Area*)	-0.342	-0.17	----	----
ENSO	-0.633***	-3.20	-0.703***	-4.99
NAO	1.700***	4.47	1.256***	3.88
1998 Dummy	4.291***	10.10	3.986***	12.06
Number of Cross Sections	48		48	
Number of Years	7		7	
Total panel observations	275		285	
Wald Chi^2	2,681***		1,673***	
Log Likelihood	-334.2644		-382.1589	

Notes: * indicates statistical significance at 10%, ** at 5%, and *** at 1%. The dependent variable is the ratio of the log of sum of number acres burned at each intensity level times the intensity level per county per year relative to total forest area. Equation estimates reported here exclude estimates of 48 county dummies, which are available from the authors.

Table 13.2. Model Parameter Estimates of Fully Specified and Parsimonious Forms of Areal Risk Functions (Source: Mercer et al. 2007).

Explanatory Variables	Full Model		Parsimonious Model	
	Parameter	Z value	Parameter	Z value
ln (*Prescribed Burn Area/ Forest Area*)	-0.262***	-3.17	-0.284***	-3.60
ln (*Prescribed Burn Area*$_{t-1}$*/ Forest Area*)	-0.051	-0.46	----	----
ln (*Prescribed Burn Area*$_{t-2}$*/ Forest Area*)	-0.373***	-3.32	-0.432***	-3.61
ln (*Wildfire Area*$_{t-1}$*/Forest Area*)	-0.266***	-4.73	-0.209***	-4.28
ln (*Wildfire Area*$_{t-2}$*/Forest Area*)	-0.239***	-4.42	-0.229***	-4.61
ln (*Wildfire Area*$_{t-3}$*/Forest Area*)	-0.186***	-3.62	-0.176***	-3.34
ln (*Wildfire Area*$_{t-4}$*/Forest Area*)	-0.238***	-3.77	-0.255***	-4.49
ln (*Wildfire Area*$_{t-5}$*/Forest Area*)	-0.193***	-3.12	-0.223***	-3.87
ln (*Wildfire Area*$_{t-6}$*/Forest Area*)	-0.160***	-2.78	-0.164***	-3.21
ln (*Wildfire Area*$_{t-7}$*/Forest Area*)	-0.013	-0.21	----	----
ln (*Wildfire Area*$_{t-8}$*/Forest Area*)	0.066	0.99	----	----
ln (*Wildfire Area*$_{t-9}$*/Forest Area*)	-0.149**	-2.25	-0.153**	-2.62
ln (*Wildfire Area*$_{t-10}$*/Forest Area*)	-0.197***	-3.19	-0.149***	-2.91
ln (*Wildfire Area*$_{t-11}$*/Forest Area*)	-0.104*	-1.61	----	----
ln (*Wildfire Area*$_{t-12}$*/Forest Area*)	-0.054	-0.93	----	----
ln (*Pulpwood Harvest*$_{t-1}$*/ Forest Area*)	0.421**	2.29	----	----
ln (*Pulpwood Harvest*$_{t-2}$*/ Forest Area*)	0.376*	1.89	----	----
ln (*Pulpwood Harvest*$_{t-3}$*/ Forest Area*)	-0.509***	-2.97	-0.470***	-3.77
ln (*Housing Density/Forest Area*)	0.834	0.59	----	----
ENSO	-0.312***	-2.51	-0.262***	-2.67
NAO	0.934***	3.81	0.906***	4.10
1998 Dummy	2.268***	8.22	2.310***	10.09
Number of Cross Sections	48		48	
Number of Years	7		7	
Total panel observations	275		285	
Wald Chi^2	2,960***		1,645***	
Log Likelihood	-228.0352		-276.6049	

Notes: * indicates statistical significance at 10%, ** at 5%, and *** at 1%. Dependent variables are natural logs of each county's annual total areal extent (acres) of wildfire (areal risk model) and the natural logs of sum of area burned (acres) at each intensity level times the intensity level per county per year. Equation estimates reported here exclude estimates of 48 county dummies, which are available from the authors.

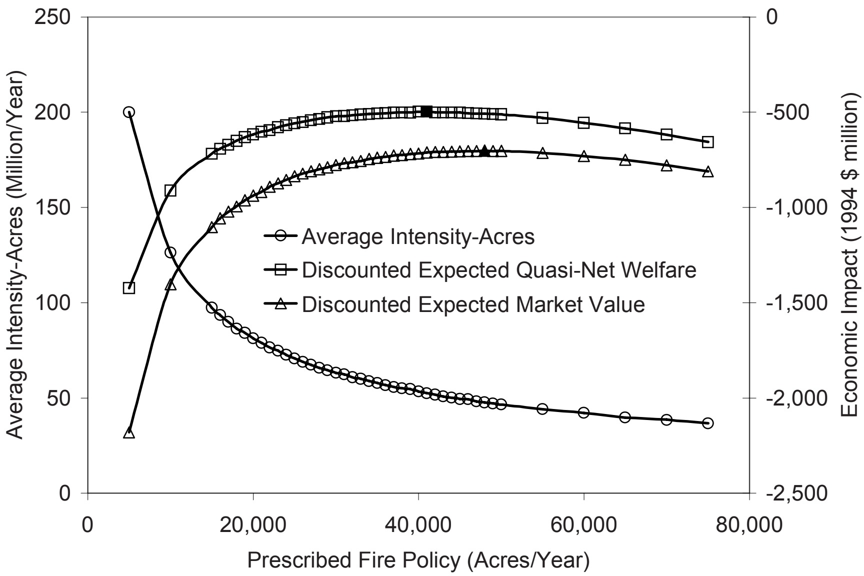

Figure 13.1. The simulated schedule of input-output combinations derived from the intensity-weighted risk model; amounts of prescribed burning yielding the maximum of net value change minus cost (symbols shaded black) are 41,000 acres/year for the quasi-net welfare analysis and 48,000 acres/year for the market value analysis (Source: Mercer et al. 2007)

and on the losses and costs associated with wildfire and prescribed fire. Figure 13.2 shows the same, but in terms of area burned related losses instead of intensity-weighted area burned related losses. Figure 13.1 shows that the expected value of losses plus costs in welfare terms is minimized when prescribed fire is set at about 41,000 acres per year and minimized in market value terms at 48,000 acres per year in Volusia County, Florida. Figure 13.2 shows that the prescribed fire area of 17,000 acres per year minimizes net value change plus costs in welfare terms and 19,000 acres per year in market value terms. The curves shown in Figure 13.1 are flatter than those shown in Figure 13.2 because the efficacy of prescribed fire on area burned and therefore economic damages is greater when the fire intensity is accounted for in the modeling. That is, the costs of progressively greater levels of prescribed burning increase at close to the same rate that wildfire damages decrease when intensity is accounted for, resulting in flatter curves in Figure 13.1. From 1994-2001, Volusia County treated about 13,000 acres per year with prescribed fire, or about 30 percent less than the optimal amount based on the area burned effect of prescribed fire and 70 percent less than the optimal amount based on the intensity-weighted area burned measure.

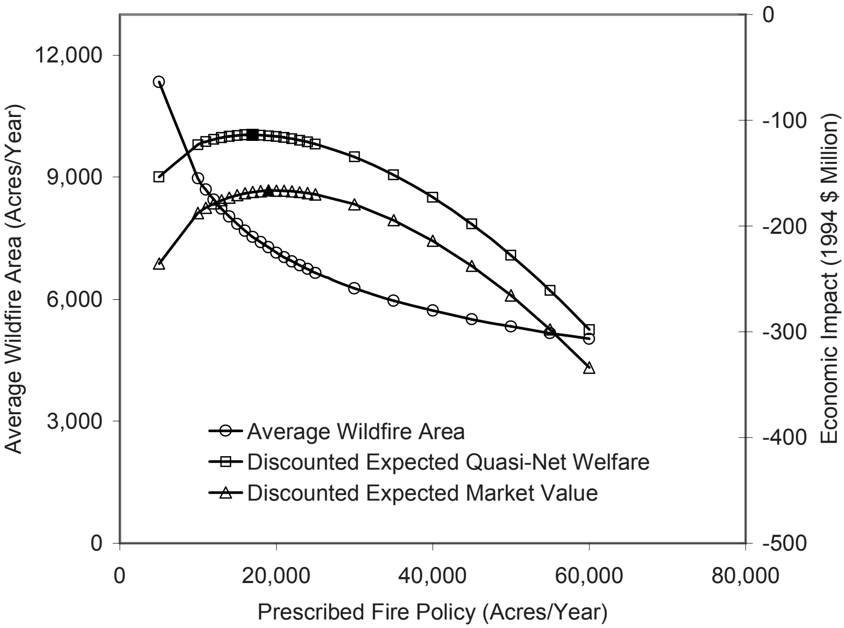

Figure 13.2. The simulated schedule of input-output combinations derived from the areal risk model; amounts of prescribed burning yielding the maximum of net value change minus cost (symbols shaded black) are 17,000 acres/year for the quasi-net welfare analysis and 19,000 acres/year for the market value analysis. (Source: Mercer et al. 2007)

3.1.3 Summary

This analysis documented that large scale programs of prescribed fire produce net economic benefits (at least in Florida). The empirical analysis of wildfire showed that the efficacy of prescribed fire appears to be greater when the effects of fuel treatments on fire intensity are accounted for. The study documented that prescribed fire levels in an already heavily treated landscape could be up to four times higher and still yield significant positive net benefits. The study also contributed to our understanding of the role of fuel treatment markets in influencing prescribed fire programs. As the amount of treatment practiced on the landscape grows, prices of prescribed fire services rise with it. Government-sponsored treatment programs on land managed by the government run the risk that they could squeeze out prescribed fire conducted on private lands. Land managers should be cognizant of these kinds of off-site impacts when making decisions about fuels management on the lands they manage.

3.2 Assessing Tradeoffs Between Fuel Treatment and Initial-Attack Investments

The following case study seeks to model the relative impacts of investments in fuel treatment and fire suppression resources. In contrast to the statistical approach taken above, the following case presents an engineering model to determine levels of investment in fuel treatment and initial attack resource deployment that minimize the expected cost of escaped wildfires. The model is based on predictions of the likelihood that a fire ignition will escape as a function of the level of fuel treatment and the number of initial attack resources that are dispatched to the fire. Results of the optimization model can be used to estimate the efficiency of fuel treatment and the tradeoffs between investments in fuel treatment and initial attack resource deployment. It provides a framework for a new kind of analysis that could be done in Florida or elsewhere, where sufficient data exist to quantify the effect of fuels management on both the cost of suppression and the losses associated with wildfire.

Fuel treatments may change wildfire behavior and enhance the effectiveness of fire suppression tactics (Finney and Cohen 2003). Deploying initial-attack resources to meet expected demands for fire suppression in the coming days, weeks, or months is an important part of wildland fire planning (Martell 1982). Deployment decisions have been incorporated in optimization models that minimize operating costs while meeting pre-defined demands for initial attack (Hodgson and Newstead 1978, MacLellan and Martell 1996) or minimize area burned or number of escapes subject to budget constraints that limit the size of the initial-attack force (Kirsch and Rideout 2005, Haight and Fried 2007). These latter models include relationships between fire behavior and fire suppression. If those relationships could be extended to include the impacts of fuel treatment, then optimization models could be used to analyze the cost-effectiveness of fuel treatment and suppression.

To demonstrate this potential, we modified the standard-response model of Haight and Fried (2007) to include the effects of fuel treatment. Their model determines where to deploy a fixed number of initial-attack resources to minimize the expected number of fires that do not receive a standard response, defined as the number of resources that must reach the fire within a maximum response time (Marianov and ReVelle 1991). The idea is that if a fire receives the standard response, the likelihood of escape is low. We modified the model to minimize the expected cost of escapes with assumptions about how fuel treatments and the number of resources dispatched affect the probability of escape. We demonstrate how the model can be used to construct cost curves for the relationship between initial attack resources in position to respond and expected cost of escapes, with and without fuel treatments. The cost curves can be used to estimate the cost savings associated with fuel treatment, in terms of reduction in expected cost of escaped fires under a given level of initial attack force.

3.2.1 A risk-of-escape model for initial attack

The optimization model is a linear-integer formulation with two objective functions: the cost of deploying initial-attack resources and the expected cost of fires that escape initial attack. A weighted sum of the objective functions is minimized, and the weight is ramped from large to small to generate a tradeoff curve showing how different levels of investment in initial attack resources affect subsequent costs of suppressing escaped fires. The model is for a single fire planning unit. The data include the locations of fire stations and representative fires. Each station has a capacity to house initial attack resources, and the time required for resources to reach each representative fire location is known. The data also include fire scenarios, each representing a set of fire locations during a single day. The model includes integer decision variables for the number of suppression resources deployed to each station and the number of resources dispatched from each station to each fire in each scenario. The probability of escape decreases with the number of resources that are dispatched to the fire within a maximum response time. Therefore, each fire is characterized by a set of parameters representing escape risk reduction for increasing numbers of resources dispatched for initial attack. In our application, the values of parameters of the risk-reduction function depend on the level of fuel treatment. The model is formulated with the following notation:

Indices:

i, I = index and set of fire stations,
j, J = index and set of potential fire locations,
k, K = index and set of suppression resource dispatch classes,
s, S = index and set of fire scenarios,

Objective functions:

Q_1 = cost of deploying suppression resources,
Q_2 = expected cost of escaped fires,

Parameters:

λ = objective weight; $0 \leq \lambda \leq 1$,
a_{jk} = escape risk reduction parameter (≤ 0) for fire location j dispatch class k,
b_i = upper bound on number of resources deployed at station i,
c^1_i = fixed cost of opening station i,
c^2_i = cost of deploying a resource at station i,
c^3_j = cost of containing an escaped fire at location j,
f_{js} = 0-1 parameter; 1 if fire occurs in location j scenario s; 0 otherwise,
p_s = probability that scenario s occurs,
t_{ij} = response time from station i to location j,
T = maximum response time,
N_j = set of stations from which resources can reach location j within the maximum response time; i.e., $N_j = \{i \mid t_{ij} < T\}$.

Variables:

v_{js} = probability of escape for fire in location j scenario s,

w_i = 0-1 variable; 1 if station i is open, 0 otherwise,

x_i = number of resources deployed at station i,

y_{ijs} = number of resources at station i dispatched to location j during scenario s,

z_{jks} = 0-1 variable; 1 if dispatch class k is used at fire location j scenario s, 0 otherwise.

The model is formulated as follows:

$$\text{Minimize}: \lambda Q_1 + (1-\lambda)Q_2 \tag{13.2}$$

subject to :

$$Q_1 = \sum_{i\in I} c_i^1 w_i + c_i^2 x_i \tag{13.3}$$

$$Q_2 = \sum_{s\in S} p_s \sum_{j\in J} f_{js} v_{js} c_j^3 \tag{13.4}$$

$$x_i \le b_i w_i \qquad \text{for all } i \in I \tag{13.5}$$

$$\sum_{j\in J} y_{ijs} \le x_i \qquad \text{for all } i \in I \text{ and } s \in S \tag{13.6}$$

$$\sum_{k\in K} z_{jks} = \sum_{i\in N_j} y_{ijs} \qquad \text{for all } j \in J \text{ and } s \in S \tag{13.7}$$

$$v_{js} = 1 + \sum_{k\in K} a_{jk} z_{jks} \qquad \text{for all } j \in J \text{ and } s \in S \tag{13.8}$$

The objective (equation 13.2) is to minimize the weighted sum of the two objective functions: the cost of deploying initial-attack resources to stations prior to the occurrence of fires (equation 13.3) and the expected cost of fires that escape initial attack (equation 13.4). The weight λ represents the decision maker's preference for the two objectives. When λ is closer to one, more weight is put on minimizing the cost of deploying initial attack resources. When λ is closer to zero, more weight is put on minimizing the expected cost of escaped fires. In equation (13.4), the inside summation is the expected cost of escapes during scenario s, where each product includes three terms: f_{js} is a 0-1 parameter for whether or not a fire occurs at location j, v_{js} is the probability of escape at location j, and c^3_j is the cost of containing an escape. In the outside summation of equation (13.4), each expectation is weighted by p_s, the probability of scenario occurrence. Equation (13.5) defines the capacity of each station, which is greater than zero only if the station is open. Equation (13.6) requires that the number of resources dispatched from each station does not exceed the number of resources deployed to the station.

Equations (13.7) and (13.8) calculate v_{js}, the probability of escape of a fire at location j, and require a bit of explanation. We assume that escape probability equals one when no resources are dispatched and approaches zero as the number of resources dispatched increases. The probability of escape is modeled as a decreasing, convex, piecewise-linear function of the number of resources dispatched so that the slope is negative and closer to zero with each additional resource dispatched. The 0-1 variables z_{jks}, $k = 1,\ldots,K$, represent resource dispatch classes where $z_{jks} = 1$ means at least k resources have been dispatched. As a result, the sum of these 0-1 variables must equal the number of resources dispatched to the fire from stations within the required response time (equation 13.7). The parameter a_{jk} is the slope of the function for probability of escape and represents the escape risk reduction ($a_{jk} \leq 0$) for dispatch class k. Because the function is convex, $a_{j1} \leq a_{j2} \leq \ldots \leq a_{jK}$. If the model dispatches any resources to fire j, the model will choose the dispatch variables z_{jks} with the most negative risk reduction parameters first to minimize probability of escape. As a result, for any k such that $z_{jks} = 1$, $z_{jts} = 1$ for all $t < k$.

It is important to recognize that the decision variables of the model take place in different time periods. Resource deployment decisions take place in the first period to meet possible resource demands in the coming days. Dispatching decisions take place in the second period once the locations of fires are known. The dispatching decisions assume that fires in a single day occur close enough in time to compete for the same resources.

The model's objectives and data requirements differ from other optimization models for initial-attack resource deployment and dispatching. Kirsch and Rideout's (2005) model has an objective of minimizing area burned and includes binary containment variables for fires based on the ratio of fire line to fire perimeter in discrete time intervals (e.g., hours) after ignition. With an objective of minimizing area burned, the model dispatches resources to contain fires as soon as possible within a budget constraint. Further, the Kirsch and Rideout model requires rates of fire line production and fire area and perimeter growth. In contrast, our model has an objective of minimizing the expected cost of escapes. As a result, a single variable representing escape risk, v_{js}, is defined for each fire along with parameters, a_{jk}, representing the reduction in escape risk per unit increase in resources dispatched to the fire. The escape risk reduction parameters are proxies for fire line production and spread rates.

The probability-of-escape model for initial attack does not explicitly include fuel treatment. In practice, fuel treatment may reduce the risk of escape by reducing fire intensity. In our model, the risk-reduction parameters, a_{jk}, will be greater in locations with fuel treatment. Making a_{jk} depend on a fuel treatment variable in equation (13.8) would create a nonlinear equation because a_{jk} is already multiplied by a variable z_{jks} representing the number of resources dispatched to the fire. To maintain linearity, we solved the probability-of-escape model for various assumptions about fuel treatment to investigate the tradeoffs between investments in fuel treatment, initial attack resources, and cost of containing escaped fires.

3.2.2 Application

The model was applied to a hypothetical problem involving a 10 X 10 grid of forest districts, each covering 6170 acres (25 km^2) and belonging to one of three fire risk classes based on daily ignition probability (fig. 13.3). Classes 1, 2, and 3 had ignition probabilities of 0.10, 0.06, and 0.02, respectively. The analysis focused on deploying fire engines in 10 stations. Each station had a capacity of 4 engines, and each engine costs $10,000 to base. Assuming that each engine traveled 31 miles/hour (50 km/hour) and the distance separating each district was a straight line between district midpoints, we calculated the time required to travel between each station and each district. Assuming a maximum response time of

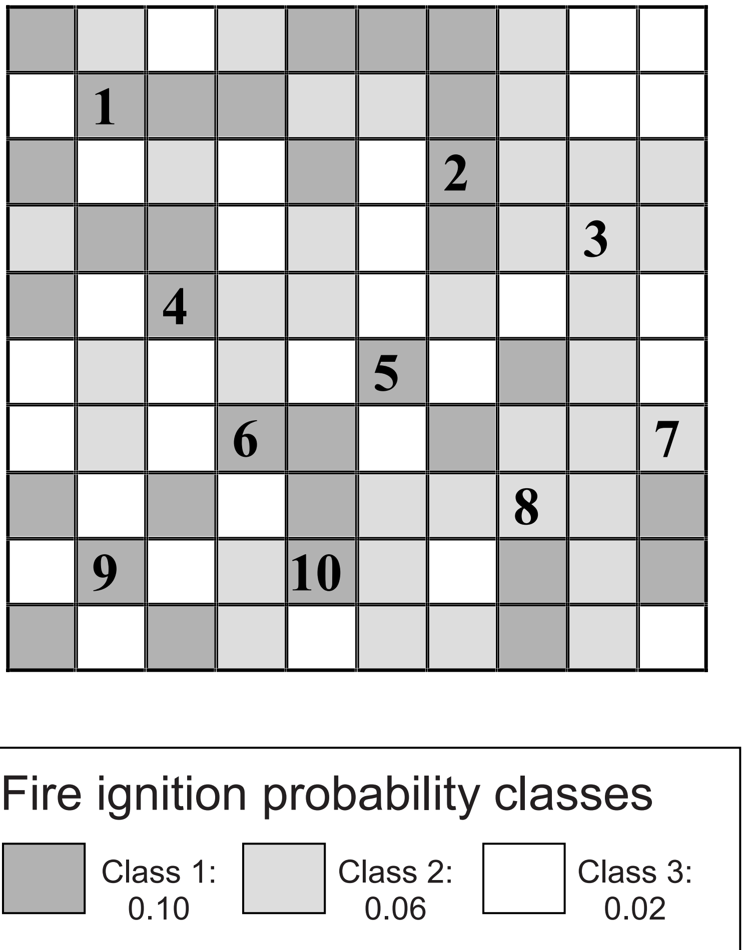

Figure 13.3. Fire districts (shaded according to probability of fire occurrence) and stations (represented by numbers) in a hypothetical planning unit.

20 minutes, we constructed a set of stations that were within 20 minutes of each potential fire location. We selected this response time threshold because fast-spreading fires tend to escape initial attack if firefighting is not well-underway within 20 minutes following a fire report.

We formulated the optimization model to determine the engine locations for days during the "high" fire season when multiple fires occur. We focused on days with multiple fires because draw-down of suppression resources on such days increases the likelihood that fires escape initial attack. We used the daily fire probabilities to construct 100 fire scenarios representing days with multiple fires. Each scenario is a list of districts in which fires occur. Each scenario, f_{js} j = 1,…,100, is a vector of 0-1 parameters where parameter $f_{js} = 1$ means that a single fire ignites in district j under scenario s. The value of each parameter f_{js} was determined by comparing a uniform 0-1 random number with the probability of ignition in district j. Because ignitions were determined randomly, each scenario had the same probability of occurrence, $p_s = 0.01$. Mean daily number of fires per scenario was 6.04 with range 2-14.

The probability of fire escape was modeled as a decreasing, piecewise-linear function of the number of engines dispatched to the fire within the 20-minute response time (fig. 13.4). We assumed that a standard response was four engines

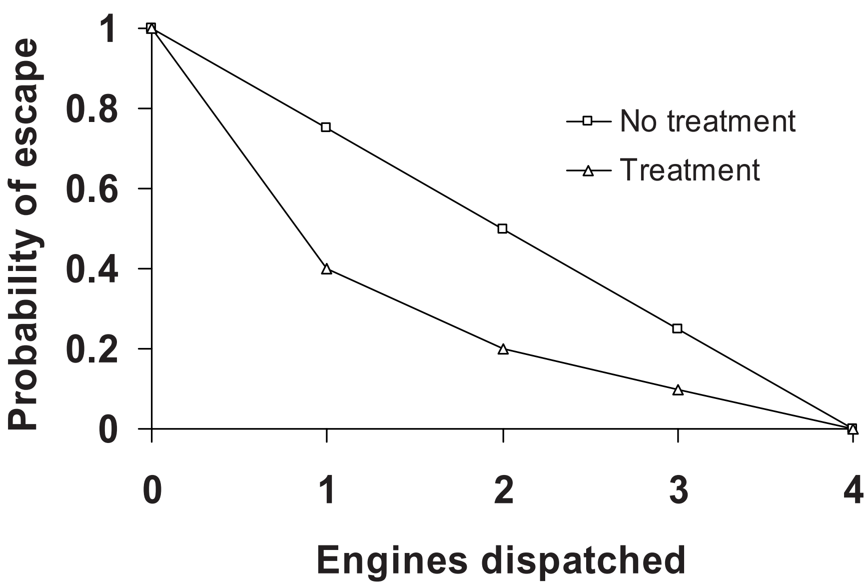

Figure 13.4. Probability of fire escape as a function of the number of engines reaching the fire within the standard response time (20 minutes) in areas with and without prior fuel treatment.

reaching the fire within 20 minutes and the probability of escape associated with the standard response was zero. The shape of the relationship between escape risk and response depended on fuel treatment. Without fuel treatment, the relationship was linear with a constant risk reduction parameter of 0.25. With fuel treatment, the relationship was piecewise linear with risk reduction parameters that decreased as the number of engines responding increased (0.6, 0.2, 0.1, 0.1). In this case, probabilities of escape are lower for each engine response category; however, the standard response of four engines is still required to achieve zero probability of escape.

The costs of escaped fires were based on observations of emergency suppression costs of 13 large fires (>300 acres) in national forests in the southeastern United States in years 2000-2003. Six fires had containment costs less than $50,000, five had containment costs of $100,000-500,000, and two had costs greater than $1,000,000. We assigned an average cost to escaped fires in each of the three risk classes in figure 13.3. Costs of escaped fires in districts with ignition probabilities of 0.10, 0.06, and 0.02 were $50,000, $100,000, and $500,000, respectively.

Our analysis focused on the trade-off between the cost of deploying initial-attack engines and the expected cost of fires that escape initial attack. We computed optimal engine locations for problems in which the objective function weight λ was decreased from 1.0 (minimize cost of deploying engines) to 0.0 (minimize expected cost of escaped fires) in increments of 0.02 subject to a capacity constraint of 4 engines per station. The baseline analysis was conducted assuming no fuel treatment. Then, trade-off curves were constructed with fuel treatment performed in districts belonging to each of the three fire risk classes.

The spatial optimization problems were solved on a Dell Pentium 4 laptop computer (CPU 2.4 GHz) with the integrated solution package GAMS/Cplex 9.0 (GAMS Development Corporation 1990), which is designed for large and complex linear and mixed-integer programming problems. Input files were created in GAMS (General Algebraic Modeling System), a program designed to generate data files in a format that standard optimization packages can read and process. Cplex solves a mixed-integer programming problem using a branch and cut algorithm, which solves a series of linear programming sub-problems.

3.2.3 Results

In the baseline case without fuel treatment, the curve showing the tradeoff between the cost of deploying engines and expected cost of escaped fires had a convex shape in which cost of escapes decreased at a decreasing rate as the total cost of engine deployment ($10,000 times the number of engines deployed) increased (fig. 13.5). The points on the curve represent non-dominated solutions and their relative performance with respect to the two objectives. For each non-dominated solution, improvement in one objective cannot be achieved without simultaneously causing degradation in the value of the other objective. As a result, the points represent a frontier below which there were no better solutions.

The best deployment of engines depended on the objective function weight. If minimizing the cost of basing engines is most important (i.e., $\lambda = 1$), the choice is solution A in which no engines are deployed and the expected number of escapes equals the average daily fire frequency of 6.04 with expected cost of $705,000 (fig. 13.5). As more weight is given to minimizing the cost of escapes, more engines are deployed resulting in higher engine deployment costs and lower costs from escapes. For example, with 24 engines deployed at a cost of $240,000 (solution B), the expected cost of escapes was $90,000, 13 percent of the expected cost of escaped fires with no engines deployed. Increasing the number of engines from 24 to 40 for a deployment cost of $400,000 (solution C) reduced the expected cost of escaped fires to $11,000, 2 percent of the expected cost with no engines deployed.

The slope of the tradeoff curve is a benefit/cost ratio showing the reduction in expected cost of escapes per increase in cost of engines deployed for initial attack. The slope was relatively steep between solutions A and B (< -1) indicating that benefits of deploying more engines exceeded costs. Between solutions B and C, the slope was relatively flat (> -1) indicating that deploying more engines was not cost-effective in terms of reducing the expected cost of escapes. The slope of the tradeoff curve was -1 at solution B, which minimizes the sum of the costs of engine deployment and escapes.

When fuel treatment was applied in risk classes 1 and 2, the curves showing the tradeoff between cost of engines deployed and expected cost of escapes were

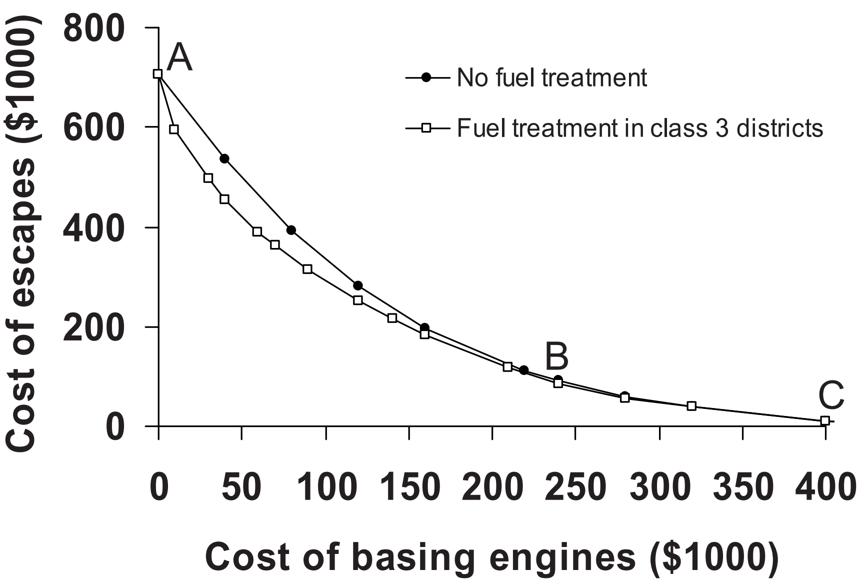

Figure 13.5. Tradeoffs between the cost of deploying initial-attack engines ($10,000 per engine) and the expected cost of fires that escape initial attack.

slightly lower than the baseline tradeoff curve computed without fuel treatment. Even though fuel treatment lowered the probability of escape, the relatively low costs of escapes ($50,000 and $100,000 per fire) in these risk classes made the economic impacts of fuel treatment relatively small.

When fuel treatment was applied in each district in risk class 3, which had a relatively high cost of escape ($500,000 per fire), the curve showing the tradeoff between cost of engines deployed and expected cost of escapes was significantly lower than the baseline curve computed without fuel treatment, especially when 1-15 engines were deployed at a cost of up to $150,000 (fig. 13.5). Fires in areas with fuel treatment have lower probabilities of escape than fires in areas without fuel treatment, especially when one or two engines are dispatched for initial attack (fig. 13.4). As a result, the greatest gain from fuel treatment in terms of reducing the expected cost of escape occurs when there are relatively small numbers of engines available for initial attack.

The tradeoff curves in fig. 13.5 can be used to evaluate the economic effectiveness of fuel treatment in districts in risk class 3. The vertical distance between points on the curves represents the reduction in expected cost of escapes resulting from fuel treatment while maintaining a given engine force. With more than 15 engines deployed at a cost of > $150,000, fuel treatment produced very little reduction in expected cost of escape because there were enough engines deployed to dispatch 3-4 engines to most fires, which resulted in a relatively low probability of escape regardless of fuel treatment. With fewer than 15 engines deployed, fuel treatment had a bigger effect. For example, with 12 engines deployed at a cost of $120,000, applying fuel treatment to risk class 3 resulted in a $32,000 reduction in expected cost of escapes. This cost saving can be compared with fuel treatment cost to determine whether this particular fuel treatment activity is cost effective. In this example, fuel treatment was applied in 31 districts (191,000 acres), and the break-even fuel treatment cost was $0.17 per acre. This is considerably lower than actual treatment costs which can be up to $250 per acre. As a result, the fuel treatment activity in this case is not cost-effective. It should be noted that this break-even analysis assumes that fuel treatment only affects the expected cost of escapes during the upcoming fire season. If the effects last more than one year, this analysis underestimates the benefits of fuel treatment.

4. CONCLUSIONS

In this chapter we have presented two case studies illustrating innovative approaches to analyzing the impacts of fuels management on wildfire outcomes and for predicting the tradeoffs between expenditures for fuels management and suppression resources. The first case study examining tradeoffs between prescribed fire treatments and damages from wildfire shows that fuels management (prescribed fire in this case) does appear to pay off, at least in Florida. At the current prescribed fire levels in Volusia County, Florida, the long-run benefit-cost ratio of prescribed fire is close to or greater than unity. This analysis, however, leaves out some additional benefits associated with prescribed fire—such as the beneficial impacts on ecosystems that depend on wildfire for their health and increased productivity of the remaining stand of timber. At the same time, the analysis omits some of the costs of prescribed fire, in terms of the risk of escape and some of the negative health impacts associated with the smoke from prescribed fires. Butry et al. (2001) showed that the asthma-related impacts of wildfire are not large, in economic terms. In contrast, Rittmaster et al. (2006), who accounted for both respiratory and cardiac-related effects, characterize the human health related losses associated with one large wildfire in Alberta, Canada, to have been substantial and far reaching spatially, with economic impacts second only to those associated with timber. But neither of these studies quantified the losses associated with the averting behavior of individuals who flee when wildfires burn near their homes. It is not clear whether individuals with respiratory problems also flee locations undergoing active prescribed fire; this is an area worthy of additional research.

The Florida case study, however, was unable to detect a significant impact of wildfire suppression on observed wildfire, because the statistical models of wildfire activity (area burned, intensity-weighted area burned) omitted a direct measure of wildfire suppression. Further research, such as that done by Butry (2006), could help to clarify those suppression impacts. Mercer et al. (2007) did not find a significant impact of housing density (a proxy for the availability of suppression resources) on observed wildfire activity; therefore, the simulation analysis simply assumed that a constant level of fire suppression is applied per unit of wildfire output, effectively assuming away any trade-off between suppression and fuels management. Butry (2006) did find that suppression could trade off for prescribed fire, but he did not attempt to quantify that trade-off in a simulation as done by Mercer et al. (2007).

The second case study examined short run tradeoffs between investments in fuels management versus increased initial-attack resources on the ground. The case study shows that decisions for basing and dispatching initial-attack resources can be formulated as a mixed-integer programming model that minimizes the cost of deploying initial-attack resources and the expected cost of suppressing fires that escape initial attack. The model is well suited to determining the tradeoffs between these objectives given uncertainties in the number and location of fires that may occur during the fire season. A key component is the relationship between the number of resources that reach a fire within a maximum response time and the probability of escape. The case study was based on a hypothetical relationship because empirical analyses of the likelihood of escape as a function of initial attack force and fuel treatment are rare. Butry (2006) identified the individual effects of suppression and prescribed fire on wildfire activity in a case study in Florida, and more work is needed to empirically model of these relationships.

Fuel treatments may increase the probability of containment of a fire during initial attack. This effect was incorporated in the model by adjusting the slope of the relationship between the number of resources dispatched to an ignition and the probability of escape. To maintain linearity of the initial-attack model, the effects of alternative levels and locations of fuel treatments were determined as one-at-a-time changes in model parameters. Analysis of these changes allows determination of the cost-effectiveness of case-specific fuel treatment activities. Given the structure of the initial-attack model, determining optimal levels and locations of fuel treatment would require a non-linear formulation and heuristic rather than exact optimization methods.

The strengths of the initial-attack model include spatial detail (e.g., locations of fire stations, suppression resources, and potential fires) and practical decision criteria (e.g., minimizing the expected cost of escape). However, this detail makes it difficult to reach general conclusions about optimal levels of investment in fuel treatment and initial attack. The results will depend on case-specific model parameters, including the number and location of fire stations, probabilities of fire occurrence, and relationship between probability of escape and resources dispatched during initial attack. Nevertheless, incorporating fuel treatment into an initial-attack optimization model is a first step toward evaluating the cost-effectiveness of these two important fire preparedness activities.

In this chapter we presented two methods for examining the strategic and tactical tradeoffs between fuels management and wildfire suppression. Separately, each approach provides essential insights into the economics of wildfire management. However, to make the most effective use of these analyses requires combining the approaches so that both the tradeoffs between fuels management and suppression expenditures and the tradeoffs between fuels management, suppression and the economic damages from subsequent wildfires can be examined simultaneously. This will require a wide array of additional research in the economics of wildfire.

At the same time, the case studies highlight the complexity of the problem of wildfire management. Wildfire management can be approached from many different angles, from fire prevention, fuels management (as described in our first example), resource pre-placement (as described in our second example), wildfire suppression (our second example, as well). Wildfires occur in time and space, and wildfire occurrence is driven by both natural and human factors. Wildfire management actions have intertemporal effects across multiple spatial scales and are inherently uncertain. Therefore, simulation models are not able to account for all the ways that managers can intervene in wildfire processes and can only roughly approximate the spatial and temporal interdependencies among both wildfire and management efforts. Likewise, economic analyses are limited by a lack of understanding of the full economic effects of wildfires on society, including public health and secondary impacts on economic sectors beyond forests. The research presented in this chapter demonstrates advancements in our understanding of the problem of designing better combinations of interventions, but they should be followed by modeling that can better account for other forms of management (e.g., fire prevention, mechanical fuel treatments) and for the interactions between fuel treatment design, fire suppression, and the landscape and how actions may affect risks in both spatial and temporal dimensions.

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CHAPTER 14

A REVIEW OF STATE AND LOCAL REGULATION FOR WILDFIRE MITIGATION

Terry K. Haines, Cheryl R. Renner, and Margaret A. Reams

1. BACKGROUND

Wildfire may result from natural processes or as the result of human actions (Ffolliott 1988, Mees 1990). As a natural phenomenon, it is important in sustaining forest health in fire-dependent ecosystems. While some wildfire may be ecologically beneficial, it poses a threat to residential communities located within or adjacent to the forest. Wildfire is considered a hazard when it endangers things that people value, such as life, property and cultural values (Burton et al. 1978). Each year the challenge of protecting Wildland-Urban Interface (WUI) communities captures headlines in American newspapers, as wildfire forces the evacuation of homes.

State governments have been granted police powers to protect the health, safety and welfare of their citizens by the Constitution. With regards to land use policy, the states pass this power to local governments enabling them to adopt regulations to control situations that pose a threat to life and property. In response to wildfire-related losses in the WUI, two states and numerous county and local governments have established regulatory programs to reduce wildfire hazards in high risk areas.

2. BASIS FOR REGULATORY PROGRAMS

Case studies of past wildfire disasters have demonstrated that some homes are more vulnerable to wildfire than others. Two factors have emerged as the primary determinants of a home's ability to survive wildfire. These are the home's roofing material and the vegetative space surrounding it. An analysis of California's Bel Air fire revealed that 95 percent of homes with both non-flammable roofs and at least 10-18 meters of vegetative clearance around the home survived the wildfires (Howard et al. 1973). In the Painted Cave fire of 1990, an 86 percent survival rate of homes with non-flammable roofs and a clearance of 10 meters or more was documented (Foote and Gilless 1996). In the Spokane fire storm of 1991, over 60 percent of the homes lost had little or no defensible space. An analysis of the

T. P. Holmes et al. (eds.), The Economics of Forest Disturbances: Wildfires, Storms, and Invasive Species, 273–293.

losses showed that most of the homes had a proximity to flammable fuels of 7 meters or less. (NFPA 1992).

Additionally, results from the Structural Ignition Assessment Model (SIAM), which includes modeling, field experiments and analysis of case studies, indicate that "the home ignition zone extends to a few tens of meters around a home, not hundreds of meters or beyond. Home ignitions and thus, the Wildland-Urban Interface fire-loss problem principally depend on home ignitibility" (Cohen 2000). Findings from the case studies of past wildfires and the SIAM model demonstrate that defensible space regulations do not have to be draconian to be effective. A minimum of 30 feet of defensible space combined with fire-resistant roofs where topographic slope is minimal, dramatically reduces a property's wildfire vulnerability. Since both the roof type and the landscaping immediately around the home are choices within the control of the homeowner, homeowner cooperation is essential to the success of wildfire risk reduction programs.

If losses can be prevented by two actions on the part of the homeowner, it seems logical that the simplest way to reduce wildfire losses is to establish mandatory requirements for non–combustible roofs and a minimum of 10 meters of clearances around homes in high wildfire risk areas. However, new regulations are often difficult to pass and, in the interest of public safety, local officials attempting to influence homeowners to reduce risk around their homes must first convince homeowners of the need to protect themselves.

3. OBSTACLES TO ADOPTION OF REGULATORY PROGRAMS

Several factors may affect the feasibility of the adoption of regulations. First, the ability to obligate financial resources and dedicate personnel for the administration and enforcement of regulations can limit their practicality for many cash-strapped local governments. Second, requiring defensible space may be unpopular with residents due to the cost of removing vegetation and, in many locations, may be politically unacceptable. Former urban residents often favor the privacy and aesthetics found in an unaltered wildland environment and may underestimate their wildfire risk exposure (Bradshaw 1987, Loeher 1985). Residents may also view defensible space requirements as infringements of private property rights (Winter and Fried 2000). Support for more restrictive regulations seems to increase after a community has experienced a wildfire (Abt et al. 1990).

Even where ordinances have been adopted, a lack of public support can stymie enforcement efforts. Fire managers strive to establish a cooperative relationship with homeowners and may view enforcement of unpopular defensible space standards as counterproductive to the overall goal of community wildfire protection. As a result, it is often more expedient to offer educational programs and homeowner assistance to motivate homeowners to reduce fuels around their homes. These homeowner education programs have been shown to be effective in encouraging private property owners to take steps to reduce risk (Hodgson 1994, Rice and Davis 1991). Where ordinances have been adopted, most jurisdictions employ a comprehensive approach to wildfire risk reduction. Fire managers use a mix of regulatory, educational, and incentive or assistance programs to motivate homeowners to take responsibility for creating defensible space and their home wildfire safety (Reams et al. 2005).

4. RESEARCH METHODS

This chapter will review state laws and local ordinances for wildfire mitigation, as well as model codes or guidelines for ordinance development. Information for the chapter is from two primary sources. First, an analysis of programs identified in the USDA Forest Service's National Database of State and Local Wildfire Hazard Mitigation Programs, www.wildfireprograms.usda.gov. The website database inventories state and local wildfire mitigation programs implemented to reduce wildfire risk on private ownerships in the WUI. Regulation is one of several program types adopted by state and local jurisdictions described on the website. Other program types identified on the website include community outreach and homeowner education, regional wildfire hazard risk assessments and mapping programs, and homeowner incentives for fuels treatment and removal. The second source of information is a survey of wildfire mitigation program administrators. The survey gathered contextual information about program adoption and implementation and provided insight into the effectiveness of regulation as a tool for reducing wildfire risk.

5. MODEL WILDFIRE PROTECTION CODES

Counties and communities at risk for wildfire need not struggle with the science and legal requirements of developing effective and enforceable wildfire risk reduction ordinances. Model codes or ordinances serve as templates for potential regulations which may be adopted by a jurisdiction. Local decision makers may select all components of a model ordinance for adoption, or may choose only those elements they believe to be most appropriate for their community. Two national organizations, the International Code Council (ICC) and the National Fire Protection Association (NFPA) have developed model Wildland-Urban Interface wildfire protection codes as standards for states and local governments to adopt. In addition, fire protection agencies in three states, California, Florida, and Utah have developed model codes for adoption by local governments in their respective states. These models have found acceptance in many fire-prone communities, where they are either adopted as separate ordinances, or incorporated into the requirements of the zoning ordinance and subdivision regulations. In California, where there are statewide regulations for defensible space, communities in fire-prone areas are required to either adopt the model code which contains the state standards or one which has more stringent requirements. The model ordinances include:

1. National Fire Protection Association (NFPA) 1144: Standard for Protection of Life and Property from Wildfire, 2002;
2. International Code Council's, International Urban-Wildland Interface Code (UWIC), 2003;
3. California's Local Responsibility Area (LRA) Model Ordinance for the Defensibility of Space and Structures, 1994;
4. Florida's Model Wildfire Mitigation Ordinance, 2004; and
5. Utah's Wildland Urban Interface Standards, 2005.

These comprehensive model ordinances include standards for roofing and the use of fire resistant construction materials, water supplies for firefighting, road, bridge and driveway design, subdivision ingress and egress, vegetative management and road clearance standards. The models generally include provisions for administration, permit requirements, and enforcement.

5.1 Defensible Space

A core concept in the model codes and the resulting wildfire mitigation ordinances is that of structure protection through the creation of defensible space. Defensible space may be defined as an area either natural or manmade where material capable of causing a fire to spread has been treated, cleared, reduced, or changed to act as a barrier between an advancing wildland fire and life, property, or resources. The following excerpt from the Urban-Wildland Interface Code (2003) provides an example of the requirements for defensible space:

> "Persons owning, leasing, controlling, operating, or maintaining buildings or structures requiring defensible spaces are responsible for modifying or removing non-fire-resistive vegetation on the property owned, leased or controlled by said person.
>
> Ornamental vegetative fuels or cultivated ground cover, such as green grass, ivy, succulents or similar plants used as ground cover, are allowed to be within the designated defensible space provided they do not form a means of readily transmitting fire from the native growth to any structure.
>
> Trees are allowed within the defensible space provided the horizontal distance between crowns of adjacent trees, and crowns of trees and structures, overhead electrical facilities, or unmodified fuel is not less than 10 feet (3048mm). Deadwood and litter shall be regularly removed from trees."

Generally, fuel reduction recommendations include both vertical and horizontal separation of vegetative fuels. Vertical separation is achieved through the reduction of ladder fuels, including shrubs and vines and low hanging branches, which might carry fire from the forest floor to the crown. Horizontal separation is achieved by thinning trees to a specified spacing.

Fire protection agencies and organizations are also providing guidance for property owners through educational publications such as Firewise brochures that offer landscaping design options for defensible space. These generally include lists of recommended fire-resistive plant species for the area, as well.

5.2 Wildfire Hazard Risk Assessment Rating Guide

Each of the model ordinances is supplemented with a fire hazard rating guide that allows inspectors to evaluate the fire hazard risk of existing homes and proposed residential developments. The hazard-rating scales differ in complexity and in the weights given to the various factors. Wildfire risk factors generally include: vegetation type, fire history, density of development, building materials, ingress and egress roads, water supply for firefighting, and presence of defensible space. The models recommend that the risk assessments be updated periodically, usually at three or five year intervals.

5.3 Roofing Standards and Defensible Space Requirements

The five model ordinances all contain requirements for both defensible space and fire-resistant roofs, however the specific standards vary. The NFPA 1144 specifications are for 30 feet of defensible space and the use of fire-resistant roofing materials. The types of roofing materials required depends on the risk classification of the property, with the least flammable roof types, Class A roofs, required for high-hazard properties. The area of defensible space required by the UWIC is 30, 50 or 100 feet depending on the hazard classification of the property–moderate, high, or extreme, respectively. Roof requirements are also based on the assessed hazard designation, as well as a property's level of conformance with defensible space standards and the availability of water supplies for fire fighting.

Of the state model ordinances, the California model recommends the most stringent standards. The code requires Class A-rated roofs, and a minimum of 100 feet of defensible space for buildings in the Very High Fire Hazard Severity Zone (VHFHSZ). The Utah Wildland-Urban Interface Standards are based on the UWIC and utilize the UWIC sliding scale for defensible space. Roofing standards are also based on the same factors as the UWIC, however, Class A roof coverings are required in all three hazard categories -- moderate, high, or extreme. The Florida model ordinance recommends Class A roofs, 30 feet of defensible space around structures, as well as 12 feet of defensible space around the perimeter of new developments.

5.4 Vegetation Management Plans

All five model ordinances require property developers to submit Vegetation Management Plans (VMP) with building plans prior to subdivision approval or issuance of building permits. The VMP is a site-specific wildfire analysis that addresses topographic and vegetative features and includes elements and timetables for the removal of slash, ground fuels, ladder fuels, dead trees and the thinning of live trees. A plan for maintaining fuel-reduction measures after initial development is also required. Regulations requiring developers to include a maintenance component in VMP's provide some assurance that wildfire protection afforded by the initial fuel reduction projects will continue to reduce the community's wildfire risk exposure.

6. STATE LEGISLATION

As of mid-2005, only two states, California and Oregon had adopted legislation requiring landowners to conduct vegetative modifications to reduce wildfire hazard. Washington and Colorado have tried unsuccessfully to pass state-level wildfire protection legislation, but have found greater acceptance through creating guidelines and offering assistance to counties and towns that enact vegetation management regulations.

6.1 California

For the purpose of fire protection, California lands are divided into two categories: State Responsibility Areas (SRA's) and Local Responsibility Areas (LRA's). The SRA is the land for which the state has the primary responsibility for preventing and suppressing fires. In LRA's, either local government or federal authorities have primary fire protection responsibility.

California uses a Fire Hazard Zoning system to identify geographic areas that are at severe risk of wildfire. Regulations apply to properties ascertained to be in Very High Fire Hazard Severity Zones (VHFHSZ) in both the SRA and the LRA. The VHFHSZ in the SRA was identified in the "Maps of Fire Hazard Severity Zones in State Responsibility Areas of California" adopted in 1984. The LRA VHFHSZ was mapped in the mid-1990's. In December 2007, California is scheduled to adopt new SRA fire hazard maps using improved mapping techniques, fire science and data. In 2008, new maps of the VHFHSZ in the LRA will be presented and adopted. The maps will form the basis of legal requirements for new wildland-interface building standards, focusing on ignition-resistant building materials for roofs, walls, windows, decks and other building elements. (California Department of Forestry and Fire Protection 2007).

Regulations pertaining to development in the VHFHSZ are found in California's Public Resource Code, the General Code, and the Health and Safety Codes. Public Resource Code (PRC) 4291 was enacted in 1985, initially requiring 30 feet of defensible space around all structures in the VHFHSZ and amended in 2005 to require a minimum of 100 feet of vegetative clearance around structures. Subsequent enactments include PRC 4290, enacted in 1991, which set additional standards for roads and access, signage and building identification, greenbelts, and private water supplies for firefighting. These additional elements continued to raise fire safety standards in the SRA.

Despite these regulations, wildfire continued to threaten homes and lives in California's ever-growing wildland-urban interface. A contributing factor was that regulations at that time did not apply to all fireprone areas of the state, only the SRA.

In 1992, California adopted the Bates Bill (General Code Sec. 51175-51189), to extend wildfire mitigation regulations to LRA. The regulations are comparable to those that existed in the SRA since 1985, and brought fire-hazard reduction regulations to all high-wildfire risk areas throughout the state.

Minimum fire safety standards for development in the VHFHSZ were set forth for local governments to adopt. A wildfire risk assessment of the state was completed in 1995, and model ordinances were drafted. Any jurisdiction located within in the VHFHSZ is required to adopt the model ordinances or demonstrate that restrictions already in place meet or exceed the Bates Bill requirements. The 2005 amendment to PRC 4291 not only extended the minimum clearance around structures in the VHFHSZ from 30 feet to 100 feet or to the property line, it also specified that state law, local ordinance, rule or regulation, or insurance company may require vegetative clearances greater than 100 feet from structures.

Local governments implement the regulations through their building permit and subdivision approval processes. The California Department of Fire and Forestry (CDF) consults with local governments and reviews all proposed construction and development, advising on wildfire mitigation issues. The CDF is responsible for enforcement of the wildfire protection regulations. They employ a force of inspectors to visit homes in VHFHSZ areas and CDF has the authority to fine landowners for failure to comply with regulatory standards.

6.2 Oregon

Oregon adopted the Forestland-Urban Interface Fire Protection Act (Act) in 1997, however, administrative rules implementing the Act were not adopted until 2002. The program, administered by the state, is being phased in slowly in designated high risk counties. To date, the Act has been implemented in two counties, Jackson and Deschutes.

In accordance with the Act, properties designated by the State as Forestland-Urban Interface (FUI) are assessed for wildfire risk based on factors such as climate, natural vegetative fuels, topography, and housing density. The Department of Forestry (DOF) notifies property owners of their assigned classification, whether low, moderate, high or extreme.

The required defensible space standards differ based on the type of roofing materials used. Minimum defensible space distances for homes with non-fire-resistant roofs are: 30, 50, and 100 feet for properties classified as moderate, high, or extreme hazard, respectively. Distances for homes with a fire-resistant roof are 30 feet in moderate and high hazard areas, and 50 feet in areas of extreme hazard.

To implement the Act, the DOF mails all owners of urban-interface forestland a property evaluation form. The form allows owners to self-assess compliance with the required standards. Accredited assessors are trained to assist homeowners with the certification process, provide prescriptions for mitigation work, and may conduct needed property treatments at the landowner's cost. Property owners have two years in which to complete the necessary wildfire risk-reduction measures and return a certification form to DOF. In counties where stricter requirements already exist, those ordinances supersede the state law.

No enforcement or inspection measures are included in the regulations at this time. In the event of a wildfire, the DOF will determine whether the ignition or spread of the fire was directly related to the owner's failure to meet the standards. If a landowner is found to have directly caused the wildfire, the costs of suppression of that fire will be assessed to the owner up to $100,000. Property classifications are updated every five years or when a transfer of ownership occurs.

7. STATE GUIDELINES

Four states: Colorado, Montana, Virginia, and Washington have developed guidance documents to assist local jurisdictions in the development of regulatory programs. The guidelines generally address firesafe subdivision design and wildfire protection measures for existing homes. The state guidelines differ from the model ordinances of California, Florida, and Utah in that they do not contain provisions for administration and enforcement. Furthermore, with the exception of the Virginia guidelines, these documents are not in a regulatory code format. Rather, they are in a less formal descriptive format, often with graphic representations of recommended wildfire protection standards. As found in the state model ordinances, state guidelines for vegetation modification are often more stringent than those provided in the UWIC and NFPA 1144 model codes.

7.1 Hazard Severity Rating and Defensible Space

Similar in content to the UWIC and NFPA model codes, many of the guidelines include wildfire-hazard-severity rating systems to evaluate the wildfire risk to individual properties and subdivisions.

For example, the Washington "Model Fire Hazard Policies and Development Standards for County and City Comprehensive Land Use Plans" establishes a Wildfire-Hazard Rating System with possible classifications of low, moderate, high and extreme fire hazard. A minimum area of defensible space of 50 feet is established for all properties classified as moderate, high, and extreme wildfire risk.

State guidelines differ in their recommendations for defensible space. The Washington and Virginia guidelines recommend a 50 and 70 foot treatment zone, respectively. Montana and Colorado establish a more complex three-zone modification scheme with varying levels of treatment recommended within each zone These guidelines correlate the extent of the defensible space area to the property's degree of slope. The Montana "Fire Protection Guidelines for Wildland Residential Interface Development" recommend increased distances of defensible space only on the upslope approach to structures (Montana Department of Natural Resources 1993), (figs. 14.1, 14.2, and 14.3). The recommended distances range from 100 feet for level terrain to 150 feet for slopes of 20-30 percent.

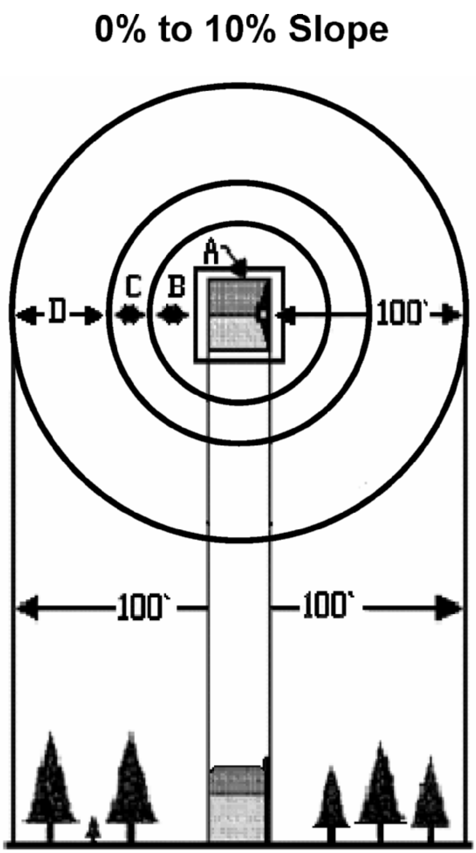

Figure 14.1. Montana Fire Protection Guidelines for Wildland Residential Interface Development.

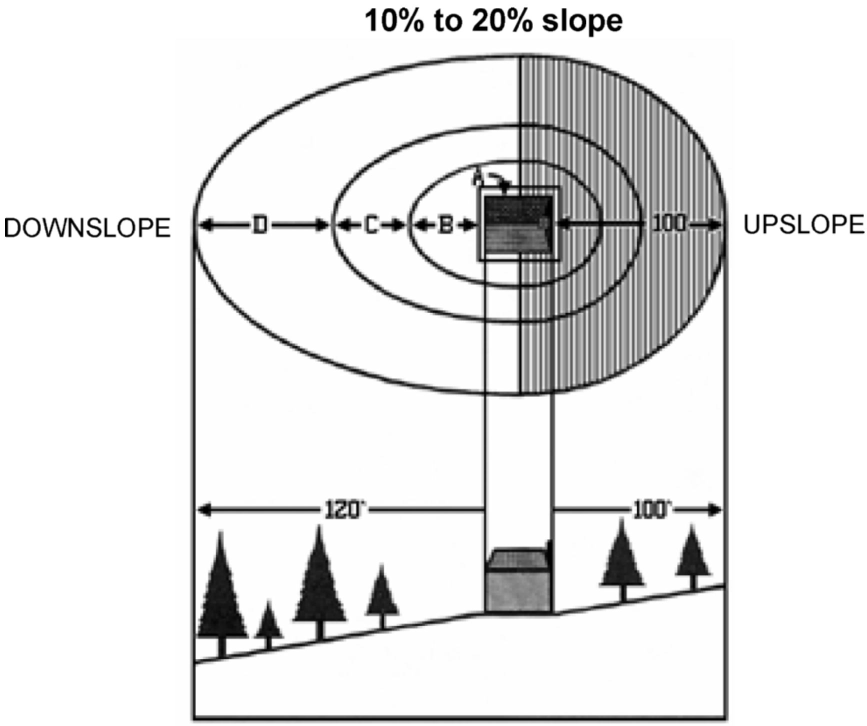

Figure 14.2. Montana Fire Protection Guidelines for Wildland Residential Interface Development; 10% to 20% slope.

The Colorado guidelines also correlate distances of defensible space to slope within a three-zone treatment area. However, the Colorado guidelines call for larger areas of defensible space for both upslope and downslope approaches to the structure with greater distances for upslope areas (Dennis 2003) (fig. 14.4). In addition, the Colorado treatment standards are fairly complex in that modifications in tree and shrub spacing are correlated to the degree of slope; thinning is intensified as slope increases (Dennis 2003) (fig. 14.5).

7.2 Goals for Growth

Washington provides leadership in its guidelines by suggesting that a wildfire protection policy statement be incorporated in high risk county Growth Management Plans. The model policy statement is exemplified in the Yakima County Growth Management Plan with the stated goal to "Protect life and property in rural Yakima County from fire hazards." Florida's guidance documents also

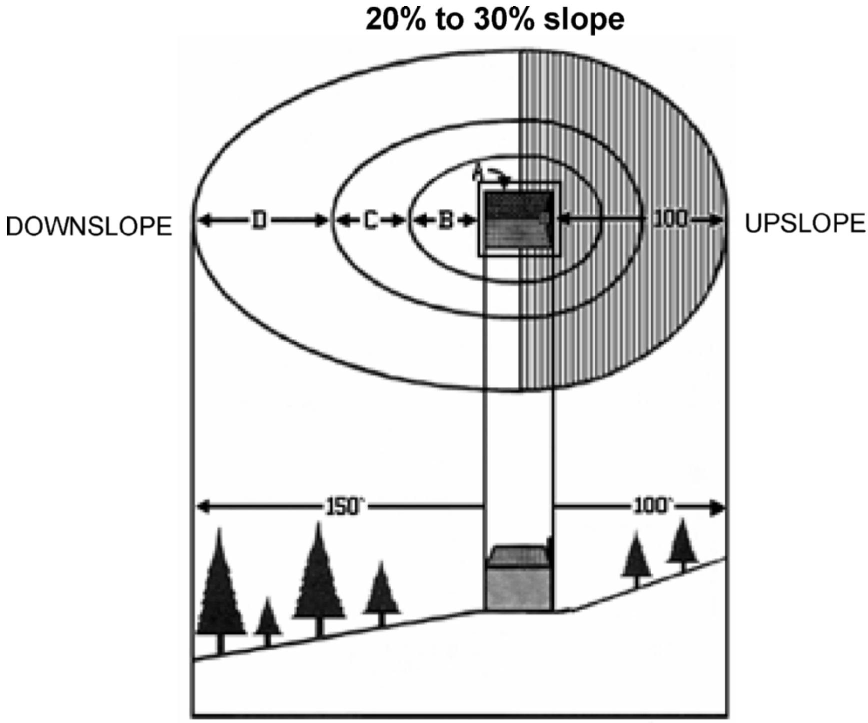

Figure 14.3. Montana Fire Protection Guidelines for Wildland Residential Interface Development; 20% to 30% slope

recommend providing a goal statement in local governments' comprehensive plans to bring protection from wildfire to the forefront for all planning purposes. The inclusion of wildfire protection goals in the vision statement for growth provides important reinforcement for the adoption of wildfire mitigation regulations.

8. LOCAL ORDINANCES

With the exception of California and Oregon, local ordinance development is a voluntary action undertaken by local leaders to address community wildfire protection. Ordinances initiated by county and municipal governments are generally based on the UWIC or NFPA 1144 model code, the respective state's recommended model ordinance, or wildfire protection guidelines. In a review of the regulations listed on the National Database of State and Local Wildfire Hazard

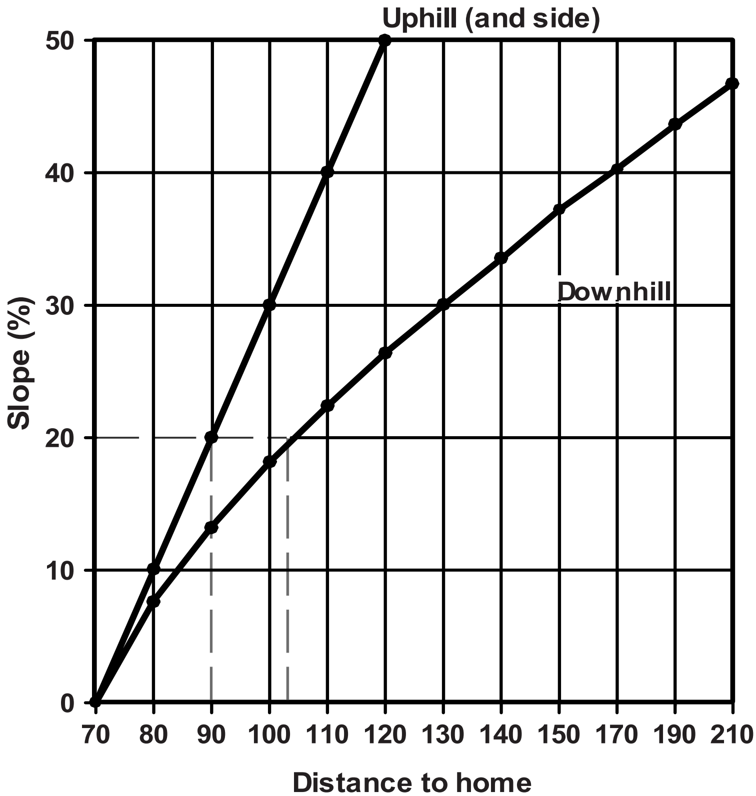

Figure 14.4. Colorado guidelines for defensible space dimension.

% slope	Tree Crown Spacing	Brush and Shrub Clump Spacing
0 – 10%	10’	2 ½ x shrub height
11 – 20%	15’	3 x shrub height
21 – 40%	20’	4 x shrub height
> 40%	30’	6 x shrub height

Figure 14.5. Colorado guidelines, tree crown and shrub spacing.

Mitigation Programs, similar standards were found in many defensible space regulations with varying ranges of requirements for the standards (table 14.1).

Local jurisdictions often modify guidelines and model codes to meet unique characteristics or specific needs of their communities. Some local ordinances focus only on standards and permitting processes aimed at creating firesafe communities as new subdivisions are developed. Others include provisions for fuel modification on existing properties, as well.

A number of jurisdictions have adopted ordinances with defensible space standards that are more stringent than those found in the state's model ordinance to achieve a particular purpose. Local governments in California have adopted some of the most restrictive ordinances in the nation. Required defensible space zones of more than 100 feet are not uncommon and fuel modification treatments can involve removing all flammable, native vegetation (including grasses and shrubs) within 100 feet of the home. For example, the city of Glendale's Hazardous Vegetation Ordinance (Building and Safety Code Vol. VI , Sec. 16, App. II-A), establishes landscape requirements to protect the visual quality of the hillsides and promote fire safety. The ordinance is unique in that a landscape/fuel modification permit must be obtained not only for new construction and significant remodels, but for re-landscaping or grading projects, as well. The selection of plant species for landscaping are also limited by the ordinance and pruning of several indigenous tree species for wildfire protection requires a permit. In addition, a four-zone fuels modification system for a total of 150 feet goes beyond the specified three-zone, 100 feet modification scheme found in the state model.

Table 14.1. Vegetation Management Components of Wildfire Mitigation Programs

Hazard rating guide — Evaluation system for assessing wildfire hazards on individual properties or subdivisions

Vegetation Management Plan — Required submission to demonstrate developer's planned actions for fuels mitigation and maintenance

Generalized Defensible Space Requirements

Fuel modification areas of 30 to 150 feet around structures and 12 feet around the perimeter of new developments including:

- Removal of flammable vegetation, excluding cultivated ground covers and single trees
- Thinning of trees to allow 10 feet of spacing between canopies
- Pruning trees to allow 10 feet of spacing between tree canopies and structures
- Pruning trees from 6 to 15 feet from the ground
- Pruning trees for a vertical clearance of 12.5–15 feet along roads
- Clearing brush for a 10-foot fuel break adjacent to roads

The adoption of an established model code with additions and deletions of some model provisions is fairly common. For example Ruidoso, New Mexico has adopted the UWIC in its Land Use Code (Art. III, Ch. 42, Sec. 70). However, requirements for vegetative modifications include unique woodland thinning provisions; thinning requirements establish minimum basal areas by species composition in a zone located 30-60 feet from structures.

Local ordinances may also be designed to specifically address removal of native combustible plant materials that create a wildfire hazard. For example, in Oregon, the Sunriver Homeowners Association's Ladder Fuel Reduction Plan (Sec. 4.01E.2, Sunriver Rules and Regulations) requires the removal of bitterbrush and manzinita, predominant flammable native shrubs. All bitterbrush, noxious weeds, dead vegetation, and other flammable shrubs within fifteen feet of a structure must be removed. In addition, bitterbrush and manzanita must be cleared three feet beyond the drip line of tree branches.

Similarly, in Monrovia, California, where highly combustible native chaparral is prevalent, required treatments include cutting all grass, weeds, and chaparral within 30 feet of homes to 3 inches in height or less; and thinning chaparral plants to an average 12 to 18 feet of separation within 200 feet of the property owner's home (M.M.C. Sec. 8.14.01-8.14.14). In a situation where the 200 feet of clearance from the home extends beyond the property line, the owner remains responsible for the vegetative clearance. To accomplish the required treatment, generally, the affected owner obtains a release from the adjacent owner and treats the property at his own expense.

Developers have a vested interest in complying with wildfire protection regulations. However, new homeowners may be less motivated to maintain fuel modifications once new subdivisions are established and the initial wildfire protection goals achieved. The procedures utilized by local governments in California illustrate one approach to achieving continuity in vegetation treatments after initial subdivision establishment. The local jurisdictions fire departments conduct inspections of all properties and send out "A Notice to Abate Fire Hazard" to owners of properties where the need for treatment has been determined. If the property owner does not complete the required treatments within the designated timeframe, the Fire Department has the authority to have the fuel modifications conducted, with the cost billed as a lien against the property.

Some local governments have included a provision requiring new developments to adopt covenants or deed restrictions for vegetation maintenance in their wildfire protection ordinances. These provisions require future homeowners and/or homeowners' associations to maintain defensible space. Upon purchase of property, the homeowner signs an affidavit accepting the restrictions on the deed. Covenants or deed restrictions typically set out criteria such as minimum square footage, type of construction, architectural style and so forth to ensure that homes built there conform to the neighborhood (Crawford 2005). Incorporating defensible space requirements in restrictive covenants is a new use of an old tool. Subdivision covenants or deed restrictions provide reinforcement of wildfire protection measures at the neighborhood level. At the time of home purchase, owners are advised of the property's vulnerability to wildfire and their responsibility to protect themselves by maintaining defensible space around the home. For enforcement purposes, the subdivision's Declaration of Covenants, Conditions and Restrictions (CC&R's), generally include a provision enabling the homeowners' association to levy fines should homeowners fail to comply with maintenance requirements.

In some localities, subdivision CC&R's for defensible space maintenance are required by local ordinances. For example, Santa Fe County, New Mexico through its Urban Wildland Interface Code (Ordinance No. 2001-11), requires vegetation management measures to be recorded in the covenants of all new subdivisions of twelve or more lots. Local code may also direct subdivision maintenance of fuel treatments in common areas. The City of Ormond, Florida addresses this need in its Land Development Code (Ch. III, Art.13A). The Code stipulates that the developer must prepare a greenbelt and/or conservation area maintenance plan that provides for the management of common areas for fuel reduction and hazard mitigation by the property owners' association. The plan must be incorporated in to the subdivision's CC&R's and recorded with the final plat.

9. INSURANCE PROGRAMS

Although insurance requirements differ from direct government regulations, they serve to reinforce wildfire protection regulatory programs by introducing a clear, economic incentive for property owners to undertake measures to reduce wildfire risk. Defensible space requirements to obtain insurance coverage can be quite stringent in some high fire hazard areas of California. For example, in Glendale, the state insurance program, the Fair Plan can require up to 400 feet of fuels treatment around structures. In addition, if brush exposure is down-slope from structures and over 30 degrees, only half of the cleared distance is counted. Under the Fair Plan, the clearance distance requirement applies to vegetation that extends beyond the property boundary. If the property owner is unable to conduct the treatment in the area extending into the neighboring ownership, a surcharge, based on the distance of the untreated area, will be applied to each $1,000 of insurance. The surcharge is removed once the treatment is accomplished.

Insurance availability for homes in high wildfire risk areas in other states is an emerging concern due to increased losses experienced by insurers in recent years. In 2003, State Farm Insurance Company began implementing a program to reduce the potential for future financial losses in some high hazard areas. The program is underway in Colorado, Arizona, and New Mexico. Over a three year period, 22,000 homes located in the targeted states will be inspected to identify fuel modifications and other mitigation measures needed to reduce wildfire hazard. Homeowners will have 24 months to complete the recommended treatments. After the allotted time period, agents will conduct follow-up inspections for compliance. If wildfire safety measures have not been completed, non-renewal of the insurance policy for the property may be considered.

Collaboration with local fire officials can facilitate the insurance company's property evaluations. For example, in Prescott, Arizona, the Fire Department's inspection reports for individual homes are being used as a basis for evaluating wildfire risk to determine policy coverage for individual homes.

A significant role exists for insurance companies in helping to create an effective framework of wildfire risk mitigation strategies. Kovacs (2001) points to areas of particular importance, beyond providing compensation for property loss. These include public education through the industry's on-going involvement in wildfire management programs, such as California's Fire Safe Council and the Firewise Community network. Second, the industry provides powerful incentives for hazard mitigation to residents of Wildland-Urban Interface communities through insurance pricing. Third, the insurance industry continues to function as active stakeholders in community wildfire reduction efforts through promotion of safer land use, along with improved building practices and standards.

10. INSIGHTS DRAWN FROM PROGRAM ADMINISTRATORS

During the spring and summer of 2005, researchers surveyed managers and administrators of wildfire risk reduction programs listed and summarized on the National Database of State and Local Wildfire Hazard Mitigation Programs website, www.wildfireprograms.usda.gov. The purpose of the survey was to gather additional information directly from program officials concerning the goals and objectives of their programs, the obstacles they have experienced in their work, and their recommendations for the most cost-effective methods to reduce risks to communities. Administrators representing 29 regulatory risk reduction programs responded to the survey.

10.1 Program Goals and Objectives

One of the attributes of interest was the extent to which regulatory-based risk reduction programs integrate other broad goals and objectives. For example, do programs that oversee the implementation of building codes also incorporate outreach and public education activities into their efforts? We found that respondents from each of the 29 regulatory-based programs include activities designed to help community residents understand, not only relevant defensible space requirements, but also the underlying wildfire risks and a variety of established mitigation strategies. Similarly, 28 of the 29 regulatory-based efforts include specific activities to help home and property owners establish and maintain a commitment to vegetation management and to assist in the removal and disposal of vegetative material. Moreover, all 29 administrators of these programs report that they examine wildfire risk criteria, attempt to evaluate the overall levels of risk to communities, and designate specific areas of high risk. Clearly, the responses of these administrators suggest that regulatory-based wildfire risk reduction efforts include a variety of related program objectives designed to provide residents with information concerning the risks they are facing, the actions they may take to reduce that risk, as well as the specific legal requirements, standards, and guidelines applicable to their communities.

Next, we were interested in the types of regulations these programs are administering. We found that the most common types of regulations are those for subdivisions and residential development, with 75 percent of respondents overseeing these requirements. Other commonly used regulations for wildfire risk reduction or mitigation included implementation of state guidelines (62 percent), building codes (65 percent), and fire codes, (59 percent). Roughly one-third of the respondents administer zoning ordinances (34 percent) and land-use codes (31 percent) that include vegetation management provisions. The least commonly administered regulations among the respondents were real estate disclosure, with about 27 percent of respondents implementing this type of regulation. Only the State of California requires disclosure of wildfire risk classification in real estate transfers.

10.2 Obstacles to Implementation

We asked the administrators to identify obstacles that they believe are impeding progress toward reducing wildfire risk within their jurisdictions. After reviewing a list of potential obstacles, they were asked to indicate the extent to which each item is an obstacle or impediment to their efforts by giving each a score from 0–5, with 5 indicating an extreme obstacle. According to the respondents, the most serious obstacles facing their programs are budgetary constraints (3.6 on a 5-point scale). In addition, respondents reported that negative attitudes among property owners are often impediments to reducing wildfire risk. These may include public apathy (3.17 on a 5-point scale) and resistance from homeowners concerning removal of dangerous vegetation and maintaining a more fire-resistant landscape (average score of 2.93). The average responses are presented in table 14.2.

10.3 Emerging Strategies for More Effective Regulatory Programs

As state and local decision makers struggle with how best to reduce wildfire risks and overcome budgetary constraints, strategies that leverage resources, such as forming collaborative relationships with other organizations are increasingly attractive. The American Planning Association (APA) recently called for increased pre-fire planning, citing the sheer volume of new development in the Wildland-Urban Interface. The authors of the APA report, Planning for Wildfires,

Table 14.2. Descriptive Statistics, Obstacles Reported by Wildfire Program Administrators

	N	Mean
Budget is an obstacle	29	3.5862
Apathy among prop. owners	29	3.1724
Homeowner resistance	29	2.9310
Inadequate enforcement of regs	29	2.5862
Tree protection ordinances	29	1.9655
Legal appeals to trt. adjacent public lands	29	1.9310
Need more technical help	29	1.7931
Constraints from env. regs	29	1.7586
Lack of qualified staff	29	1.5172
Inadequate public input into program	29	1.4138
Low coop. among stakeholders	29	1.3793
Valid N (listwise)	29	

state that the rapid growth of many of these communities makes it imperative that residents, business owners, developers and local decision makers adopt strategies for safer designs for new neighborhoods and risk-mitigation for existing developments. “Safe Growth” has become an important element of the anti-sprawl, environmentally friendly “Smart Growth” movement among professional planners. In addition, they point to an increasing federal emphasis on mitigation planning as a way to reduce the damages associated with catastrophic wildfire. This emphasis is seen in the Disaster Mitigation Act of 2000, as well as the Healthy Forests Restoration Act of 2003 (Schwab and Meck 2005).

There is ample evidence that administrators of regulatory-based programs are working with other agencies from various levels of government to formulate more effective pre-fire plans to reduce wildfire risks. According to our survey of administrators, twenty-seven of the twenty-nine reported that they participate in collaborative partnerships, with a mean of three different levels of government–local, county, state, or federal–represented. This indicates that most administrators regularly interact with multiple public decision makers, thus increasing the likelihood of more coordinated implementation of current regulations, as well as more coherent planning for future risk reduction standards and requirements.

In addition, program administrators recommend several specific program activities they have found to most valuable in reducing risk within their jurisdictions. We asked administrators to indicate, on a scale of 1-5, with 5 indicating “extremely cost effective”, the specific program activities they have found to be most cost-effective. According to their responses, these risk reduction activities have been most effective:

Regulations for fuels treatment in new developments (3.96),
Meeting with neighborhoods and communities (3.72),

Public education (3.52),
Cost/Share assistance for homeowners' fuels treatment (3.52), and
Demonstration Projects (3.44).

From these recommended program activities, it is clear that administrators favor a more comprehensive approach to reducing risk that entails implementing legal requirements while also offering specific instruction and assistance to property owners.

These survey results and the information compiled for the National Database of State and Local Wildfire Hazard Programs website suggest that regulations play an important role in a comprehensive approach to reducing wildfire risk at the state, local and community levels throughout the nation. Furthermore, mitigation efforts are often developed from collaborative plans that incorporate goals of multiple stakeholders to achieve continuity in mitigation practices across high fire risk landscapes. Comprehensive regulatory programs include: 1) state laws or guidelines to direct local governments, 2) growth management or comprehensive plans that incorporate wildfire risk reduction goals at the regional level, 3) county and municipal ordinances that establish specific requirements for developers and property owners, and 4) mechanisms for maintaining defensible space such as inspection/notification programs or the use of deed restrictions to drive homeowners' mitigation efforts at the subdivision level. However, as demonstrated in the survey results, for regulatory-based efforts to be effective, administrators need adequate funding, appropriate technology to assess risk to communities, clear guidelines to implement, and the support of a public that is often skeptical about the benefits of vegetation management and enhanced building codes.

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CHAPTER 15

ECONOMIC ANALYSIS OF FEDERAL WILDFIRE MANAGEMENT PROGRAMS

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1. INTRODUCTION

In the past two decades, there has been a significant increase in the number of acres burned by wildland fires and in the amount of money being spent to suppress these fires (Calkin et al. 2005a). With expenditures on suppression alone climbing to more than a billion dollars in four of the past seven years (2000-2006), the federal land management agencies are coming under ever increasing pressure by Congress and government oversight agencies to manage fire in a cost efficient manner. Economic analysis can benefit all fire-related programs and activities, and ignoring economic analysis in the wildland fire decision-making process, whether on a strategic or tactical level, can lead to wasted resources, poor outcomes, and higher-than-warranted expenditures.

A full economic analysis of federal land management agencies' wildfire programs would address activities undertaken: (1) before the fire, (2) during the fire, and (3) after the fire. The chapter begins with a description of the generally accepted model for evaluating wildland fire programs, the cost-plus-net value change or cost-plus-loss model. Though the cost-plus-loss model has been extended from its initial focus on presuppression to address all of these activities, most research to date has focused on a single aspect of the wildland fire program. We then turn to a discussion of where and how fire economics currently enters wildfire program decision making and indicate where additional applications are possible. We conclude with noting the issues and complications specific to conducting analyses of Federal wildfire management programs and suggestions for future research.

Early theoretical models of fire management determined the efficient level of the fire management program by minimizing the sum of program costs and fire damages (Headley 1916, Sparhawk 1925). As noted by Rideout and Omi (1990), these models have evolved over time to incorporate fire benefits as well. These cost-plus-net-value-change (C+NVC) models recognize that damages must be subtracted from benefits to arrive at the net gain or loss from a fire. The C+NVC model, whether set up to maximize net benefits or minimize costs plus loss, yields

T. P. Holmes et al. (eds.), The Economics of Forest Disturbances: Wildfires, Storms, and Invasive Species, 295–322.

the standard requirements for economic efficiency: the correct level of the program will be where the marginal benefits of fire management are equal to the sum of the marginal costs of fire management and the marginal net loss from fire.

Solving the C+NVC model determines the levels of presuppression inputs (the expenditures on placing equipment and personnel prior to a fire season) and in-season suppression inputs (Donovan and Rideout 2003). In addition, this model can be used to develop optimal levels of fuels treatment, prevention activities, and recovery and restoration activities (chapters 8 and 13-18 of this book). The main difficulty in implementing the C+NVC model, either from an operational standpoint or as an analytical tool, lies in determining net value change. The absolute change in human and natural values in the absence of management is unobserved. Even at the margin, the reduction in net damages from increasing suppression resources to fight a fire is unobserved or only observed with error. Further, the values-at-risk from wildfire include both market and non-market goods. Valuing non-market resources in monetary terms can be accomplished with well-established techniques such as travel cost or hedonic models. However, the effect of fire on these values is often unclear. These techniques are also data-intensive and time-consuming; qualities that make them ill-suited for use at the fire level where conditions change rapidly. Alternative techniques for monetizing non-market goods are discussed later in this chapter.

The net value change from wildfire can be broken down into three components, where each component reflects a change in a different set of values resulting from a potential condition where management does not try to influence fire behavior. The first is the monetized change in human values and services due to wildfire, such as damages to structures, timber losses, and damaged or destroyed recreation opportunities. The second is the monetized future change in ecosystem function and services from wildfire. As more suppression is applied, the beneficial effects of fire are diminished in fire-adapted ecosystems. However, reducing ecosystem damages from catastrophic fires by applying suppression inputs may increase future ecosystem function and services. A third element is the change in future fire management due to changes in fire regimes that may result from the over- or under-suppression of wildfire. For instance, if current suppression creates fuel buildups that depart from the natural fire regime and result in larger and more intense fires in the future, the discounted future value of increased suppression expenditures and damages should be included in NVC.

Applications of the C+NVC model over the years have primarily focused on budgeting for presuppression and determining the optimal level for suppression once the presuppression budget has been determined. For years, a computer-based, simulation modeling system based on the C+NVC model entitled NFMAS (National Fire Management Analysis System) was used by several federal and state fire management agencies to support fire program budget requests (NARTC 1997, Lundgren 1999). Another fire management tool developed using the C+NVC model was the Fire Economics Evaluation System (FEES) (Mills and Bratten 1982). The C+NVC-based systems were used primarily by agencies whose missions included resource utilization and commodity values. Different fire management tools (FIREPRO and FIREBASE) were used by the National Park Service and the Fish and Wildlife Service whose missions focused more on non-market values. These systems quantify staffing and financial requirements for fire management activities based upon an analysis of program workload and complexity including initial attack readiness, wildland fire use, and fuels management using historical information and average annual workload (Botti 1999).

Although these early fire management budgeting tools focused primarily on developing a presuppression budget, economic analysis is important for all fire-related programs and activities. In the pre-fire stage, economic analysis enters the wildland fire program through (1) land management planning, which provides overall direction for federal wildland management, (2) presuppression budgeting and the determination of location and quantities of physical fire suppression resources, and (3) fuel treatment programs to reduce fuel loads and thereby reduce damages from future wildfires. Economic analysis during the fire assists in tactical level planning during the season to determine appropriate management responses, such as allowing fires to burn for wildland restoration or suppressing the fire. In the post-fire stage, economic analysis is necessary to determine the appropriate level and type of expenditures for rehabilitation projects and for evaluating suppression performance ex-post after the season. Note that while we discuss these programs as though there are linear stages, one following the other, in fact the pre-fire stage is also the post-fire stage and, thus, may be better thought of as a circle. In addition, while we have defined the stages in reference to fire events, this is for discussion only, as a wildfire management program may involve activities of equal or greater consequence than the fires themselves.

2. ECONOMIC ANALYSIS OF ACTIVITIES BEFORE THE FIRE

Activities undertaken before a fire occurs include planning, pre-positioning of suppression resources, prevention, and fuels management (including mechanical fuel treatments and prescribed fire). Each of these is discussed in more detail below.

2.1 Planning

Current direction is for land management plans (long range planning documents that guide the management of individual forests) to recognize the role of fire, particularly where fire has historically been part of the ecological process. The Federal Fire Policy Implementation Strategy (USDA and USDI 2003) states that:

> Overall direction is provided to the wildland fire management program by land and resource management plans (L/RMP) . . .. The paramount policy is firefighter safety. Fire regime dynamics must also influence land and

> resource management objective development in the L/RMP. The L/RMP's desired future condition will incorporate the desired mix of Condition Classes by fire regime (page 9).

From the standpoint of desired future condition, wildland fire is seen as either a disturbance that can thwart attempts to achieve the desired future condition or a tool that can be used to make progress towards that condition.

More specific fire management plans (FMPs) identify and integrate all wildland fire management and related activities within the context of approved land management plans. Cost considerations in the FMP are most often addressed through the use of aggressive initial attack, to put out fires before they become large and costly, and appropriate management response (AMR) (discussed in section 3.2), which is suppose to consider both the costs of suppression and the values at risk. This is accomplished through the wildland fire situation analysis (WFSA) (also discussed in section 3.2) that is done at the time the fire escapes initial attack efforts.

In their 2004 report, the Strategic Issues Panel (Strategic Issues Panel 2004) stated that none of the L/RMPs that they looked at included any consideration of the costs of suppression. They further stated that because the plans were centered on gains, in terms of meeting desired future conditions, they provide little help in the area of loss aversion, which the panel felt was central to wildfire management. They recommended that the land management agencies "set policy and direction on agency land/resource management planning to incorporate cost management on large wildfires." In response to this, direction has been given to include discussions of the costs of suppression in these plans; however, there is currently no requirement for any sort of economic analysis.

Ideally, economic analysis at the planning stage would include some determination of values at risk, potential suppression expenditures, and possible benefits of wildland fire. Activities specifically addressed, in addition to suppression, would include (1) location and numbers of initial attack resources, (2) prevention and detection programs, and (3) fuel treatment options. Some of the new tools currently under development for budgetary planning (see FPA discussion in section 2.2) and wildland fire decision support (section 3.2) could be modified or extended for use in land management and fire management planning.

2.2 Resource Pre-Positioning

Pre-positioning of suppression resources has long been seen as a way to minimize damage from wildfires and constrain suppression costs. Having sufficient resources available to aggressively attack fires shortly after ignition was, and still is, seen as a way of keeping fires small, therefore mitigating damage and avoiding the large expenditures associated with attempting to suppress a wildfire once it has gotten large. Many initial attack planning models were designed around the general concept of C+NVC discussed in an earlier section and have included a mix of deterministic and simulation models. These models have recently come under criticism for a number of reasons including their inability to account for nonmarket values and to deal with more complex fire situations since they are based on average fire seasons and single fire starts.

More recent models have attempted to address some of these shortcomings. The California Fire Economics Simulator (CFAS) (Fried 2006) was designed to deal with these more complex situations including multiple fire starts, drawdown of resources, differing fire fighting tactics, and so forth. The Fire Program Analysis system (FPA 2005), currently being developed by the federal land management agencies, uses cost effectiveness analysis to better account for the value of nonmarket resources. The numbers, types, and locations of suppression resources prior to the fire season can be optimally determined by cost effectiveness analyses to evaluate the efficient allocation of presuppression resources by optimizing weighted acres managed, a non-monetized measure based on expert opinion.

2.3 Prevention

Wildland fire prevention programs are aimed at reducing the occurrence of human-caused wildland fires and mitigating the damages caused by those fires that do occur (reducing the cost plus loss of wildfires). In fact Stephen Pyne has been quoted as saying "an ounce of prevention is worth several pounds of fire damages and fire suppression expenses" (Doolittle and Donaghue 1991).

Prevention programs are targeted at ignition sources with the potential to cause the greatest losses. This potential is evaluated through an assessment of risk, hazard, and value with these terms being defined as follows: (1) risk–uses, human activities, or events with the potential to result in wildfire ignitions, (2) hazard–the fuels and topography of an area, and (3) values–natural or developed areas where losses by fire are unacceptable. These elements are evaluated using historical fire information and are tied to land management and fire management plans. Prevention programs include such things as education programs aimed to prevent human-caused fires by changing people's behavior (such as the Smokey Bear program), visible patrolling of fire prone areas, and enforcement of fire regulations and ordinances. Early fire detection is also part of the fire prevention program by catching fires before they become a problem, thereby reducing losses. Removal of hazardous fuels around homes and using fire resistant materials when building homes in fire prone areas can help homeowners protect their property when fires do occur and the Firewise communities program has been developed to aid communities and homeowners in designing safer communities and homes (NIFC 2007, Firewise Communities 2007).

Research related to the economics of fire prevention, however, is virtually nonexistent and has primarily focused on identifying areas at risk from wildland fires to aid in the planning process or reducing the risk of wildland fires through the use of fuels management (next section). Some research has been done related to wildland arson fires, which is described in detail in chapter 7.

2.4 Fuels Management

The development of the National Fire Plan produced an increased interest in the economics of fuel reduction treatments as land managers attempt to deal with high fuel loads. The primary purpose of the Healthy Forest Restoration Act of 2003 (Public Law 108-148) is: "to reduce wildfire risk to communities, municipal water supplies, and other at-risk Federal lands through a collaborative process of planning, prioritizing, and implementing hazardous fuel reduction projects." Therefore, the USDA Forest Service (USFS) and the U.S. Department of Interior (DOI) have committed to a significant increase in hazardous fuel treatments. The cost and effectiveness of different fuel treatments in different forest settings must be understood, however, before desired outcomes can be achieved in a fiscally responsible manner,

While there is overall agreement that fuel treatments can affect fire behavior by reducing intensity and/or size of fire (Graham et al. 2004, Agee and Skinner 2005, Stephens and Moghaddas 2005), it is still uncertain whether the benefits of these treatments outweigh the costs. Benefits include the restoration of ecosystem health, which is difficult to value, as well as avoided costs such as reduced suppression expenditures and reduced property damage. Costs of fuel treatments, whether prescribed burning or mechanical treatments, have been shown to be a function of fuel loads, slope, and fuel treatment location (wildland urban interface or not) (Berry and Hesseln 2004, Berry et al. 2006, Calkin and Gebert 2006, Skog et al. 2006). In addition, administrative factors and managers' risk aversion were found to influence treatment costs (González-Cabán and McKetta 1986, González-Cabán 1997). Nearly all empirical studies have found that larger treatments (in acres) result in lower costs per acre (Rideout and Omi 1995, Berry and Hesseln 2004, Calkin and Gebert 2006). However, data on actual fuel treatment costs, for both mechanical and prescribed burning treatments, are limited, in spite of continued interest in this topic for more than 20 years (González-Cabán and McKetta 1986, Cleaves et al. 1999, Berry and Hesseln 2004, Calkin and Gebert 2006).

Recently, simulations have been used to augment actual data, including simulations of mechanical fuel treatment costs (Skog et al. 2006), and simulation combined with data to calculate reductions in suppression costs from implementing landscape scale fuel treatment projects (Prestemon et al. 2007, Mercer et al. 2007). Decision support systems have also been developed that use optimization to select the spatial arrangement and timing of treatments to best meet ecosystem and economic objectives (Jones et al. 1999, Barrett et al. 2000, Chew et al. 2003). Additional research is needed to develop methods to evaluate the overall costs and benefits from fuel treatments.

3. ECONOMIC ANALYSIS OF ACTIVITIES DURING THE FIRE

Suppression expenditures have dominated discussion of wildland fire management in recent years because of the huge amounts of money being spent to suppress those fires that escape initial attack efforts. Expenditures by the USFS alone have topped $1 billion in four of the last seven years (2000-2006) and suppression expenditures are making up an increasingly larger part of the overall USFS budget. This has brought increased scrutiny of the suppression program and has led to a greater emphasis on cost containment and the need to make economical decisions.

There are two times that suppression activities and expenditures are analyzed. First is when the fire escapes initial attack and a plan for suppression must be developed (currently called the wildland fire situation analysis or WFSA). The second time is when suppression is analyzed after the fire (ex post). The information gained from these ex post examinations can then be applied during the suppression planning process in subsequent years. So, while the WFSA process happens first for any given fire, in the context of the fire program, we first analyze suppression expenditures ex post and then apply the lessons learned to individual fires during the suppression planning process. Therefore, we discuss ex post suppression cost analysis first followed by suppression planning for individual fires.

3.1 Ex Post Suppression Cost Analysis

The ex post analysis of suppression costs can be performed at several different levels of the organization (national, regional, forest level, individual fire level) and can differ in the types of costs being analyzed. The analysis can focus solely on the money spent to suppress the fire or it can attempt a broader focus, by trying to assess the costs and benefits of wildland fire and suppression activities. Due to the magnitude of the money being spent to suppress wildfires as of late, much of the recent work has focused on analyzing suppression expenditures themselves.

In the past, wildfire suppression expenditures were justified as a necessary part of the business of land management. However, recent severe fire seasons and their associated costs have spurred interest in the costs and value of wildland fire suppression programs, and government oversight agencies such as the Office of Management and Budget and the Government Accountability Office (GAO) have responded with increased scrutiny of Federal wildland fire programs.[1]

[1] GAO was formerly known as the General Accounting Office. Examples of reports that address wildland fire programs include Wildfire Suppression: Funding Transfers Cause Project Cancellations and Delays, Strained Relationships, and Management Disruptions (GAO 2004) and Wildland Fire Management:Timely Identification of Long-Term Options and Funding Needs is Critical (GAO 2005) and Wildland fire Mangement: Lack of Clear Goals or a Strategy Hinders Federal Agencies' Efforts to Contain the Costs of Fighting Fires (GAO 2007). These and other reports are available online at www.gao.gov.

In 2003, the USFS, the DOI, and the National Association of State Foresters issued a Large Fire Cost Action Plan recognizing that:

> Our culture and incentive system are not oriented toward reducing the costs of large fires. Currently, the local Agency Line Officer and Incident Commander have three primary objectives: 1) ensure firefighter and public safety, 2) suppress the wildland fire, and 3) respond to community needs (USFS, DOI, and NASF 2003, page 7).

While the Action Plan recommended that cost containment be elevated so that it is commensurate with these other objectives, a more recent USFS conference concluded that cost containment is essential; however, it is not a primary objective of fire management.[2] Specifically, "cost management is a very significant component of meeting fire suppression objectives, but is not an overriding goal in itself." The Action Plan also states that managers should "expend only those funds required for the safe, cost effective suppression of the wildfire incident," a difficult task to accomplish without sufficient knowledge of the cost and benefits of suppression actions.

Cost containment of wildland fire suppression expenditures generally refers to controlling expenditures on wildfires. Figure 15.1 shows federal spending on wildfire suppression by each federal agency from FY 1995–2005 (adjusted for inflation to 2005 dollars). The USFS has been responsible for 73 percent of expenditures on average. All of the Federal agencies have experienced statistically significant

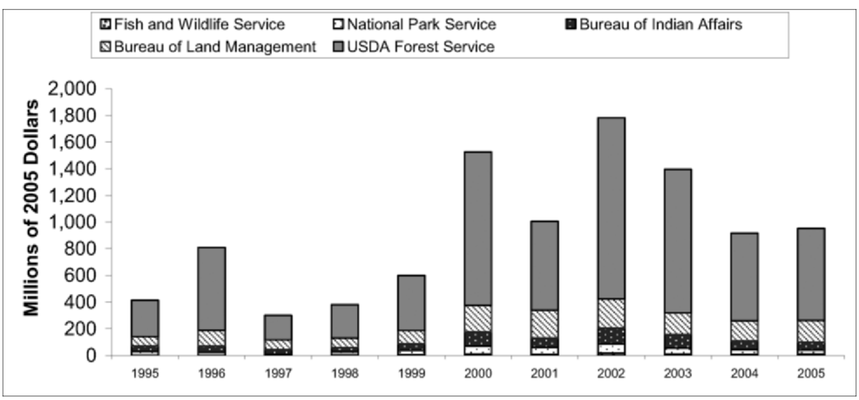

Figure 15.1. Wildland fire suppression expenditures (2005$) by Federal land management agencies, FY 1995-2005.

[2] The Pulaski conference was a weeklong workshop sponsored by USDA Forest Service, Fire and Aviation Management, designed to develop foundational doctrine with regard to wildfire suppression (USDA Forest Service 2005).

upward trends in expenditures. The National Park Service and the Bureau of Land Management experienced annual real growth rates of approximately 10 percent. The Bureau of Indian Affairs was slightly lower, with an annual real growth rate of 8 percent, and USFS suppression expenditures grew at an annual real rate of more than 12 percent. However, the Fish and Wildlife Service experienced an annual growth rate of 61 percent, though in total their share of suppression expenditures was extremely small. This large percentage change is largely due to extremely low expenditures for the Fish & Wildlife Service in FYs 1996 and 1997 rather than substantial increases in later years (the rate of growth would have been 13 percent if those years were excluded).

The Budget Object Classification Code (BOC) system is used by the federal government to record financial transactions when obligations are first incurred. Analysis at the BOC level indicates no substantial change in expenditures by general categories such as personnel versus supplies and services over the past decade (table 15.1). These percentages remained fairly stable during the period from FY 1996-2004 even though suppression expenditures were rising. Expenditures on contracts are often blamed for rising expenditures. However, it does not appear that the percentage of expenditures in this category has shown much change since 1996. Unfortunately, there is insufficient data for an analysis of the effect of contractual services on suppression expenditures. Even at its finest scale, BOC information does not provide sufficient detail for a full analysis of suppression expenditures (data discussion later in the chapter).

A variety of explanations for the increased severity and associated expenditures on wildland fire suppression have been suggested including fuel accumulation due to past fire suppression (Arno and Brown 1991, Arno et al. 2000), a more complex fire fighting environment due to expanding private development within the wildland urban interface (WUI) (Snyder 1999), and severe drought

Table 15.1. Suppression expenditures by category for fiscal years 1996-2004

Fiscal Year	Salaries and benefits	Travel	Contracts	Supplies	Other
	Percent				
1996	34	4	54	9	0
1997	39	2	49	8	2
1998	38	5	44	13	0
1999	29	3	60	7	1
2000	29	4	58	8	1
2001	33	4	54	9	1
2002	28	3	59	9	0
2003	31	3	61	5	0
2004	37	3	56	4	0

caused by long term weather phenomena (Agee 1998, Westerling et al. 2003, Calkin et al. 2005a, Westerling et al. 2006). Numerous fire reviews have tried to assess the reasons for increased suppression expenditures (USDA Forest Service 1995a, 1995b, NAPA 2002, Strategic Issues Panel 2004), citing many of the same reasons for the rise in expenditures. However, concerns remain that Federal suppression resources are not being utilized efficiently, perhaps due to an incentive system that discourages risk taking and thus encourages excessive resource use (for example, chapter 16).

Using regional fire suppression data, Calkin et al. (2005a) provide evidence that recent increases may largely be weather-driven. Using data from FYs 1971-2002, they state that unit expenditures (cost per area burned) have not been increasing, but the number of large fires, the average size of large fires, and the overall area burned have been increasing along with the increase in expenditures. Additionally, they showed that area burned can be modeled successfully using current and past regional drought indices and that area burned and suppression expenditures are highly correlated (r = 0.70). They state that "simply put, suppression expenditures increased and became erratic when acres burned increased and became erratic" (fig. 15.2). Since 1995 the USFS has experienced an annual rate of growth in total suppression expenditures of around 15 percent. However, unit expenditures during that time have not significantly increased (fig. 15.3). Westerling et al.

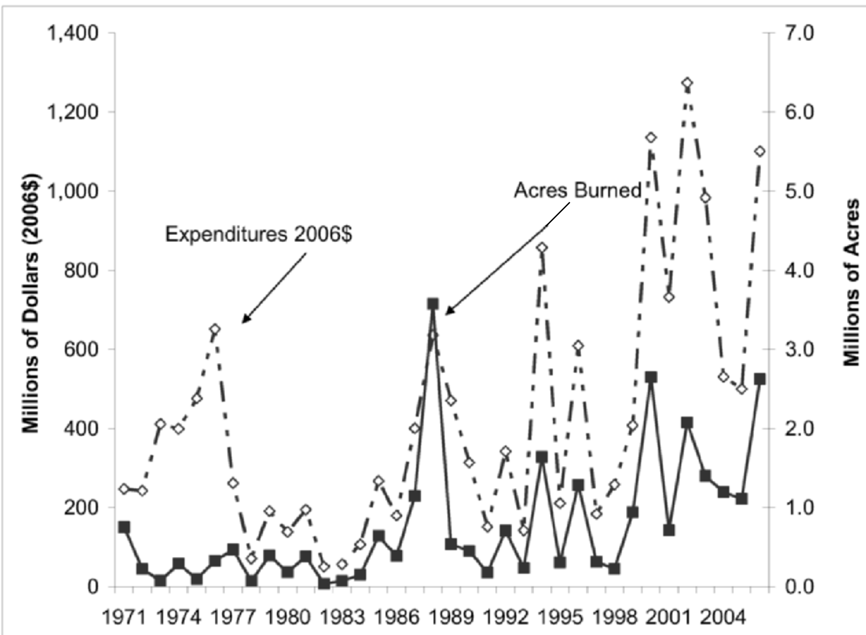

Figure 15.2. Forest Service wildland fire suppression expenditures (2006$) and acres burned for large fires (>= 300 acres), FY 1971-2006.

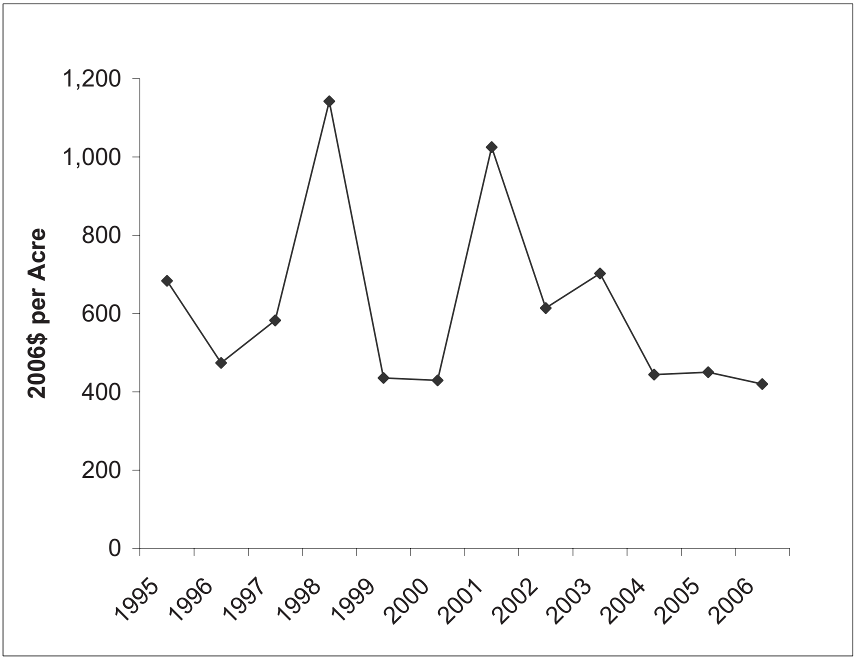

Figure 15.3. Forest Service suppression expenditures per acre burned.

(2006) also provides evidence that the severe fire seasons, and related expenditures, of recent years may be largely climate driven.

A study by Canton-Thompson et al. (2006) focused on how incident management teams (IMTs) make suppression decisions regarding the resources used on the fire and other factors affecting suppression expenditures. Data for this study were collected from 48 in-depth interviews with Incident Management Team (IMT) command and general staff personnel from federal, state, and local land management agencies throughout the country. Some important cost issues that have emerged from preliminary analysis of the interviews include increased risk aversion on the parts of both IMTs and agency administrators, the inability of IMTs to make major cost decisions due to limited decision space defined by the Agency Administrator, the significant increase in rules and regulations in recent years which constrain the IMT's flexibility, external (often politically driven) decisions about what resources to use on a particular fire, contracting-related issues including quality of contracted resources, and substantial increase in technology and associated expenditures.

Several studies have found evidence of the link between increasing values at risk, especially in terms of private property, and suppression expenditures. Gebert et al. (2007) compiled a large dataset of USFS fires in the Western United States (USFS Regions 1 through 6) to estimate a predictive suppression expenditure model and did find that higher home values within 20 miles of a fire ignition increase total fire cost. Among the other variables found to influence cost were variables associated with extreme fire behavior and drought conditions, with increasing fire intensity levels and energy release component values associated with higher costs. Liang et al. (in review) studied USFS fire suppression expenditures for 100 large fires occurring in the Northern Region of the USFS and the influence of 16 potential spatial factors including fire size and shape, private properties, public land attributes, forest and fuel conditions, and geographic settings. They found fire size and private land had a strong effect on suppression expenditures.

Economic analysis is not solely concerned with the expenditures for suppressing wildfires but also with the efficient use of resources to achieve the maximum benefit to society, taking into account both costs and benefits. A key in evaluating the economic efficiency of wildfire suppression is estimating the benefit of suppression efforts, a component of net value change in the C+NVC model. This involves identifying the area that would have burned had suppression activities not occurred. Once the appropriate analysis area for value change estimation is determined, the problem then turns to valuing those resources at risk. In the current wildfire environment, private resource values and public infrastructure are frequently the strategic drivers of suppression decisions both from a values at risk standpoint, and often, more importantly, a political standpoint (NAPA 2002, Canton-Thompson et al. 2006, Gebert et al. 2007, Liang et al. (in review)). Structures, specifically homes at the wildland-urban interface, are among the most obvious values at risk from wildland fire. Threatened structures significantly influence suppression decisions and are potentially the most difficult, dangerous, and expensive resource to protect.

Many of the resource values protected by wildland fire suppression are non-market resources, and monetizing these values can be difficult and controversial. Although non-market valuation techniques such as contingent valuation, travel cost models, and hedonic pricing have identified monetary values for many non-market resource values (Englin et al. 2001, Koteen et al. 2002, Hesseln et. al. 2003 and 2004, Huggett 2003, Donovan et al. 2007), issues specific to individual locations of a given large fire, uncertainty associated with the effect of fire on the resources of interest, and fire induced changes to the forest ecosystem over time make application of these results very challenging. A direct monetary comparison of values protected with suppression expenditures has not been done. When monetized values are not available or appropriate, alternative valuation methods can be used, including cost effectiveness analysis, reference to historic range of variation, analytical hierarchical processes, and other expert opinion based methods. However, each of these methods has associated challenges.

3.2 Suppression Plan for an Individual Fire

Once a fire has started, the first role of economic analysis is to help choose the appropriate management response. Historically, federal land management agencies have drawn a distinct line between (1) suppression responses, where the manager's objective is to contain/control the fire as soon as possible with maximum efficiency and with the highest regard for safety (USDA Forest Service 2005) and (2) management responses referred to as wildland fire use, where the manager's objective is to monitor the fire to ”protect, maintain, and enhance resources and, as nearly as possible, allow it to function in its natural ecological role (paraphrased) (USDA and USDI 2005).[3] One of the reasons for this distinction is that managing a wildfire for resource benefits (wildland fire use) subjects the fire to National Environmental Policy Act (NEPA) requirements which must be completed prior to designating the fire as wildland fire use (FSM 5103.2-6). Suppression fires have no NEPA requirements as they are considered an emergency situation.

The Wildland Fire Use Implementation Procedures Reference Guide (USDA and USDI 2005) restates current policy outlining the separation of suppression and wildland fire use:

> "Only one management objective will be applied to a wildland fire. Wildland fires can either be managed for resource benefits or suppressed…Human-caused wildland fires will be suppressed in every instance and will not be managed for resource benefits…(page 3)

In 1995, the Federal Wildland Fire Management Policy and Program Review was completed and introduced the concept of appropriate management response (AMR) where fires receive "management actions appropriate to conditions of the fire, fuels, weather, and topography to accomplish specific objectives for the area where the fire is burning" (Zimmerman 1999). In the 2001 review of the 1995 Federal Fire Policy (Interagency Federal Wildland Fire Policy Review Working Team 2001) one recommendation was to "base responses to wildland fires on approved fire management plans and land management plans, regardless of ignition source or the location of the ignition", and they advised that barriers to achieving this goal should be eliminated. They also stated that "determination of the appropriate response will include an evaluation of such factors as risks to firefighter and public health and safety, weather, fuel conditions, threats, and values to be protected."

With the rising costs of wildland fire suppression, increasing emphasis is being placed on the use of AMR. It is hoped that the use of less aggressive suppression strategies, where appropriate, will result in suppression expenditure savings (OIG 2007). Though AMR does not represent a change in policy, as it was first introduced in the 1995 Fire Policy (USDA and USDI 1995), there has been some

[3] As of 2006, only 29 percent of FS wilderness areas had approved fire management plans that allowed for the option of wildland fire use somewhere within their boundaries (Carol Miller, Pers. Comm., Aldo Leopold Wilderness Research Institute, January 20, 2004).

confusion surrounding its implementation, and steps are being taken to clarify the existing policy. Several directives and clarifying guidelines were issued in FY 2007 to clear up the confusion surrounding appropriate management response and to give the forest land managers guidance as to the different types of suppression tactics available for both wildland fire use and suppression (USDA Forest Service 2006, USDA and USDI 2007, NRCG 2007). These documents emphasize that even though the distinction is still made between wildland fire use and suppression (with only one of these management objectives allowed on a fire), the tactics used on the fire may not be much different. With both types of fires, except for special circumstances dictated by law, land management plans, or fire management plans, "managers have the options of suppressing a fire, confining a fire with natural barriers, conducting a large scale burnout to contain a fire, or even just monitoring the fire" (USDA Forest Service 2006, page 2).

Two separate processes have been used for evaluating response strategies depending upon whether the fire is deemed suitable for wildland fire use or the fire should be actively suppressed. When deemed a wildland fire use fire, a Wildland Fire Implementation Plan is used to examine the available response strategies. See Wildland Fire Use Implementation Procedures Reference Guide (USDA and USDI 2005) for an explanation of these plans. The role of economics in developing the response strategy is largely subjective. Users are asked to exercise their judgment in evaluating the values at risk. These values include "ecological, social, and economic effects that could be lost or damaged by a fire" and are assessed with a risk scale of low, moderate, or high. Additionally, proximity of the fire to these values is assessed with a scale of distant, moderate, or adjacent.

If a fire is not extinguished in the first operational period (usually no more than 24 hours), either through the use of initial or extended attack forces, and a larger and more complex firefighting organization is deemed necessary, the fire is declared "escaped" and a wildland fire situation analysis (WFSA) begins. Through this process, the land management agency evaluates alternative suppression strategies defined by different goals and objectives, suppression costs, impacts on the land management base, and values at risk (MacGregor and Haynes 2005). Although the analysis provides a mechanism for valuing resources at risk, these valuations are subjective and have only taken into account resource values on public lands. Though land managers are aware that private resource values play a large role in fire management decisions, "there is no clear direction pertaining to the preparation of WFSA's and the inclusion of private land and community values (structures and infrastructure) as an element of benefit cost relating to suppression costs when WUI or occluded communities are at risk from a fire originating on National Forest land."[4]

In an effort to alleviate some of the aforementioned problems, the Wildland Fire Decision Support System (WFDSS: http://wfdss.nwcg.gov) is currently in

[3] Fire and Aviation Management briefing paper dated April 23, 2003, available from the authors.

development and beta testing. Use of the new system has become a required process for fires deemed to be of national significance during the 2007 fire season (per a letter from Chief Abigail Kimbell, June 11, 2007). For fires expected to cost more than $5 million but less than $10 million, the use of the new WFDSS system is recommended. For fires expected to cost more than $10 million, its use is required. In 2006 two promising components for WFDSS were prototyped; a new fire spread probability prediction application (FSPro–Fire Spread Probability) and an application that uses FSPro outputs to assess values at risk (RAVAR–Rapid Assessment of Values at Risk).

The Fire Spread Probability model, FSPro, is a new fire modeling tool that calculates the probability of fire spread from a current fire perimeter or ignition point for a specified time period assuming no suppression. The model simulates the 2-D growth of the fire across the landscape (fuels & topography) using a computationally efficient form of the *FARSITE* calculations (see Finney 1998 for a description of the *FARSITE* program). FSPro differs from *FARSITE* in that it simulates fire growth for thousands of possible weather scenarios using the latest recorded perimeter (or point). Different weather possibilities are developed statistically using the data from the weather station (fuel moisture, wind speed and direction). FSPro can assist managers to prioritize fire fighting resources based on probabilities of fire spread by assessing a fire's growth potential, informing appropriate strategy and tactics development and allocation of resources

RAVAR identifies the primary resource values threatened by ongoing large fire events. RAVAR is directly integrated with the new FSPro model to identify the likelihood of different resources being affected by an ongoing fire event. RAVAR spatially maps the location of threatened structures, public infrastructure, and high priority natural and cultural resources. RAVAR was designed to assist agency administrators, incident managers, and fire planners develop wildfire suppression strategies by rapidly identifying and quantifying the significant resource values and relative threat to those resources from an ongoing fire event. Additionally, RAVAR can help support development of the Wildland Fire Situation Analysis (WFSA) and could be used to help prioritize large fire needs during periods when area command is convened.

Developing estimates of suppression expenditures, the cost portion of the C+NVC model, is another critical piece of the economic analysis of the fire stage. Estimated expenditures on fire suppression in the WFSA system were based upon cost estimates using either historical cost per acre or by selecting the fire fighting resources to be used and arriving at an aggregate cost for these resources. Both methods are flawed. Per acre cost estimates are often based upon a small number of fires, whose characteristics might vary dramatically from the fire in question. Aggregating fire suppression resources does not take into account the large overhead costs often associated with these larger fires. Suppression cost functions estimated ex-post could be used by the analyst during the fire to forecast expenditures.

Donovan et al. (2004) used regression analysis to identify variables affecting suppression expenditures for 58 fires that occurred in Oregon and Washington in 2002. The only significant variables were fire size and terrain with measures of housing density, a focus of this study, not showing up as a significant predictor of costs. Steele and Stier (1998) developed a series of regression equations to estimate suppression costs for Wisconsin wildfires managed by the State Department of Natural Resources. Significant variables included final fire size and burning index. González-Cabán (1984) estimated suppression expenditures based on the number and type of the different resources used on the fire, and they found considerable variation among fires and regions of the country. With the exception of Steele and Stier (1998) all of these studies indicated they were hindered by the lack of reliable data.

The Stratified Cost Index (SCI) was developed in FY 2006 and has been adopted as a performance measure and incorporated into the WFDSS system. Per the Chief's letter of June 11, 2007, its use is required on all fires estimated to cost more then $5 million. The Index determines average suppression costs based on fire characteristics such as fuel types, fire intensity, topography, region, and values at risk. After the characteristics of a current fire are entered into the WFDSS system, the user is given a range of possible expenditures for fires with similar characteristics and shown where the cost of their fire falls within that range. If the cost of the current fire falls within one of the upper ranges, it is likely that the fire will be reviewed post-season (Gebert et al. 2007).

4. ECONOMIC ANALYSIS OF ACTIVITIES AFTER THE FIRE

Activities that occur after the fire include rehabilitation and restoration, performance measures, and impact studies. Impact studies are specifically addressed in chapter 8; thus, they are not further addressed here.

4.1 Rehabilitation and Restoration

Following containment of a large fire, immediate rehabilitation and restoration projects may be required to prevent additional catastrophic resource damage from floods, erosion and landslides. Burned Area Emergency Response (BAER) teams are sent to assess and initiate the required rehabilitation work. Working under tight timelines, BAER teams are required to demonstrate that the value of resources to be protected by emergency response treatments exceeds the costs of the treatments. However, several limitations compromise effective calculation of values-at-risk including: (1) inadequate scientific knowledge and data to support calculation of the market values, (2) methods to account for non-market resources are controversial and difficult to apply, and (3) the extent of the area that should be included as at-risk is unknown. Currently, the final calculations developed within BAER assessments are best described as estimates based upon professional judgment (Robichaud et al. 2000).

Calkin et al. (2006) note that studies addressing how fire and post-fire erosion affect values at risk are limited and that benefit transfer issues (i.e., the transferability of study results from one location or resource value to another) reduce the applicability of these studies. They have proposed new methods to assess the economic value of treatments designed to protect non-market values at risk. These recommendations were developed through a study that included direct field observation, surveys with BAER personnel, a literature review of non-market resources typically encountered, and recognition of the challenges of the BAER analysis environment. They concluded that an implied minimum value approach may be most appropriate for valuing the nonmarket values associated with the BAER environment. Implied minimum value is defined as valuation based on the amount that is spent to avoid a negative outcome and the amount of risk reduction received for the money spent.

4.2 Performance Measures

Alternative performance metrics are under development in response to a current push towards evaluating the performance of government programs, including wildland fire suppression. Conference Report on HR 4818, Consolidated Appropriations Act, 2005 required the Secretaries of Interior and Agriculture to promptly establish appropriate performance measures for wildland fire suppression and develop a report on interagency performance measures planned for implementation in fiscal year 2006. This report was completed and highlighted the work being done for the WFDSS system and the SCI as potential performance measures. This was in line with the report by the Strategic Issues Panel on Large Fire Costs (2004), which recommended that federal agencies responsible for wildland fire should "Develop and use a benefit cost measure as the core measure of suppression cost effectiveness." In addition, the Forest Service Strategic Plan (USDA Forest Service 2004) outlines a performance measure, under the goal of "reducing the risk of catastrophic wildfire", of the "percent of large fires in which the values of resources protected exceeds the cost of suppression" with a target of 55 percent in FY 2008.

An approach that follows the benefit-cost logic of the C+NVC model would relate suppression expenditures to the values of the resources protected, not the area that did burn. Wildland fire suppression efforts could be seen as economically justified if the values protected are worth at least as much as the amount of money spent to suppress the fires. Therefore, an outcome-based performance measure should compare suppression expenditures to estimated resource value change on unburned areas that would have burned in the absence of suppression activities.

Calkin et al. (2005b) used break-even analysis to compare the value of structures threatened, identified from cadastre data, against suppression expenditures for two fires during the 2003 fire season in Montana. The authors examined an interface fire adjacent to the city of Missoula (the Black Mountain Fire) and a more rural fire that began in a USFS designated wilderness area (Crazy Horse Fire). The authors found that suppression expenditures on the Black Mountain Fire were economically justified by the value of structures protected if these efforts reduced the potential fire perimeter a modest amount (approximately 50 percent). However, the potential fire perimeter for the Crazy Horse Fire would have had to be considerably larger (greater than 180 percent) than the actual fire perimeter for the associated suppression expenditures to be economically justified from a structures at risk standpoint. The vegetation types and fire regime condition class associated with the Crazy Horse Fire suggest that the impact of the fire on non-market values at risk may not have been severe and in some cases may have provided ecological benefits. However, fire managers familiar with this fire suggested that the intermix of private and state timberlands adjacent to federal lands may have triggered an aggressive and, therefore, expensive suppression response.

Another performance measure currently being used by the USFS and in the development stage for the DOI is the stratified cost index (SCI) discussed in section 3.2. SCI assesses a variety of factors that influence suppression expenditures, including energy release component. Regression equations were developed to estimate fire specific expenditures given fire characteristics such as size, the fire environment, values at risk, and location (Gebert et al. 2007). When used as a performance measure, the SCI regression equations are used to calculate the expected suppression cost of a large fire (>= 300 acres) given its characteristics. The expected cost is then compared to actual suppression expenditures and a list of outliers (fires where actual cost is one or two standard deviations above expected cost) is provided to USFS Fire and Aviation Management. This effort will result in a common metric to normalize large fire suppression cost which can be used for reviews, evaluations, planning, and reporting.

The development of new performance metrics for the wildland fire program reflect the need for information on what the increasing levels of spending are buying the American taxpayer. With the wide array of possible reasons for escalating expenditures, economic analysis becomes very complex, and the answer as to whether or not these rising expenditures are the result of uncontrollable events and increased values at risk or due to overspending is difficult to assess.

5. DATA CONSTRAINTS

Any analysis of the economic efficiency of the fire program requires accurate expenditure data. Historically, this data has been difficult to obtain. Large fires most often involve multiple agencies, with multiple systems for recording both expenditures and fire characteristics, and little or no linkage between systems, making identification of actual costs difficult. Changes in record keeping over time make time series analysis difficult. Fire expenditure data is plagued by accounting practices that were not designed to aid systematic investigations of suppression expenditures. Problems are associated with specific fire expenditures as well as expenditures at national or regional levels.

The Budget Object Classification Code (BOC) system is used by the federal government to record financial transactions when obligations are first incurred. Unfortunately, the current BOC system makes it nearly impossible to obtain useful information on specific high cost items, such as contractual services. In FY 2002, $412 million of the $1.19 billion fire suppression expenditures were categorized as "Contractual Services–Other". Determining exactly what this money was spent on requires accessing the detailed transaction records. The "Contractual Services–Other" category includes such items as caterers, showers, tents, toilets, and can even include aviation contracts, if someone miscodes these to the more general category (Schuster et al. 1997, page 16). More specific BOC categories could provide valuable information in discerning the effectiveness of cost containment measures.

Another problem associated with tracking costs at a regional level is the mismatch between the tracking of expenditures versus fire activity. Expenditures are tracked according to the administrative unit spending the money, not according to the geographic area were the activity occurred. For example, if the USFS Northern Region sends crews to fight a fire in the Southern Region (as often happens early in the season or when suppression resources are scarce), in the financial system these expenditures are ascribed to the Northern Region (the resource's home unit), not the Southern Region. However, the fire activity is associated with the Southern Region. This is not an issue when dealing with aggregate national expenditures or even fire-specific expenditures (which can be traced to the region where the fire activity occurred through the accounting code), but it can cause problems for regional analyses. To assess the extent of this disconnection, we analyzed a subset of fires for which expenditures could be obtained both by and in region, which consisted of collecting accounting code-level expenditures from the financial system by region. The accounting code, in many instances, can be used to discern where the fire occurred (in-region expenditures) since the accounting code for a specific fire contains a region designator. The financial system also contains a field for the home unit of the resources charging to the fire (the by-region designator). An analysis of fires from FY 1995–2006 shows that the percentage of regional expenditures spent on fire activity in that region ranged from a high of 95 percent for Region 5 to a low of 11 percent for Region 10 (table 15.2). This process is being changed in FY 2007, with fire expenditures being tracked in the financial system according to the region where the fire started.

Other problems occur with fire-specific data. Until recently, reliable data on fire suppression expenditures on individual large fires has not been widely, or easily, available for several reasons. First, prior to FY 2004, obtaining actual individual fire suppression expenditures on an interagency basis was difficult

Table 15.2. Percent distribution of fire organization expenditures by location of fire, FYs 1995-2006

The In-By Problem	Fire Location (in)									
	Region									
1995-2006	1	2	3	4	5	6	8	9	10	Total
	Percent									
Regional (by)										
Region 1	91.5	0.9	1.4	2.0	1.3	0.8	1.7	0.2	0.2	100
Region 2	2.0	84.6	1.3	6.3	1.8	1.4	2.4	0.2	0.1	100
Region 3	2.1	1.2	82.8	5.2	3.4	1.9	3.3	0.1	0.0	100
Region 4	1.7	1.2	1.2	91.4	2.4	0.7	1.1	0.1	0.2	100
Region 5	1.5	0.9	2.3	3.7	94.8	1.3	-4.6	0.0	0.2	100
Region 6	1.2	0.7	0.9	1.8	1.6	92.3	1.1	0.0	0.4	100
Region 8	2.7	0.8	0.8	3.8	2.1	3.1	86.4	0.2	0.0	100
Region 9	9.0	2.4	1.9	11.2	4.8	6.7	6.8	57.0	0.1	100
Region 10	28.7	4.5	2.2	31.6	8.8	8.8	4.3	0.2	10.9	100
Region 13+	5.6	7.5	6.6	43.8	13.2	14.9	7.6	0.5	0.3	100

Note: Region 13 refers to non-region specific expenditures, such as national contracts, and the National Interagency Fire Center in Boise, ID.

due to different accounting systems and processes for tracking suppression expenditures. As of FY 2004, the federal land management agencies began using “FireCode” as a response to the House Appropriations Subcommittee for National Fire Plan Agencies’ directive to “develop a method to standardize fire incident financial coding for fire suppression and subsequent emergency stabilization…. to provide the capability to effectively track and compile the full cost of a multi-jurisdictional fire suppression effort” (NIFC 2004). Second, matching actual fire-specific expenditures with fire characteristic information has been problematic due to the lack of a common field between (and among) fire characteristic databases and the financial systems used to track fire-specific expenditures. However, in FY 2007, the FireCode was made a required field in the Forest Service’s fire occurrence database (NIFMID—National Interagency Fire Management Integrated Database), which should alleviate this problem in the future. Another problem associated with tracking suppression expenditures on individual fires, at least for the USFS, is that smaller fires (< 300 acres) are lumped into one FireCode for a region or forest. This makes it impossible to ascertain expenditures on individual small fires or analyze the costs and characteristics of small fires, as opposed to large fires.

Finally, a great deal of information is collected on large wildland fires, but there is not a single data repository or a common fire identifier among fire information systems. For instance, the USFS’s fire occurrence database, NIFMID, contains fire occurrence information for fires reported by the USFS. The ICS-209 Incident

Status Summary is used for reporting information on "incidents of significance" (USDA Forest Service 2004b) and contains additional information on individual fires that differs from that found in NIFMID, such as resources used, structures threatened or destroyed, and so forth. However, there is no common identifier at this time between the two systems, so it is difficult to compile information from the two systems. Additionally, the DOI has its own fire occurrence reporting system, actually two systems, one for the Fish and Wildlife Service (Fire Management Information System—FMIS) and another for the other three agencies (Wildland Fire Management Information—WFMI). Both of these systems collect slightly different information on individual fires.

Steps are being taken to make access to fire data easier and more reliable. An interagency fire occurrence database (Fire Occurrence Reporting System—FORS) is currently being developed by the USFS and DOI. The vision for this system is that it will contain a set of critical and common fire occurrence data elements to be used by the interagency fire community. This will allow for easier access to fire information across the federal agencies, but at this point, is not expected to include non-federal wildland fires. Additionally, the FAMWEB (Fire and Aviation Management Web Applications) web site will soon contain a data warehouse for accessing historical fire, weather, and aviation databases through a variety of query and report options, and it will also contain some information related to non-federal wildland fires. The site will also include a web-based geospatial tool to enable users to view interactive maps of fire-related information. However, at this point it is unclear whether these systems will attempt to improve the quality of the data available or will just provide easier access to current systems.

Currently, much of the wildland fire data available is non-spatial. However, the technology and information to collect data more conducive to spatial analysis has grown considerably in recent years, and fire data that could be analyzed from a spatial perspective would likely help provide answers that non-spatial data cannot. Examples of such information would be location and type of fire line, ownership patterns, property and resource values adjacent to the fire perimeter, and location of past fuel treatment areas. Additionally the production of spatial data describing vegetation, wildland fuel, and fire regimes at a 30-meter grid resolution by the LANDFIRE project (Landscape Fire and Resource Management Planning Tools Project) will provide consistent data across the United States that can be used to help analyze fire management activities (Rollins et al. 2006).

6. DIRECTIONS FOR FUTURE RESEARCH

Though research to date has made substantial progress in many areas related to the economics of wildland fire management, there are still many unanswered questions. For example, little is currently known about the relative effectiveness of alternative suppression resources (e.g., crews versus aircraft). If aircraft are deemed too expensive, what are the consequences of using other less expensive, but perhaps less effective, resources? The encroachment of populations into the wildland-urban interface is often blamed for the rising costs of suppression, but more research is needed to examine this issue and its cost implications. Statements have been made that 50 to 95 percent of the money to suppress fires is spent protecting structures in the wildland urban interface (USDA Forest Service 1995b, OIG 2006), but no research studies have been done to assess the validity of these statements. Policy, both formal and informal, and social/political pressures can affect the suppression strategies and resources used on a fire, but how much of a factor this plays is largely unknown.

Fire suppression is, however, only one piece of the fire program. The effectiveness of fuel treatments on affecting fire behavior and suppression expenditures is largely unknown. The theoretical relationship of presuppression and suppression is largely untested. In reality, does more spending on presuppression lead to less spending on suppression? How do suppression expenditures relate to expenditures on emergency rehabilitation and stabilization? Does more wildland fire use mean less money spent on suppression and/or fuel treatments? Further research into these issues is needed to create a wildland fire program that will reduce the risk from catastrophic wildland fire in a cost efficient and effective manner.

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CHAPTER 16

INCENTIVES AND WILDFIRE MANAGEMENT IN THE UNITED STATES

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1. INTRODUCTION

A recent series of severe fire seasons in the United States has contributed to sharply rising wildfire suppression costs. These increasing costs have caught the attention of policy makers, but so far the responses have not focused clearly on the incentive structures that allow or encourage rising costs (National Academy of Public Administration 2002). We analyze the problem of rising suppression costs by examining the incentive structures faced by fire managers. Specifically, we examine the influence of wildfire suppression funding mechanisms on managers' behavior. The rationale for this approach is that fire managers have a good deal of control over suppression costs; they, like other people, respond to incentives; and thus it is through a change in those incentives that fire managers are most likely to change their behavior.

To understand how the current incentive structure for fire managers developed, we begin with a brief history of wildfire management in the United States and an introduction to wildfire suppression budgeting. We then examine how the government-wide movement for increased accountability has led to the use of performance measures in the wildland fire program, and we explore how these have shaped the incentive structures faced by fire managers. Finally, we suggest an alternative incentive structure created through changes in wildfire suppression budgeting.

2. HISTORY

In the late 19th Century, a series of severe fire seasons in the Northeast and the lake states, plus the failure of local efforts to adequately respond to these events, contributed to a call for the federal government to manage wildfire suppression on public land (Pyne 2001). This responsibility initially fell to the Department of the Interior, which received help from the U.S. Army. However, in 1905, Theodore Roosevelt transferred responsibility for wildfire suppression to the Bureau of Forestry, which soon became the U.S. Forest Service headed by his friend Gifford Pinchot. The main mission of the new agency was conservation and the provision of

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a secure timber supply. One of the main dangers posed by wildfire was the disruption of that supply. In addition, wildfire threatened community water supplies.

Although there was general agreement on the values at risk from wildfire, there was considerable debate about the best way to manage the risk. One approach, often referred to as light burning, advocated fire use to achieve a variety of objectives including hazardous fuels reduction, land conversion for agriculture, and the improvement of game habitat. Light burning was particularly prevalent in the Southeast. In contrast, some, including Gifford Pinchot, advocated a policy of fire control, which emphasized fire suppression and had no place for fire use. This debate over the role of fire on public lands might have continued for longer or resulted in a different outcome had it not been for the 1910 fire season, during which 5 million acres of national forest land burned and 78 people were killed (Pyne 2001). This extreme fire season caused the Forest Service to adopt a policy of strict fire protection and influenced a generation of foresters.

The Forest Service's commitment to fire protection was intended to protect timber and community water supplies. Because of the huge amount of land under its control, the Forest Service had to be selective about the areas it protected. The criterion used to determine which areas to protect, and how much to spend protecting them, was therefore based on timber and water values. This economic principle, that suppression expenditures should be commensurate with values at risk—first formally presented by Sparhawk (1925)—became known as the "least-cost-plus-loss" model. Simply put, the most efficient level of fire management expenditure is the one that minimizes the sum of all fire-related costs and damages.

The late 1920s and early 30s saw more extreme fire seasons (Gorte and Gorte 1979), the losses from which led fire managers to the conclusion that they had not been sufficiently aggressive in fighting fires. They reasoned that because the values at risk from wildfire were so high, a more aggressive fire suppression effort, with a focus on strong initial attack, would be consistent with the least-cost-plus-loss model (Hornby 1936).

This shift in attitudes may not have been sufficient to fundamentally alter wildfire management, had it not coincided with the Great Depression and Roosevelt's subsequent New Deal. The New Deal had two profound impacts on wildfire management. First, the Forest Service acquired significant new land holdings. Second, the Civilian Conservation Corp (CCC) provided a huge increase in manpower available for wildfire suppression. This expansion in manpower allowed the Forest Service to extend fire protection to previously unprotected and newly acquired land. However, much of this land had little if any market value, as it was often abandoned farmland or cutover forestland. Therefore, if the Forest Service was to make use of the influx of manpower provided by the CCC, it would explicitly have to set aside the economic principal of protecting land commensurate with the values at risk. This example of the resource availability tail wagging the policy dog is succinctly summarized by Pyne et al. (1996, p. 248): "…the means at hand were often so powerful as to dictate to some extent the ends to which they might be applied".

This change of policy was codified in 1935 by the 10 AM policy (Gorte and Gorte 1979, p. 2):

> *"The approved protection policy of the National Forests calls for fast, energetic, and thorough suppression of all fires in all locations, during possibly dangerous fire weather. When immediate control is not thus attained, the policy calls for prompt calculating of the problems of the existing situation and probabilities of spread, and organizing to control every such fire within the first work period. Failing in this effort, the attack each succeeding day will be planned and executed with the aim, without reservation, of obtaining control before ten o'clock the next morning."*

This new policy of aggressive suppression, which mentions neither suppression costs nor resources at risk, was embodied in 1944 by the successful Smokey the Bear public education campaign. A more emotive example of the prevailing attitudes to wildfire was provided by the death of Bambi's mother in 1943. Interestingly, Walt Disney intended Bambi to be an anti-hunting film: careless hunters started the fire. However, the potency of the fire imagery overwhelmed the original message.

The period following the Second World War provided a further example of resource availability driving wildfire policy and practices. The Forest Service received numerous war-surplus vehicles and aircraft under the federal excess equipment program and was able to increase its use of fire engines and bulldozers. In 1955 converted aircraft were used to drop fire retardant for the first time (Pyne 2001, Andrews et al. 1996). As with the earlier use of the CCC, this increased use of vehicles and aircraft was driven by resource availability, not by any analysis showing that these increased expenditures would result in a commensurate reduction in resource damages. Aggressive suppression of wildfire was also consistent with the cold war social contract: people expected the government to protect them from harm.

Not until the 1960s did the Forest Service waver from its policy of aggressive wildfire suppression. As reflected by the passage of the Multiple-Use Sustained-Yield Act (1960), the Wilderness Act (1964), and the National Environmental Policy Act (1970), attitudes concerning public lands management had begun to shift (Dale et al. 2005). These changes in public attitudes may or may not have been sufficient to change Forest Service suppression policies. However, the Forest Service was also facing scrutiny for a more prosaic reason—decades of increasing suppression expenditures had not resulted in a decrease in resource damages. The inability of the agency to demonstrate a sufficient return on its investment in fire suppression resulted in a series of policy changes in the 1970s (Gorte and Gorte 1979).

In 1971 the 10 AM policy was amended, the new goal being to contain all fires before they reached 10 acres, and then the entire policy was scrapped in 1978. Also in 1978, Congress eliminated emergency funding for pre-suppression. Although the agency still relied on emergency funds to pay for large fire suppression, the new protocol required the Forest Service instead to conduct a cost-benefit analysis on all future presuppression budget requests. This led to the 1979 development of the National Fire Management Analysis System (NFMAS), a computerized fire planning and budgeting tool. As a further incentive, the Forest Service's budget was reduced by 25 percent until it could more rigorously support its budget requests. Other public land management agencies either adopted all or part of NFMAS (Bureau of Land Management and the Bureau of Indian Affairs) or developed their own tools (National Park Service and the Fish and Wildlife Service). NFMAS was the first widely adopted computerized fire management tool (Donovan et al. 1999).

The realization that not all suppression expenditures could be economically justified, along with an increasing awareness of the ecological importance of wildfire, led the Forest Service to adopt the Wilderness Prescribed Natural Fire Program in 1972 (Dale et al. 2005). Under the program some wildfires in wilderness areas were allowed to burn. In 1968 the National Park Service recognized the natural role of fire, and adopted a wildfire use program beginning in Sequoia Kings Canyon National Park. Since then, several high profile examples of prescribed or wildfires being managed for resource benefit have escaped management control and become destructive wildfires (e.g., Yellowstone in 1988 and Los Alamos in 2000). These well-publicized incidents have tempered enthusiasm for wildfire use both within the agency and among the public at large.

The success of decades of fire suppression has deprived fire-dependent forests of their natural fire cycle and has led to an accumulation of fuels in many locations (Arno and Brown 1991). Furthermore, the country has seen a dramatic increase in the number of houses and other structures being built in the forest, expanding the extent of the wildland-urban interface (NAPA 2002). Both of these stresses have tended to make fires more difficult and expensive to control. And recently, a severe drought in the Western United States exacerbated the situation. In 2000, total federal wildfire suppression expenditures exceeded $1 billion for the first time, and they have done so twice since (table 16.1).

In recent years appropriated dollars for fire suppression have fallen far short of total suppression expenditures. In addition, emergency appropriations, which take place after final appropriations bills have been released, often failed to make up the shortfall. As a result, agencies have been forced to borrow money from other programs to fund their suppression activities. Although attempts are made to reimburse programs that have had funds transferred, these repayments are rarely in full (chapter 17). The disruption and uncertainty associated with suppression funding transfers have compounded inefficiencies and led to numerous cancelled projects.[1]

[1] The Government Accountability Office, analyzing this trend, concluded, "Despite Forest Service and Interior efforts to minimize the effects on programs, transferring funds caused numerous project delays and cancellations, strained relationships with state and local agency partners, and disrupted program management efforts" (GAO-04-612. *Wildfire Suppression: Funding Transfers Cause Project Cancellations and Delays, Strained Relationships, and Management Disruptions,* p. 3).

Table 16.1. Wildfire suppression expenditures by land management agency, fiscal years 1994 to 2004, in thousands of nominal dollars

Fiscal Year	Bureau of Land Management	Bureau of Indian Affairs	Fish and Wildlife Service	National Park Service	USDA Forest Service	Total
	thousand dollars					
2004	147,165	63,452	7,979	34,052	637,585	890,233
2003	151,894	96,633	9,554	44,557	1,023,500	1,326,138
2002	204,666	109,035	15,245	66,094	1,266,274	1,661,314
2001	192,115	63,200	7,160	48,092	607,233	917,800
2000	180,567	93,042	9,417	53,341	1,026,000	1,362,367
1999	85,724	42,183	4,500	30,061	361,000	523,468
1998	63,177	27,366	3,800	19,183	215,000	328,526
1997	62,470	30,916	1,957[a]	6,844	155,768	257,955[a]
1996	96,854	40,779	2,577[a]	19,832	521,700	681,762[a]
1995	56,600	36,219	1,675	21,256	224,300	340,050
1994	98,417	49,202	3,281	16,362	678,000	845,262

Source: National Interagency Fire Center (2005).

[a] Corrected, as provided by special request, from the Fish and Wildlife Service, December 21, 2007.

The wildfire situation—accumulating fuels, rising costs, and budget disruption—suggests that significant changes in the management of wildfire suppression are warranted. Rising federal budget deficits and shrinking discretionary spending make the need for reform more urgent. Before we examine proposed changes, we review the current funding structure.

3. FUNDING WILDFIRE SUPPRESSION

Wildfires can have both negative and positive effects. Negative effects, or damages, may include loss of timber, damage to structures, loss of tourism revenue, and temporary reduction in water and air quality. Positive effects, or benefits, may include nutrient cycling, enhancement of the long-run success of native fire-adapted trees and plants, and a reduction in fuel loads. A wildfire that reduces fuel loads reduces the severity of future wildfires,[2] thereby reducing future wildfire-related damages and suppression costs. Conversely, wildfire suppression allows fuel loads to grow, thereby increasing the sum of future wildfire damages, suppression costs, or fuel treatment expenditures. Thus, an increase in the level of suppression leads to a decrease in *both* the damages and the benefits of wildfire.

The relation between fire fighting cost (C, the sum of presuppression cost and suppression cost), fire damage, and fire benefits can be represented graphically as in figure 16.1. In the figure, the value of all wildfire damages minus the value of all wildfire benefits is shown as the net value change (NVC) (Donovan and Rideout 2003, chapter 18). Holding presuppression cost fixed at \$X, the optimal amount of suppression minimizes the sum of C and NVC (C+NVC) at level S*.

This minimization problem is represented mathematically as:

$$MIN: \quad C + NVC = W^p P^e + W^s S^e + NVC(P^e, S^e)\,, \qquad (16.1)$$

where P^e denotes presuppression effort, S^e denotes suppression effort, W^p denotes the wage of presuppression, and W^s denotes the wage of suppression. Differentiating with respect to P^e and then S^e gives the following first-order conditions:

$$\frac{\partial(C + NVC)}{\partial P^e} = W^p + \frac{\partial NVC}{\partial P^e} = 0\,, \qquad (16.2)$$

$$\frac{\partial(C + NVC)}{\partial S^e} = W^s + \frac{\partial NVC}{\partial S^e} = 0\,. \qquad (16.3)$$

[2] Wildfires can increase the severity of future wildfires in the short run, if the wildfire causes significant mortality and an increase in fuel loads. However, in the long run, regular wildfires will generally reduce wildfire severity.

Rearranging the terms yields:

$$\frac{\partial NVC}{\partial P^{e}} = -W^{p}, \tag{16.4}$$

$$\frac{\partial NVC}{\partial S^{e}} = -W^{s}. \tag{16.5}$$

Of course, figure 16.1 and equations (16.1)–(16.5) are an abstraction. Fire managers face uncertainty about fire behavior, suppression effectiveness, and resource damages, and therefore must base their decisions on less information than is implicitly contained in figure 16.1 or equations (16.1)–(16.5). However, the model does provide a rule of thumb: an additional dollar should only be spent on presuppression or suppression if it averts at least one dollar of NVC.

The Forest Service funds presuppression and suppression efforts in different ways. Until approximately 2005 presuppression budgets were developed using NFMAS, a simulation model that allows users to compare the effect of alternative suppression strategies on historical wildfires. Since that time, the Forest

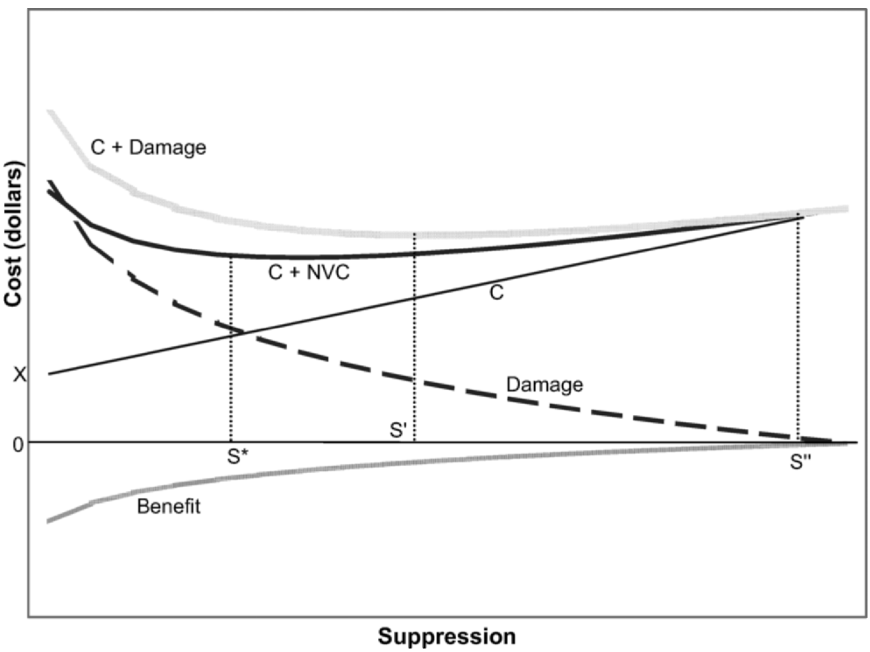

Figure 16.1. A comparison of optimal suppression expenditure and suppression expenditure under current incentive structure (C=suppression plus presuppression cost, X = fixed presuppression cost. Benefits are negative relative to costs).

Service and other land management agencies have begun the process of developing and implementing a new interagency wildfire planning and budgeting tool called the Fire Program Analysis system (FPA). Under NFMAS or the FPA System, presuppression budgets are set early in the year and are generally spent regardless of the number of fires that ultimately occur. The suppression budget, however, begins with an initial appropriation that in a mild fire year may not be totally spent and that in a severe fire year can be supplemented with emergency suppression funds. The Forest Service bases its suppression budget request on a 10-year moving average of total suppression expenditures (chapter 17).

Most wildfires are contained by suppression efforts within the first 12-hour burning period (often referred to as initial attack). Those that are not are considered "escaped fires". Although few fires escape, those that do account for the great majority of burned acres and suppression expenditures (USDA 2003b).

The suppression of large escaped wildfires is undertaken jointly by local land managers and incident command teams. Incident command teams assume responsibility for tactical wildfire suppression decisions, although local land managers provide overall strategic guidance. To determine the appropriate suppression strategy, local land managers are required to perform a wildland fire situation analysis (WFSA). A WFSA requires a manager to consider different suppression strategies, associated costs and damages, probability of success, and the compatibility of these strategies with established land management objectives. For example, in a situation where significant volumes of commercial timber are at risk and the weather forecast predicts hot, dry, windy weather, a manager may recommend that the incoming incident command team use an aggressive suppression strategy. However, if a wildfire does not significantly threaten resources of particular management concern or the weather forecast is favorable, a less aggressive strategy may be recommended. Anecdotal evidence suggests that managers may sometimes use WFSA more as a decision justification tool. That is, managers decide in advance what strategy they wish to employ and then use the WFSA process to justify this decision (Donovan and Noordijk 2005).

A WFSA provides the incoming incident command team with strategic guidance and a non-binding estimate of suppression cost, which can be reassessed if fire conditions change. Notably, when preparing a WFSA, managers are directed not to consider the potential beneficial effects of wildfire. Incident commanders also are directed not to consider beneficial fire effects when planning or executing suppression activities. But even if land managers and incident commanders were free to consider the beneficial effects of wildfire, it is unlikely that they would accord them sufficient weight. Wildfire damages are immediate, and both land managers and incident commanders face intense pressure to minimize those damages. In contrast, the benefits of wildfire are only partially understood, nearly impossible to quantify, and occur in the future.

Disregard of the beneficial effects of wildfire creates an incentive to increase suppression expenditures beyond the efficient level (S*) shown in figure 16.1. When the beneficial effects of wildfire are ignored, the optimal level of suppression becomes S'. The magnitude of the difference between S* and S' will of course depend on the functional relationship between wildfire damages and benefits and suppression expenditures.

Funding wildfire suppression with an emergency suppression budget provides fire managers with an additional incentive to over-utilize suppression resources, as the opportunity cost to fire managers of suppression expenditures is zero. If fire managers were to forgo some increment of suppression spending, the savings would not remain within the fire or fuels management budget. Therefore, unless suppression resources are simply unavailable, fire managers may continue to spend on suppression as long as their efforts decrease damage by even a small increment. If all needed resources are available, suppression expenditures may reach S" in figure 16.1: the point where all damages are theoretically averted.

In summary, the current Forest Service mechanism for funding wildfire suppression has two related problems. First, the benefits of wildfire are ignored. Second, the opportunity costs of wildfire suppression expenditures are not fully considered. Both problems encourage fire managers to use inefficiently high levels of suppression expenditure. In essence, the budget (B) that fire managers have to use in a given year, say year 1, is:

$$B_1 \leq P_1 + S_1 + E_1 \tag{16.6}$$

where P is the presuppression budget, S is the suppression budget, and E is emergency spending. The budget in year 1 (B_1) is independent of expenditures in other years, and surplus budget is returned to the treasury. Thus, in a year with unexpectedly few wildfires E equals 0 and suppression expenditure is less than S, and in a year with more wildfires than expected all of S is spent and E is greater than 0. Because of the possibility of emergency funding, the fire manager's choice of suppression effort on one fire is independent of suppression decisions (or, more precisely, expected suppression decisions) on all other fires during the current fire season or any future fire season. Since 2002 managers have been required to meet some of the cost of emergency wildfire suppression with transfers from other programs. Although this may affect managers' decisions on a given wildfire, suppression decisions across wildfires likely remain independent.

We do not mean to imply that fire managers completely ignore opportunity costs. There is considerable emphasis on cost containment within the agency, and undoubtedly fire managers give consideration to complying with this request. However, current wildfire suppression budgeting policy provides little or no serious incentive for fire managers to consider the costs of suppression resources (or the beneficial effects of wildfire). In the heat of battle, the pressures are to contain damage, not costs, and funds are generally made available when requested. It is within this light that we consider two alternatives for improving the incentive structure. First we look at the use of performance measures, a process that has recently been introduced to various federal government agencies to improve budget decision making and accountability. Then we examine an idea that has not yet been implemented, but which may hold promise as a way to raise the importance of both cost containment and consideration of the benefits of wildfire.

4. PERFORMANCE MEASURES

In 1993, Congress passed the Government Performance and Results Act to improve congressional oversight of federal expenditures by tying appropriated dollars to measurable results (Office of Management and Budget 2007). Under this act, all federal agencies have (since 1997) been required to submit five-year strategic plans, which identify planned accomplishments and the performance measures used to judge progress towards these accomplishments; an agency's future budget may be dependent on being able to demonstrate such progress. For example, field staff track the "number of acres treated" for hazardous fuels reduction. These numbers serve both to assess progress in the program to date and to help budget planners predict necessary funding for future budget cycles. Although performance measures are intuitively appealing, designing effective performance measures is not easy. We first discuss these difficulties in general and then use this discussion to illustrate problems with performance measures used to evaluate the Forest Service's wildfire suppression program.

There are two main problems encountered when designing performance measures. First, a performance measure may require data that are difficult if not impossible to collect. If progress towards a goal cannot be readily measured, then managers have little incentive to direct resources towards that goal. Second, a performance measure may not be a good indicator of progress towards the goal it was intended to measure. To illustrate this point, consider the difference between *outputs* and *outcomes*. Outcomes are the desired goals that an agency is working to achieve. However, these outcomes are often long-term and difficult to quantify, for example, increasing forest health or reducing wildfire risk. Therefore, an agency often selects a more easily measurable intermediate output as a performance measure, for example, the number of acres that receive a certain land management treatment. If the performance measure is not a good indicator of progress towards a particular outcome, then the agency may not allocate its resources efficiently—emphasizing the intermediate output at the expense of actions that would better achieve the desired final outcome.

The Forest Service currently uses 17 performance measures to assess its wildland fire management program. Of these, three directly address wildfire suppression costs (table 16.1), and are intended to minimize those costs subject to safety and resource constraints. The first performance measure encourages managers to spend suppression resources commensurate with the values at risk. However, to demonstrate that a suppression strategy does this, a manager must show what a fire would have done in the absence of the suppression strategy and must place a monetary value on resource damages. The data required to complete both of these tasks are incomplete at best, and, therefore, managers may not be able to reliably demonstrate progress under this performance measure. Problems implementing this performance measure illustrate the need to develop better resource value measures. The second and third performance measures encourage aggressive initial attack, as fires are easier and cheaper to control when they are small. However, these two performance measures do not encourage managers to consider the benefits of wildfire. A century of aggressive wildfire suppression has led to elevated fuel loads on the nation's forests and has contributed to the recent increase in suppression costs (Calkin, Gebert et al. 2005). Therefore, the type of wildfire management encouraged by these two performance measures, although it may minimize short term suppression costs, may contribute to an increase in long term suppression costs and may adversely affect forest health. In other words, the desired outcome of minimizing long term suppression costs becomes subordinate to reducing short run suppression costs.

The Forest Service's current performance measures are not final. As the Office of Management and Budget noted in its 2004 program review, "discrete targets and baseline data have not been developed for either annual or long-term goals, and some performance measures are vague and in need of greater definition."[3] Another analyst has noted that measures being tested by the Forest Service lack clarity, combine activities, miss important qualitative achievements, fail to capture important activities, count only a portion of what has been accomplished, and provide data that is difficult to interpret.[4] These problems demonstrate that designing a system of performance measures that provides the desired incentives for managers is not straightforward, and that some performance measures may introduce unintended and unwanted incentives.

5. INCENTIVES AND THE WILDFIRE SUPPRESSION BUDGET

Recognizing that performance measures may not provide adequate incentives to reduce suppression costs, we examine another approach to the two identified problems—lack of attention to the benefits of wildfire and the opportunity cost of emergency suppression expenditures. This approach uses the budget process to fashion manager incentives.

Consider first simply eliminating emergency spending, which produces the following annual budget:

$$B_1 \leq P_1 + S_1 \tag{16.7}$$

[3] FY04 Budget Fall Review, OMB PART Review.

[4] Gorte, Ross. 2000. Testimony before the House Resources Committee Subcommittee on Forests and Forest Health: *Forest Service Performance Measures* June 29, 2000.

Under this funding mechanism, if S_1 is low enough to constrain suppression expenditure, the budget provides fire managers with an incentive to consider the tradeoff between suppression costs and damage averted; managers would seek to use limited funds where they were most effective. However, this simple approach ignores a major problem—uncertainty about the severity of a fire season would make it impossible to properly set the optimal level of S_1 in advance.

A solution to the problem of determining a suppression budget for an uncertain fire season is to set P and S constant for multiple seasons and allow fire managers to carry over surpluses and deficits from year to year. Therefore, savings from a mild fire year could be used to supplement suppression expenditures in a severe fire year:

$$B_1 = P_m + S_m + \left(C_0 - C_1\right) \quad (16.8)$$

where C_0 is carryover from the previous year and C_1 is carryover from this year to the next. For example, if a manager receives \$5 million in carryover from the previous year (C_0) and carries over \$2 million to the next year (C_1), then the net carryover (C_0- C_1) is \$3 million. As long as managers expected their base funding (P_m + S_m, *where* the *m* indicates multiple season amounts) to remain constant (in real terms) from year to year, this funding mechanism would provide an incentive to consider the tradeoff between suppression costs and damages averted, and would address the issue of budgeting for an uncertain fire season. P_m and S_m would have to be carefully set based on fire cost history and the overall objectives of the fire suppression policy. (We defer to later the consideration of what is perhaps the more difficult task for the agency and Congress—to stay within the preset budget levels.)

We turn now to the other deficiency of the current system—the lack of an incentive to consider the beneficial effects of wildfire. Various approaches can be imagined for attempting to achieve this aim. We present one offered by Donovan and Brown (2005), which is to add a severity adjustment based on the number of acres burned in a fire season:

$$B_1 = P_m + S_m + \left(C_0 - C_1\right) + bA \quad (16.9)$$

where *b* is a constant and *A* is acres burned. *A* is a function of suppression effort. It is assumed that $dA/dS \leq 0$. The Forest Service currently uses a severity adjustment if a region is expected to experience a severe fire season. However, this adjustment is to presuppression budgets and is based on anticipated severity not actual acres burned.

To illustrate how this severity adjustment provides an incentive to consider the beneficial effects of wildfire, consider the cost of suppressing a specific wildfire (c) implied by equation (16.9):

$$c = W^s S^e + b(q - A) \quad (16.10)$$

where q is the number of acres the wildfire would burn in the absence of suppression (so $A \le q$). Equation (16.10) shows that as suppression expenditures reduce the number of burned acres (i.e., as A gets smaller), a fire manager's budget is reduced. For example, consider a forest with a base suppression budget of \$50,000, where a wildfire starts that would burn 1,000 acres in the absence of suppression. If b were chosen to equal the per-acre benefit of wildfire, which in this case we assume is \$50, then the fire manager's maximum suppression budget would be \$100,000 (\$50,000 base + \$50 • 1,000). If the manager spent \$20,000 suppressing the wildfire, reducing the total number of burned acres to 900, the manager's total suppression budget would be reduced to \$75,000 (\$50,000–\$20,000 + \$50 • 900). Therefore, the cost of suppressing the wildfire would be \$25,000: \$20,000 in direct suppression costs and \$5,000 in reduced budget. The reduction in budget of \$5,000 is a proxy for the wildfire benefits that were foregone by suppressing fire on 100 acres of forest. Therefore, although the fire manager does not directly consider the benefits of wildfire, the manager does consider the reduction in budget from reducing the number of burned acres. In this simple example we assume that all acres have equal wildfire benefits. However, the proposed incentive structure can accommodate different levels of wildfire benefits in different areas. Of course, this reinforces the need for a better understanding of resource values and how these values are affected by wildfire.

To illustrate this point more formally, consider the fire manager's benefit function for a specified wildfire:

$$TB = r(q - A), \tag{16.11}$$

$$TC = W^s S^e + b(q - A) \tag{16.12}$$

where r denotes the per-acre value of resources at risk. Differentiating equations (16.11) and (16.12) with respect to S^e yields the following expressions for marginal cost (*MC*) and marginal benefit (*MB*) of suppression:

$$MB = -\frac{dA}{dS^e} r\,. \tag{16.13}$$

$$MC = W^s - \frac{dA}{dS^e} b\,, \tag{16.14}$$

Equating equations (16.13) and (16.14) and rearranging terms yields the following equilibrium condition:

$$\frac{dA}{dS^e} r - \frac{dA}{dS^e} b = -W^s\,. \tag{16.15}$$

The first term on the left hand side of equation (16.9) is the product of the marginal physical effectiveness of suppression and the per-acre value of resources at risk, and is, therefore, the marginal benefit of suppression. Now consider the second term on the left hand side of equation (16.15). If b is chosen to be the per acre benefit of wildfire, then this expression, with its negative sign, becomes the marginal loss of wildfire benefits, and the left hand side of equation (16.15) becomes the marginal effect of suppression on NVC. Therefore, if b is set to equal the per-acre benefit of wildfire, equation (16.15) is the same as equation (16.5)—the first order condition for optimal level of suppression expenditure—and, at the margin, the proposed incentive structure will promote an efficient level of suppression expenditure. Assuming as stated earlier that $dA/dS \leq 0$, fire managers would—via the incentive to maintain budget for suppressing future, potentially more destructive fires—consider the opportunity cost of preventing an acre of land from burning. That is, when the value of the potential damage was judged to be less than direct suppression costs plus the value of the funds that suppression would remove from future suppression activities, managers would avoid the cost and let some acres burn.

We have shown that if b is chosen to be the per acre benefit of wildfire, then, at the margin, the proposed incentive structure promotes the efficient use of suppression resources. However, the benefits of wildfire are difficult to accurately quantify, and in any case, because they are in part nonmarket goods, they are difficult to value. A practical approach to setting b is to determine how much it would cost to achieve these benefits by different means. The two main management tools for mimicking the beneficial effects of wildfire are prescribed fire and mechanical treatment. The cost of these tools can vary from less than \$100 per acre for prescribed fire to over \$1000 per acre for mechanical treatment (Hesseln et al. 2006). Therefore, the optimal value for b varies by site depending on whether prescribed fire is an option and on the difficulty of applying whichever treatment is chosen.

A logical extension to this incentive structure would be to remove the artificial delineation between wildfire and fuels management. In this case, the budget constraint would apply to both wildfire and fuels management, so that A represents burned acres plus treated acres. Therefore, if the increase in budget from burning or treating an additional acre (b) were larger than the sum of the treatment cost and any damages caused, the manager would treat that acre of land. Another extension to the model would be to remove the distinction between presuppression and suppression budgets. Fire managers would receive a single fire management budget, which could be used to finance suppression, presuppression, or fuels management. These extensions can be incorporated into the model presented; although they do make the model somewhat cumbersome, they do not fundamentally change the results.

An incentive structure that encourages fire managers to increase the number of burned acres would increase the possibility of wildfires or prescribed fires escaping management control and causing unexpected damage. Measures may need to be taken to encourage fire managers to accept this increased risk. Managers should not face undue consequences if a wildfire or prescribed fire they are managing causes unexpectedly high damages. In addition, managers may need additional decision support tools to help them identify and mitigate risk.

Implementing the proposed incentive structure would also require institutional changes. Currently, the local land manager cedes tactical fire management decisions to the incoming incident command team. But for the proposed incentive structure to work, the local land manager must maintain control over suppression decisions. However, incident command teams have far more experience than local land managers in managing large wildfires. Therefore, the proposed incentive structure would require establishing some form of principal-agent relationship between the local land manager and the incident command team.

6. DISCUSSION

The current system for funding wildfire suppression in the United States has evolved over one hundred years in response to changes in the forest landscape, changes in societal values, increasing development in the wildland urban interface, and a number of exogenous factors such as the increased availability of mechanized equipment following the Second World War and the Korean war. We identify two problems with the current system. First, funding wildfire suppression with emergency suppression funds provides little incentive for cost containment. Second, the benefits of wildfire are not given adequate consideration. Both problems contribute to inefficiently high levels of suppression expenditure, which contribute to elevated fuel loads, leading to future wildfires that are more difficult and expensive to suppress.

Performance measures (table 16.2) are being adopted by the Forest Service partly to help control suppression expenditures. However, as these performance measures encourage aggressive wildfire suppression, they may contribute to increased suppression costs in the long term. Furthermore, because progress toward desired outcomes is difficult to quantify, the performance measures often focus instead on intermediate outputs that may not adequately represent the desired outcomes.

There are additional problems with the use of performance measures as a cost containment strategy for wildfire suppression. It is not clear how managers will be rewarded for meeting performance targets. Conceptually, performance measures should tie closely to budget allocations; in the case of fire suppression, there is no evidence that this feedback loop exists. Managers may receive budget increases, but nowhere is this explicitly stated. It is also possible that if managers are seen to meet performance targets too easily, then their budgets may be reduced. Therefore, managers may not try to exceed a performance target, because this might lead to an increase in the performance target or a decrease in budget. An additional problem is that performance measures are insensitive

Table 16.2. Performance measures relavant to fire suppression costs.

Objective: "Consistent with resource objectives, wildland fires are suppressed at a minimum cost, considering firefighter safety, benefits, and values to be protected."[1]

Performance Measure	Line-up with Outcome?	Incentives created?
The percent of large fires in which the value of resources protected exceeds the cost of suppression.	*No.* This measure is not accurately assessed, is done *ex post facto* with computer models, fails to account for fire benefits, and does not provide any useful tool for line managers during a fire.	Unclear. Fire managers are unlikely to stop fighting a fire just because they notice that there is little commercial timber value on site.
The number of acres burned by unplanned and unwanted wildland fires.	*Yes.* Extinguishing fires while they are small will keep acreage numbers low.	Strong incentive for initial attack.
The percent of unplanned and unwanted wildland fires controlled during initial attack.	*Yes.* Fires are most easily stopped with a strong initial attack while they are small. *No.* Although overall costs may be reduced, spending a lot of money in initial attack translates into very high cost/acre numbers.	Aggressive initial attack on all fires. This measure also functions as a disincentive for fire use, since a "wait and see" management response is specifically discouraged here.

[1] USDA Forest Service (2003, p. A12)

Source: USDA Forest Service (2002).

to unique local conditions. For example, consider the measure that encourages strong initial attack. A manager might have the opportunity to let a wildfire burn a larger area to generate resource benefits, but under the performance measure this would be considered a failure. Similarly, moderate weather conditions at the end of a fire season may make it cost effective to let a fire burn for longer than would be the case earlier in the fire season, but again this would be considered a failure.

We present an alternative approach that does not involve measuring managers' performance. Rather, we suggest a funding mechanism that encourages managers to spend their wildfire suppression budgets efficiently in light of the costs and benefits of wildfire suppression. Many of these costs and benefits are not easily measured in monetary terms. However, local land managers are best placed to make these tradeoffs; they are familiar with the diverse benefits generated by the land they administer and how these benefits may be affected by wildfire. In addition, they have an understanding of the relationship between the land and local communities. In summary, we suggest providing land managers with incentives to use suppression budgets efficiently, and then allow them to use their professional judgment when making suppression decisions.

The proposed funding mechanism for wildfire suppression encourages efficient resource use at the margin, but does not help determine the optimal *total* wildfire suppression budget. However, identifying the optimal wildfire suppression budget is perhaps more of a political than an economic question. For example, a reduction in wildfire suppression spending will result in an increase in wildfire damages at least in the short term. These damages would include loss of homes, increased smoke, and a reduction in recreational opportunities. Tools exist for estimating the economic cost of these damages, but what is probably more important is the extent to which the public and their political leaders will tolerate these losses. The task of determining the optimal total suppression budget would, therefore, become an iterative political process balancing wildfire damages and suppression budgets. Wildfire budgets would be determined in much the same way as other public programs such as education or defense.

This chapter dealt exclusively with wildfire management; however, some of the lessons learned may be applicable to other natural disasters. For example, although fixed budgets are not appropriate for natural disasters, we can still provide incentives to encourage the efficient use of resources. In addition, we should consider the impact of current mitigation efforts on the probability and intensity of future natural disasters. For example, protecting one community from flooding may make flooding worse for a downstream community. Finally, it is important to consider the effect of mitigation on development patterns. For example levees may encourage development in a flood plain putting more homes at risk of flooding should the levees fail.

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CHAPTER 17

FORECASTING WILDFIRE SUPPRESSION EXPENDITURES FOR THE UNITED STATES FOREST SERVICE

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1. INTRODUCTION

The wildland fire management organization of the United States Forest Service (USFS) operates under policy and budget legacies that began nearly 100 years ago and a forest fuel situation that is all too current. The confluence of these three factors contributes to increased burning and firefighting costs for the agency, and increased concern from both the U.S. Congress and the public. Historically, the 10-year moving average of suppression expenditures has been used in USFS annual budget requests to Congress. But in a time when fire activity and costs are steadily rising, the 10-year moving average budget formula has translated into shortfalls in available suppression funds nearly every year since the mid-1990s. When the budgeted amount is insufficient, the agency continues to suppress fires by reallocating funds from other land management programs and by making subsequent requests to Congress for additional funding. A recent report from the U.S. General Accounting Office (renamed the Government Accountability Office in 2004) recommended a reevaluation of the budgeting system for wildfire suppression expenditures by the federal land management agencies (U.S. GAO, 2004). While many of the issues and critiques made by GAO are beyond the control of the agencies, the USFS has explored alternatives to current practices used in developing out-year budget requests for emergency fire suppression.

We have two primary objectives in this chapter. First, we seek to evaluate candidate forecast models of wildfire suppression expenditures. These time series models are constructed to allow suppression budget forecasts up to 3 years in advance of a coming fire season. These models are evaluated for their suitability for budget documents presented to Congress. The structure of estimated models highlights the importance of accounting for intertemporal dynamics and stochasticity in wildfire suppression expenditures. Second, we demonstrate a method from the forecasting literature that quantifies some of the factors potentially important in choosing among alternative models. The method applies loss functions to errors in forecasts, and our comparisons are between the 10-year moving average and our estimated time series models.

T. P. Holmes et al. (eds.), The Economics of Forest Disturbances: Wildfires, Storms, and Invasive Species, 341–360.

Budget requests for emergency fire suppression are of particular importance in part because these expenditures are high (they can exceed $1 billion per year) and in part because they are inherently uncertain. Currently, the USFS requests a wildfire suppression budget for a future fiscal year as part of their overall fiscal year budget request for the entire agency, which also includes monies for managing national forests and grasslands, research, and providing assistance to state governments and landowners. If actual wildfire suppression expenditures exceed this budget, the agency is allowed to sequester money from other USFS budgeted programs to continue to suppress wildfires. Congress may, and sometimes does, refund some or all of the sequestered funds, but this leads to uncertainty in the other USFS programs as their funding may be cut partway through the fiscal year. Until recently, Congress made midyear allocations to pay for suppression during high-expenditure years, and the USFS borrowed money from agency controlled trust funds, which are dedicated to activities unrelated to wildfire, to pay for suppression. Over the past few years, however, Congress has made fewer midyear allocations. Additionally, the USFS no longer borrows from the trust funds, such as the Knutsen-Vandenberg Fund, because the agency has not always been able to reimburse these funds immediately due to continued severe fire seasons.

Under the current budget process, the USFS begins to develop a budget request more than two years before the start of the fiscal year in question. Initial budget development uses the best available expenditure data to generate a 10-year moving average for inclusion in the initial budget document. In June 2005, for example, the agency began development of a budget for FY 2007 (October 1, 2006 to September 30, 2007). This budget was based on the ten years of expenditures from FY 1995 to FY 2004 as the FY 2005 fire season had not yet concluded. This is a 3 year out forecast. While the USFS resubmitted a revised budget to USDA in December 2005, the emergency fire suppression budget request was not revised, even though a 2 year out forecast could have been developed as soon as the expenditure accounting for FY 2005 was completed (sometime between October and December of 2005).

Forecasts can lead to costs and benefits for both the agency and the public. Benefits arising from improved accuracy of the forecast may accrue to the public if the total federal budget is assumed to have a maximum—a better forecast will ensure that other USFS programs are completed as originally funded by the U.S. Congress and yet money is not diverted from non-USFS programs to hold for the (poorly forecasted) suppression expenditures. Costs to the USFS could result if the total USFS budget is not allowed to vary with the variation in forecasts for suppression expenditures, in which case a more accurate forecast may mean that funds are diverted from other agency programs to hold for potential wildfire suppression expenditures. These costs and benefits can be represented in loss functions, where values can be assigned to the errors in forecasting. In the final section of this paper, we evaluate the effect of different loss functions on the choice of forecasting models and on the development of the budget request.

2. MODEL OF WILDFIRE SUPPRESSION EXPENDITURES

Wildfire is inherently unknowable in time, place, and extent, but it may be somewhat predictable if fire occurrences are added together up to arbitrary degrees of temporal and spatial aggregation. Uncertainty stems from the randomness of ignitions (whether lightning or human caused), both in terms of frequency and location, as well as from uncertainties associated with fuels and weather. Fuels are, in turn, influenced by longer term trends in both climate and anthropogenic fuel alterations (logging, grazing, prescribed fire, etc.). Historically, overall suppression expenditures have been greater when burned acres are greater, even if economies of scale occur and large fires cost less per acre to suppress than small fires. Figure 17.1 demonstrates the close relationship between acres burned and suppression expenditure for the USFS. However, this relationship is ill-defined, in part because suppression expenditures and burned acres are contemporaneously determined. Nonetheless, this relationship implies that if area burned could be forecast two or three years in advance, then forecasts of expenditures could be developed using area burned forecasts.

The model underlying agency fire management is commonly referred to as the cost plus loss model, or more recently, as the least cost plus net value change model (for example, chapters 13-18 in this book). The agency's objective is to

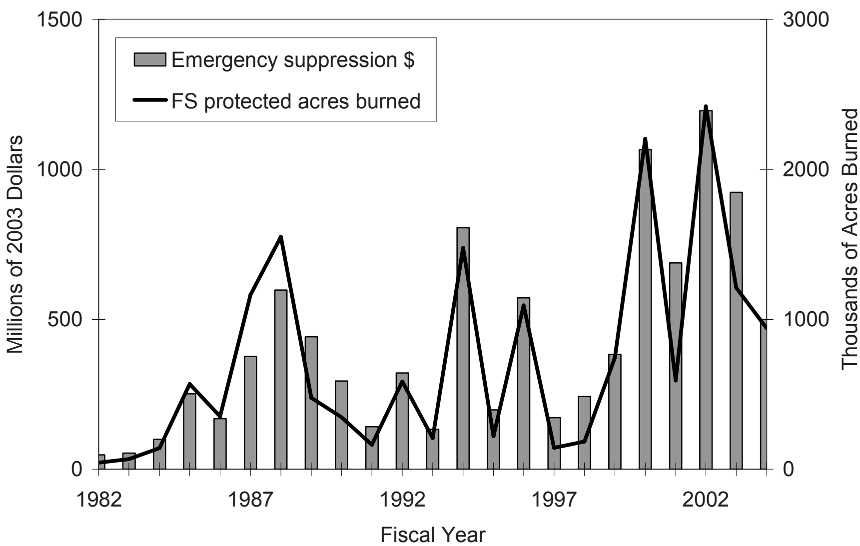

Figure 17.1. Wildfire suppression expenditures by USFS (2003 dollars) and thousands of USFS-protected acres burned, FY 1982 to FY 2004. Data sources: Expenditure data from USFS accounting records managed by the Rocky Mountain Research Station. Fire data are from FS 5100-9 records managed by the National Wildfire Coordinating Group.

minimize the sum of the expenditures associated with fire suppression and the losses associated with fires. Within the overall model, suppression expenditures are minimized along with losses resulting from fires and are subject to ignitions, weather, fuels and human presence, or other factors.

Over time, fuels, ignitions and weather vary as climate and anthropogenic factors change, leading to changes in the forests themselves. To use this type of model for forecasting two and three years out, we would need forecasts of all the included variables, including prices of inputs and values of protected and damaged resources, as well as forecasts of ignitions, fuels, weather and human factors. Even if the model were fully defined and the functional form known or approximated, we do not have the forecasts of independent variables to develop forecasts of costs and losses. Thus, we developed simple time series models with lags of suppression expenditures and a time trend to model suppression expenditures by national forest region.

3. DATA

Suppression expenditure data used in the time series models were based on USFS accounting databases as compiled by the USFS Rocky Mountain Research Station. These data were available beginning in FY 1977 for the nine land management regions, as well as the for the remainder of the Forest Service (RFS), which includes the National Offices, Research Stations, and the National Interagency Fire Center expenditures related to USFS fires.[1] Wildfire suppression expenditures include all costs incurred by the USFS and not reimbursed by other agencies for suppressing wildfires including salaries, contracts, equipment and supplies. Prior to FY 1994, these data can only be obtained as summaries of expenditures *by* region. After FY 1994, the data can be obtained as summaries of expenditures *for* a region, but this time series was deemed too short as yet to provide reliable statistical results. This issue may be of little importance here as we are using time series models without acres as a covariate (see chapter 15 for more discussion of this issue).

The data series were evaluated and most regional series were log-transformed to approximate a normal distribution. The Southern and Alaska regions and the RFS were estimated untransformed. Note that three of the observations have negative numbers (Pacific Southwest for FY 1998, RFS for FY 1983 and Alaska for FY 1999). These numbers are recorded in the official database as negative because of accounting adjustments that were made after the end of the fiscal year. Unfortunately, we are not able to resolve these negative values, and we must accept them as is. Because the Pacific Southwest data were log-transformed, the FY 1998 value was set to a small positive number and then a dummy variable

[1] Data are available beginning in FY 1971. However, due to the change in FY in 1976 (changed from a July start date to an October start date), only data from FY 1977 on were used in model estimation.

was added for the Pacific Southwest model for FY 1998 to exclude any effects of this variable change on the model and forecasts.

4. TIME SERIES FORECASTING MODELS

We estimated time series models for the nine land management regions and the RFS. The models were estimated and then evaluated at two- and three-year forecast horizons. The three-year forecast horizon may be useful for the initial budget request submitted by the agency, while the two-year horizon could be used to update a budget request several months after the initial request is made. In this analysis, we assume that the budget request can be revised when updated forecasts are available, so that changes in the forecast will affect the budget request and also the budgeted appropriation from the U.S. Congress. Estimated time series models for all regions, summarized in table 17.1, employed various lags of expenditures and a time trend. Where the data had been log-transformed, the forecasts were back-transformed and bias-corrected using a method recommended by Karlberg (2000). In our model selection, we also experimented with unlogged time series models, but out-of-sample performance and non-normality of residuals of these equations in most cases argued for logged versions. The preferred version typically took the natural logarithm of suppression costs for the dependent variable.

For individual models, a model selection procedure which minimizes a model fitting criterion (the Schwarz Information Criterion) was used to identify parsimonious versions of the equations. These models were selected first individually, as least squares regressions of costs as functions of lags of costs (beginning with k=8 lags) and time trends. To account for the unusually low fire expenditures occurring in the Pacific Southwest in FY 1998, where regional costs were reported near zero, a dummy variable was also included.

In most cases, the best-fitting models included one or more lags of costs and a time trend, although selected models for the Rocky Mountain and Alaska regions included just a time trend; for these regions, lagged variables did not significantly explain inter-annual cost variations. Once individual models were determined, the ten regional cost equations were estimated simultaneously, using multiple equation (generalized) least squares, in a Seemingly Unrelated Regression (SUR) (Greene 2003). The SUR method exploits cross-regional correlations in unexplained variation in costs to reduce uncertainties about the values of estimated parameters. The R^2's of individual equations estimated within this system ranged from 0.41 for the Northern Region (Montana and northern Idaho) to 0.92 for the Pacific Southwest Region (California).

Coefficients on many lags of suppression costs included in most of the regional cost equations are estimated to be negative. Negative signs indicate that costly years are followed by cheaper years, and vice versa. This kind of pattern could be capturing cycles in climate, and it may also be demonstrating more

Table 17.1. Model coefficients and fit statistics for the time series forecast model for 10 USFS regions.

	Variables (coefficient; standard error below in parentheses)												
			Year lags										
Region*	Constant	Trend	-1	-2	-3	-4	-5	-6	-7	-8	1998 dummy	R^2	Adjusted R^2
Northern	46.674 (8.346)	**0.177** **(0.036)**	**-0.339** **(0.137)**	-0.120 (0.137)	-0.087 (0.148)	-0.144 (0.134)	**-0.460** **(0.134)**	-0.174 (0.153)	-0.166 (0.152)	-0.492 (0.152)		0.409	-0.075
Rocky Mountain	14.235 (0.329)	**0.084** **(0.016)**										0.446	0.425
Southwestern	26.092 (4.788)	**0.138** **(0.024)**	0.060 (0.143)	-0.167 (0.135)	**-0.291** **(0.126)**	0.031 (0.147)	**-0.332** **(0.134)**					0.732	0.637
Intermountain	16.937 (2.637)	**0.097** **(0.020)**	0.133 (0.112)	-0.038 (0.114)	-0.214 (0.117)							0.552	0.466
Pacific Southwest	16.996 (0.234)	**0.064** **(0.011)**									-9.470 (0.392)	0.917	0.910
Pacific Northwest	15.310 (2.604)	**0.066** **(0.018)**	**0.162** **(0.107)**	0.136 (0.103)	**-0.245** **(0.107)**							0.528	0.438
Southern	41.324 (7.342)	**0.212** **(0.044)**	-0.117 (0.143)	**-0.254** **(0.137)**	-0.471 (0.138)	0.068 (0.130)	**-0.597** **(0.126)**	-0.098 (0.127)	0.036 (0.124)	**-0.368** **(0.122)**		0.642	0.349
Eastern	19.814 (3.056)	**0.089** **(0.024)**	-0.004 (0.133)	-0.051 (0.129)	**-0.369** **(0.130)**							0.422	0.312
Alaska	-11.198 (3.233)	**0.883** **(0.222)**	**-0.425** **(0.154)**	**-0.682** **(0.161)**	**-0.736** **(0.189)**	**-0.606** **(0.212)**	**-0.744** **(0.204)**	**0.467** **(0.236)**				0.697	0.556
Rest of USFS	-109.748 (36.450)	**11.775** **(2.815)**	**-0.455** **(0.138)**	-0.170 (0.150)	-0.265 (0.155)	**-0.599** **(0.157)**	-0.115 (0.164)	**0.267** **(0.163)**	**-0.492** **(0.167)**			0.632	0.406

*The costs used in all models besides Southern, Alaska, and Rest of USFS are expressed as natural logarithms. Coefficients significant at 5 percent are bolded.

complicated intertemporal budget reallocations. Additional research into the underlying climate factors driving fire activity and the economic factors driving costs could improve our understanding of these results. It is notable that the trend is positive and statistically significant (at 5 percent) for all models. This finding documents the universally positive trend in real suppression costs for the USFS over the estimation period.

4.1 Forecast Confidence Intervals

The estimated models can be used to develop not only a point forecast, but can be used to develop a distribution that will provide an estimate of the confidence intervals for the forecasts. To develop the distributions, we employed techniques described by Krinsky and Robb (1987). These distributions account for the uncertainties in the equations estimated and reported in table 17.1. Uncertainties include those associated with parameter estimates and equation residuals. The Krinsky and Robb (1987) approach accommodates, as well, correlations across estimated parameters and equation residuals. In the discussion below, the forecasts are in both constant (2003) dollars and inflated (current 2007 or 2008) dollars for the year being forecasted.

Table 17.2 shows the FY 2007 forecast, using the time series models and data available in early October, 2005. The median forecast of FY 2007 USFS suppression expenditures, implying a 50 percent chance that expenditures will exceed the value, is $1,096 million (in projected 2007 dollars), with the 95 percent confidence interval ranging from $436 million to $2,904 million. The probability-weighted (expected or mean) forecast expenditure is $1,242 million, while the most likely single value (point forecast) is $1,015 million. The distribution of the forecast for FY 2007 (in constant 2003 dollars only) is shown in figure 17.2.

Table 17.3 and figure 17.3 show the FY 2008 forecast using the time series model and data through early October, 2005, as the input data set, creating the 3 year out forecast. The median forecast of FY 2008 suppression expenditures

Table 17.2. Confidence interval table for forecast of total USFS wildfire suppression expenditures for FY 2007.

	FY 2007 forecast millions of 2003 dollars	FY 2007 forecast millions of 2007 dollars
Point forecast	907	1,015
Mean	1,111	1,242
Median	980	1,096
95% confidence interval lower bound	389	436
95% confidence interval upper bound	2,596	2,904
90% confidence interval lower bound	453	507
90% confidence interval upper bound	2,208	2,469

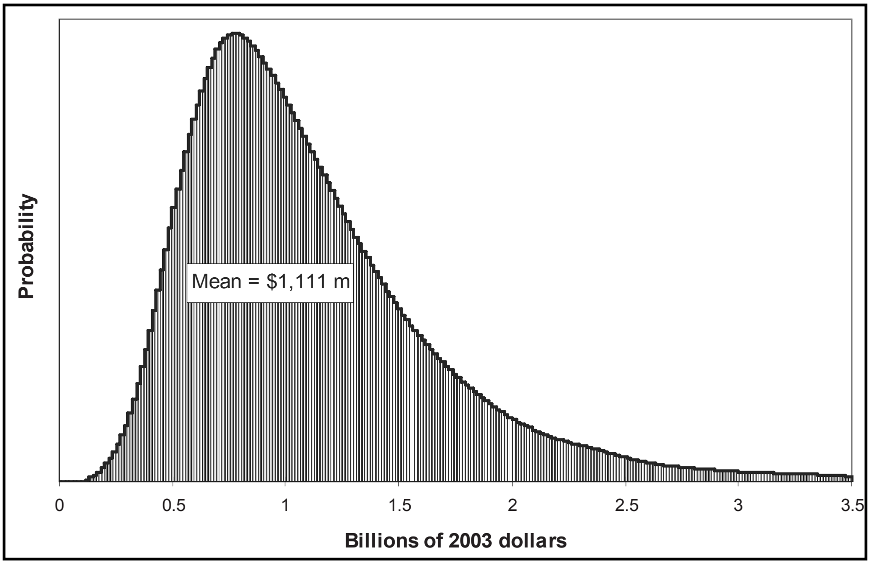

Figure 17.2. Empirical probability density of the forecast of total USFS wildfire suppression expenditures for FY 2007, in constant (2003) dollars.

Table 17.3. Confidence interval table for forecast of total USFS wildfire suppression expenditures for FY 2008.

	FY 2008 forecast millions of 2003 dollars	FY 2008 forecast millions of 2008 dollars
Point forecast	1,039	1,197
Mean	1,230	1,417
Median	1,096	1,262
95% confidence interval lower bound	493	567
95% confidence interval upper bound	2,760	3,179
90% confidence interval lower bound	556	641
90% confidence interval upper bound	2,345	2,701

is $1,262 million with the 95 percent confidence interval ranging from $567 to $3,179 million. The mean forecast is $1,417 million, while the most likely single value is $1,197 million.

4.2 Forecasts and Model Comparisons

To evaluate the time-series models in conditions approximating real-world budget forecasting, we developed out-of sample, cross-validated forecasts of agency-wide (total) suppression expenditures. Agency-wide totals of suppression are simply

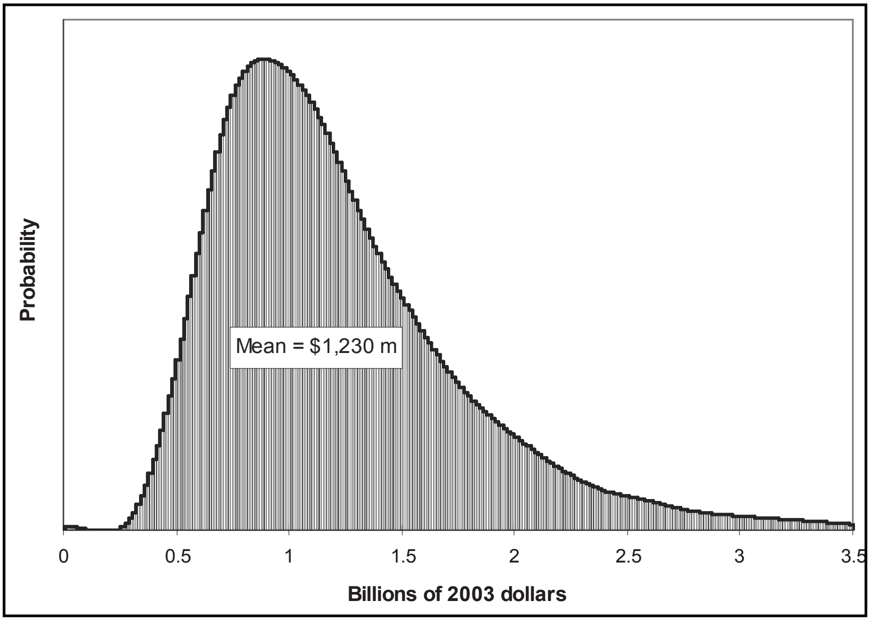

Figure 17.3. Empirical probability density of the forecast total USFS wildfire suppression expenditures for FY 2008, in constant (2003) dollars.

the sums of the forecasted expenditures for each region in each year forecasted. The cross-validation is achieved by leaving out the forecast year observation, estimating the model, and then forecasting the left-out observation, for all years. This cross-validation is done region by region, and the cross-validated forecast values are then summed across all ten regions. These forecasts and actual expenditures are displayed in figure 17.4 and used in the model evaluations presented in tables 17.4 and 17.5.

We find that the agency-wide actual expenditures are more volatile than the forecasts, regardless of the model used (fig. 17.4). This implies that there is important information not captured in the time trends or lags of costs. Figure 17.4 shows the total USFS expenditures and the total USFS forecasts using the 10-year moving average and the 2 year out and 3 year out time series models. Also included is the time series of actual expenditures . The 10-year moving average is calculated and shown for both two- and three-year horizons, which allows for a more direct comparison between the time series and 10-year moving average models. In other words, the 2 year out time series model should be compared with the 2 year out moving average model, and the 3 year out time series model should be compared with the 3 year out moving average model.

Fitness statistics document the performance of the alternative forecast models (table 17.4) as quantified by the root mean squared error (RMSE) and the R^2 of

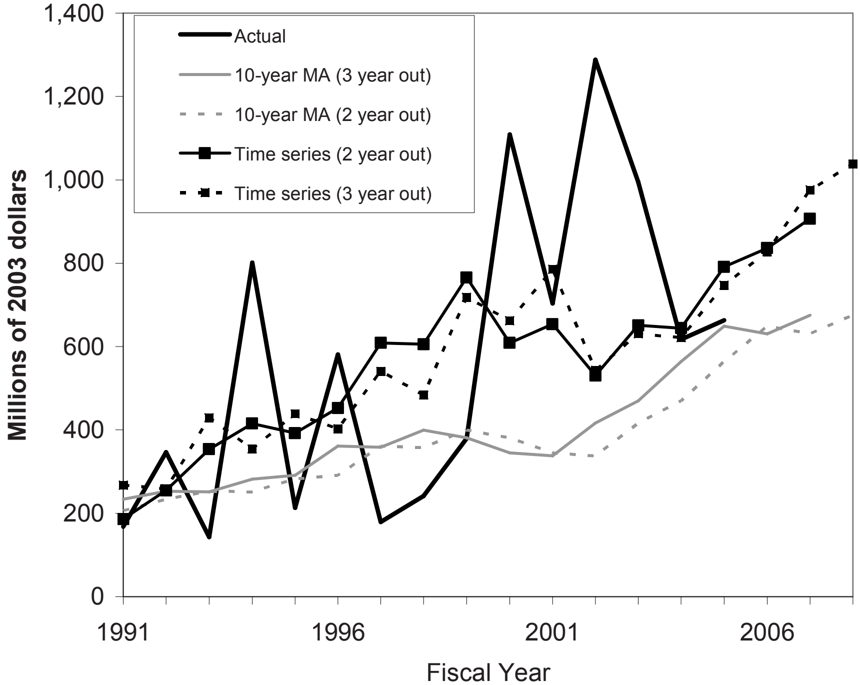

Figure 17.4. Actual total USFS wildfire suppression expenditures compared to forecast total USFS expenditures for the 10-year moving average (MA) and time series forecast models for 2 and 3 years out.

Table 17.4. Root mean squared error (in millions of dollars squared) and cross-validated (jackknife) R^2 for suppression forecasting models for 15 years (FY 1991-FY 2005) and 5 years (FY 2001-FY 2005).

	FY 1991-FY 2005 RMSE	FY 2001-FY 2005 RMSE	FY 1991-FY 2005 R^2	FY 2001-FY 2005 R^2
Time series—2 year out	346	486	0.73	0.80
Time series—3 year out	337	419	0.76	0.82
10 year moving average—2 year out	376	619	0.69	0.67
10 year moving average—3 year out	413	591	0.64	0.65

Table 17.5. Significance tests (F) for RMSE between 2 and 3 year out models and between time series and moving average models for 15 years (FY 1991-FY 2005) and 5 years (FY 2001-FY 2005).

	FY 1991-FY 2005	FY 2001-FY 2005
Critical F Statistic	2.48	6.39
Degrees of freedom	14	4
Time series (2 year out vs. 3 year out)	1.05	0.61
10 year moving average (2 year out vs. 3 year out)	1.09	1.75
2 year out (time series vs. 10 year moving average)	1.07	1.31
3 year out (time series vs. 10 year moving average)	0.93	1.23

cross-validated forecasts. These statistics are calculated for the longest common comparison set of years and more recent years, the latter set containing higher average levels of observed suppression expenditures (fig. 17.4) and providing some evidence of model performance in the most recent years of forecasting. Over both sets of years, the 3 year out time series model has the lowest error (RMSE of 346 million) and highest jackknife R^2 (0.76) of all the models, followed by the 2 year out time series model (RMSE of 346 million, jackknife R^2 of 0.73) over FY 1991-FY 2005. Ten year moving average forecasts have RMSE's that are higher by at least 30 million over the comparison set of years. All four models demonstrate a poorer fit in more recent years, when costs have been generally higher. That, however, would be expected, even if the percentage errors were to remain the same. Indeed, the jackknife R^2's for all models are higher over the FY 2001-FY 2005 period.

Unfortunately, we have a short time series of forecasts with which to compare our forecasts with observed costs, and this shortness makes it difficult to discern whether any model is superior to the other in terms of these two measures of fitness. F-tests can be used to compare the RMSE's (table 17.5). This simple ratio of variances test, distributed as $F_{n-1,n-1}$ where n is the number of observations used to calculate the mean squared errors (assuming normality), produces test statistics for the comparisons between 2 and 3 years ahead for both model types, and for the comparisons between the time series and 10-year moving average at both 2 and 3 years ahead. We find that none of the models is statistically significantly superior to any other model at the 5 percent significance level.

Comparisons such as those shown in tables 17.4 and 17.5 are meaningful for policy makers seeking to select among competing models when the policy maker places value only on forecast accuracy and when errors in forecasts generate losses to the agency that are linear in these errors. In contrast, if a decision maker cares about the direction (positive, negative) of these errors, then the RMSE is not informative. Likewise, if the decision maker cares relatively more about large errors than small errors (per dollar of error, say), then the

RMSE is not informative, and neither is the jackknife R^2. More specifically, these error comparisons based on the RMSE assume that (1) the losses associated with suppression expenditure forecasting are symmetric and linear, (2) the only losses are those associated with error, and (3) the point value of the forecast was used for budget allocations. These assumptions and the effects of altering these assumptions (shape of loss functions, multiple loss functions, and choice of forecast level for budgeting) are discussed further in the following section.

5. LOSSES ASSOCIATED WITH FORECASTS

As Lawrence and O'Connor (2005) summarize well, the primary objective in most economic forecasting is accuracy—minimizing forecasting error. But, they state, for some organizations, the economic or non-economic costs (or losses) associated with forecasting errors can differ according to highly nonlinear valuation criteria. Sometimes, the error associated with forecasting too high may have different implications than the same size error associated with forecasting too low. Some organizations may identify an error band within which forecasting errors have lower losses associated with them compared to errors that surpass the band. Alternatively, the losses associated with forecasting errors may steadily increase, or may increase in a nonlinear fashion per unit of forecasting error as the absolute size of the error goes up.

Losses associated with forecasting errors may arise from a variety of mechanisms. They may arise out of investment errors caused by a forecasting error—over- or under-investing in anticipation that the state of the world will be at a certain level in the future. They can also arise from the opportunity costs associated with diverting scarce administrative resources to making accounting adjustments across budget categories of an organization's overall budget. A challenge for an organization seeking to improve forecasting is to quantify how forecasting errors translate into losses for the organization.

The losses associated with these kinds of errors have been identified as significant in the case of budget under-predicting (budgeted less than actual) for wildfire suppression expenditures (U.S. General Accounting Office 2004). Losses associated with under-predicting occur when either (1) the agency must reallocate internal funds to pay for higher than forecast suppression expenditures, or (2) insufficient crews (contract or agency) are available to suppress fires at the start of the season leading to increased contracting or hiring costs during the season (Donovan 2005). A second type of loss results from over-predicting (budgeted more than actual), where budgets for other programs and activities are reduced in order to maintain sufficient funds for potential suppression expenditures when those expenditures are less than predicted.

Thus, although there are potential costs associated with inaccurate budgeting for fire suppression, what constitutes an improved forecast will depend on how the losses and budget request decisions are defined. The suppression expenditure forecast models presented above were developed using classical statistical criteria of minimizing the sum of squared errors and minimizing bias (the tendency to over- or under-predict). However, optimal budget allocation models do not need to be so narrowly (or perhaps naïvely) focused, and sometimes other measures of forecast fitness (e.g., mean absolute percent error of forecast) may be more important indicators of a model's usefulness.

To reiterate an earlier point, forecasts above assumed that (1) the point forecast value is used to make a budget request and the budget request is met at that same point forecast value, (2) perceived losses associated with budgeting errors are symmetric (e.g., a budget created by the forecast that exceeds the observed amount by $100 million creates the same loss for the agency as a budget that is $100 million less than the observed amount), (3) perceived losses are linear (a $100 million error is twice as bad as a $50 million error), and hence that (4) error minimization is the only forecasting objective. Nonetheless, Lawrence and O'Connor (2005), Granger and Pesaran (2000), and evidence described above about the agency decision on suppression fund allocations imply that decision makers may have loss functions that depart from purely statistical criteria. Below, we discuss the shape and multiplicity of loss functions. We follow that with a discussion of how these losses can be used to develop a tool for choosing an optimal forecast value to use for budget requests. The forecast models used in the following discussion include the 2 year out models for both the time series and the 10 year moving average.

5.1 Shape of Loss Functions—Symmetry and Linearity

Sometimes the consequences of over-predicting versus under-predicting are the same (symmetric loss), while other times it is more problematic to over-predict than to under-predict, or the opposite (asymmetric loss). From the USFS perspective, it may be worse to under-predict as this leaves the agency without sufficient funds for suppression (and they violate anti-deficiency regulations) unless funds are borrowed from other USFS programs. However, in some cases, the agency may determine that it is worse to over-predict because it leads to sequestering of funds in advance of the fire season that could be utilized elsewhere.

A second factor affecting the calculation of losses from forecasting is the presence of thresholds—levels above or below which losses per unit of error change. Some forecasts may be associated with no losses for small errors, or may have distinct losses associated with specific ranges of errors. For example, a forecast that is off by $10 million may cause few or no problems, but as the error increases above that level, problems increase. There are infinitely many forms that these loss functions could take and values that could be attributed to the various losses.

5.2 Multiple Loss Functions

Baumgarten and James (1993) suggest that federal budgets are adjusted only incrementally for most years, until some external change occurs that leads to jumps in particular budgets, leading to a new epoch of cost levels. The USFS budget also demonstrates this incremental-epochal characteristic, where most years have budget changes of less than 6 percent, net of inflation dollar terms. The three years following the 2000 fire season and the implementation of the National Fire Plan can be characterized as epochal, where the total USFS budget increased by more than 40 percent. The trend seems to have reverted to incremental for 2003-2005, with annual changes of less than 5 percent for the last two years, but costs in 2006 were again among the highest ever observed for the agency.

These trends could be important for emergency fire suppression forecasting because a forecast made using the time series model is usually more variable than a forecast made using the 10 year moving average. If the entire USFS budget (for emergency fire suppression and all other activities) is limited in incremental years to increases of no more than 6 percent, say, then forecasted suppression budgets that exceed this rate of increase will have to be accommodated internally by reducing budgets for other agency programs. Thus, one possible objective of a preferred budget allocation model could be to maintain stability in the suppression budget requests made by the USFS to Congress. While stability would seem to imply that agency spending would occasionally need to be reallocated late in a budget year, some of the losses associated with late budget year reallocations can be avoided through special supplemental budget requests by the agency to Congress. The 10 year moving average model dampens budget request volatility and hence could continue to be used to achieve this stability.

Alternatively, another objective for agency planners could be to improve the accuracy of budget requests using a statistical budget forecasting tool. This would require the agency to submit more volatile annual budget requests (in total, suppression plus other agency spending). It is unclear how the oversight agencies and Congress would respond to this type of budget request volatility. Depending on the shape of the loss function, and the values associated with the losses, either model (time series or moving average) could be preferred if losses are lower with one than the other.

5.3 Designing an Improved Budget Request Tool

A budget request tool should include the values of the losses, for both over- and under-predicting (symmetry), stability, accuracy, and threshold values (linearity). If values were known for these losses (e.g., one unit of loss of accuracy could cost \$1) an optimal choice, or even combination, of forecast models could be used to develop a forecast value. Alternatively, the probability that the budget is sufficient to cover expenditures could be varied, allowing a value to be chosen that would reflect an optimal budget request amount given the losses. The RMSE and R^2 model evaluations are based on an assumption that the point forecast is used for budget requests and appropriation. For the evaluations of shape and objectives, we used the median forecast, which implies that there is a 50 percent chance that the budget request will be exceeded by actual costs.

Table 17.6 reports the results of a series of simulations where loss values, functions and objectives were varied. The simulations were conducted for both the time series and the 10 year moving average forecast models. We assumed in our simulations that (1) stability and accuracy were the two objectives jointly sought by the agency; (2) stability was defined as variation between the budgeted amount in year t and year t-1; (3) accuracy was defined as variation between budgeted and actual amounts in year t; (4) stability was given a symmetric loss (i.e., the absolute value of an under-prediction generated the same loss as an equivalent absolute value of over-prediction), but was evaluated with both (i) constant loss values and (ii) a threshold absolute value, $100 million, below which errors had a loss of zero, (5) accuracy was valued both symmetrically and asymmetrically, and for both linear losses (constant loss per dollar of under-or over-prediction) and nonlinear losses (threshold=$100 million, below which loss=0).

The first salient result of our simulations, shown in table 17.6, is that when over-budgeting (actual < budgeted) for the fire season is more costly than under-budgeting (actual > budgeted), then the 10-year moving average model would be preferred. Similarly, if instability in budgets is more costly, then the 10-year moving average model outperforms the time series model. However, when under-budgeting is more costly (actual > budgeted), the time series models are preferred. There are, however, many alternative objectives, shapes and loss values that do not result in a clear preference for one model over the other. Estimating or collecting loss values, objectives and shapes of functions could be used to develop a budgeting tool, either by choosing the best performing model, or possibly by developing a combination (or ensemble) forecast from the two models that would outperform either model independently.

A second method of developing a budget request tool could utilize the forecast distribution for a particular fire season's expenditures (e.g., figs. 17.2 and 17.3 above) that can be developed for both forecast models. Using this distribution for the time series models and the loss values for accuracy, the optimal budget request may be a figure different from either the point, mean or median forecast from the model. Table 17.7 shows the budget request that minimizes the losses (FY 1991 to FY 2005) from over- and under-budgeting. The first simulation assumes that each dollar of loss has a value of $1 to the agency. With a 50 percent (median) budget request, the losses (the same as in table 17.6) are $4,424 million. By making a budget request that would provide a lower probability (44.3 percent) of being exceeded (i.e., a lower "probability of ruin"), this value could be reduced to $4,375 million. Similarly, total losses resulting from other assumed loss values and shapes can be reduced by either increasing or decreasing the probability of ruin.

6. CONCLUSIONS

The current forecast model used for out-year budgeting for the USFS is the 10-year moving average. Development of more sophisticated time series models,

Table 17.6. Simulations of losses from variations in loss functions resulting from forecasting suppression expenditures using the 10 year moving average and time series models.

Definition of alternative		Total Losses from Forecasting (mm$), 1991-2005		Values of losses ($/unit)					
				Accuracy				Stability	
				Asymmetric Actual>predicted		Asymmetric Actual<predicted		(always symmetric)	
				Above or below threshold (linearity)					
Shape of loss function	Objective	If 2 year out time series model used for budgeting	If 2 year out 10 year moving average model used for budgeting	Below	Above	Below	Above	Below	Above
Symmetric and linear	Accuracy only	4,424	4,433	1	1	1	1	0	0
Symmetric and linear	Stability only	1,320	629	0	0	0	0	1	1
Symmetric and linear	Accuracy & Stability	5,744	5,062	1	1	1	1	1	1
Asymmetric (accuracy) and linear	Accuracy only	7,104	8,274	2	2	1	1	0	0
Asymmetric (accuracy) and linear	Accuracy only	6,128	5,024	1	1	2	2	0	0
Asymmetric (accuracy) and linear	Accuracy & Stability	8,424	8,903	2	2	1	1	1	1
Asymmetric (accuracy) and linear	Accuracy & Stability	7,488	5,653	1	1	2	2	1	1
Symmetric and nonlinear	Stability only	2,490	1,118	0	0	0	0	1	2
Symmetric and nonlinear	Accuracy & Stability	9,852	8,577	1	2	1	2	1	2
Symmetric and nonlinear	Accuracy & Stability	8,682	8,089	1	2	1	2	1	1
Asymmetric (accuracy) and linear	Accuracy & Stability	9,744	9,532	2	2	1	1	2	2
Asymmetric (accuracy) and nonlinear	Accuracy & Stability	8.808	6,282	1	1	2	2	2	2

(continued)

Table 17.6. Simulations of losses from variations in loss functions resulting from forecasting suppression expenditures using the 10 year moving average and time series models. ***(continued)***

Definition of alternative		Total Losses from Forecasting (mm$), 1991-2005		Values of losses ($/unit)					
				Accuracy				Stability	
		If 2 year out time series model used for budgeting	If 2 year out 10 year moving average model used for budgeting	Asymmetric Actual>predicted		Asymmetric Actual<predicted		(always symmetric)	
				Above or below threshold (linearity)					
Shape of loss function	Objective			Below	Above	Below	Above	Below	Above
Asymmetric (accuracy) and nonlinear	Accuracy & Stability	10,002	8,718	1	2	1	2	2	2
Asymmetric (accuracy) and nonlinear	Accuracy & Stability	9.504	9,392	2	2	1	1	1	2
Asymmetric (accuracy) and nonlinear	Accuracy & Stability	8,658	6,152	1	1	2	2	1	2
Asymmetric (accuracy) and nonlinear	Accuracy only	7,362	7,459	1	2	1	2	0	0
Asymmetric (accuracy) and nonlinear	Accuracy only	5,910	5,838	2	1	2	1	0	0

Table 17.7. Simulations of losses from variations in budget request percentage (probability of ruin) resulting from forecasting suppression expenditures using the 10 year moving average and time series models.

Shape of loss function	Total loss (mm$)	Asymmetric Actual>predicted		Asymmetric Actual<predicted		Budget request percentage (probability of ruin)
		Below	Above	Below	Above	
Symmetric and linear	4,375	1	1	1	1	44.3
Asymmetric (accuracy) and linear	6,199	2	2	1	1	35.6
Asymmetric (accuracy) and linear	5,906	1	1	2	2	64.5
Asymmetric (accuracy) and nonlinear	7,317	1	2	1	2	45.4
Asymmetric (accuracy) and nonlinear	5,807	2	1	2	1	44.3

and particularly time series with covariates that can explain variations in fire activity and costs beyond variation explained by lags of costs and time trends, may improve the accuracy of expenditure forecasts. These models could be informed by research reported by Swetnam and Betancourt (1990), Westerling et al. (2002, 2003) and Collins et al. (2006). Our modeling faced severe data constraints when seeking to understand the time series nature of wildfire suppression costs. We have short time series, which limited our inferential and model selecting abilities. Symptomatic of the data constraints was the fact that neither of the proposed time series out-year models provided forecasts that were statistically significantly better (using variance ratio tests) than the current agency approach to budgeting. However, there are likely to be other factors that will, and should, influence the choice of optimal forecasts, such as the costs to the agency, the public and the oversight agencies resulting from the always imperfect forecasts. We have described a set of procedures, involving loss functions, that could help agency decision makers to design a budget request tool that balances desires for both accuracy and stability or that minimizes the costs associated with over- or under-budgeting. For example, if the cost of having a budget that is insufficient to cover costs is very high, then the time series models may be preferred. Alternatively, if the cost of having too high a budget is very high or if the cost of a variable budget is very high, then the 10 year moving average is preferred. Because the 10 year moving average is the currently selected budget tool, this may reflect the possibility that the agency or U.S. Congress loss function has high costs of over-budgeting or variability.

Ultimately, advances in our understanding of how to forecast wildfire activity at multiple spatial and temporal scales can be achieved only through additional research and observations. Each new season of wildfire and suppression generates new information that can be used to identify better performing models. Research has shown that fire activity is closely related to droughts, precipitation levels, temperatures, and length of seasons (Schoennagel et al. 2003, Schoenberg et al. 2003, Westerling 2006, Kitzberger et al. 2007), many of which are forecastable using ocean temperatures, sea level pressures, and other ecological indicators. As climate science advances, longer-term climate forecasts may become available and be potentially useful for forecasting fire season activity and expected costs with greater accuracy.

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CHAPTER 18

TOWARD A UNIFIED ECONOMIC THEORY OF FIRE PROGRAM ANALYSIS WITH STRATEGIES FOR EMPIRICAL MODELING

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1. WHY A UNIFIED THEORY

Recent United States federal wildland fire policy documents including the 2001 policy update (US Department of Agriculture and US Department of the Interior 2001) call for integrated approaches to the national fire program. An important theme of these inter-agency policies is to encourage planning and budgeting across the major fire program components (e.g., suppression, fuels, prevention) in a consistent way. This means, for example, that planning and budgeting for the fuels (suppression) component is informed by the planning and budgeting of the suppression (fuels) component. In this chapter we specify the economic structure of a planning and budgeting system, as opposed to a component-by-component analysis. This structure shows, for example, that budgeting a federal system by program component is unlikely to promote efficiency. The structure also shows that the components can be managed in concert to capitalize on the complementary impacts they are likely to have on each other.

Implementing a unified theory in planning constitutes a major challenge across uncharted waters. Current planning approaches are largely based upon component specific models and budgeting is often executed as incremental adjustment to precedent. This chapter reaches beyond by deriving the essential principles of an integrated fire system in support of cost effective planning and budgeting. While some of our analytics used to derive the principles are complex, we do not intend to imply that budgeting systems need to reflect such complexity: only the essential principles.

Previous planning and budgeting models have focused on individual program components such as fuels management, suppression, or prevention with no direct or simultaneous consideration of the other components. This means that managing and budgeting the system of components in concert has been largely unattainable. Current models were not intended to directly address how the plans for initial attack (fuels treatment) are affected by a simultaneous consideration

T. P. Holmes et al. (eds.), The Economics of Forest Disturbances: Wildfires, Storms, and Invasive Species, 361–380.

of the plans for fuels treatments (initial attack). For example, the initial attack models used in the U.S. such as the National Fire Management Analysis System (NFMAS) and the California Fire Economics Simulator (CFES2) are specific to initial attack. Fuels models including advances designed by Hof and Omi (2003) are not intended to directly incorporate initial attack or suppression effects. In some instances these component-specific models can use the output from one component as input to another. This sequential approach to program interaction has serious limitations that can be improved upon by a fuller development of a system level analysis that attempts to more holistically address the problem.

While previous conceptual models (such as the least cost plus loss or cost plus net value change) address the balance between damage (net value change) and fire program level (preparedness), this chapter addresses wildland fire management at the system level by specifying each program component as part of a unified system. For further development of current management approaches, see other chapters in section IV of this book. Section two provides critical background on the fire program components and the key ways that they interact. Section three develops the core analytics of the unified theory at the system level. This structure serves as a potential foundation for addressing the principles of management and budgeting of the fire program components within a cohesive and unified system. For example, we show how the productivity of the fuels component changes the productivity of the suppression component. This section concludes with an application of the envelope theorem revealing a potentially refutable proposition regarding program cost effectiveness. In the last section, we identify alternative modeling approaches. These approaches inform the balance between the advantages of the unified theory and the pragmatic concerns of viable modeling and implementation. Implementation of a truly unified approach is perhaps impractical, but development of the theory will identify important principles, conditions, and implications related to policy analysis, budgeting and program implementation. We start by establishing the structure of the key relationships between the program components.

2. RELATING THE PROGRAM COMPONENTS

In this section, we review the three basic kinds of interactions among the program components:

- the *budgeting* process,
- *cost* structures,
- physical interactions among the *productivity* of the components.

In *budgeting*, funds allocated to one component often reduce funds available for another component. For example, allocating more funding to prevention may reduce funding available for suppression. This form of interconnectedness directly reflects scarcity through the fire program budget and appropriation processes. The economic principle often used to address budget scarcity across the components is to require equal improvement in each component per additional dollar spent. This is an application of what economists refer to as the equi-marginal principle (for example, Samuelson and Nordhaus 2001). While this is an important consideration, it does not provide a singularly compelling reason for developing a unified economic theory. The reason for this is that a common budget does not directly affect the underlying benefit or cost structure of the program. Separate program component levels could be independently adjusted up or down to conform to the equi-marginal principle[1].

Cost analysis by component is complicated and often frustrating because fire management resources (engines, aircraft, personnel, etc.) are interrelated through the cost function. A fire management resource, such as an engine, is often used to support multiple program components. For example, the purchase cost of an engine used in both fire protection and in fuel management would be joint, making it impossible to logically divide the purchase price of the engine between these two program components. In economics this is the well-known problem of joint cost allocation. Such cost considerations are not well addressed through a separate, or sequential analysis of program components. It is unlikely that a separate consideration of the program components will enable the planning or budgeting process to take advantage of the cost savings available in fire resources that are common across components. This can lead to redundant funding.

Interconnectedness in the *productivity* of the components has been long recognized, but it has not been well analyzed. For example, a major rationale for hazardous fuel reduction is to positively affect suppression efforts by reducing flame lengths, slowing fire growth rates, and enabling faster fireline construction. Budgeting and physical interactions among fire program components enjoy both a longstanding and consistent recognition. Sparhawk (1925) recognized the interaction between preparedness and suppression for fire management planning. More recently, Pyne et al, (1996, page 386) stated

> *"All of these activities and all these levels of management require planning. Especially as fire management enters a period of consolidation, plans by which to integrate program with program, agency with agency, region with region will assume ever greater importance."*

The most widely used fire economics model for planning and budgeting, known as least cost plus loss, or cost plus net value change, was not intended to address multiple program components. While numerous fire management models have been designed to address individual fire program components (McGregor 2005), none of those designed for U.S. federal lands have directly attempted to integrate multiple fire program components into an overall unified system. Additionally, advances in geographic information systems (GIS) and computing

[1] This would require the potentially awkward enumeration of a full set of funding levels for each program component, each to be compared to select the optimal mix of component levels.

resources applied to fire (Miller 2005, Finney 1998) enable a fresh look at the problem of system-wide resource allocation across the various fire program components.

The next section takes a philosophical departure from traditional approaches to fire management economics to provide the core analytics of a unified system.

3. A UNIFIED PROBABILISTIC ECONOMIC MODEL

To formulate a unified economic model of fire program analysis we assert that federal managers exhibit behavior consistent with cost minimization. While such behavior may not always reflect reality, this assertion has withstood the test of time for modeling purposes and we suggest that modeling such behavior is desirable at least as a benchmark of comparison to alternative behavior. The cost minimization assertion also aids in understanding and in recognizing a cost-effective fire program.

We begin by specifying a series of important conditions and assumptions. First, while recognizing that fire program management involves a full suite of program components, we develop our analysis with just two: hazardous fuel treatments and suppression. We will discuss prevention in this context without a substantive development. We define suppression broadly as the activities involved in extinguishing wildland fires while recognizing that for pragmatic purposes, suppression may be separated into initial and subsequent attack categories such as "large fire." Focusing our discussion and analysis on two components greatly simplifies and improves our ability to illustrate the underlying economic relationships. Expanding the analysis to include additional components such as fire prevention and ecologically based fuel treatments for site condition improvement is straightforward.

Our second assumption represents a major departure from many previous approaches. Here, we recognize that program planning and budgeting is performed in the context of managing for future fires and fire seasons that are *unknown* with respect to fire incidence, intensity, size, etc. The usual assumption that specific individual fire events expressing a future fire season workload can be modeled from historic events has been widely used and appears in models such as the Interagency Initial Attack Analysis system (IIAA) and in CFES2 (Fried et al. 2006). It has also been customary to model placement of fuel treatments based upon assumed ignition locations such as the Monte Carlo simulations of Hof and Omi (2003) or as in FlamMap (Finney 2005). The Monte Carlo simulations and the FlamMap application both use stochastic processes to establish the location of modeled fires. However, once the locations are established, even if they are established using a stochastic process, they become a "known" set or distribution of fires and all information on the likelihood of having a fire in a given location is lost. The "known fire" assumption introduces two important challenges. First, assuming knowledge of the ignition point plays an important role in affecting the solution as to placement of fuel treatments (Hof et al. 2000). Second, modeling individual known ignitions suggests, or may require, tactical management and modeling of the individual fire event(s).

Event-based modeling introduces a potential philosophical inconsistency between program-level and the tactics of event-level analysis. For example, current event-based models require management of individual fire events that belong to a set of events. This potential inconsistency can be overcome by incorporating a probabilistic production function. Although future fires and fire seasons are unknown, we assume that the probability of fire occurrence by intensity and location can be estimated using established probabilistic methods. For example, recent research such as that by Prestemon et al. (2002) introduced econometric approaches to wildland fire occurrence that also focused on probabilistic functions. For a fuller development of how to model disturbances at broad spatial and temporal scales using a probabilistic framework, see chapter 3.

Abstracting to a probabilistic production function eliminates the potential philosophical inconsistency of modeling individual events to analyze an entire program and it better conforms to the scale of analysis often needed to address the wildfire program. Abstraction to a probabilistic production function introduces new challenges regarding the availability of information. We therefore assume that there is (or could be) the technology and resources to generate credible information regarding the probability of fire incidence and behavior to create spatially explicit "probability" maps of burn probability across the landscape.

Our fourth assumption is that the productivity of each fire program can be represented by changes to the landscape probability map. Representation of program productivity is essential to any production based economic analysis; it is unavoidable. Because our probabilistic approach abstracts away from the individual fire event, it symmetrically abstracts away from the individual fire resource. Our intent is to focus on the fire program and its relation to the program components. We therefore concentrate on how changes in each program component would, in principle, change the unifying probabilistic production function that would ultimately be represented as a landscape map.

Finally, we recognize that spatial and temporal interrelationships are important. Spatial relationships are important because the probability of fire at a given location is influenced by conditions at neighboring locations. For example, the probability of fire in a given location (e.g., a geographic information system (GIS) raster cell) is a function of that cell's fire producing attributes and of the attributes of neighboring cells. Recent advances in GIS technology and in fire applications of GIS technology reflect this concept well (Finney 1998 and Miller 2005). Temporal considerations have historically been associated with fuels management because investments in treatments provide returns over time and they affect the structured pattern of optimal treatments over time. Other program components including suppression often provide benefits or impacts for many years and thus are equally well suited for temporal analysis. For example, aggressive suppression is commonly asserted to have led to a long-term accumulation of fuels.

While a probabilistic production function enables a more robust integration of the spatial and temporal interactions, we use a static model as a simplifying first step[2] because our focus is on the theory and its related principles. In the same way that the well established "theory of the firm" provides a theoretic framework that reveals principles and structure, as opposed to operational detail, we formulate a static economic model to capture the key underlying structure of the wildfire problem across program components. We note that firms face important intertemporal choices, including long-term investments. While this limits the applicability of the static theory, the static theory continues to provide a rich foundation for intertemporal analysis including the development of capital theory (for example, see the classic by Hirshleifer 1970).

We begin with the most general structure in (18.1a) to minimize a budget constrained expected loss (Z) from wildfire where program components, fuels (F) and suppression (S) are modeled as decision variables.

$$MinZ = \Lambda[P(F,S)] + \lambda(B - C(F,S)) \tag{18.1a}$$

Where:

> Λ (capital lambda) denotes a general loss function of burn probability P across the program.
>
> P(F,S) denotes the probabilistic production function for the program.
>
> C denotes the cost function of the fire program
>
> B denotes the fire program budget
>
> λ (Lambda) denotes the Lagrange multiplier for the program budget constraint.

First, we note that if performance is measured in the same units as cost, such as dollars, then the budget constraint could be omitted assuming the objective would be to solve for *the* optimal levels of program components to minimize the total cost. However, we include the budget constraint as a central feature for two important reasons. First, regardless of analysis aimed at identifying economically efficient levels of program components, public budgets are appropriated at levels that are dependent upon the appropriations process and there is no evidence to suggest that appropriations are economically optimal. That is, even if we solved for the optimal level of the program, we should not expect it to be appropriated. Instead, it is more realistic and useful to incorporate the budget as a "hard" constraint to illuminate the economic principles required for efficient allocation of a fixed, but unknown budget. Modeling a budget constraint

[2] Static formulations of the theory are customary in fire management (Rideout and Omi 1990)

better demonstrates how alternative appropriations affect decisions and performance. Secondly, since the Government Performance and Results Act (1993), federal agencies, including the agencies entrusted with wildfire management, are required to engage in performance based planning and have increasingly relied upon physical measures of performance[3] that are problematic to measure in dollars.

The general function Λ(P) translates the physical impact of fire into a present value of expected loss. Therefore, Λ(P) depends upon the resources affected, the fire intensity, seasonality, and potentially on the extent of risk aversion. The function Λ(P) allows for risk aversion where increasing probabilities by intensity level may be non-linearly related to the value of loss from fire. We typically expect increases in fire probability to cause increases in expected fire loss. For more on the economic impacts of wildfire see related chapters in section III. In a risk-neutral program, increases in probability would increase the expected value of loss at a constant rate, or price (L), such that $\frac{\partial \Lambda}{\partial P} = L$ and $\frac{\partial^2 \Lambda}{\partial P^2} = 0$. In a risk neutral program, L denotes a constant "price" of fire risk. For risk-averse management, the second derivative of Λ with respect to P is positive, $\frac{\partial^2 \Lambda}{\partial P^2} > 0$, indicating that the importance of loss (or loss mitigation) increases with increasing loss probability. Risk aversion suggests that fire program managers would be willing to disproportionately allocate fire resources in an effort to avoid higher expected losses resulting from higher fire probabilities. Risk aversion may be especially prevalent with respect to the probability of high intensity fires and for fires threatening highly valued resources such as fires in the wildland urban interface.

While fire managers may be risk averse, and this topic deserves further investigation, we continue with the customary simplification of risk neutrality in public management so that the expected loss can be expressed linearly as:

$$MinZ = L \bullet P(F,S) + \lambda(B - C(S,F)) \tag{18.1b}$$

Here we substitute *L* for Λ to denote the customary but special case of risk neutrality. The cost function in (18.1a) and (18.1b) is generalized to support appropriate specification as needed. An important consideration in program component analysis is the economic problem of joint costs discussed earlier.

[3] Our development reflects a single program appropriation that we assume is observable. In the event that the program components are separately appropriated, we would introduce a separate budget constraint for each component. Independent program appropriation is problematic to the extent that costs are joint between the program components.

When program components share fire management resources, jointness in cost will inevitably occur. We therefore add structure to (18.1b) to accommodate the joint and separable costs of the program components. Equation (18.2) includes terms for the separable costs for each program component (SCS only for suppression and SCF only for fuels) and for the program joint cost (JC).

$$C(F,S) = SCS(S) + SCF(F) + JC(F,S) \tag{18.2}$$

By substituting the cost function from (18.2) into (18.1b) we arrive at (18.3) that includes our probabilistic production function, the assertion of risk neutrality, the recognition of jointness between the program components and a fixed budget or appropriation that would be fully allocated to the components to minimize overall program loss.

$$MinZ = L \bullet P(F,S) + \lambda(B - SCS(S) - SCF(F) - JC(F,S)) \tag{18.3}$$

Also note that L denotes a constant price of fire loss and P denotes the probability function of burns under the fuel treatment level F and suppression level S. Using subscripts to denote partial derivatives, the first order conditions for minimization of (18.3) are expressed as:

$$Z_S = L \bullet P_S - \lambda\ (SCS_S + JC_S) = 0 \tag{18.4a}$$

$$Z_F = L \bullet P_F - \lambda\ (SCF_F + JC_F) = 0 \tag{18.4b}$$

$$Z_\lambda = B - (SCS + SCF + JC) = 0 \tag{18.4c}$$

The first-order conditions reflect the usual marginal benefit-cost condition that the change in expected loss ($L \bullet P_S$ or $L \bullet P_F$) is equal to the marginal cost of the program component ($SCS_S + JC_S$ or $SCF_F + JC_F$) adjusted for the shadow price of the budget restriction (λ). For example, suppression would be applied (18.4a) until the increase in cost (joint plus separable), adjusted for the shadow price of budget restriction (λ), equals the decrease in expected loss. A parallel interpretation is made for (18.4b).

It might be tempting to interpret the fuel and suppression first-order condition as stating that allocations are made to each component until the components marginal cost (adjusted by λ) equals the reduction in loss, but this would be incorrect. The terms JC_S and JC_F denote the increase in joint cost from an increase in suppression (fuels) effort; not from an increase in suppression (fuels) cost or appropriation. The implications of this are important in today's inclination to budget by program component. Where joint costs matter, budgeting for fuels is budgeting for suppression and visa-versa. As discussed above, there is no logical way to divide the joint portion of cost by component. Instead, recognizing that costs cannot be fully divided by component, at least logically, implies that appropriating the system instead of the component deserves consideration.

Dividing (18.4a) by (18.4b) yields (18.5) revealing the familiar equi-marginal principle.

$$\frac{P_S}{P_F} = \frac{(SCS_S + JC_S)}{(SCF_F + JC_F)} \tag{18.5}$$

In this context, the principle is interpreted as stating that for a constant loss rate, L, and budget, B, the ratio of the reduction in fire probabilities of the two programs must equal the ratio of their respective marginal costs. Setting the marginal costs equal to one dollar directly produces the usual interpretation of the principle. Here, minimization requires the addition of a dollar to each program component to yield equal reductions in wildfire probabilities. While directly observing such a ratio is unlikely, this interpretation provides a powerful conceptual tool for understanding a fundamental condition required for cost minimization under multiple program components.

The marginal cost of each program is assumed to be positive, requiring that the sum of costs in parentheses is positive. The "strong" case for this is that the marginal joint and separable costs are each positive for each component. Condition (18.4c) requires the budget to be fully allocated to the program component separable costs plus the program joint cost. Because increases in program components reduce the expected loss ($L \bullet P_S$), the marginal value of the program component $L \bullet P_S$ is negative because P_S is negative. This relationship is illustrated in figure 18.1 where the reduction in expected loss is shown to diminish with increasing suppression holding all else constant.

While (18.5) focuses on comparisons between the program components, λ addresses the value of another dollar to the program as defined in (18.6).

$$\lambda = \frac{\partial Z}{\partial B} = \frac{P_F}{C_F} = \frac{P_S}{C_S} < 0 \tag{18.6}$$

We can think of λ as having two equivalent interpretations: the first defines the value of another dollar to the fire program while the second defines the equilibrium condition for expenditures between program components. From left to right, the second term $\frac{\partial Z}{\partial B}$ denotes the marginal value of an increase in budget in reducing program loss. Because increases in program budget are fully expended on program components to reduce loss, λ is negative[4]. This is consistent with the signs of the ratio of partials P_F/C_F and P_S/C_S. The equality of the ratios explains that a minimum is achieved when the probability reduction per unit cost increase

[4] Further, to the extent that Z is strictly concave, as in fig. 18.1, the rate of change of λ* with respect to the budget would also be negative denoting declining marginal benefit of increasing budgets.

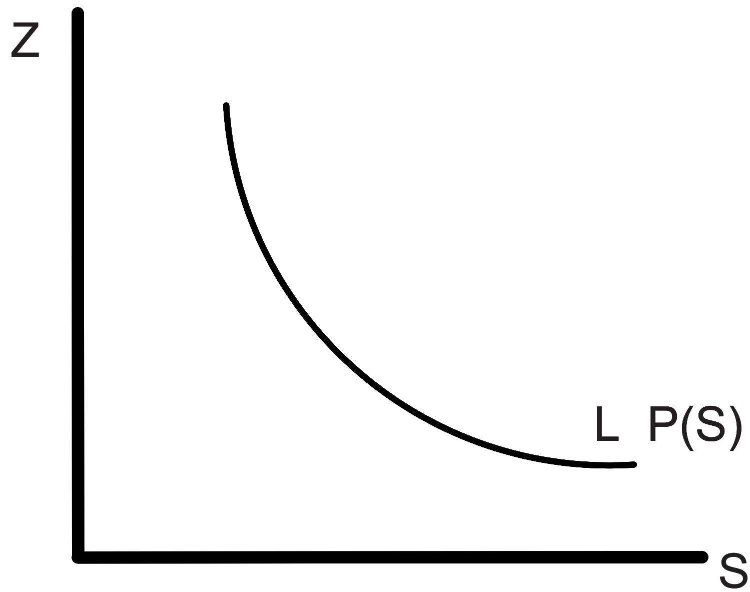

Figure 18.1. Loss as a function of suppression.

is equilibrated across the program components. Note that these ratios are negative as the reduction in probability from a program increase is negative while the marginal cost of each program component is positive.

Prevention is often considered as another important program component that is managed under the overall system. Consider that a key role of any prevention and education program is to reduce the probability of human-caused ignitions. Because prevention can be conveniently expressed as affecting probability, it fits well into the probabilistic framework. Specifically, the probabilistic production function (18.1a and 18.1b) is directly modified to include the prevention component "V" such that P = P(F,S,V). All of the conditions developed above, and below, for the relationship between fuels and suppression can be directly expanded to consider the prevention program component and its interactions.

The second order conditions are of particular interest because they reveal the conditions for program component complementarity and ultimately reveal the program supply condition. These conditions are often assumed to hold, and then swept away to simplify the development. However, they capture the interactions that are the theme of this chapter so we encourage an extra dose of caution and persistence on the part of the reader. It will enhance the marginal product to economic knowledge. Conditions (18.7) list the second order conditions while ignoring the redundant cross partials[5].

[5] For a minimum we assume the sign of the bordered Hessian is negative.

$$Z_{SS} = L \bullet P_{SS} – \lambda\ (SCS_{SS} + JC_{SS}) \tag{18.7a}$$

$$Z_{FF} = L \bullet P_{FF} – \lambda\ (SCF_{FF} + JC_{FF}) \tag{18.7b}$$

$$Z_{SF} = L \bullet P_{SF} – \lambda\ JC_{SF} \tag{18.7c}$$

$$Z_{\lambda\lambda} = 0 \tag{18.7d}$$

$$Z_{\lambda F} = -SCF_F – JC_F \tag{18.7e}$$

$$Z_{\lambda S} = -SCS_S – JC_S \tag{18.7f}$$

First we note that Z_{SS}, Z_{FF}, P_{SS} and P_{FF} are each positive reflecting diminishing returns (convexity consistent with fig. 18.1) and that the sum of cost terms would be positive so long as marginal cost increases with increases in the level of each program component. Therefore, to the extent that the budget constraint is binding, $L \bullet P_{SS}$ exceeds λ $(SCS_{SS} + JC_{SS})$, indicating that the marginal value product function will be increasing faster than the budget adjusted (λ) marginal cost function with respect to increases in each program component.

For complementary program components Z_{SF} is negative defining component "synergism." This is the fundamental rationale for managing program components within a unified program. The "strong" case for Z_{SF} being negative is that program components are known to be complementary (see discussion above) in production (indicating that P_{SF} is negative) and in cost making JC_{SF} also negative. Complementarity in the production function is represented by a negative P_{SF} indicating that the marginal product of suppression (fuels) is enhanced by increases in fuels (suppression). Because the positive function Z is being minimized, improvement is denoted by reducing Z such that complementarity is represented by the negative cross-partial. A weaker argument for component complementarity is achieved through complementary production (cost) so long as it is not overwhelmed by a substitution effect in the budget constrained cost (production) function[6]. For example, if the programs are complementary overall ($Z_{S,F} < 0$) and complementary in reducing wildfire probabilities $P_{S,F} < 0$, but if the cost of one component adversely affects the marginal cost of another component the cost function ($JC_{S,F} > 0$, which we suspect is unlikely) then $P_{S,F}$ would need to exceed $JC_{S,F}$ to preserve complementarity.

Synergism in the components with respect to productivity (the probabilistic loss function) addresses the interaction between suppression and fuels and is illustrated in figure 18.2.

In figure 18.2 the effect of suppression on reducing expected loss before a fuel treatment is denoted by F_O. Following the fuels treatment, the marginal productivity of suppression is improved by reducing expected loss at all levels

[6] While program component substitution (Z_{SF}) is unlikely, substitution is also best managed through combining components into a common system. Only $Z_{SF} = 0$ suggests separating the components into independent management.

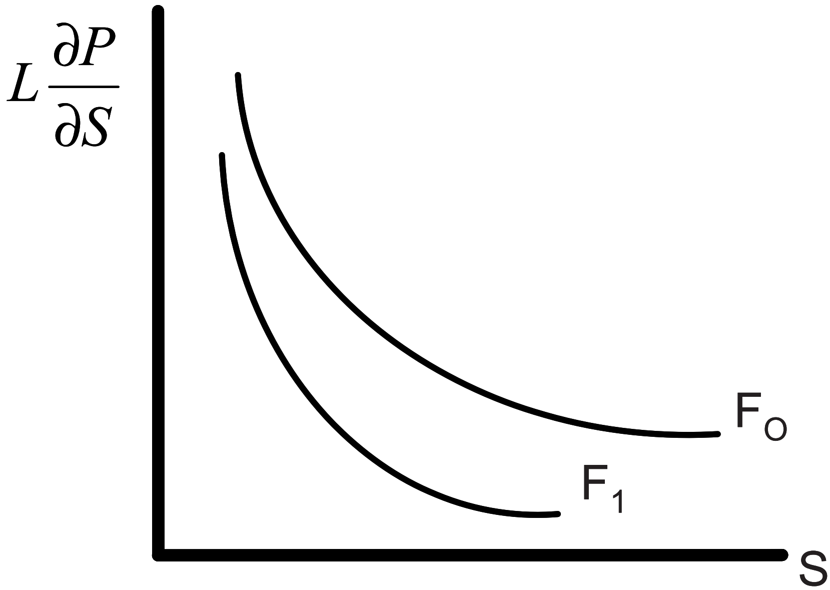

Figure 18.2. Marginal value product of suppression at two levels of fuel treatments.

of suppression and, importantly, by increasing productivity of suppression forces as denoted by the steeper slope of the F_1 function. This illustrates the normal complementary relationship between inputs fuels and suppression.

Increasing fuels treatments (suppression) may enhance the marginal product of the suppression (fuels) program. This can be seen in the example of a fuels treatment that both reduces hazardous fuels, but also increases the marginal productivity of the suppression activities by improving the physical environment for suppression as in figure 18.2. Fuel treatments may make suppression forces more able to move through the forest, improving line production and they may make fire lines hold better. Consider a fuel treatment designed to reduce the intensity of wildfire with the assumption that suppression forces are best designed to contain low intensity fires. This crucial interaction between the productivity of the program components remains largely unquantified. However, management of the components in common federal programs provides evidence of a complementary interaction at the program level and there are many landscape level examples where fuel treatments have stopped or diverted the progression of intense wildfires, e.g., the 2002 Rodeo/Chediski Fire in the Apache-Sitgreaves National Forests in Arizona (Schoennagel et al. 2004). Additional evidence is provided by fireline production rates that are adjusted by fuel type. For example, fuel treatments are often intended to change fuel types in ways that reduce fireline intensity while improving fireline production rates (Haven et. al. 1982, Hirsch et. al. 2004).

How often have you heard that increases in fuels treaments will reduce suppression expenditures? The statement is difficult to evaluate without considering the implied level of damage. Perhaps implicit in such statements is the notion of a constant level of damage. By holding the expected value of loss constant (Z°), we can envision the tradeoff between the two components using the construct of the iso-loss function shown in figure 18.3.

For a constant level of expected loss (Z°), consider alternative mixes of S and F. The slope of the Z° function is known as a "marginal rate of substitution" and it does not imply that the program components are properly considered substitutes. Program component substitution is shown in figure 18.2. For additional material on the tradeoff between program components, see chapter 16.

Understanding why we can substitute fuels for suppression for a given level of expected loss (fig. 18.3), while the program components themselves can be defined as complements (18.7c and fig. 18.2) is crucial to understanding the economic structure of a wildfire program. Planning documents often promote increasing fuels treatments as a means of reducing suppression costs. While this may apply for a constant level of expected loss (fig 18.3), if the fuels treatment improves the productivity of suppression (fig 18.2), then the optimal level of suppression may actually increase. Therefore, to the extent that the fuels and suppression components exhibit normal complementarity, the promise of reduced

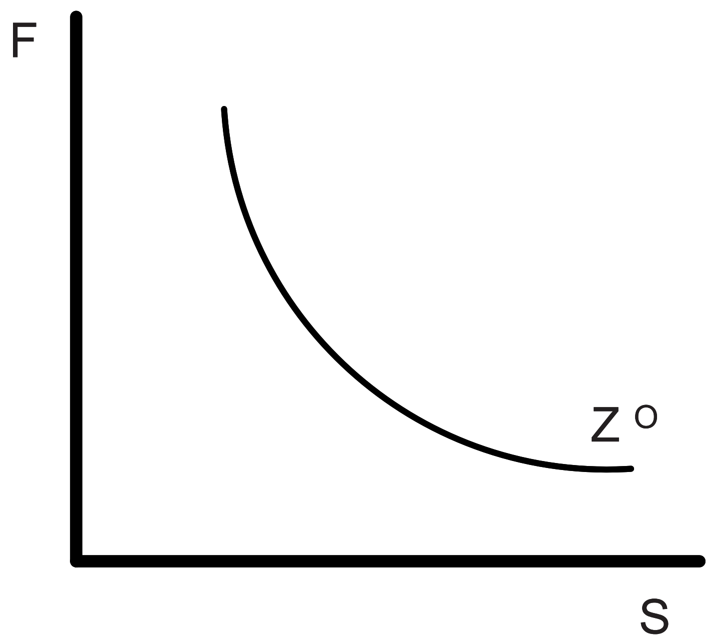

Figure 18.3. Varying fuels and suppression levels to produce a constant expected loss.

suppression expenditures resulting from fuels treatments should be made with copious caution and qualification. For example, behavior that is inconsistent with cost minimization, such as maximization of initial attack success rate, might not reflect such complementarity.

Finally, the cross partials with respect to the budget constraint, $Z_{\lambda F}$ and $Z_{\lambda S}$ in (18.7e) and (18.7f), each reveal that the marginal value of the budget is affected by its marginal cost of each program component. For example, (18.7e) denotes that an increase in the fuels component affects the marginal value of the budget (λ). The implication of this is that λ^* changes with changes in the program component marginal cost. When a program component level is altered, minimization requires re-equilibrating between program components (when changing the component involves changing the marginal cost). The marginal value of a dollar added to the program changes with changes in the component's marginal cost.

Further application of the second-order conditions reveals the interesting comparative static result when the slope of the minimized loss function Z*(L,B) in figure 18.4 is analyzed.

Consider the cost minimizing indirect loss function (Z*) where optimal levels of program components F* and S* have been applied.

$$Z^*(L,B) = L \bullet P(F^*,S^*) + \lambda^*(B - C(S^*,F^*)) \tag{18.8}$$

Differentiating Z* with respect to L once and twice yields the envelope result denoted in (18.9a) and (18.9b) respectively.

$$\frac{\partial Z^*}{\partial L} = P^* > 0 \tag{18.9a}$$

$$\frac{\partial^2 Z^*}{\partial L^2} = \frac{\partial P^*}{\partial L} < 0 \tag{18.9b}$$

Figure 18.4 illustrates the indirect loss function Z* that forms a strictly concave envelope. Setting Z equal to Z* we see that slope of the Z and Z* functions each equal P*, but that the <u>slope</u> of the Z* function is declining while the <u>slope</u> of the Z function is constant at P*. Because this change in slope is equal to the change in probability with respect to the change in the unit value of loss (18.9b), our model reveals that cost minimizing behavior will reduce the probability of loss (P*) in response to an increasing the price of wildfire damage (L). This would reflect an upward sloping supply function for damage reduction. This comparative static result is of some importance because it provides a testable and potentially refutable proposition, regarding fire management that has not been investigated. Accepting or rejecting (18.9b) as a hypothesis provides evidence for accepting or rejecting the unobservable assertion of cost minimization in fire program management. Equation (18.9b) is the only comparative-static result available because the other model parameter, B, enters the constraint[7] and not the loss function.

Movement toward a unified economic theory of wildfire analysis provides many economic principles that enhance our ability to understand and model fire systems. The theory reveals many new and important principles in the context of fire program analysis. The theory also provides guidelines to develop specific fuel treatment and suppression resource allocation models. Compared with previous approaches focused on individual program components, the unified approach emphasizes the importance of integrating multiple program components by utilizing their joint productivity and joint costs in order to improve the overall fire management efficiencies. However, integrating multiple components in a fire management project remains a challenge from both theoretical and practical aspects. To address this challenge, we next discuss related empirical modeling strategies. Through the discussion, we hope to gain better understanding about how different integration strategies can be used to fully or partially capture the joint production and cost that are often ignored by previous modeling strategies focused on separate program component.

4. EMPIRICAL STRATEGIES FOR INTEGRATING PROGRAM COMPONENTS

While development of the theory provides a framework for analysis and for understanding the principles at work in program management, it does not show how such principles could be implemented. Consequently, this section will explain the basic approaches that could be considered in formulating applications of the unified theory. In defining empirical strategies, we will assume the landscape can be spatially represented as a raster map.

4.1 Individual Components

We review the fire program components individually and then address approaches for formulating models for an integrated system.

4.1.1 Suppression component

We assume fire suppression resources reduce the fire probability in a given cell. This represents an important departure from the traditional method of modeling specific fires or fire events. Consider a landscape where the existence of suppression resources at particular dispatch points could be used to decrease the expected fire loss (L•P) within a certain distance from that point. If we call the area under

[7] Envelope results with respect to B would require analyzing changes in B while program component levels changed. This would violate the constraint.

the influence (control) of suppression resources at a certain dispatch point an influence zone, an operations research based fire suppression model could be used to allocate suppression resources to reduce the expected fire losses. As suppression resources are allocated to particular dispatch locations, they would reduce the fire probability inside the influence zone and mitigate the accumulated fire losses across the landscape.

4.1.2 Fuel treatment component

Fuel reduction programs such as prescribed burning or mechanical treatments are frequently used to reduce hazardous fuels under the consideration of spatial aspects of fire spread (Loehle 2004). Fuel treatments decrease the expected fire loss by changing the fire behavior, including fire intensity and spread rate. An approach consistent with the unified economic model would be reflected in a probability based allocation under the assumption that fuel treatments are not aimed at specific fire events, but focus on the likelihood of ignition locations and spread patterns. Effective fuel treatments need to be located in places that can efficiently decrease the overall expected fire losses in a landscape.

4.1.3 Prevention component

Much like the suppression component, consider a landscape where the probability of human caused ignitions can be potentially reduced through prevention activities. By spatially locating prevention activities, such as signage and law enforcement efforts, the spatial area can be mapped as an influence zone. Considering the effects of prevention activities on the probability surface, prevention activities can be coordinated with fuels and suppression programs.

4.2 Integrating the Components

Under the framework of a unified economic model, fuel treatments and suppression combine to mitigate the fire probability while sharing a common budget. The difference between the approaches discussed below depends on how the interactions are addressed. We focus on five broad strategies that might apply to a variety of specific model formulations: 1) non-linear, 2) total enumeration, 3) serial, 4) joint impact and 5) additive. They are presented in an order of decreasing complexity.

4.2.1 Non-linear approach

The nonlinear approach recognizes the inherent dependencies of the interactions between the fire program components. To the extent that fuel treatments and suppression are complementary, a "synergism" between them is denoted by the negative cross partial ($P_{SF} < 0$). The negative cross partial indicates that either the fuel program or suppression program could potentially increase the marginal productivity (more effective at decreasing the probability of fire) of the other. This negative cross partial supports the development of a unified economic system for efficient fire management. Theoretically, the non-linear approach fully accounts for the joint cost and joint production of all fire program components. The difficulties associated with the non-linear approach are in the generation of practical formulations and solutions. Non-linear formulations are notoriously difficult to solve and often require linear approximations.

4.2.2 Total enumeration

Total enumeration accounts for the non-linear dependencies and avoids the complex non-linear formulation. Here, all of the possible combinations of management actions on a given cell are identified as a possible solution. The model would then choose the "best" combination of management components. The exhaustive list of possible combinations for each cell accounts for all of the synergies and interactions between the components but is often impractical because of the large number of possibilities. Subsets of this approach are the serial and the order indifferent approaches explained below. Solution techniques such as Bayesian Belief Networks and Influence Diagrams might also be considered here.

4.2.3 Serial

To reduce the number of possible combinations, the serial approach can limit the interactions to just one dependency. For example, for practical purposes, we might assume that fuels treatments have a large impact on optimal positioning of suppression resources. Initial attack models that use a given fuel model as an input provide an example of this. We might assume that locating suppression resources has negligible impact on our fuels planning. This assumption reduces the number of combinations that need to be considered for modeling purposes while accounting for the increased marginal productivity of one component.

4.2.4 Joint impact with no interaction

This simplifying approach assumes that a cell can receive treatment from either or both components but that treatments do affect each other. Suppression or fuels treatment impact cell probability individually. However both may be applied to capture the full effect of the combination. The strength of this approach is that it keeps the application linear while enabling the model to capture any joint (synergistic or detracting) effect on the reduction of probability. The number of possible solution combinations is reduced while accounting for the individual or total impact of treatment application by component.

4.2.5 Additive approach

The additive approach assumes that the reduction in the probability of loss from both fuel treatments and suppression actions can be added linearly. Diminishing returns in each component (P_{SS} and $P_{FF} > 0$) and potential nonlinear interrelationship between components ($P_{SF} = P_{FS} \neq 0$) are both ignored by assuming $P_{SS} = P_{FF} = 0$ and $P_{SF} = P_{FS} = 0$. Because this approach only considers the additive relationship between components, it addresses the joint impact on the probabilistic production function as a simple linear approximation. The difference between this approach and the joint impact approach is that there is no opportunity here for jointness, e.g., synergism, if impacts are strictly linear.

5. CONCLUSIONS

The unified economic theory represents a potentially important advancement in the economic modeling of wildland fire. The principles of program component interaction illuminate the advantages of managing suppression, fuels and prevention under a single program. To develop the unified theory we focused on three key interactions among the program components: cost, productivity (substitutes and complements) and a common budget/appropriation. Each requires careful consideration in any implementation effort. While these interactions can be modeled in the comparative-statics framework used above, developing operationally meaningful strategies and model formulations constitutes an enormous challenge across uncharted territory.

Modeling these interactions in the context of a unified economic theory will likely prove to be a key challenge in this new era of strategic fire management and planning. We outlined five modeling strategies to illustrate key considerations of the implementation problem. By assessing these strategies, we conclude each has strengths and weaknesses and none, except the non-linear strategy, fully capture the interactions analyzed in our unified economic theory. The choice of modeling strategy may ultimately depend upon the scale of application, the information needs of managers and upon the need to demonstrate cost effectiveness in the program at the federal level. Regarding scale, it is likely that cost and product interactions would be less prevalent at coarse scales. While coarse scale modeling moves beyond meaningful interactions at the landscape level, there are still strategic or national wildland fire management resources, such as smoke jumpers and air tankers to consider that will involve joint costs and product interactions. Smaller scale analysis, such as provided by the landscape level analysis, will involve extensive consideration of joint costs and product interactions such that approaches resembling the non-linear strategy may have more appeal. Consequently, we reach two additional and potentially important conclusions. First, the unified theory provides a powerful tool for addressing and evaluating the design of integrated program components. Secondly, because there currently is no empirically based modeling approach that will fully capture the problem, difficult choices are required regarding modeling strategies.

While the pragmatics of implementation often require sacrifices in the theory, the theory represents an important advancement beyond the basis used for current planning and budgeting systems. Even with the more robust theoretical foundation of the unified theory, much enhancement will be required by enriching the spatial integration and inter-temporal choice analysis.

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CHAPTER 19

ECONOMIC ASPECTS OF INVASIVE FOREST PEST MANAGEMENT

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1. INTRODUCTION

The past decade has evidenced growing concern with the causes and consequences of biological invasions, many of which are economic in nature (Perrings et al. 2002). The risk of a new pest introduction is positively correlated with world trade flows (Costello and McAusland 2003, Margolis et al. 2005) and new invasions threaten the productivity and biological diversity of native ecosystems (Mack et al. 2000). A recent study reports that roughly 50,000 exotic species are established in the United States and annual domestic costs and annual losses from invasive species (forest and non-forest) may exceed $120 billion (Pimentel et al. 2005). The passage of Executive Order 13112 (Clinton 1999), which enhances federal coordination and response to invasive species, and the creation of the National Invasive Species Council (NISC 2001, NISC 2005), are evidence of the federal government's substantial concerns with these emerging threats to terrestrial and aquatic ecosystems.

Forests provide suitable habitat for an assortment of invading organisms (Liebhold et al. 1995) and invasive species have been ranked as one of the four critical threats to our Nation's forest ecosystems by the Chief of the U.S. Forest Service (USDA Forest Service 2004). Although most people might argue that it is laudable to counter threats to the structure and functioning of forest ecosystems, relatively few exotic organisms become a major pest (Williamson 1996). It is the main thesis of this chapter that decisions regarding budget allocations and the targeting of forest protection efforts would benefit from a clear understanding of the costs and benefits of invasive forest pest management. Interventions designed to mitigate damages from exotic forest pests are costly—the Forest Service spent $95.1 million dollars for the management of invasive forest pests in fiscal year 2005 (USDA Forest Service 2005, p. 14-55). However, very little is known about the magnitude of economic damages caused by exotic forest pests, or the efficacy of the money spent on pest control. This lack of knowledge impedes economic analyses of pest management programs and it

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remains unclear whether current expenditures on invasive forest pests are too little, too large, or about right.

The goal of this chapter is to provide an overview of some salient economic aspects of invasive forest pest management. Our synopsis begins with a broad economic characterization of the invasive species management problem. Following this, we provide a brief review of management strategies that have been applied to combat select invasive forest species in the United States. We then turn to a case study of a recent threat to forests in the eastern United States, the hemlock woolly adelgid, and emphasize the importance of public awareness and private action to strengthen links in forest health protection.

2. ECONOMIC ANALYSIS OF INVASIVE SPECIES

To begin, a generalized biological invasion can be described by a sequence of ecological states (fig. 19.1). Economic analysis is relevant to the design of invasive species management programs because each ecological state is associated with one or more management actions and a vector of economic costs and losses. An economic approach to invasive species management seeks to minimize the sum of management costs plus losses to trade, domestic market production, and non-market economic values.

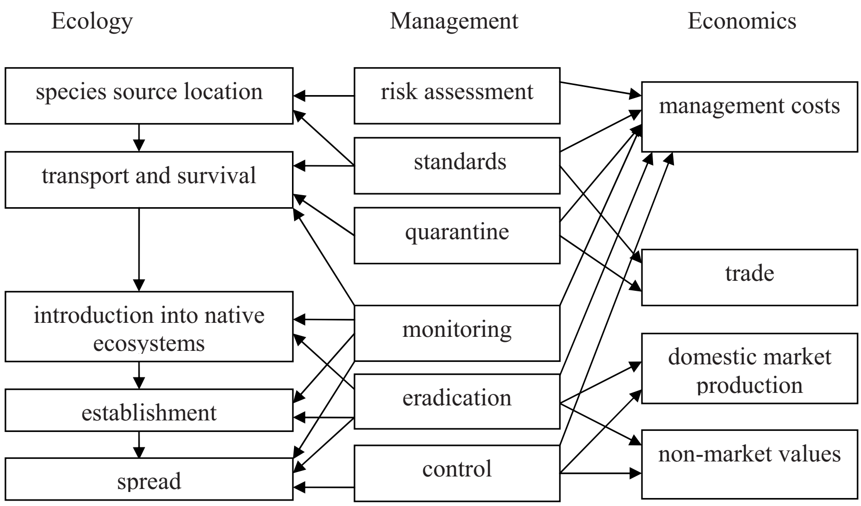

Figure 19.1. A general, conceptual model of the relationships between the ecological stages of a biological invasion, management responses, and economic impacts. Social welfare is optimized by minimizing expected management costs plus the expected losses to trade, domestic market production, and non-market values.

In this section, we describe two prominent issues in the design of optimal economic programs for invasive species protection. First, forest protection programs provide services that are public goods, and private landowners cannot be expected to provide the socially efficient level of forest protection. Thus, government has an essential role in the provision of forest health. Second, the time lag between the investment of capital and labor targeted at forest protection and the observation of a change in physical or economic damages to forest ecosystems introduces substantial uncertainty into both public and private decision-making. Although economic frameworks have been developed to improve decision-making under conditions of uncertainty, major challenges remain in the implementation of optimal economic programs. In such an environment, Bayesian methods provide a promising approach to adaptive management given uncertainty regarding models, parameters, and data (see section 2.2.3).

2.1 Weakest-Link Public Goods

Any evaluation of the optimal level of investment in forest health protection, either from the perspective of private or public forest owners, needs to recognize that forest health protection is a public good. The benefits of a quarantine, for example, are non-excludable (if quarantine benefits are made available to one person, they are available to every member of the community), and non-rival (the benefits received by any individual do not decrease the level of benefits available to others). As is well known, the standard theory of public goods argues the private provision of public goods is sub-optimal if self-interested individuals equate the marginal cost of their investment in public good provision with their marginal private benefits, but do not account for the benefits received by other members of society (Samuelson 1955).[1] Because the level of private provision is socially sub-optimal, governments have a key role to play in providing the socially optimal level of forest health.

Government-sponsored provision of forest health protection proceeds using an assortment of strategies (section 3), some of which necessitate the involvement of private landowners and households. In the standard public goods model, the socially available amount of a public good such as forest health protection (H) is the sum of the amounts (h_i) produced by community members (i): $H = \Sigma h_i$. As highlighted by Hirshleifer (1983), other social composition functions are possible. Of particular importance to the design of invasive species management programs is the concept of a weakest-link public good (Shogren 2000, Perrings et al. 2002). The weakest-link aspect of biological invasions arises from the condition that each member of a social group (say, forest landowners) has a "kind of veto power

[1] A game theoretic formulation of the provision of public goods by self-interested individuals is a Prisoner's Dilemma where the dominant strategy, not providing the public good, is sub-optimal to both players providing the public good (Harrison and Hirshleifer 1989).

over the extent of collective achievement" (Hirshleifer 1983, p. 373). Just as the strength of a chain depends upon its weakest link, or the protection provided by a system of levees depends upon its lowest height, the aggregate provision of forest health protection is compromised by forest landowners within a community who fail to take actions to protect their land from invasive forest pests, thereby increasing the risk for other forest landowners in the community. Hirshleifer (1983) argues that the weakest-link social composition function is given by the level of protection provided by the weakest member. Thus, in the case of forest health protection, the weakest-link social composition function is $H = \min(h_i)$.

When the weakest-link character of the social provision function is understood by each member of the community, the value of $\min(h_i)$ can be raised, perhaps dramatically. Anecdotal evidence supports the proposition that improvements to the weakest-link occur when a social threat is perceived to be overwhelming, as is sometimes evidenced in the aftermath of a natural disaster (Hirshleifer 1983). Further, experimental evidence has demonstrated that, when people understand that the social provision of public goods is of the weakest-link variety, the propensity of individuals to free-ride is greatly reduced (Harrison and Hirshleifer 1989). This result can be understood by examining table 19.1, which shows the payoffs to self-interested individuals from either protecting or not protecting their forest land. Letting b represent the forest protection benefit received by each player if both players protect their forest land, and letting c represent the cost of forest protection, the weakest-link model applies if $b > c$. It can be seen that if either player does not protect their forest, then no benefits are forthcoming to either player and the net economic payoff is either $-c$ or zero. However, if either player invests c in forest protection, then the best strategy for the other player is to likewise invest c, and the net economic payoff to each player is $b-c$. This formulation of the forest protection problem provides a rationale for the development of public programs that raise the awareness among stakeholders that the benefits of forest health protection critically depend upon the contributions made by each member of the community.[2]

Table 19.1. Two-player, private forest landowner payoff table illustrating a weakest-link social composition function.

	Protect forest 2	Not protect forest 2
Protect forest 1	$b-c_1$, $b-c_2$	$-c_1$, 0
Not protect forest 1	0, $-c_2$	0, 0

[2] A weakest-link interpretation might be applied to the best known slogan in forest protection: "Remember—only you can prevent forest fires!" Assuming that the avoidance of careless behavior entails a cost, Smokey Bear can be thought of as reminding the public that the benefits of forest protection are conditional upon the (costly) actions taken by each member of the community.

The weakest-link concept can be applied to specific stages of invasive species management such as early detection and citizen response. A good example is provided by the hemlock woolly adelgid (*Adelges tsugae*, or HWA), an exotic insect that is responsible for widespread mortality of hemlocks throughout the eastern United States from Georgia to Maine. To contain the spread of HWA in Maine, the state has mounted a public awareness campaign (see section 4). Early detection and removal of HWA infected trees can reduce the risk for other landowners, and this strategy has been pursued by informing landowners of the visible symptoms of HWA infestation and what to do in case a suspected infestation is identified, and by providing maps of areas of known infections. A second example is provided by the emerald ash borer (*Agrilus planipennis*), an exotic insect that is responsible for the death of millions of ash trees, primarily in Michigan. A major emphasis of the current control program is to contain this pest in the Lower Peninsula of Michigan and eradicate it from Ohio and Indiana. A major focus of this program is to change the behavior of the weakest-link—residents who move firewood from infested areas to summer homes or campsites in uninfested parts of the states. Although the cost to individuals or households of changing their behavior may appear to be relatively small, the forest protection benefits will accrue only if everyone subscribes to this program.

2.2 Decision-Making Under Risk and Uncertainty

One of the most challenging obstacles to the development of forest health protection programs, both within public agencies and with broad-based private landowner participation, is the prevalence of risk and uncertainty. Although the risk (which we define as a probability, π) associated with each stage of a biological invasion is rather low[3], the uncertainty (θ) associated with each risk estimate may be quite large.[4] In this section, we provide a broad overview of the ways in which economists have modeled risk and uncertainty, and illustrate these concepts in the context of forest invasive species. We argue that, because the risk and consequences of a biological invasion can be influenced by management actions, and because the characteristics of an invasion might be of a kind not seen before, novel management approaches may be required.

The most general economic approach to decision-making under risk and uncertainty is the state-preference approach (Deaton and Muellbauer 1980,

[3] Williamson (1996) notes that roughly 10% of exotic species arriving in a non-native habitat become introduced into the wild, 10% of introduced species become self-sustaining, and that 10% self-sustaining species become pests. The so-called *10-10-10 rule* indicates the inherent difficulty of predicting which exotic forest organisms will ultimately become invasive forest pests.

[4] We use the term uncertainty to refer to limited knowledge of fixed quantities such as model parameters. As sample sizes increase, for example, uncertainty will decline. This allows learning as data accumulate.

p. 383-386). Three concepts are essential to this framework—states of the world, acts, and consequences. Acts (such as invasive species management programs) must be taken before the state of the world (such as the true ability of a new organism to invade an ecosystem) is known.[5] Consequences result after actions are taken and the true state of the world is revealed. From the perspective of invasive species management programs, consequences represent the sum of program costs and economic losses as well as other ecological or social impacts which cannot be monetized. At the social level, the major categories of economic damages are the loss of trade benefits, losses to agricultural, forest, and range productivity and losses to non-market economic values.

2.2.1 Expected utility

The major state-preference paradigm developed since World War II is expected utility theory (Shoemaker 1982). The conceptual framework can be visualized as a two-dimensional matrix where rows represent management actions, columns represent states of nature, and matrix cells represent economic consequences (table 19.2). The implementation of this framework necessitates two strong assumptions: (1) only consequences matter to the decision-maker—states of nature do not, and (2) the decision-maker cannot influence the probability of various states of nature—they are exogenous to human control (Deaton and Muellbauer 1980, p. 389). Given this framework, a rational decision-maker should choose the action (α) that maximizes expected utility:

$$\underset{\alpha}{Max}\, U = f\left[\pi^1 \upsilon(\gamma_1(\alpha)) + \pi^2 \upsilon(\gamma_2(\alpha)) + \ldots + \pi^n \upsilon(\gamma_n(\alpha))\right] \qquad (19.1)$$

where π^i is the probability of occurrence of state i, υ is a sub-utility function, γ_i is the consequence associated with state i, f is a function that aggregates sub-utilities into total utility, and $\Sigma_i \pi_i = 1$, that is, all states of nature are assigned a probability. The utility function shown in equation (19.1) not only expresses the decision-makers' preferences over various possible outcomes but also includes their assessment of the relative likelihood of the various states of the world that might occur. The decision-maker would choose the action (or, more generally, management program) that maximizes their expected utility.

As suggested by the example illustrated in table 19.2, application of expected utility (EU) theory to the optimal economic design of invasive species management programs is complex and requires a wealth of detailed information including estimates of the probability that each state of nature will occur, a list of feasible

[5] Decisions taken to prevent the occurrence of unwanted states of the world, such as terrorist acts or ill health, are more akin to solving a mystery than a puzzle (Treverton 2007). Preventive actions must be taken before states of the world are revealed and it is often not clear whether non-events are due to prevention efficacy, luck, or some other cause.

Table 19.2. Social payoff table showing hypothetical economic costs and losses of public management programs and ecological states associated with a biological invasion. Management programs are undertaken before ecological states are revealed. Other combinations of ecological states and management actions are possible.

	Ecological state				
Management acts	Transport unviable	Transport viable; establish unviable	Transport viable; establish viable; slow spread; low virulence	Transport viable; establish viable; slow spread; high virulence	Transport viable; establish viable; rapid spread; high virulence
Assess risk + quarantine	medium cost + low loss (t)	medium cost + low loss (t)	medium cost + medium loss (t,m,nm)	medium cost + high loss (t,m,nm)	medium cost + very high loss (t,m,nm)
Monitor ports + ecosystems	medium cost	medium cost	medium cost + low loss (m,nm)	medium cost + medium loss (m,nm)	medium cost + high loss (m,nm)
Monitor ports + ecosystems + aggressive eradication	medium cost (no eradication)	medium cost (no eradication)	very high cost + low loss (m,nm)	very high cost + medium loss (m,nm)	very high cost + high loss (m,nm)
Monitor ports + delayed control	low cost (no control)	low cost (no control)	low cost + low loss (m,nm)	medium cost + high loss (m,nm)	high cost + very high loss (m,nm)
Beliefs as to states	Π^1	Π^2	Π^3	Π^4	Π^5

Note: t refers to trade losses, m refers to market losses, and nm refers to non-market losses.

management actions, and estimates of the costs and losses associated with each combination of ecological state and management act. Table 19.2 also illustrates various economic trade-offs that must be considered when designing a biological invasion protection program. Understanding the tradeoffs between program costs and economic losses is a major challenge in the design and development of programs to counter the threat of biological invasions.

2.2.2 Endogenous risk

The standard EU model cannot address a class of economic phenomena known as moral hazards, which are acts people undertake that alter the risks they face. Insurance companies, for example, are concerned with moral hazard because people who buy insurance might then engage in extra risky behavior. Ehrlich and Becker (1972) used state-preference theory to evaluate insurance purchase decisions recognizing that risk can be influenced by decision-makers, and that alternatives to market insurance exist that can reduce the consequences of undesirable states of nature. They defined self-protection as actions designed to decrease risk, and self-insurance as actions designed to reduce consequences. This so-called endogenous-risk approach has been applied to analyses of optimal programs for invasive species management by integrating economic and ecological information (Shogren 2000, Leung et al. 2005).

Analytical endogenous risk models typically simplify states of the world to be dichotomous—either a state of the world occurs or it doesn't. For example, Leung et al. (2005) present an invasive species model where nature is either invaded or uninvaded. As with the EU model described in equation (19.1), social utility in the endogenous risk model is the welfare associated with a state multiplied by the probability of being in that state. The public decision-makers' problem is to invest in mitigation (M, or self-protection) and adaptation (A, or self-insurance) programs so that social welfare is maximized:

$$\begin{aligned} \underset{M,A}{Max}\, U = \pi^u(M)[\upsilon^u(w - L^t(M) - C(M,A)] + \\ (1-\pi^u(M))[\upsilon^i(w - L^p(A) - C(M,A))] \end{aligned} \tag{19.2}$$

where u is the uninvaded state, i is the invaded state, ν^u (ν^i) is the utility associated with being in the uninvaded (invaded) state, w is endowed forest wealth (market and non-market values), L^t is the loss to trade from quarantines and standards, L^p is the loss to the production of market and non-market values in the invaded state, and C is a cost function. Equation (19.2) illustrates that some costs (such as risk assessments and port inspections) and losses (such as trade losses due to quarantines) will be incurred even if an invasion does not occur. Likewise, investments in adaptation programs are needed prior to the state of nature being revealed so that resources are in place in the event of an invasion.

In this simple model, the first-order condition for the optimal level of investment in mitigation programs yields the following expression:

$$\pi^u\left[-\left(\frac{\partial \upsilon^u}{\partial L^t}\frac{\partial L^t}{\partial M}\right)-\left(\frac{\partial \upsilon^u}{\partial C}\frac{\partial C}{\partial M}\right)-\left(\frac{\partial \upsilon^i}{\partial C}\frac{\partial C}{\partial M}\right)\right]=\frac{\partial \pi^u}{\partial M}\left[\upsilon^u-\upsilon^i\right]. \tag{19.3}$$

The left-hand side of the expression is the expected marginal welfare loss from trade reduction plus mitigation expenditures. This is equated with the change in welfare induced by the marginal effectiveness of mitigation efforts in altering the risk of an invasion. As welfare losses from trade reductions and mitigation expenditures constitute social costs, and the welfare changes induced by mitigation effectiveness are benefits, Equation (19.3) simply states that mitigation should be undertaken up to the point where expected marginal costs equal expected marginal benefits. Similarly, the first-order condition for the optimal level of adaptation programs is:

$$\frac{\partial \upsilon^i}{\partial L}\frac{\partial L}{\partial A}=\frac{\partial \upsilon^i}{\partial C}\frac{\partial C}{\partial A}+\frac{\partial \upsilon^u}{\partial C}\frac{\partial C}{\partial A} \tag{19.4}$$

providing the result that adaptation investments should be made up to the point where the marginal benefits of adaptation (the reduction in losses) equal their marginal social costs.

2.2.3 Uncertainty and subjective probability

One of the critical variables highlighted in equations (19.1) through (19.4) is π, the probability that a well-defined state of nature occurs. Assigning an accurate value to π is difficult because biological invasions are novel events. Although estimates of the average risk that an introduced species will become a pest can be computed using lists of introduced species for which their success or failure is known (Reichard and Hamilton 1997), it is not known how well past invasions can realistically predict the risk of future invasions.[6]

Invasion probability might be considered as a degree of belief, which is applicable to both unique and repetitive events (Pratt et al. 1964). Treating probability as a degree of belief is the Bayesian approach to decision making and allows the analyst to incorporate both prior information and uncertainty in a model of subjective probability. Given limited information on ecological states for biological invaders, the decision-maker chooses a prior distribution to represent their degree of belief regarding the stages of a biological invasion (Clark 2005, Wikle 2003). As new information is acquired, the prior probability distribution can be updated using Bayes' rule:

$$\pi(\theta \mid y,x) \propto \pi(y \mid \theta,x) \bullet \pi(\theta) \tag{19.5}$$

where $\pi(\theta)$ is the prior probability, $\pi(\theta|y,x)$ is the posterior probability, y is the dependent variable of interest (such as the extent of a biological invasion), x is an explanatory variable (such as the level of adaptation effort), and $\pi(\cdot)$ now represents a distribution rather than a scalar value. The posterior distribution describes the subjective uncertainty about the probability which is proportional to the likelihood of observing y (given x and parameter θ) times the prior probability. As data accumulate, the posterior probability becomes the prior probability and learning occurs.

Uncertainty can be incorporated in the endogenous risk model of optimal invasive species management. For example, Shogren (2000) presents an economic model where uncertainty is an integral part of the decision-making framework:

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[6] However, Reichard and Hamilton (1997) found that the single best predictor for invasive plants is whether a species was known to invade elsewhere in the world.

$$\underset{M,A}{Max}\,U = \left(\int_a^b \begin{Bmatrix} \pi^u(M;\theta)\upsilon^u[w - L^t(M;\theta) - c(M,A)] \\ +(1-\pi^u)(M;\theta)\upsilon^i[w - L^p(A;\theta) - c(M,A)] \end{Bmatrix} dF(\theta;\beta)\right) \quad (19.6)$$

where most variables are as defined in equation (19.2), θ is a random variable reflecting uncertainty about parameter values, and F is the cumulative distribution bounded over the support (a,b) of the random variable θ. Note this model introduces uncertainty not only in the probability of observing states of nature but also in the level of realized damages. Although stringent data requirements would render this model difficult to operationalize, Shogren (2000) demonstrates a manager will maximize expected welfare by selecting levels of M and A that equate the marginal cost of influencing the severity and probability of an invasion with the marginal wealth acquired (or damages avoided). Perhaps of greater interest are the implications that a lower value of π^u will always increase investment in adaptation activities (A) and may decrease or increase investment in mitigation activities (M).

Although equations (19.2) through (19.6) were presented to represent the social welfare maximizing problem of a public decision-maker, the expected utility model is quite general and can be applied to the decisions facing private forest landowners. Linking an expected utility decision-making model for private forest landowners with the weakest-link social composition function introduces a new source of uncertainty—will all landowners in a community make investments in forest protection? A general expression for the private forest landowners' decision can be written:

$$\underset{M,A}{Max}\,U_i = \int_a^b \begin{Bmatrix} \pi^u(M_i \mid \sum_{j=1}^{n-1} M_j(\theta))\upsilon^u[w_i - c_i(M_i, A_i)] + \\ (1-\pi^u)(M_i \mid \sum_{j=1}^{n-1} M_j(\theta))\upsilon^i[w_i - L_i^p(A_i \mid \sum_{j=1}^{n-1} M_j(\theta)) - c_i(M_i, A_i)] \end{Bmatrix} dF(\theta;\beta)) \quad (19.7)$$

where the summation over $M_j(\theta)$ expresses the idea that the mitigation expenditures made by each member of the community influences both the probability of a biological invasion and the losses if an invasion occurs. Subjective uncertainty about the forest protection behavior of one's neighbors is a key element in decision-making by each individual (i) in the community of n landowners (equation 19.7). In the case study presented in section 4, we will note how the weakest-link character of private forest health protection decisions are influenced by recognition of the positive externalities created by individual investments in invasive species control. We also note the cost of control c_i (M_i, A_i) might include an argument for externalities (E_i), such as the unanticipated effects of mitigation and adaptation on non-target species.

2.2.4 Predictability and the base rate effect

Over time, as data accumulate, the subjective posterior probabilities converge towards some objectively correct probabilities. Even armed with correct probabilities, however, decision-makers face the problem that invasion probabilities are quite low. For example, Williamson (1996) has argued that roughly 0.1 percent of introduced exotic species eventually become pests. The rarity of an event greatly complicates predictability, even if predictions are accurate (Smith et al. 1999).[7] This is known as the base rate effect, and can be illustrated as follows. Suppose that an invasive species screening system is 90 percent accurate in identifying true invasive and true noninvasive organisms, and that the risk of an introduced species becoming a pest is 0.1 percent. If 10,000 organisms are screened, then roughly 1,000 true noninvasive species will be incorrectly identified as invasive. This is roughly two orders of magnitude greater that the number of true invasive species that are correctly identified. Although the screening system is quite accurate, the predictions of which organisms are truly invasive are quite poor (Smith et al. 1999, Keller et al. 2007). The base rate effect may therefore induce risk reduction policies that are overly restrictive. On the other hand, the potential for catastrophic forest damage from a novel invader suggests that application of the precautionary principle may be warranted.

A final issue that needs to be raised is the fact that rational decision-making under conditions of risk and uncertainty requires effort. When decision-makers are confronted with events that have a very low probability of occurrence, they often rely on *ad hoc* decision rules, or heuristics, rather than fully rational responses (Camerer and Kunreuther 1989). In particular, the *threshold effect* posits that, if probabilities fall below some threshold, they are treated as though they are zero (Slovic et al. 1977). Consequently, some individuals might entirely discount the risk of a biological invasion, thinking that “it can’t happen to me”. We expect that this behavior is especially likely when private forest owners are confronted with forest protection decisions, and may be a contributing factor to weakest link behavior in communities.

3. MANAGEMENT OF INVASIVE FOREST PESTS

As previously mentioned, each stage of a biological invasion can be linked with a strategy for mitigation or adaptation (fig. 19.1). In this section, we provide an overview of some major forest pest management programs in the United States that were undertaken to combat invasive species. Unfortunately, very few economic analyses have been conducted to assess the relative success or failure of these programs.

[7] Accuracy here refers to a situation where true invaders and true non-invaders are correctly identified.

3.1 Risk Assessments, Standards, and Quarantines

The first line of defense against a biological invasion is to prohibit potential invaders from crossing national borders. This strategy is implemented by conducting risk assessments of potential invaders that may hitch-hike in products of international trade (USDA Forest Service 1991) and by establishing mitigation standards that would assure the destruction of unwanted organisms either at the port of origin or the port of entry. In some cases, quarantines may be warranted.

Prior to the late 19th century, the idea of protecting agricultural and forest systems from biological invasions was not seriously considered (Popham and Hall 1958). The first legislation used to protect plant resources in the United States from potential biological invasions was the Quarantine Act of 1912. This Act was passed largely in response to the devastating effects resulting from two forest diseases—the chestnut blight and white pine blister rust (Anderson 2003), and Quarantine Number 1 prohibited the importation of 5-needle pines (Maloy 1997). Further protection to agricultural and forest producers was provided by the 1957 Plant Protection Act which allowed the USDA to make predeparture inspections of plant material at sites such as Hawaii and Puerto Rico, and impose quarantines without a public hearing and without notice (Bryson and Mannix 2000).

By definition, quarantines limit trading opportunities between countries and they have long been accused of functioning as tariffs to protect favored industries (Campbell 1929). The Uruguay round of talks on the General Agreement on Tariffs and Trade include Sanitary and Phytosanitary Standards (SPS) which are designed to limit the risk posed by trade in agricultural and nursery products. Although trade liberalization has generally reduced tariffs on agricultural and nursery products, it is widely acknowledged that SPS can restrict trade, especially for developing countries that cannot afford the means to attain imposed standards (Henson and Loader 2001). The benefits of quarantines to the country that impose them directly depend on their effectiveness in preventing new invasions.[8] However, quarantine effectiveness is difficult to evaluate due to the scarcity of comparative data that would permit scientific analysis, and it has been noted that many damaging pests have been introduced into the United States since the Quarantine Act of 1912 (Mathys and Baker 1980).

[8] The U.S.D.A. agency charged with responsibility for implementing plant inspections and quarantines is the Animal and Plant Health Inspection Service (APHIS). This agency was created in 1970 by removing the regulatory functions from the research oriented Agricultural Research Service and creating an independent agency. The Plant Protection and Quarantine Program was formed that year and placed under the new agency. Also during that year, the United States became a signatory to the 1952 International Plant Protection Convention.

3.2 Eradication

If an exotic organism slips through a quarantine, plant inspection or treatment, the second line of defense is to initiate an eradication program with the intent of forcing the extinction of a newly introduced organism before it becomes established in native ecosystems. The processes by which exotic organisms become established are highly stochastic (Liebhold et al. 1995), and strongly influenced by propagule pressure (Von Holle and Simberloff 2005). Forced extinctions are more likely to result from early and aggressive suppression efforts while population numbers are limited and Allee effects may be used to advantage.

As noted by Maloy (1997), "it is human nature to do something in a crisis even if it is a long shot" (p. 105). The largest invasive forest pathogen eradication program undertaken in the United States, in terms of time, money and labor, was in response to white pine blister rust, which was first discovered in 1906 on pine seedlings imported from Europe (Maloy 1997). White pine blister rust requires an alternate host to complete its life cycle—cultivated and wild currants and gooseberries (*Ribes* spp.)—and control was focused on destroying these extensively distributed hosts. Destruction of wild *Ribes* was labor intensive, especially in the remote and rugged terrain of the western U.S. A federal government eradication program wasn't initiated until 1933, some 27 years after the disease was first discovered. Federal involvement in white pine blister rust eradication may have been as much of a political decision as a forest management decision, as the initiation of the program coincided with the Great Depression. During the years 1933-40 the program rapidly expanded due to low-cost labor provided by the CCC. Although *Ribes* eradication efforts were ultimately applied to more than 20 million acres nationwide, the success of the eradication program was often called into question. An economic analysis of the program in the Lake States was very critical (King et al. 1960) and the program was discontinued in 1966. Over the roughly 34 years of the program, it is estimated that $150 million (in nominal dollars) was spent on control (Maloy 1997).

The first forest insect eradication program implemented in the United States was the attempt to wipe out the European gypsy moth. Although the pest was accidentally introduced in 1869, the initial governmental appropriation, made by the Massachusetts legislature, did not occur until 1890. Some have argued that the aggressive eradication program over the next 10 years was successful, and that eradication was nearly achieved. However, perhaps due to the apparent success of the program, funding was discontinued and the range of the insect spread rapidly. Subsequent to World War II, DDT was sprayed on outlying infestations which led to the successful eradication of the pest on nearly 4 million acres in Michigan, Pennsylvania and New Jersey, and complete eradication was considered a possibility (Popham and Hall 1958).

3.3 Control

Once an exotic organism becomes established in a native ecosystem, eradication becomes difficult, if not impossible, and control programs can be attempted to limit the growth and spread of the organism. Such adaptation programs buy time for both public and private forest owners to alter their management strategies (Waring and O'Hara 2005) and allow scientists the opportunity to discover new methods for eradication (Hain 2006). Control programs attempt to alter the spatial and/or temporal population growth dynamics of an invasive species while recognizing that complete eradication is unlikely.

With the elimination of DDT as an eradication tool in the United States, the gypsy moth has steadily continued to expand its range. Recent efforts have shifted from a strategy of eradication to control, by "slowing the spread" (STS) of the organism.[9] The contemporary STS gypsy moth program focuses control efforts on creating a barrier zone along the leading edge of the population front by targeting isolated insect communities. Sharov and Liebhold (1998) conclude that the STS program has recently slowed population spread in the Appalachian region of the United States by 59 percent.

Another important strategy for controlling invasive forest insects is the use of biological controls. Classical biological control is the control of exotic pests by means of importing their natural enemies from their country of origin.[10] The identification of potential biological control organisms is a complicated and lengthy process (Pschorn-Walcher 1977) and concerns have been raised about risks to native ecosystems (Simberloff and Stiling 1996).

3.4 Cost-benefit Analysis

Our review of the literature reveals that economic analyses of forest invasive species programs are rarely conducted. Consequently, the efficiency of investments in these programs cannot be evaluated. Commonly used measures of economic damages to forests from invasive species are solely focused on lost timber values, and are computed by multiplying the price of timber by an estimate of the annual quantity of timber destroyed (Pimental et al. 2000). This approach does not include the broader impacts of exotic forest pests on non-market economic values and is therefore biased downwards, perhaps severely. For example, *P. ramorum* infections in California and Oregon are causing enormous mortality to oaks and other tree species on public and private landscapes and yet none of the impacted species have important uses as timber (Rizzo et al. 2005). We anticipate that understanding the non-market economic impacts of *P.*

[9] This strategy was initially attempted in 1923 by creating a barrier zone from Long Island to Canada along the Hudson River.

[10] The National Biological Control Institute was established in 1990 to provide leadership for biological control and functions under the auspices of APHIS.

ramorum and other exotic forest pests on forest ecosystems will make a major contribution to cost-benefit analyses of invasive forest pest programs (Holmes and Kramer 1996, Rosenberger and Smith 1997, Kramer et al. 2003, Holmes et al. 2006).

4. PUBLIC AWARENESS AND THE HEMLOCK WOOLLY ADELGID

The risk and uncertainty associated with most biological invasions, combined with the weakest-link public good characteristics of forest health protection programs, may help to explain why mitigation and adaptation strategies have lagged far behind the initial arrival and establishment of exotic species. One of the key factors in developing a rapid response to invasive forest species is the participation of the public (U.S. Government Accountability Office 2006). This is especially true in the eastern United States where private forest land dominates the forest landscape.

Ongoing research funded by the USDA Forest Service to better understand the economic impacts of the hemlock woolly adelgid (HWA), an exotic forest insect inadvertently introduced from Japan, demonstrates how economic analysis can be used to support management responses to invasive forest pests. In this section, we bring attention to the results of a pilot project completed as part of this ongoing research project, and focus attention on the role of public awareness in private forest protection actions.

4.1 The HWA Problem

The HWA is currently spreading across the eastern United States and threatens the widespread decline of eastern hemlock forests (Orwig and Foster 1998). The spread of HWA is facilitated by wind as well as the movements of birds, mammals, humans and the leading edge of an infestation travels at an approximate rate of 30 kilometers per year (McClure 1990). Roughly twenty-five percent of the 1.3 million hectares of eastern hemlocks in the United States are currently infested with HWA and experts predict that the remaining 75 percent may become infested within 20 to 30 years (Rhea 2004). There are no known effective native predators of this insect and eastern hemlock has shown no resistance to HWA, nor has it shown any recovery following heavy, chronic infestation (Orwig and Foster 2000). Eastern hemlock forests provide a suite of public and private goods that have economic value, including wildlife habitat (Benzinger 1994, Evans et al. 1996), aesthetic quality in residential areas (Holmes et al. 2006, chapter 11 of this book), sales of nursery stock (Rhea 2004), and commercial timber (Howard et al.1999). As the impacts of this invasion accrue, forest managers' demand for information increases.

4.2 HWA Management

The management of HWA relies on an integrated system of mitigation and adaptation activities. State-level quarantines have been imposed to regulate the transport and sale of infested ornamental stock and infested hemlock logs (Bofinger 2002, Gibbs 2002). Eradication of HWA requires treatments to individual hemlock trees and is not considered a forest-wide option. Arborists eradicate HWA infestations on individual trees through semi-annual drenching with horticultural oils and insecticidal soaps.[11] Trunk injections of chemical insecticide are also effective over the short-term in eliminating HWA on individual trees (Pais and Polster 2000). At the forest level, biological control is attempted via release of an exotic predatory beetle, and experimental releases of beetles have been authorized by federal and state agencies in limited areas of highly infested forest since 1988 (Pais and Polster 2002).[12] Although these experiments have revealed the potential of biological control, the effectiveness of this approach remains uncertain (Knauer et al. 2002, McClure and Cheah 1999).

4.3 HWA in Maine

A pilot study undertaken by Byrne (2004) examined public awareness of HWA and its role in household control decisions for residential landscapes in Maine. Household control of HWA through spraying and tree removal might play an important role in reducing the risk of spread to uninfested areas ($\pi^u(M_i|\Sigma M_j(\theta))$ in equation (19.7) and evidence of whether or not an informed public can effectively aid in the control of HWA is of great value to forest resource managers. The weakest-link characteristic of controlling HWA to protect eastern hemlock raises the question of whether increased awareness can improve the effectiveness of management efforts or policy outcomes. Because public involvement is typically contingent on knowledge or awareness (Janicke 1997), an investigation of the factors that influence levels of awareness is warranted.

HWA was first discovered in Maine in 1999 as an isolated spot infestation resulting from infested nursery stock shipments, which, as of 2000, are quarantined by the State of Maine. HWA was not observed in a natural setting in Maine until 2004. Existing infestations have been controlled and monitored by state forest management agencies since 2000. A 2-year public awareness campaign consisting of newspaper and television announcements has proven to be critical

[11] According to a leading insect control company that specializes in treating hemlock for HWA, trunk and soil injections range from $35 up to $75 per tree depending on tree diameter, and foliar spraying costs approximately $550 per acre depending on the location of the infestation.

[12] Biological control efforts in the Great Smoky Mountains National Park cost approximately $6,000 per acre, and are being applied in old growth and interior backcountry areas (U.S. National Park Service, *personal communication*).

in the identification of HWA infestations (Ouellette 2002). As part of its management response to HWA infestations on residential properties, the Maine State Forest Service has compensated homeowners for the cost of treatment which, in most cases, involved removing and destroying infested hemlock trees.

4.3.1 Maine pilot study

In 2004, survey responses were collected from a sample of Maine residents using a multi-mode survey method that employed a web-based survey instrument and a mail survey instrument, identical in questions and format. A sample of 415 Maine residents was drawn from a list maintained by the Maine State Forest Service Entomology Lab consisting of residents who had contacted the Maine Forest Service within the previous 3 years and whose interactions with staff had been classified by staff as HWA-related. This sample is expected to have less variation in awareness and control responses than would a random sample of Maine households. However, the lack of information about the presence and location of hemlock trees on residential lands in Maine necessitated the use of this informed sample within the limited time frame of the pilot study.

At the end of the 9-week data collection period, which was supported by the Maine State Forest Service, a total of 81 surveys were completed either online (61) or by mail (20), resulting in a response rate of approximately 25 percent. Of the 81 households responding to the survey, 63 reported having hemlock trees on their property, and of this number 21 households reported that actions had been taken to control or eradicate HWA in their yard. When asked about the extent to which various motives influenced the decision to control or eradicate HWA, the two most influential motives selected were (1) "To maintain the health of hemlocks on my property" (16 households), and (2) "To maintain the health of other hemlocks in my neighborhood" (16 households). These responses indicate that, of the households that have acted to control or eradicate HWA on their property, the majority were motivated to a "very great extent" by their awareness that their actions could affect the health of other trees in their neighborhood.[13] Although the evidence is limited, this response indicates a degree of awareness among landowners regarding the weakest-link nature of household forest protection decisions.

Two empirical models were estimated using survey responses. The first approach employed an ordered logit model of categorical responses ranking self-reported awareness, along a Likert scale, of HWA. This model assumes that individuals are able to make meaningful distinctions between awareness levels in self-reports when asked to what extent they are aware or knowledgeable of HWA. The second empirical analysis employs a binary logit model of household

[13]The degrees of motivational influence included in the survey question were categorized as "very great extent", "to some extent", "a small extent", "not at all", and "don't know".

management decisions featuring individual household awareness as an explanatory variable. The dependent variable in this model is based on responses to the hypothetical question of whether or not a household would control or eradicate HWA if there were an infested tree in their yard.

Variables selected to explain public awareness levels include socio-economic characteristics of individual household members (income, gender, age, education, employment), the types of media they use to learn more about HWA (television, newspaper, radio, internet, and magazine), sources they may have used to gain information about HWA (state government agencies, university extension staff, landscaping firms or nurseries), characteristics expected to affect perceived awareness (membership in an environmental organization, gardening or tree club, prior control/eradication experience with HWA) and household landscape characteristics (acreage, percent tree coverage and the presence of hemlock trees).

As expected, most respondents reported some degree of HWA awareness. Four percent reported being aware to a very great extent, 46 percent reported being aware to some extent, 36 percent reported being aware to a small extent, and 14 percent were not at all aware. Table 19.3 displays results of the ordered logit model, which are largely consistent with research findings in the political science literature examining environmental and public policy knowledge (Steel et al. 1990, Steel 1996).

According to the empirical estimates (table 19.3), socio-economic characteristics play a significant role in HWA awareness. This finding, in combination with spatial proximity of households to hemlock resources, can be used to help target public information campaigns. Reported awareness is positively correlated with income, male gender, age, and membership in an environmental organization. Contrary to expectations, however, the empirical results suggest that respondents with college degrees are less likely to report higher awareness levels.[14] As anticipated, the effect of prior control/eradication experience with HWA is also a significant factor that positively effects reported awareness levels. This factor is important because, as described above, households with prior control/eradication experience were motivated to a "very great extent" by the awareness of their forest protection actions on the health of other hemlocks in their neighborhood.

When asked the question "If you had an infested hemlock tree in your yard, would you control or eradicate hemlock woolly adelgid", eighty-eight percent of the sample responded "yes" while the remaining 12 percent responded "don't know". Presumably, survey participants will respond "yes" if the expected net

[14]A careful examination of the data suggest the possibility that more education might induce greater caution about overstating perceived awareness.

[15]The cost of foliar spraying to control HWA in Maine is roughly $260/tree/year, which can be quite expensive if several hemlocks are located on a landowners' property. For example, more than one-half of the respondents in our sample reported 11 or more hemlocks were located in their yard, indicating that annual treatment per household could cost thousands of dollars.

Table 19.3. Ordered logit regression parameter estimates for variables influencing public awareness of the hemlock woolly adelgid in Maine.

Variable	Coefficients (t-value)	Variable	Coefficients (t-value)
College degree	-1.1773* (-1.82)	Environmental Org	1.2438* (1.83)
Income	0.0207*** (2.58)	Garden/Tree Club	-0.1451 (-0.20)
Male	10.3643*** (2.90)	Prior HWA Experience	1.6793** (2.40)
Age	0.1360** (2.49)	Employed in Forestry	-1.5556 (-1.11)
Age*Male	-0.1791*** (-2.89)	State Government	0.6840 (1.06)
Television	-1.0045* (-1.62)	University Extension	0.8262 (1.29)
Newspaper	0.3824 (0.61)	Landscape/Nursery	-0.9839* (-1.60)
Radio	-0.0048 (-0.01)	Yard Size (acres)	-0.6158*** (-3.04)
Internet	0.5826 (0.91)	Percent Tree Coverage	0.0052 (0.40)
Magazine	0.6113 (0.93)	Have Hemlocks	0.0895 (0.13)

Likelihood Ratio 40.6280
χ^2 Probability 0.0042
Observations 78

Note: * 10% significance, ** 5% significance, *** 1% significance.

benefits from management are positive.[15] The fact that not a single respondent answered 'no' might be interpreted in two ways. First, respondents might truly be uncertain about a variety of factors associated with the scenario including the potential damage that would be incurred by an infested tree and the possibility that an infestation could spread to trees in their yard or neighbors' yards. Second, respondents may be exhibiting compliance bias, a situation where respondents' consciously or unconsciously rationalize that to answer 'no' is "socially irresponsible" (Kemp and Maxwell 1993). Given that sampled households had previous contact with the Maine State Forest Service, it is not surprising they many may have felt it irresponsible to answer 'no'. For purposes of this analysis, 'don't

Table 19.4. Binary logit regression parameter estimates for the household control of hemlock woolly adelgid in Maine.

Variable	Coefficients	t-value
Intercept	-2.1930	-0.93
Awareness	2.2383**	2.23
College Degree	2.8549	1.54
Income	-0.0285*	-1.82
Environmental Org	0.4343	0.26
Garden/Tree Club	-3.1211**	-1.96
Yard Size (acres)	-0.1162	-0.29
Tree Coverage (%)	-0.0125	-0.43
Number of Hemlocks	0.1738	1.49
Driveway/Border	-3.0015*	-1.92
Likelihood Ratio	25.7520	
$\chi 2$ Probability	0.0022	
Observations	63	

Note: * 10% significance, ** 5% significance.

know' responses are interpreted as reflecting uncertainty about intended actions, while 'yes' responses are interpreted as statements that the household will invest in HWA control with certainty.

Only respondents with hemlocks on their property were used in the logit analysis (63 observations). The key finding is the positive, statistically significant effect of awareness on the household control decision (table 19.4). Consistent with results reported by Miller and Lindsay (1993) for a study of gypsy moth control in New Hampshire, this result indicates that individuals who reported higher awareness levels are more likely to invest in the control of invasive species. This result suggests programs designed to increase public awareness about HWA can encourage household control and reduce the risk of spread. We also note the statistically significant, negative signs on "garden/tree club" and "driveway/border" may reflect concern with the effect of chemical treatments on non-target organisms.

Of course, our use of an informed sample does not allow us to generalize these results to the entire Maine population. Nonetheless, this case study identifies characteristics associated with household awareness levels and a stated intention to pursue private adaptation behavior in the context of HWA, and establishes a positive relationship between the two. It also demonstrates how economic theory and methods can be used to support management responses to HWA. Our use of social, economic, and landscape data suggests that, if more extensive data were collected for a larger number of respondents, model results could help target forest protection efforts to areas characterized by low awareness and high uncertainty.

5. CONCLUSIONS

Invasive forest pests have been a persistent problem plaguing forest managers in the United States for more than a century. Despite the fact that hundreds of millions of dollars have been spent on eradication and control of exotic forest pests, comprehensive economic analysis of the costs and benefits of these programs are almost non-existent. The lack of comprehensive economic assessments of the effects of invasive forest species on the production of market goods (such as timber) and non-market economic values (such as aesthetics) has stymied meaningful economic analyses. We see this gap as one of the greatest issues facing the development of more rational and effective forest pest management systems.

This chapter reviewed four key economic concepts that we think are integral to the design of socially optimal programs for combating invasive forest pests. First, forest health protection is a public good. Private provision of forest health protection is expected to be sub-optimal because self-interested individuals do not account for the benefits that flow to other members of society when they make forest protection investments. This context provides the justification for government intervention in forest health protection.

Second, forest health protection is a weakest-link public good. The weakest-link character of forest health protection relegates the level of forest protection attained by a community to the weakest members of the community. Consequently, effective forest health protection programs require that the weakest links be strengthened. We argue that forest health protection programs can be enhanced by targeting information to those most likely to engage in risky behavior. In particular, information describing the weakest-link nature of forest protection should be targeted at private landowners to enhance the likelihood that they will participate in forest protection programs. As indicated in our case study, weakest links can be identified using economic surveys of household behavior.

Third, the design of optimal strategies for managing invasive species is highly complex due to the trade-offs that must be evaluated between the costs of management actions and the economic losses to trade, the production of market goods, and the provision of non-market economic values. Data are costly to obtain and until decision-makers recognize the value of economic information, they are unlikely to invest in its collection.

Finally, each biological invasion represents a novel situation. However, mitigation and adaptation investments must be made prior to the time at which the true state of nature is ultimately revealed. Pervasive uncertainty regarding the parameters of economic and ecologic models argues for the necessity of treating uncertainty in as pragmatic a fashion as possible. Bayesian methods provide a useful approach for incorporating uncertainty about data, parameters, and processes in models of inference and prediction. As new information is realized, and uncertainty is reduced, economic models of optimal decision-making can be updated. We anticipate that Bayesian approaches to learning about the risks and economic consequences of biological invasions will provide a substantial contribution to the adaptive management of invasive forest pests.

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