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Volumes 2



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Advances in Threat Assessment and Their Application to Forest and Rangeland Management

Volume 2

**John M. Pye, H. Michael Rauscher, Yasmeen Sands,
Danny C. Lee, and Jerome S. Beatty, Technical Editors**

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Abstract

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In July 2006, more than 170 researchers and managers from the United States, Canada, and Mexico convened in Boulder, Colorado, to discuss the state of the science in environmental threat assessment. This two-volume general technical report compiles peer-reviewed papers that were among those presented during the 3-day conference. Papers are organized by four broad topical sections—Land, Air and Water, Fire, and Pests/Biota—and are divided into syntheses and case studies. Land topics include discussions of forest land conversion and soil quality as well as investigations of species' responses to climate change. Air and water topics include discussions of forest vulnerability to severe weather and storm damage modeling. Fire topics include discussions of wildland arson and wildfire risk management as well as how people perceive wildfire risk and uncertainty. Pests/biota topics include discussions of risk mapping and probabilistic risk assessments as well as investigations of individual threats, including the southern pine beetle and *Phytophthora alni*. Ultimately, this publication will foster exchange and collaboration between those who develop knowledge and tools for threat assessment and those who are responsible for managing forests and rangelands.

Keywords: Environmental threats, threat assessment, environmental risk analysis, disturbance, wildfire, pests, forest and rangeland management.

Preface

Danny C. Lee¹ and Jerome S. Beatty²

In July 2006, more than 170 researchers and managers from the United States, Canada, and Mexico convened in Boulder, Colorado, to discuss the state of the science in environmental threat assessment. The 3-day conference explored the latest information on environmental threats, bringing together people who develop knowledge and tools for threat assessment and management and those responsible for managing forests and rangelands. The event included more than 100 oral and poster presentations on topics ranging from severe weather and climate change to risk mapping and forest pests.

The year preceding the conference, 2005, was one of extremes and provided an ideal backdrop for a discussion of environmental threats. More hurricanes were tracked that year than had ever before been reported, including Katrina, one of the costliest U.S. hurricanes on record and one of the deadliest. Also in 2005, entomologists confirmed the capture of a female sirex woodwasp in a sample collected in New York in late 2004. It was the second of its kind identified in the United States and has pine managers across the country understandably concerned given the species' ability to cause severe tree mortality. The wildfire season that year also made history, claiming more than 8 million acres and breaking the record that had been set in 2000 for total acreage burned.

Given this context and, more generally, the complexity of assessing and managing the myriad threats that face North America's wildlands, the conference's organizers were committed to continuing the exchange of information and collaboration fostered during the event long after its final session. To that end, nearly 50 of the synthesis and case study papers presented were adapted to form the initial content of the Encyclopedia of Forest Environmental Threats (<http://www.threats.forestencyclopedia.net>)—an online resource that promises to deliver to researchers, land managers, and policymakers the scientific knowledge about environmental threats they need to achieve their objectives. The threats encyclopedia—like the others in the Forest Encyclopedia Network, of which it is part—connects scientific results, conclusions, and impacts with management needs and issues. Designed for scientists and practitioners alike, the encyclopedia serves as a growing online compilation of scientific knowledge relating to environmental threats and their assessment and management.

The conference's content is also preserved here, in a traditional and more permanent form. The peer-reviewed papers featured in this volume represent the scope of environmental threats and underscore the complexity of their assessment and management. As these papers show, environmental threats often act in concert and with no regard for land ownership and administrative boundaries, making them as difficult to identify and anticipate as they are to manage and control. In response, researchers and managers are developing a growing foundation of knowledge, which can help to assess or minimize these threats. This volume represents a significant contribution to this effort.

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The “Advances in Threat Assessment and Their Application to Forest and Rangeland Management” conference and this publication were made possible by the contributions of many people and organizations. Participating organizations include the USDA Forest Service’s Pacific Northwest (PNW) and Southern Research Stations (SRS); the Western Wildland Environmental Threat Assessment Center (WWETAC); the Eastern Forest Environmental Threat Assessment Center (EFETAC); the Cooperative State Research, Education, and Extension Service; Southern Regional Extension Forestry; and the Southern Forest Research Partnership. The conference’s organizing committee members included John M. Pye, Jerome S. Beatty, Danny C. Lee, H. Michael Rauscher, Yasmeen Sands, Gregg DeNitto, Charles G. Shaw, James P. Shephard, and David A. Weinstein. Special thanks to the moderators who oversaw the extensive peer-review process—Gregg DeNitto, William Bechtold, Charles G. Shaw, Borys Tkacz, William D. Smith, Kurt Riitters, David Weinstein, Becky Kerns, Alan Ager, Charley Luce, and Kerry Overton—and to the many peer reviewers who helped to ensure the quality of the science. This publication would not have been possible without the editing expertise of Alan Salmon, Gary Benson, Aimee Tomcho, Charmaine Rini, and Sharon DeLaneuville. Matt Howell and his University of Georgia team provided excellent Web development and hosting for the Web version of this publication’s content, available online at <http://www.threats.foresten-cyclopedia.net>. Funding for the conference, this publication, and its Web counterpart was provided by WWETAC, EFETAC, PNW, and SRS.

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FIRE

Syntheses

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Ecological Risk Assessment to Support Fuels Treatment Project Decisions

Jay O'Laughlin

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Abstract

Risk is a combined statement of the probability that something of value will be damaged and some measure of the damage's adverse effect. Wildfires burning in the uncharacteristic fuel conditions now typical throughout the Western United States can damage ecosystems and adversely affect environmental conditions. Wildfire behavior can be modified by prefire fuel treatments, thereby reducing risks to firefighters, structures, and ecosystems, but such projects pose their own environmental risks. To support fuels treatment decisions, environmental analysis of alternatives is generally required, including taking no action. How can managers determine whether risks of actively treating fuels are greater than risks posed by no action? The risk-reduction benefits of fuel treatment are often overlooked in decision processes for comparing wildfire effects with and without fuel treatment. To fill the void, a comparative ecological risk assessment conceptual model is presented. Both prefire fuels treatment and postfire events produce sediment that can adversely affect water quality and aquatic organisms. Similarly, both prescribed fire and wildfire can adversely affect air quality. The model's tradeoff diagram tests a risk management hypothesis: The benefits of restoring natural (historical) fire regimes and native vegetation in a particular location, plus the benefits of reducing the severity of wildfire effects, balance favorably against any adverse effect, either short- or long-term, from fuels treatment. Managers may believe this hypothesis, but policies require environmental analysis to support it. A tradeoff diagram illustrates the conceptual model and graphically replies to the question: Which produces more sediment, wildfire burning under untreated conditions, wildfire burning after fuels are reduced, or the treatments designed to reduce wildfire risks? Similarly: Which situation would produce more fine particulate matter (PM_{2.5}) air pollution? Tradeoff diagrams of such situations may contribute to sustainable

resource management decisions by improving communications between risk assessors, public agency managers, and interested nongovernmental parties.

Keywords: Comparative ecological risk assessment, conceptual model, hazardous fuel reduction, policy, risk management, wildfire.

Introduction

Compared to not taking any action, fuel treatments may or may not reduce adverse ecological and environmental effects that accompany wildfires. Risk management is central to many human enterprises and is the foundation for all fire management activities (USDA-FS/USDI and others 2001). The U.S. Department of Agriculture, Forest Service (USDA-FS), and the U.S. Department of the Interior, Bureau of Land Management (BLM), have been advised to adopt a systematic risk-based approach to target fuel reduction projects across landscapes and to make fully informed decisions about fuels treatment project alternatives and their effects (GAO 2004, USDA-OIG 2006).

Risk is usually defined as having two components: a measure of adverse effects, and the probability of the adverse effects' occurrence. Many people have difficulty comprehending risk, and its quantification has challenged and confused lay persons and professionals (Haimes 2004). A simplifying approach for fire-adapted ecosystems is to use the fire return interval as the time horizon for analyzing environmental effects. By definition, this ensures that a wildfire will occur during the analytical period, and different conditions that affect fire behavior can be compared more readily by eliminating the need for knowing the probability distribution of fire occurrence. The risk management question then becomes: What is the desired condition for a particular ecosystem and location when the inevitable lightning bolt strikes?

Wildfire risk management is a synthesis of scientific and nonscientific concerns. Information from ecological, social, managerial, and policy sciences is integrated into a decision process framework that incorporates social values and concerns. These include democratic process and

institutions for governing collective decisions in our society. Risk management depends on effective communication of information between risk assessors in regulatory agencies, risk managers in land management agencies, and interested members of the public who may be directly or indirectly affected by wildfire.

Risk is related to each of the three components of the general sustainable forestry model—risk of losing ecosystem components or damaging environmental values, economic investment risk, and social risk in communities facing forest-based change (O’Laughlin 2004, 2006). This synthesis takes a problem-oriented approach (see Clark 2002) to wildfire risk management and deals primarily with the ecological and environmental aspects of wildland fire management embedded in decisionmaking and social contexts and reflected in various institutions. Whether fuels treatment should take place on public lands is a collective decision.

Managers may lack the tools and information to demonstrate the beneficial effect that fuel treatment projects could have on environmental quality. To fill the void, a conceptual model of decision tradeoffs has been developed (O’Laughlin 2005a, 2005d). Herein, it is applied to the multiobjective fire/fish risk management problem in order to compare postfire sedimentation with and without fuel treatment. This same approach can be used to assess other environmental effects including fine particulate matter (PM_{2.5}) air pollution from either wildfire or prescribed fire smoke. The model is based on the framework of the Environmental Protection Agency’s Guidelines for Ecological Risk Assessment (EPA 1998). In a comparative risk assessment framework, the integration of sediment, smoke particles, and other environmental risks with land management objectives is accomplished not by science, but via social process, primarily the public involvement processes required by environmental and land use planning laws. Risk assessors and risk managers interact with stakeholders to determine which risks are most important. Risks to firefighters, structures, scenery and aesthetics, vegetation species and age classes, wildlife habitat, and air and water quality should be considered. Using cause-and-effect modeling results developed by risk assessors, risk managers can

demonstrate for each risk whether prefire fuels reduction could potentially reduce postwildfire adverse effects.

This section is concerned with all aspects of risk analysis, including risk communication as well as risk assessment and management. First, appropriate terminology is presented. Then the wildfire situation in the Western United States is defined as a fuels management problem and placed within its decision process and social contexts. Complications from spatial and temporal dimensions, as well as risk governance, arise from policy requirements and social perceptions of risk that may require institutional changes to produce sustainable improvements in forest ecosystem conditions. This synthesis rationalizes the choice of parameters selected to adapt the EPA’s ecological risk assessment framework for the purpose of supporting fuels treatment decisions by comparing wildfire risk management alternatives using a with-or-without framing consistent with National Environmental Policy Act (NEPA 1969) requirements. A conceptual model diagram is the core of ecological risk assessment and the principal product of this synthesis, where it is used to compare wildfire effects on sediment with and without fuel reduction.

Risk Analysis Terminology

Risk terms can be a barrier to effective communications if not properly defined. **Risk** is a combined statement of the probability that something of value will be damaged and some measure of the damage’s adverse effect. Risk simply gives meaning to the things, forces, or circumstances that pose danger to people or what they value (NRC 1996). Risk can mean both the probability of loss and the hazard or threat that might cause that loss (Harwood 2000). A **hazard** or **threat** is something that poses danger or can cause an adverse effect. **Stressor**, an EPA term, seems to be synonymous with hazard or threat and is any physical, chemical, or biological entity that can induce an adverse response in an ecological **risk assessment endpoint**. The endpoint is an explicit expression of the environmental value that is to be protected, operationally defined by an ecological entity and its attributes (EPA 1998). **Ecological risk assessment** is a process that evaluates the likelihood that adverse ecological effects may occur or are occurring as a result of exposure to

one or more stressors (EPA 1998). **Ecological risk assessments** are developed within a risk management context to evaluate human-induced changes that are considered undesirable and are used to support many types of management actions, including the management of watersheds or other ecosystems affected by multiple nonchemical and chemical stressors (EPA 1998).

Adverse effects and their potential damages and consequences are real components of risk (Haimes 2004). The meaning of risk has always been inherently controversial and political because it depends on value-based (i.e., non-scientific) judgments about adverse effects (Slovic 1999). Changes are considered undesirable because they alter valued structural or functional characteristics of ecosystems or their components. An evaluation of adversity may consider the type, intensity, and scale of the effect as well as the potential for recovery from the risk-inducing event (EPA 1998). The probability component of risk is an imagined, mathematical human construct (Haimes 2004). Although most people understand probability in its simplest form—the likelihood of outcomes from tossing a coin or rolling dice—conditional and joint probabilities can be perplexing.

Risk analysis is usually considered to be the process of assessing, characterizing, communicating, and managing risk (e.g., Haimes 2004, NRC 1996). Effective risk analysis is integrated into decisionmaking processes, not treated as a gratuitous add-on task (Haimes 2004). Risk assessment asks: What can go wrong, and what are the consequences? Human and organizational failures are sources of risk and may be caused by environmental and institutional elements. Risk management asks what can be done, and what are the impacts on future options? Risks cannot be managed until they have been assessed, and some form of model is necessary for that (Haimes 2004).

Fuels Problem and Context

The accumulation of fuels over time can lead to uncharacteristic wildfires and associated problems. Alteration of historical fire regimes often causes serious changes in forest ecosystem processes, resulting in unusually intense, large fires. Current forest conditions and wildfire regimes pose

risks to many environmental and socioeconomic values and threaten human communities (USDA-FS 2004). Wildfires near human communities can have devastating effects, as can postfire floods (May 2008). Wildfire poses immediate threats of damage or loss to nearby structures, and endangerment of public safety is an even more important concern during a wildfire incident and for many years thereafter. Wildfires also pose secondary threats to ecological and environmental resources (Summerfelt 2003).

The acreage affected by wildfire nationwide has steadily increased over the past four and a half decades, with a trend towards uncharacteristically severe and uncontrollable fire behavior (NIFC 2006b). Trends of increasing fire size and severity have emerged over the past 20 years (USDA-FS 2004) as wildfires in the Western States have increased. The trend is influenced by changes in climate, extreme droughts, and, in some forests, overabundant fuels (Westerling and others 2006).

The combined effects of increased fuel accumulations, lengthened fire seasons, and intensified burning conditions are expected to contribute to larger and more extensive wildfires in the near future (Covington and others 1994), with increases of 74 to 118 percent in wildfire burn areas expected over the next century (Running 2006). These expectations underscore the urgency of fuels management to reduce wildfire hazards to human communities as well as actions to mitigate wildfire impacts in forests that have undergone substantial alterations from past land uses (Covington and others 1994, Westerling and others 2006).

An understanding of the decision processes (“Decision Process”) and social context (“Social Context”) for wildland fire management is necessary if risk analysis is to be integrated into land and resource management decision-making. Decision processes are defined by public policies and laws, which are a function of the social environment within which wildland fire management occurs. Although various agencies and organizations may perceive wildfire risks differently, management decision processes are affected by the same evolving institutional framework of laws and policies. By engaging collaboratively with stakeholders in formal decision process forums, managers can integrate different perspectives regarding environmental

risks and thereby arrive at socially acceptable decisions regarding wildland fire management and fuels reduction.

Decision Process

The ultimate utility of decision analysis is not necessarily articulating the best policy option, but avoiding extreme events (Haimes 2004), such as large-scale, uncharacteristically severe wildfires. Risk analysis traditionally has been used for other purposes, but it can address forest management issues in a transparent way and disclose risk tradeoffs that are often not accounted for in other decision analysis techniques (Hollenstein 2001). Land and resource management decisions always involve risk, including the decision not to take action (Thomas and Dombeck 1996).

The Federal Wildfire Policy (USDA-FS/USDI and others 2001) recognizes that sound risk management is a foundation for all fire management activities. Together with the National Fire Plan (NFP), this policy provides the institutional framework for Federal agencies, States, Native American tribes, local governments, and communities to manage wildfire risk, improve land conditions, and reduce impacts to communities while ensuring sufficient firefighting capacity for the future. The NFP has established a long-term hazardous fuels reduction program in which treatments are designed to reduce wildfire risks to people, communities, and natural resources while restoring forest and rangeland ecosystems to closely match their historical structure, function, diversity, and dynamics (USDA-FS/USDI 2006a). The NFP is implemented through a collaborative framework agreed upon by Federal and State agencies, tribes, and other parties (WGA 2006). The collaborative framework is a way to address fire and fuel management problems within their social context.

The National Environmental Policy Act (NEPA 1969) is the cornerstone of our environmental laws. It requires that Federal agencies analyze the short- and long-term adverse environmental consequences of a range of proposed management alternatives, including no action. The result of the NEPA process is some type of an environmental impact assessment document. According to an EPA scientist (Fairbrother and Turnley 2005), the NEPA process has various shortcomings that have hampered decisionmaking

and significantly reduced public acceptance of fuels treatment. Risk assessment integrated into the NEPA process could result in more meaningful environmental impact analyses, thereby providing a more technically sound and robust means for assessing and comparing potential adverse outcomes of proposed management alternatives (Fairbrother and Turnley 2005).

The healthy forests policy—composed of the President’s Healthy Forests Initiative and the Healthy Forests Restoration Act (HFRA 2003)—has led to modification or streamlining of decision processes (see O’Laughlin 2005b). The HFRA requires that before a court can issue an injunction, there must be a weighing of the environmental effects of doing fuels treatment against not doing fuels treatment. A legacy of HFRA therefore may be the stimulation, if not institutionalization, of comparative ecological risk assessment. Courts will look to land management agencies for such analysis. At this writing, it is unclear what the analysis might look like. One option is adapting the EPA (1998) Guidelines (“EPA’s Risk Assessment Framework”) to a wildfire risk management problem (“Fire/Fish Risk Management Application”).

Social Context

Risk is a social construction, combining science and judgment with psychological, social, cultural, and political factors (Slovic 1997). What risks are taken into account, how they are framed, and what constitutes a solution to a risk problem are all matters that go beyond scientific inquiry (De Marchi 2003). Environmental risk involves value judgments that reflect much more than just the probability and consequences of the occurrence of an event (Kunreuther and Slovic 1996). Because risk assessment brings specific values into consideration, it can help reveal which values are at greater risk (Molak 1997). Interested parties can contribute knowledge that risk assessors would otherwise overlook (NRC 1996). Serious attention to citizen participation and process issues may eventually lead to more satisfying and successful ways to manage risks (Slovic 1997). If done systematically and transparently, risk assessment can help build trust (Slovic 1993) primarily by adding transparency to forest decisionmaking processes (Hollenstein 2001).

Social interaction is the phase of the risk analysis process during which integration of different risk assessment endpoints can be addressed. To enable management actions to improve the fire and fuel problem on Federal lands, managers should focus on the things people care about. These include:

1. The condition of forests relative to forest values associated with management objectives (i.e., forest health).
2. Forest values at risk of damage from nature's forces as well as human actions.
3. The environmental and socioeconomic effects of wildland fire management policy that protects some values while putting others at greater risk (O'Laughlin 2006).

For example, trying to protect fish habitat in fire-adapted ecosystems by not allowing a riparian vegetation management project simply because it will produce a small amount of sediment may be counterproductive in the long term. Fire is inevitable, and the magnitude of adverse effects is more important than when the effects occur (O'Laughlin and others 1998).

When interacting with stakeholders, risk assessors and managers should use qualitative and quantitative approaches as appropriate, while disclosing assumptions and potential for errors. Risk assessment can help risk managers compare the environmental effects of management alternatives. Clear objectives are needed, consistent with a long-term vision of what the land should look like. Effective communication of short- and long-term risks and risk-reduction benefits can help build trust with stakeholders (O'Laughlin 2005c). The thought process that goes into evaluating a particular hazard is more important than the application of some sophisticated mathematical technique or formula (Molak 1997). More emphasis on the risk management aspects of risk analysis would mean greater stakeholder involvement and de-emphasis on quantitative characterization of risk and uncertainty (Power and McCarty 1998). Fuels management projects on several California national forests emphasize the importance of stakeholder collaboration and modeling tools to help facilitate communications (see "Strategic Fuel Treatment").

Improving the Fuels Problem

The goal for improving the wildland fire and fuels problem is to reduce the long-term risk wildfire poses to human and ecological communities. Managers need to weigh the short-term risks posed by active management against the long-term risks posed by continued inaction, and to communicate these risks in a meaningful way to the public (Bosworth, Dale. 2003. Risk assessment for decisionmaking related to uncharacteristic wildfire. Unpublished keynote address to conference in Portland, OR, November 17. On file with Jay O'Laughlin, College of Natural Resources, University of Idaho, P.O. Box 441134, Moscow, ID 83844-1134). Managers cannot change weather or topography, but fuels can be modified to change the burning and value-loss characteristics at specific locations as well as across large landscapes. This not only reduces the negative impacts on those forests but the wildfire itself may also provide benefits (Finney 2005). Benefits include environmental risks prevented by management actions (Davies 1996). For example, before enjoining "an agency action under an authorized hazardous fuel reduction project, the court reviewing the project shall balance the impact to the ecosystem likely affected by the project of (1) the short- and long-term effects of undertaking the agency action; against (2) the short- and long-term effects of not undertaking the agency action" (HFRA 2003, title 1, section 106).

Fire hazard in a given area is partly a function of the combustible materials located on site. Thinning and prescribed burning are the primary fuel management activities and repeatedly have shown reduced fire intensities and increased survival of some forest types (Finney 2005). Considering anticipated changes in interior West forests, including climate, Covington and others (1994) concluded that the undesirable consequences of inaction far exceed those of action. Without active management of fuels, many forests will continue to be subject to uncharacteristically severe fires, and the costs of firefighting will continue to increase (Stephens and Ruth 2005). Active management can mitigate wildfire risks to watersheds in some situations, but, in others, forest management may not be effective (Bisson and others 2003, Schoennagel and others 2004, Westerling and others 2006).

Benefits from prefire management are most likely to come from prioritizing treatment areas (Dunham and others 2003). Priorities can be based on ecological value, evolutionary significance, and the risk of loss (Bisson and others 2003). The scale of the problem, however, is enormous. High-priority treatment areas cover 397 million acres of forests and grasslands across all ownerships, public and private, an area three times the size of France. Some 73 million acres of forests in the low- and mixed-severity regimes are far denser than they ought to be, increasing their vulnerability to stand-replacing fires. These have been identified as high-priority treatment areas (USDA-FS 2006a).

Successful projects for reducing fire hazard depend on taking many factors into account and developing protocols for deciding which stands should be thinned and by how much, with each situation evaluated on its own merit and operations planned carefully to ensure that the cure is not worse than the disease (NRC 2000). In risk management, avoiding actions in which the cure is worse than the disease means avoiding extreme events, i.e., the worst and the most disastrous situations (Haimes 2004).

Problem fires are today's parlance for extreme fire events (NIFC 2006b). Of all ignitions, 2 to 3 percent escape initial attack and become the problem fires that damage resources, threaten communities, and cost millions of dollars in suppression efforts. Whereas not all wildland fires grow to such proportions, problem fires are those events that are large, destructive, dangerous, and costly to manage. Problem fires are the symptoms of a larger forest health issue, where ecological realities conflict with social expectations and economic limitations (NIFC 2006b). Spatial fire behavior models used in collaborative settings ("Strategic Fuel Treatment") offer some promise in dealing with social issues at various spatial and temporal scales ("Spatial Scale Issues" and "Temporal Scale Issues"). Especially on National Forest System lands ("National Forest System Issues"), institutional improvements ("Institutional Improvement") may be necessary to help put such technologies in place.

Spatial Scale Issues

Wildland fire risk reduction is a national goal that depends on landscape-level planning and project-level actions (Barbour and others 2005). The EPA (1998) guidelines have been used to compare risks at a regional scale (Landis 2005). To date, such efforts have not included forest threat assessment in general or wildfire in particular. Although sustainable forest management issues involve multiple scales, achieving the national goals of sustainability rest, in large part, on actions that are carried out at the local or forest management unit scale (USDA-FS 2004). A variety of modeling approaches are now available to meet the landscape-level planning needs. However, there is a lack of explicit guidance about how to connect perceptions of risk across vast spatial expanses of geographic or ecological regions to the outcomes of specific management activities at the project scale of a few to tens of thousands of acres (Barbour and others 2005). The guidance document for Federal healthy forests project implementation (USDA-FS/USDI 2004) recognizes the importance of scale and assumes assessments at scales larger than individual projects have been done before fuels treatment projects are initiated.

Tying small area project-level analyses to larger scales helps managers think about the importance of different resources through space and time (Barbour and others 2005). Refining analyses at the midscale helps to understand how different resources interact. Considering resource conditions and management objectives at very broad scales can help managers understand where they might concentrate efforts (Barbour and others 2005). Broad-scale assessments should set priorities for reducing the risk to social and ecological values caused by uncharacteristically dense vegetation. To reduce risk, the assessments should evaluate the potential for vegetation treatments, such as mechanical treatments and prescribed fire. A tactical schedule of priority vegetation-treatment projects should result from strategic assessments of the need for fuel treatments conducted at appropriate landscape scales (USDA-FS/USDI 2004).

Spatial data and risk-based methods are available to analyze 2-million-acre watersheds in order to prioritize treatments at the subwatershed scale of 20,000 acres

(Hessburg and others, this volume). At the subwatershed scale, it is not necessary to treat an entire landscape for effective risk reduction (Ager and Finney 2007). Instead, strategic placement of fuel treatments (SPOTS) can be used to attain the desirable outcome of modifying fire behavior (McDaniel 2006).

Strategic Fuel Treatment

Wildland-urban interface (WUI) areas are generally recognized as high priorities for fuels treatment (Pyne 2004). However, treating only WUI areas alone will not achieve the wide range of human and natural resource benefits forests provide (Summerfelt 2003). The large-scale fires of 2002—Hayman (Colorado), Rodeo-Chediski (Arizona), and Biscuit (Oregon-California)—caused considerable damage and disruption in WUI areas. These fires began miles beyond the WUI where excessive fuel loadings had accumulated (USDA-FS/USDI 2006b). In addition, at higher elevations outside the WUI, wildfire can damage riparian areas and associated watershed benefits and values (Dreesen 2003, Obedzinski and others 2001).

Firesheds are large (thousands of acres) landscapes, delineated based on fire regime, condition class, fire history, fire hazard and risk, and potential wildland fire behavior. Fireshed assessment refers to an interdisciplinary and collaborative process for designing and scheduling site-specific projects (NIFC 2006b). The purpose of fireshed assessments is designing the most effective fuel treatment program with the resources available for reducing the likelihood of a large, severe problem fire (McDaniel 2006).

SPOTS is an interagency, interdisciplinary, collaborative landscape-scale GIS-based tool that has emerged from fireshed assessment efforts on several national forests in California. The strength of the process lies in purposeful dialogue between interested parties (McDaniel 2006). The important thing is to focus on exploring everyone's ideas and not trying to find one right answer. Project leaders stress this collaborative approach and active learning by participants as the key strength. Although most people have a perception of the fireshed assessments as a set of modeling tools, project managers indicate that the models really are used to promote dialogue (McDaniel 2006).

The SPOTS concept contributes to an overall understanding of the spatial dynamics of fuel and related fire behavior by employing fire modeling tools that describe fire potential on a specific landscape. The SPOTS approach considers tradeoffs between multiple treatment options by gaming fire scenarios with fire behavior and spread modeling software. The SPOTS framework meets the need identified by the U.S. Government Accountability Office (e.g., GAO 2004) to establish a consistent way to define risk and test potential solutions. SPOTS was developed in California, and, in 2005, the Forest Service and BLM tested it in eight pilot areas across the country, including central Oregon.

SPOTS analysis approaches should dovetail with the Fire Program Analysis (FPA) system, which is a new interagency planning and budgeting tool for evaluating the effectiveness of alternative fire management strategies (NIFC 2006a). The SPOTS approach also may allow managers time to implement long-term management strategies to restore ecosystems, perhaps including effective decision support for wildland fire use (WFU) (Gercke and Stewart 2006). WFU allows lightning-ignited fires to burn in order to attain planned resource management objectives.

Temporal Scale Issues

There is a lack of explicit guidance about how to consider changes in conditions that occur over the decades or even centuries required for ecological processes to play out on the landscape (Barbour and others 2005). Proposed projects that could produce benefits by reducing risks over the long term are sometimes considered unacceptable because they pose a small amount of risk in the short term. For example, in some situations prefire hazardous fuel reduction treatments can reduce sedimentation and smoke in the long term, but, in the short term, such treatments produce additional quantities of sediment and smoke that some people may consider unacceptable, no matter how small.

Compliance with the NEPA (1969) requirement for short- and long-term effects analysis raises the issue of appropriate time horizon selection. Comparison of environmental risks should be done within the same time period, and risks prevented by management programs should be included in the comparison (Davies 1996). Extinction

should be viewed over hundreds of years so that short-term considerations do not create long-term problems (NRC 1995). For risks to native fish, 100 years is a minimum (Rieman and others 2003a). By selecting the fire return interval as the minimum time horizon, the probability of a fire on the landscape is assured, and risk analysis can proceed to focus on reducing adverse effects.

National Forest System Issues

The buildup of forest fuel and changes in vegetation composition are particularly problematic on National Forest System lands (O’Laughlin 2006, O’Laughlin and Cook 2003). According to former USDA Forest Service Chief Dale Bosworth (2003), the situation on National Forest System lands is not sustainable—ecologically, economically, or socially. Active management of Federal lands that maintains forest cover and structure within a range consistent with long-term disturbance processes can reduce the potential for severe fire behavior, maintain and enhance long-term ecological integrity, and provide the mix of goods and services people want from ecosystems (Quigley and others 1998).

To improve forest health conditions on Federal lands, managers generally must support project-level decisions with NEPA (1969) analysis documents and demonstrate the short- and long-term effects on environmental values other than woody vegetation. These project-level analyses allow managers to consider protecting or enhancing specific resources (Barbour and others 2005). For example, wildfire risk and northern spotted owl (*Strix occidentalis caurina*) habitat suitability are complex issues requiring site-specific assessment and management (Lee and Irwin 2005). Managers need tools at the project level to help them work through the project approval process (USDA-FS/USDI 2004). In addition, projects need to be prioritized so that scarce resources can be used effectively (Bisson and others 2003).

The effect of NEPA—in combination with the Clean Water Act, Clean Air Act, Endangered Species Act, Federal Land Policy and Management Act, and the National Forest Management Act—has been to create increasingly difficult decisionmaking on Federal lands. The USDA Forest Service process predicament report presents an argument that because of policy-driven delays, the agency is hindered

from producing on-the-ground results, including improving forest health (USDA-FS 2002). Breaking decisionmaking gridlock is one reason for applying formal risk assessment (Lackey 1994).

Institutional Improvement

Managing ecological risks depends on an integrated approach because risks arise from many sources—hydrologic, forest, rangeland, and aquatic as well as economic and social—and reducing risks from one source may increase risk to another ecological component (Quigley and others 1998). The integration will come through social process. One such approach is illustrated by the fire/fish risk management problem (“Fire/Fish Risk Management Application”).

Ecologists generally recognize that barriers to improving ecological conditions may be more social or institutional than scientific (Szaro and others 1998). Improving the wildfire problem will be a complex, lengthy, expensive, and risky process, not only because of the ecological legacy on the land, but also the institutional legacy (Busenberg 2004). The framework for implementing the National Fire Plan is dependent on effective collaboration between Federal agencies, other levels of government, and interested parties or stakeholders (WGA 2006). Finding ways to meaningfully incorporate risk analysis—especially cooperative or collaborative risk assessment and risk management—into decisionmaking processes seems to be the most direct institutional path to on-the-ground improvements in ecosystem conditions.

Risk can be thought of as a game in which the rules must be socially negotiated within the context of a specific problem. This contextual approach highlights the need for interested parties to define and play the game, and emphasizes the importance of institutional, procedural, and societal processes in risk management decisions (Kunreuther and Slovic 1996). Risk assessment methods, assumptions, and conclusions differ dramatically across the Federal government (Cantor 1996). Standardization of policies and procedures among Federal agencies is an ongoing objective in wildland fire management (USDA-FS/USDI and others 2001). Nevertheless, different agencies can

be expected to have different perceptions of risk based upon their agency missions and policies. Unless there are appropriate forums for reconciling differences in risk perceptions among all interested parties, information developed in risk assessments is unlikely to change the way land and resource management decisions are made.

Comparative ecological risk assessment can play a role by facilitating communications between risk managers, risk assessors, and interested parties. Two things need to be accomplished with stakeholders: (a) identify the things they care about, i.e., risk assessment endpoints; and (b) communicate what is known and unknown about the cause-effect relationships of factors (“stressors”; i.e., threats or hazards) affecting those endpoints. Slovic’s (1999) advice to wildland fire managers is to forgo attempting to determine what stakeholders think may be an acceptable level of risk, and, instead, focus on demonstrating the benefits from risk management actions.

Participative governance for managing risks requires a shift of mentality, broad changes in professional and institutional practices, and the design and implementation of new instruments and procedures (De Marchi 2003). These are difficult things to change. One opportunity to incorporate societal concerns in the governance of risks is to encourage public participation from the beginning of decisionmaking processes. The twofold challenge in risk governance is first providing the forums where citizens present and debate their interests and ideas about public matters and then making such deliberations a meaningful part of democratic decisionmaking (De Marchi 2003). The design of the EPA (1998) Guidelines explicitly addresses both challenges.

EPA’s Risk Assessment Framework

Spurred by considerable political interest in the 1990s, a substantial body of literature exists on environmental and ecological risk analysis (Molak 1997). A review of laws and policies supports the conclusion that Federal land and resource management agencies, and agencies responsible for environmental protection, must use some form of risk assessment in their decisionmaking processes (O’Laughlin 2005b). Neither laws nor policies prescribe how agencies

should do risk analysis or what the end result should look like. The EPA’s (1998) *Guidelines for Ecological Risk Assessment* provides a useful starting point.

The EPA risk assessment process estimates the likelihood of the occurrence of an unwanted adverse effect (Fairbrother and Turnley 2005). At least nine Federal agencies, including the USDA Forest Service, have used the EPA (1998) “guidelines and agreed that they provide a common basis for analyzing risks” (CENR 1999). The EPA framework (Figure 1) recognizes that the interface among risk assessors, risk managers, and interested parties at the beginning (during planning) and end of the risk assessment process (during risk communications) is crucial for ensuring that the results of the risk assessment can be used to support a management decision (EPA 1998).

The first step in the EPA framework is a well-defined problem formulation built on the involvement of stakeholders as well as scientific information about the magnitude of wildfire effects. Next, risk characterization makes a comprehensive statement about risk, including assertions about uncertainty, and clearly communicates results to resource managers and interested stakeholders (Fairbrother and Turnley 2005).

Decision analysis and other structured problem-solving methods emphasize the need for clearly articulated objectives, along with criteria to evaluate how well various alternatives might meet those objectives (NRC 1995). Sustainable resource management depends on clear objectives describing desired future conditions. Objectives provide managers with targets and others with benchmarks for holding managers accountable for their actions. For risk analysis objectives, called assessment endpoints, EPA “Guidelines recommend specific ecological entities and their attributes, and caution against the use of vague ideas such as sustainability and integrity” (EPA 1998).

Fire/Fish Risk Management Application

To enable risk-reducing fuels treatment projects, managers need to take a problem-oriented approach to reducing fuels without causing irreparable harm to fish populations. I call this integrated multiple-objective situation the fire/fish risk management problem (O’Laughlin 2005a, 2005d).

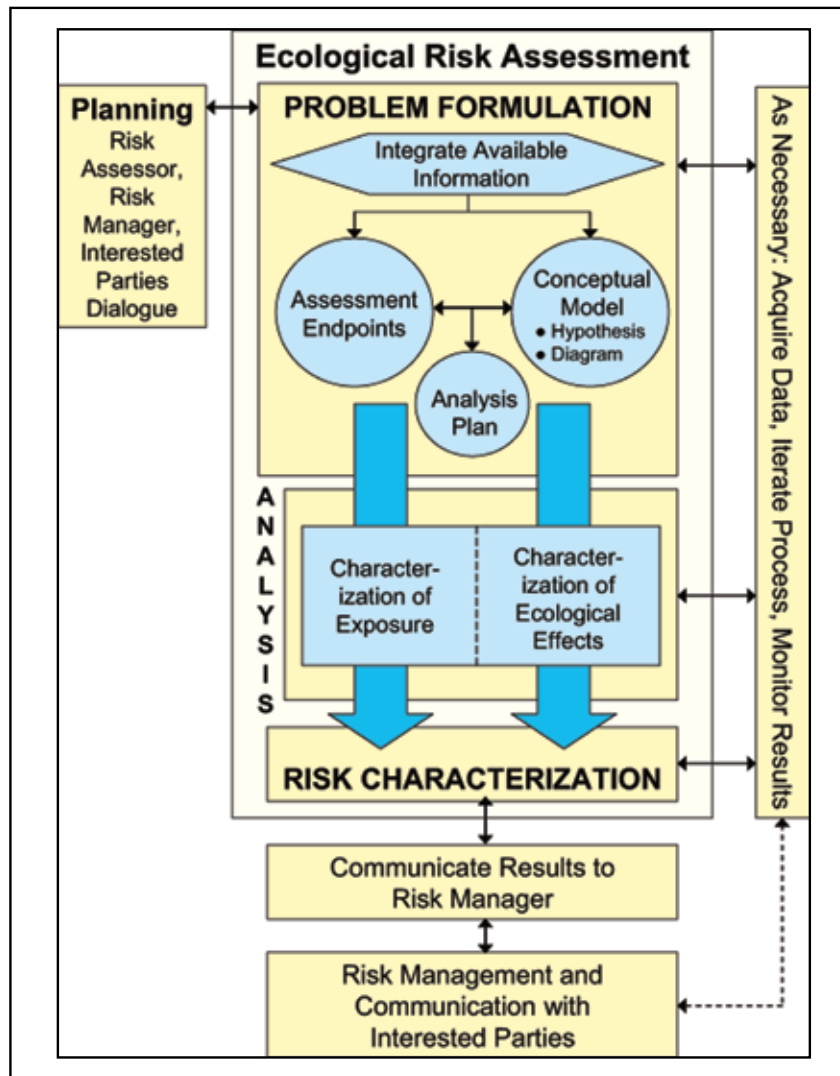


Figure 1— Framework for ecological risk assessment.
Source: Redrawn from EPA (1998)

Risk assessment can be used to support many sustainable forest management decisions, including comparing wildfire risks to various environmental values with and without fuels treatment. Herein, only the problem formulation phase of the EPA framework is covered in detail. Problem formulation (“Problem Formulation”) involves understanding the situation well enough to develop a conceptual model (“Conceptual Model”), which consists of a risk management hypothesis and a conceptual model diagram. Both of these model components clearly document the risk assessor’s thought process regarding cause and effect relationships. This approach facilitates risk characterization and communication with interested parties.

Fish are selected as the risk assessment endpoint, and the stressor adversely affecting them is sediment from logging or wildfire burning or both under different conditions that vary according to fuel loadings. A quantitative example is provided (“Quantitative Application”), and uncertainties are explicitly addressed (“Uncertainty”).

Problem Formulation

The first phase of ecological risk assessment is problem formulation, and a conceptual model is an essential part of the process (Figure 1). The inability of management and regulatory agencies and the public to articulate common goals and conceptual approaches to land management is

part of the problem, and until there is improved coordination and recognition of a common conceptual framework for management actions, conflicts are likely to continue (Bisson and others 2003).

The underlying structures of belief, perception, and appreciation people have toward situations are called frames (Schön and Rein 1994). Framing resource management problems as questions is a clarifying exercise. Lackey (1997) asked: If ecological risk assessment is the answer, what is the question? In the fire/fish context, Rieman and others (2003b) replied: Which is worse, new fires that may result from past management, or new management intended to mitigate those fires? These are good questions. Providing answers to the wrong questions and missing the relevant aspects of a problem because of inaccurate framing of a risk issue should be avoided (De Marchi 2003).

To consider the “which is worse” question in a fire/fish decision model, the relevant parameters are the adverse environmental effects of fire with and without fuel treatments and the beneficial effects of treatments. Two risk analysis experts suggest focusing on risk management benefits. Haimes (2004) cautioned that suboptimal decisions are likely unless the beneficial as well as adverse effects of current decisions on future options are assessed and evaluated to the extent possible. Slovic (1999) recommended focusing on the benefits of managing wildland fire, instead of trying to determine the acceptable level of risk from adverse effects.

Many factors adversely affect fish populations. Wildfire can cause fish mortality directly and indirectly by modifying habitat quality (Rieman and Clayton 1997). By affecting vegetation, wildfire can accelerate soil erosion rates and sediment delivery to streams (Wondzell and King 2003). Although closer integration of terrestrial and aquatic management is necessary, the lack of a common understanding or conceptual foundation is a fundamental challenge to progress (Rieman and others 2003a).

Conceptual Model

Risk cannot be managed unless it has been properly assessed, and some form of model provides the best assessment process (Haimes 2004). The EPA framework relies on

a conceptual model, and it has two principal components:

(a) a risk hypothesis describing predicted relationships among stressor, exposure, and assessment endpoint response, along with rationales for their selection; and (b) diagrams illustrating these relationships (Figure 1).

By highlighting what we know and do not know about a system, a conceptual model provides an opportunity for others to evaluate explicit expressions of the assumptions underlying decisions. Conceptual models can represent many relationships, including exposure scenarios qualitatively linking land-use activities to stressors (EPA 1998). A conceptual model for the fire/fish risk problem compares short-term effects of fuel treatment project implementation to long-term effects with and without fuel treatment, including project benefits from reducing post-wildfire environmental damage. Sediment production is the environmental effect analyzed. The idea that active management can improve conditions is a testable risk hypothesis that can be visualized and communicated in a conceptual model diagram.

Cause-and-Effect Hypothesis—

In the problem formulation phase of the EPA framework (Figure 1) the objective of the analytical phase of the assessment is called the endpoint. The stressor’s ecological effects on the endpoint are described in stressor-response profiles. The EPA (1998) guidelines illustrate these concepts using salmon reproduction and age class structure as a risk assessment endpoint and logging sediment as a stressor. A key assumption in the fire/fish risk management conceptual model is that hazardous fuel reduction treatments will reduce wildfire intensity and subsequent severity of environmental effects by reducing postfire sediment delivery.

The relationship of a sediment-causing disturbance and a fish population is illustrated in a conceptual model diagram (Figure 2) describing the relationship of quantities of sediment delivered to the stream and fish biomass (Rieman 2003). This model diagram serves as the stressor-response profile called for in the EPA’s approach to risk analysis. In the diagram, fish biomass is reduced almost immediately in response to a disturbance event, which could be either wildfire or logging. Sediment rapidly returns to the pre-event level, but it may take decades for fish biomass to return

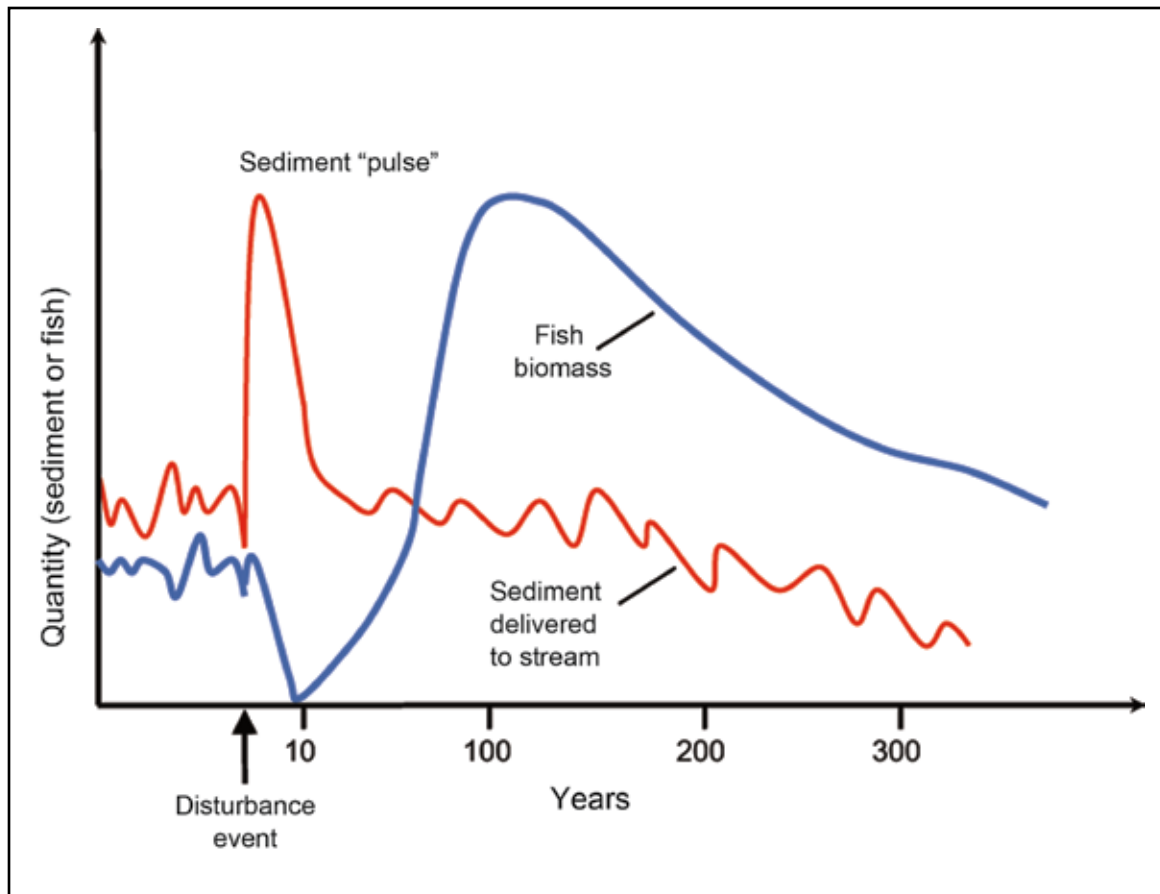


Figure 2—Conceptual model for sediment and fish relationship.
Source: Redrawn from Rieman 2003.

to the pre-event level. Over the long term, fish biomass becomes higher than before the event and remains there for centuries (Figure 2). In other words, sediment produced by a disturbance will have short-term adverse effects on fish, offset by long-term benefits.

A risk hypothesis is a fundamental component of an ecological risk assessment model (EPA 1998). Hypothesis: the benefits of restoring natural (historical) fire regimes and native vegetation on a particular site, plus the benefits of reducing the severity of effects from stand-replacing wildfires, balance favorably against any adverse effect, either short or long term, from hazardous fuel reduction treatments. The hypothesis is derived from language in a memorandum from the directors (Williams and Hogarth 2002) of the two Federal agencies charged with implementing the Endangered Species Act (ESA 1973). The memo provides guidance for ESA regulatory personnel engaging

in interagency consultation with land management agencies. It is consistent with NEPA (1969) requirements that Federal agencies analyze and document short- and long-term environmental effects of proposed major actions, including the no-action alternative.

Formulating the problem as a temporal comparison of adverse effects, however, often results in decisions to reject fuels treatment projects near imperiled species habitat. Adverse effects from fuels treatment are certain in the short term, whereas wildfire occurrence in the short term is uncertain.

An alternative problem formulation focuses on the relative magnitude of adverse and beneficial effects from wildfire burning under different fuel conditions. By selecting a long-term planning horizon corresponding to fire return interval, wildfire becomes a certainty. The magnitude of postfire effects remains an uncertainty, but such effects are

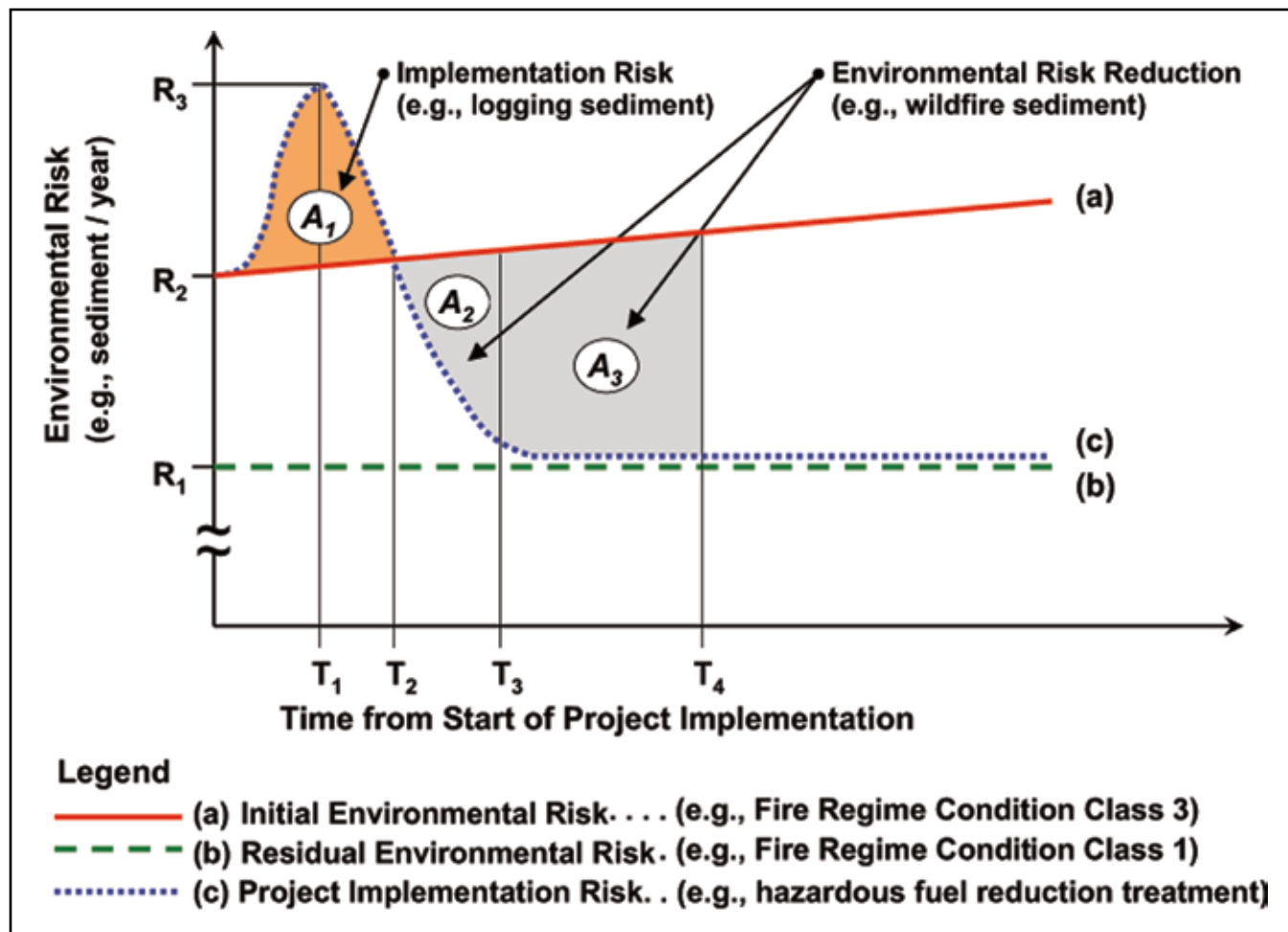


Figure 3—Conceptual model for comparing short-term fuel treatment implementation risk with long-term environmental risk reduction. Source: Redrawn from O’Laughlin (2005a); modified from U.S. Dept. of Energy (2002)

certain to occur at some level. Instead of trying to confront the landscape-level uncertainties of if, when, and where an uncharacteristically severe wildfire will occur, the environmental analysis question in the project area simply becomes “Which prefire condition produces the more desirable post-fire effect—fuel treatment or no fuel treatment?” Managers may accept the fuels treatment hypothesis, but they need to present evidence in NEPA documents to convince others who may be skeptical.

Conceptual Model Diagram—

The objective of fuel treatment is modification of fire behavior (Stephens and Ruth 2005). One way to do that is to move from a higher fire regime condition class (FRCC) (Hann and others 2003) to a lower one (see, e.g., USDA-FS/USDI

2006b). In 2000, approximately 151 million acres of Federal forest land was in FRCC 2 or 3 (USDA-FS 2001). Some of these lands could be improved by restoring FRCC 1 conditions through fuels reduction. On Federal multiple-objective lands, fuels management projects must meet a variety of objectives, including water and air quality standards. How does moving vegetation from a higher to lower FRCC affect other values? In the fire/fish example, sediment that adversely affects fish habitat is portrayed as environmental risk on the vertical axis of the tradeoff diagram (Figure 3). Other effects such as increased stream temperature from reduced shade, or $PM_{2.5}$ from different smoke regimes, could be analyzed similarly.

Line (a) is the initial environmental risk of sediment produced by a wildfire burning under uncharacteristic fuel

conditions (FRCC 3). Point R_2 is at the origin of line (a) and represents the current risk of postfire sediment; as fuels continue to accumulate over time, the postfire sediment load on line (a) increases without fuel treatment. In NEPA (1969) terminology, line (a) is the postfire effect of the “no-action” alternative. Line (b), a constant at R_1 , is residual environmental risk, which is postfire sediment associated with the management target fuel reduction objective (FRCC 1). When a wildfire occurs at any future time, the environmental risk from the condition represented by line (a) is considerably greater than that of line (b). This reflects the difference in prefire forest conditions and the severity of postfire effects as measured by sediment production from the different FRCCs.

The project described by line (c) results in postfire environmental risk reduction. Line (c) traces over time the effect of implementing a fuels treatment project. Shortly after project initiation, the implementation risk of additional postfire sediment from logging rises above and exceeds that of the initial environmental risk on line (a). At time T_1 , implementation risk is maximized at R_3 , and then it begins to decline. At T_2 , environmental risk reduction commences as the benefit of reduced postfire sediment from the fuel treatment project on line (c) drops below the amount of postfire sediment on line (a) that would occur without fuel treatment. At T_3 , project benefits continue to increase, but implementation risk still exceeds environmental risk reduction ($A_1 > A_2$). Over time, environmental risk reduction continues, and, at T_4 , project benefits exceed implementation risk ($A_1 < A_2 + A_3$). Sediment from the project results from management actions to change an ecosystem from the condition represented by line (a), or initial environmental risk, to that of line (b), or residual environmental risk.

The decision whether to undertake the management project conceptualized in Figure 3 depends on the decision-maker’s time horizon, the decision rule, and the relationship of lines (a), (b), and (c). For this discussion, the contours of the lines are similar to those in the source document (U.S. Department of Energy 2002) from which the diagram and terminology are derived. The lines may be expected to take on different configurations for specific forest types, fire regime conditions, and sediment production relationships.

For example, there is no particular reason to expect that line (a) would be linear.

Based on this conceptual diagram, it would be difficult to argue that a decision using only information at time T_3 or earlier would be more sustainable than a decision using information at time T_4 or later. Sustainability is about many things, but first among them is the consideration of inter-generational equity. Fairness of current decisions for future generations of fish or people cannot be determined with a short-term outlook.

Quantitative Application

Sediment production can be quantitatively modeled using Forest Service Watershed Erosion Prediction Project (WEPP) tools. An Internet interface for WEPP Fuel Management Erosion (FuME) is capable of providing the necessary sediment estimates for the conceptual model described in “Conceptual Model.” The sediment prediction model estimates fuel treatment sediment and simulates postfire precipitation with or without treatment, averaging the outcomes. WEPP FuME also has a road sediment feature. The WEPP FuME user interface is currently under revision (see USDA-FS 2006b).

To estimate sediment in Western United States ecoregions from fuels treatment opportunities, researchers used WEPP tools to compare effects of thinning and prescribed burning to those of wildfire in several different representative forest ecosystems. Based on their results, the relationships in Figure 3 seem reasonable. The average of predicted results, on a per unit area affected basis, was that wildfire would yield 70 times as much sediment as thinnings employed during hazardous fuels reduction efforts (USDA-FS 2005). In lieu of empirical data on sediment production from different FRCCs on a particular forest site, consider a hypothetical example based on this 70:1 relationship. In Figure 3, $R_2 = 70$ units of postfire sediment under current conditions in a project area. A thinning project would add one unit of sediment, i.e., $R_3 = 71$. By visual inspection, the target fuel reduction goal [line (b)] to produce the depicted relationship would be a very modest 2-percent postfire sediment reduction, i.e., $R_1 = 68.6$.

The BlueSky project tools could be used to similarly estimate PM_{2.5} air pollution. BlueSky is a short-term planning tool to aid land managers using fire on the landscape in making go/no-go/go-slow smoke management decisions. It is a Web-based modeling framework for predicting cumulative impacts of smoke from forest, agricultural, and range fires, including prescribed fire and wildfire. By combining data and models for fuels, fire, smoke, and weather, BlueSky makes emission, dispersion, and weather prediction model outputs easily accessible to the operational fire and air quality management communities. It provides hourly predictions of PM_{2.5} concentrations based on information available from multiagency tracking systems, wildfire reports, and, in some cases, from manually entered burn data (USDA-FS/EPA 2006).

Uncertainty

Any approach to integrating fire, fuels, and aquatic ecosystem management has inherent risks and uncertainties (Bisson and others 2003). It is not safe to ignore uncertainty because it may be important to our decisions (Morgan and Henrion 1990). However, many events that affect ecosystems (e.g., disease outbreaks, fire patterns, weather) and human systems (e.g., innovation, changes in preferences, political change) cannot be predicted in advance (NRC 2004). Owing in part to uncertainty, and in part to inadequacies in risk assessment techniques, risk analyses often have failed to meet expectations that they can improve decisionmaking (NRC 1996).

Managing risks on public lands necessitates communicating the results of risk assessment with interested parties. Risk characterization is an intermediate step (Figure 1), but one of paramount importance in risk analysis (NRC 1996). Ultimately, the condition of land and resources is more important than the terms used to describe various situations, alternatives, and outcomes. However, to avoid adding another source of uncertainty, risk assessors and managers should choose their terminology carefully. Ambiguous terms like forest health and sustainability are useful to draw people into discussions, but when deliberation about management alternatives commences, clarity is more important than ambiguity.

In the fire/fish risk management problem, nuances of definitions are less important than the risk management question—Which effect on fish is worse: (1) wildfires burning uncharacteristically under high fuel load conditions, e.g., FRCC 3; or (2) wildfires following management designed to reduce wildfire intensity to a level corresponding with FRCC 1? As discussed earlier, the question can be converted during the problem formulation phase of ecological risk assessment to a risk management hypothesis and visualized in a diagram (Figure 3). The diagram presumes that in fire-adapted forests typical of the Western United States, fire is inevitable. If the analytical time horizon is far enough in the future, fire is a certainty, and its environmental effects are realized. The magnitude of the effect, however, is uncertain and affected by many variables, including fuels (Finney 2005).

If risk can be quantified, or at least qualitatively ranked, ecosystems under greatest threat can be identified and efforts to improve these situations prioritized; few ecological risks, however, can be measured accurately (Lackey 1994). The probability of wildfire occurrence in fire-adapted forests is an exception. Fire is certain to occur within the fire return interval period, but we do not know precisely when or what the magnitude of the effects will be. Molak (1997) cautions that if uncertainty is not clearly spelled out, the numbers derived by risk analysis can be misleading. Haimes (2004) concludes with a risk analysis paradox: “To the extent that risk assessment is precise, it is not real. To the extent that risk assessment is real, it is not precise.”

Summary: Risk Assessment for Managing Wildland Fire Effects on Ecosystems

Table 1 lists some ideas risk assessors and risk managers should consider when adapting the EPA (1998) framework (Figure 1) and *Guidelines for Ecological Risk Assessment* for wildland fire risk management. This approach could be adapted to fit situations other than the fire/fish risk problem, such as PM_{2.5} emissions in prescribed fire and wildfire smoke. The conceptual model diagram (Figure 3) can enhance communications between risk managers, risk assessors, and stakeholders by graphically demonstrating whether the reduction in environmental risk following a

Table 1—A risk assessment approach for managing wildland fire effects on ecosystems

1. Keep it simple. Resource managers need project-level decision support models. Start with the essentials and add complexity only as necessary to fit the management situation.
2. Determine desired future forest conditions. The management objective is attaining a specific forest condition that will reduce wildfire risks, expressed clearly in terms the manager can be held accountable for. Many areas in the Western United States have accumulated levels of fuels that represent uncharacteristically hazardous conditions that can be categorized by fire regime condition classes (FRCC) (Hann and others 2003). An appropriate objective would be historical species composition and stand structure prior to the time fire suppression policy was implemented.
3. Select risk assessment endpoints consistent with management objectives. An appropriate endpoint for assessing logging effects is viable salmonid fish populations and appropriate quality of spawning and rearing habitat conditions (EPA 1998). The EPA's guidelines caution against using vague concepts like "sustainability" and "integrity."
4. Develop a stressor-response profile. One effect of either wildfire or management on population viability is additional sediment, a stressor from either logging management options or wildfire disturbance events. The response is an effect on fish biomass (Figure 2). A similar approach could be taken with reduced riparian vegetation and shade effects on stream temperature, or vegetation change and wildlife habitat effects, or fine particulate matter (PM_{2.5}) from prescribed fire and wildfire smoke.
5. Use an appropriately long time horizon to compare postwildfire sediment production with and without fuel treatment. If the fire/fish risk management situation involves imperiled species, anything less than 100 years is inappropriate (NRC 1995, Rieman and others 2003a).
6. Wildfire is certain to occur in fire-adapted forest ecosystems. This can be assured in risk analysis by selecting an appropriate long-term time horizon, such as the fire return interval of natural (historical) or characteristic fire regimes. This deterministic approach avoids the difficulty of assessing and communicating a probability distribution of fire risk potential, but probabilistic refinements such as the relationship of precipitation and sediment production could be used to add stochastic elements if appropriate data are available.
7. Compare the magnitude of adverse environmental effects (e.g., sediment production or PM_{2.5} emissions) on aquatic habitat quality from wildfire with and without fuel treatment.
8. Include the benefits of fuel treatment in the analysis. Fire regime condition class (FRCC) categorizations are useful for this. The effects of wildfire with fuel treatment (e.g., restoring to FRCC 1 or historical fuel load levels) should reflect the reduction of adverse postfire environmental effects, such as sediment or PM_{2.5}, that can be attributed to fuel treatment. The effect of wildfire without fuel treatment is the no-action alternative represented by current FRCCs 2 or 3. The benefit is obtained by management actions that change forests from a higher to a lower FRCC.
9. Focus on the benefits of preemptive or prefire management of forests instead of trying to determine safe or acceptable levels of risk. The problem of determining the level of risk society is willing to accept is avoided altogether, and analysis focuses on comparing two options, one against the other, rather than against a nonexistent or elusive, value-laden, socially determined standard of acceptable risk.
10. Avoid the difficulties involved in discounting future ecological effects to the present time by not discounting (Davies 1996). If economic or social considerations are added, discounting may be appropriate. To reduce bias against future generations, use a very low discount rate (Solow 1994).
11. Use quantitative data when they exist. Qualitative assessments and comparisons of ecological risks can provide useful insights for environmental decisionmaking, even if the scientific understanding of them is poor (NRC 1996).
12. Display relationships in a conceptual model decision diagram, e.g., Figure 3 that compares adverse and beneficial effects over time.

wildfire, represented by a change in sediment production, would exceed the implementation risk of pre-emptive fuel treatment. Fire management programs require repeated treatments (Franklin and Agee 2003). The model can be modified to include a series of fuel treatments to maintain desired conditions over the long term.

All the points in Table 1 have been covered in previous sections, except 10 (discounting) and 11 (quantitative and qualitative analysis). The caution on discounting is self-explanatory. Quantitative models appropriate for a given risk management situation should be used along with whatever data may be available. Deferring decisions until quantitative models and data are available, however,

may create additional risk from inaction, and qualitative approaches may be necessary.

Risk management decisions rely on a mix of science and policy; the most important role for science is providing information to be used in environmental decisionmaking (Power and McCarty 1997). Ecological risk assessment parameters can be represented quantitatively with existing data or qualitatively with expert opinion. Scientific quantification exists to aid judgment, not to supplant decisions (Clark 2002). Qualitative assessments of relative ecological risks can provide useful insights for environmental decisionmaking (NRC 1996). None of the scientific difficulties of estimation negate the importance for policy decisions of considering ecological outcomes. Interested and affected parties may want to take account of ecological effects even if the scientific understanding of them is poor (NRC 1996), as in the fire/fish risk management problem.

In conclusion, simple conceptual models used in decision analysis frameworks can be powerful communication tools (EPA 1998). The tradeoff diagram in Figure 3 is capable of demonstrating to the public, regulatory agencies, and the courts the long-term net benefits of active forest management designed to modify fire behavior. The transparency and clarity of such models can help people think through the questions of if, where, and when hazardous fuels reduction projects should be undertaken. Further development and use of conceptual models may help guide us along the path to sustainable resource management.

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Wildland Arson: A Research Assessment

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Abstract

Wildland arson makes up the majority of fire starts in some parts of the United States and is the second leading cause of fires on Eastern United States Federal forests. Individual arson fires can cause damages to resources and communities totaling over a hundred million dollars. Recent research has uncovered the temporal and spatial patterns of arson fires and their long- and short-term drivers. In statistical analyses, explanatory variables include those associated with general economic conditions and law enforcement. Research findings indicate that wildland arson ignitions are consistent with other kinds of crimes, in terms of their relationships to hypothesized factors. Arson is predictable in short and long timespans, as its rate is heavily influenced by weather, climate, fuels, and recent information on other nearby and recent arson fires. These results could be used to enhance the effectiveness of law enforcement and wildfire management resources.

Keywords: Arson, autocorrelation, crime, spatio-temporal, wildfire prevention.

Introduction

Over 0.5 million fires are set by arsonists each year in the United States, resulting in over \$3 billion in damages (Tri-Data Corporation 1997). Arson is a leading cause of wildfire in several heavily populated States, including California and Florida. Furthermore, arson fires are concentrated in urban interface areas (Butry and others 2002), where values at risk are likely to be high. Several recent large wildfires were intentionally set, including the Hayman Fire near Denver in 2002, which caused damages exceeding \$100 million (Kent and others 2003). In the Eastern United States, wildland fire is primarily a human-initiated phenomenon. Data provided

by Schmidt and others (2002) show that for 92 percent of the area burned 1986-1996 in 18 Eastern U.S. States, the causes were attributed to human-related ignition sources. During the same period, 74 percent of area burned on all Eastern Federal forests was from human-related ignition sources (18 percent by arsonists). Wildland arson ignitions in Florida compose a quarter of all fire starts. Arson fires are set for a variety of reasons, but a primary feature of these fires is that they are ignited close to high values at risk, and, hence, also threaten human safety. In spite of the potentially large economic losses associated with such events, wildland arson has received scant attention in the refereed literature, with some exceptions (e.g., Donoghue and Main 1985, Prestemon and Butry 2005). The vast majority of research into wildland fire management and policy in the United States has been concerned with wildfire suppression, fuel treatments, fire physics, and overall economic efficiency questions.

Worldwide, wildland fire setting has been a common practice of rural residents for centuries (Gamst 1974), and at least in the 20th century in parts of the United States South (Doolittle and Lightsey 1979, Kuhlken 1999). In early parts of the European settlement of America, fires were often set intentionally, for prescriptive purposes. Fires were set to shape vegetation communities, enhance forage for grazing animals, reduce pests, and clear land for agriculture (Doolittle and Lightsey 1979). However, some fires were set in the same context as many are today—for revenge against a landowner, as an act of protest, as an attempt to cover up another crime, or as vandalism. It is unclear whether the kind of relatively innocuous, managerial-type fire setting persists today in the United States. However, as we explain in this article, current wildland arson ignition patterns appear to closely align with certain behavioral patterns found with other criminal activities.

Long-term studies of intentionally set fires are non-existent. Instead, the analyst has to resort to statistical tests based on short time series to establish the validity of proposed theories. Here, we attempt to provide an overview of what is known about wildland arson today, as it has manifested itself since 1970, based on such research. We

focus on wildland arson within the context of crime, generally describing how it follows patterns similar to violent and nonviolent crimes. In this, we exclude consideration of fires ignited by children, which we consider accidental (although this carries with it several assumptions that we leave for other analysts for the moment).

Wildland Arson Background

Spatial and Temporal Scope of the Wildland Arson Problem at the National and State Levels in the United States

Wildland arson fires on national forests have exhibited conflicting trends. Arson ignitions on national forests clearly trended downward from the mid-1980s to 2004 (Figure 1), but no identifiable trend exists on the area burned by wildland arson (Figure 2). The ignition share of arson (proportion of all ignitions attributed to arson) appears to be less correlated with the ignition shares of lightning and other fire sources than those sources are to each other (Figure 3). There is, however, no research that provides statistical evidence that the apparent decline in arson ignitions has led to an overall decline in wildfire activity. Prestemon and others (2002) showed how wildfire area burned in Florida by wildland arson responds to the same kinds of factors as do fires started by other means. But Mercer and Prestemon (2005) indicate that wildfire ignitions by arsonists respond differently to influential factors—especially socioeconomic variables—compared to other fire causes. The overall decline in the trend of arson fire ignitions and nontrending area burned, however, translates into a relative decline in arson area burned compared to other fire sources (Figure 4), as total area burned by all causes has apparently trended upward.

Information from the State of Florida, which includes fires on both public and private lands, also demonstrates that arson appears to be on a negative time trend. Prestemon and Butry (2005) suggested that rising policing levels, rising wage rates, and falling poverty rates explain some of the negative trend. However, more research is needed to establish the general validity of such findings.

Wildland Arson in the Context of Crime Crime and Criminology

Wildland arson is classified as one subset of arson, a serious crime that is tracked by State and national authorities. Icove and Estepp (1987) reported that wildland arson is the third most common type of arson behind arson in residential and educational structures. These crime categories are each summarized at local, State, and national levels into an index of the number of crimes of each type per 100,000 individuals in the reporting location. Indices reported across States (and smaller geographical and political jurisdictions) include the violent crimes of murder, rape, robbery, and aggravated assault and the property crimes or nonviolent crimes of burglary, larceny, motor vehicle theft, and arson. Historically, arson has not been consistently tracked across States, so a long-term nationwide picture of arson is not available. However, a nationwide picture is available for all index crimes besides arson.

National Crime Trends

Between 1972 and 2004, nationwide index crimes have undergone a rise and then a fall (Figure 5). Crime trends have been used to empirically test economic theories of crime and to uncover the primary drivers of the observed, aggregate time trends, and the reasons for differences in rates across space (e.g., Burdett and others 2003; Corman and Mocan 2000; Gould and others 2002; Grogger 1995, 1998). Beginning from foundational work on the economics of crime by Becker (1968), research has shown that aggregate crime rates follow socioeconomic and law enforcement variables, which are linked to the opportunity costs experienced by criminals. Thus, crime-rate fluctuations are related to changing wage rates, unemployment rates, intensities of law enforcement, length of prison sentences, and the proportion of the population incarcerated.

Criminology Theories

Many theories have been used to explain levels of crime and their variations, and an extensive review of these is not possible here. Instead, we mention Becker's (1968) crime

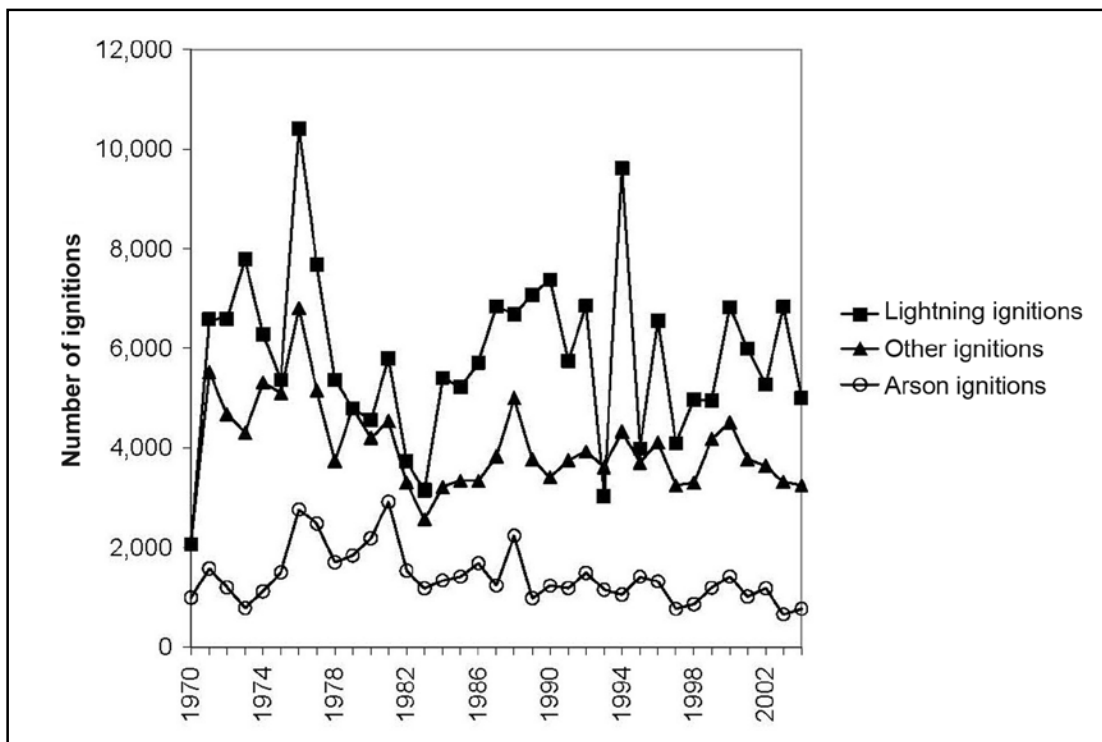


Figure 1—Wildland arson and other wildland fire ignitions on all national forests, 1970-2004. (Source: USDA Forest Service 2007)

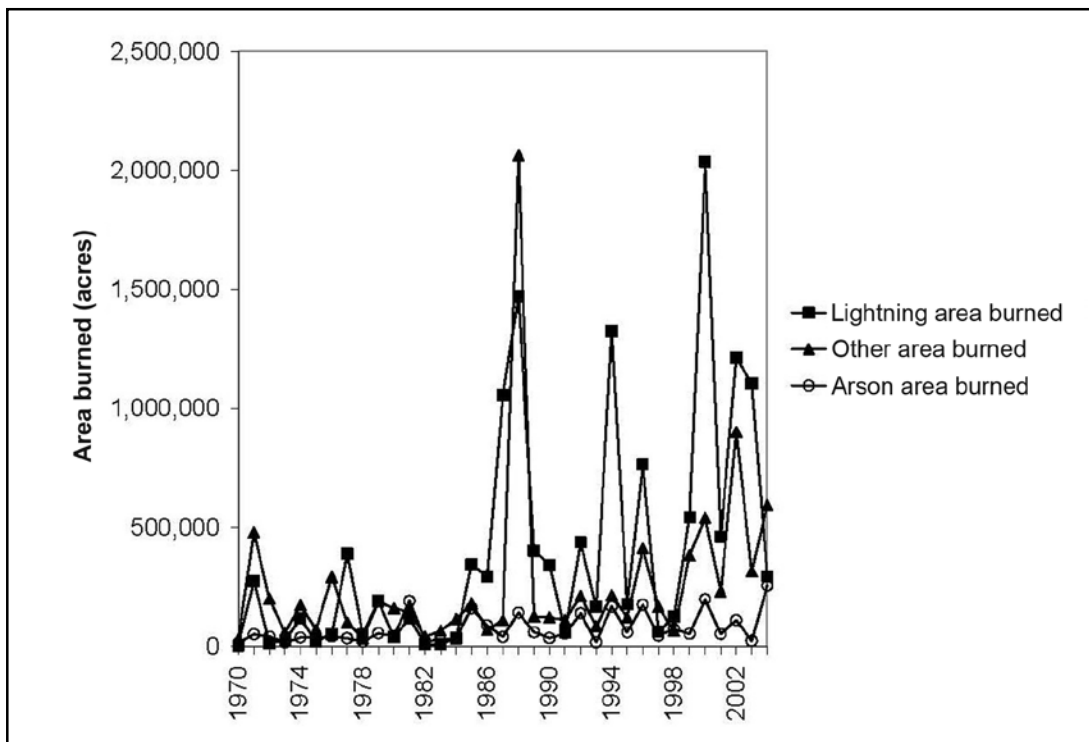


Figure 2—Wildland arson and other wildland fire areas burned on national forests, 1970-2004. (Source: USDA Forest Service 2007)

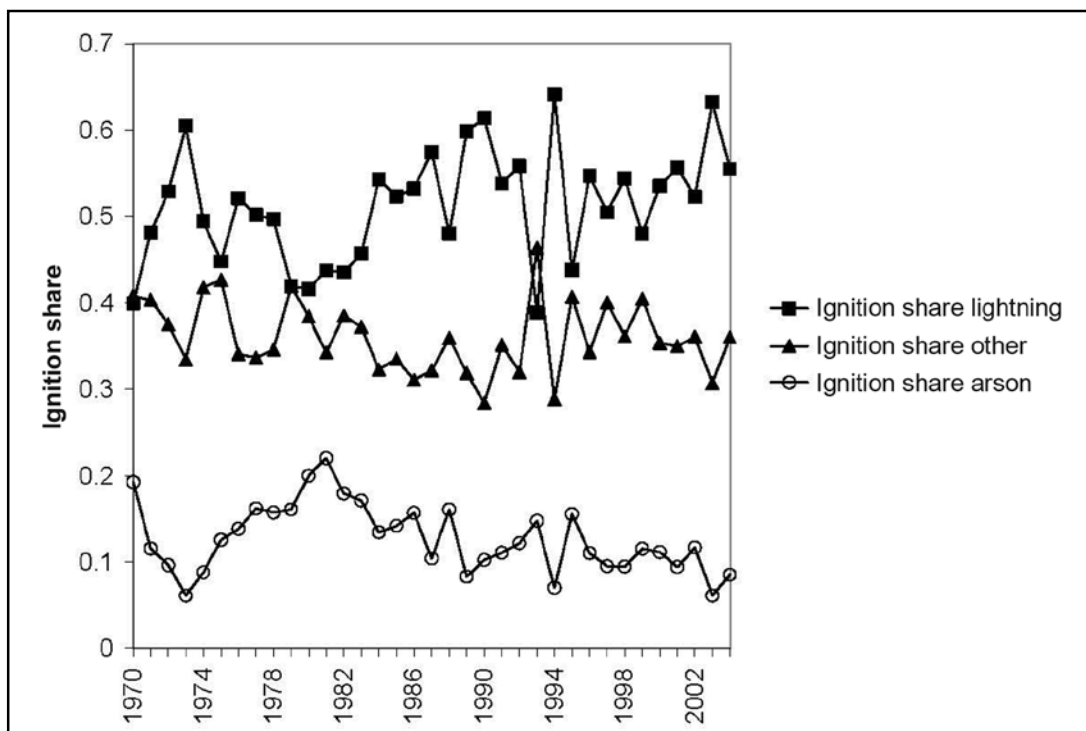


Figure 3—Share of wildland arson, lightning, and other wildfire ignitions on national forests, 1970-2004. (Source: USDA Forest Service 2007)

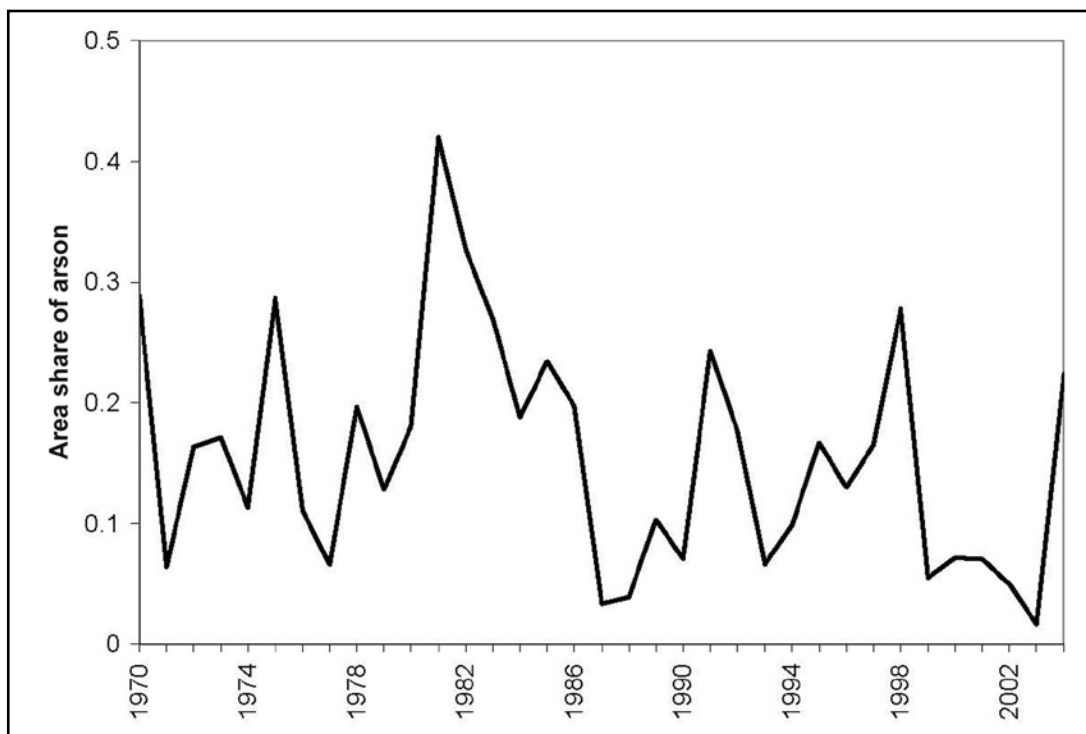


Figure 4—Share of wildfire area burned attributed to arson on national forests, 1970-2004. (Source: USDA Forest Service 2007)

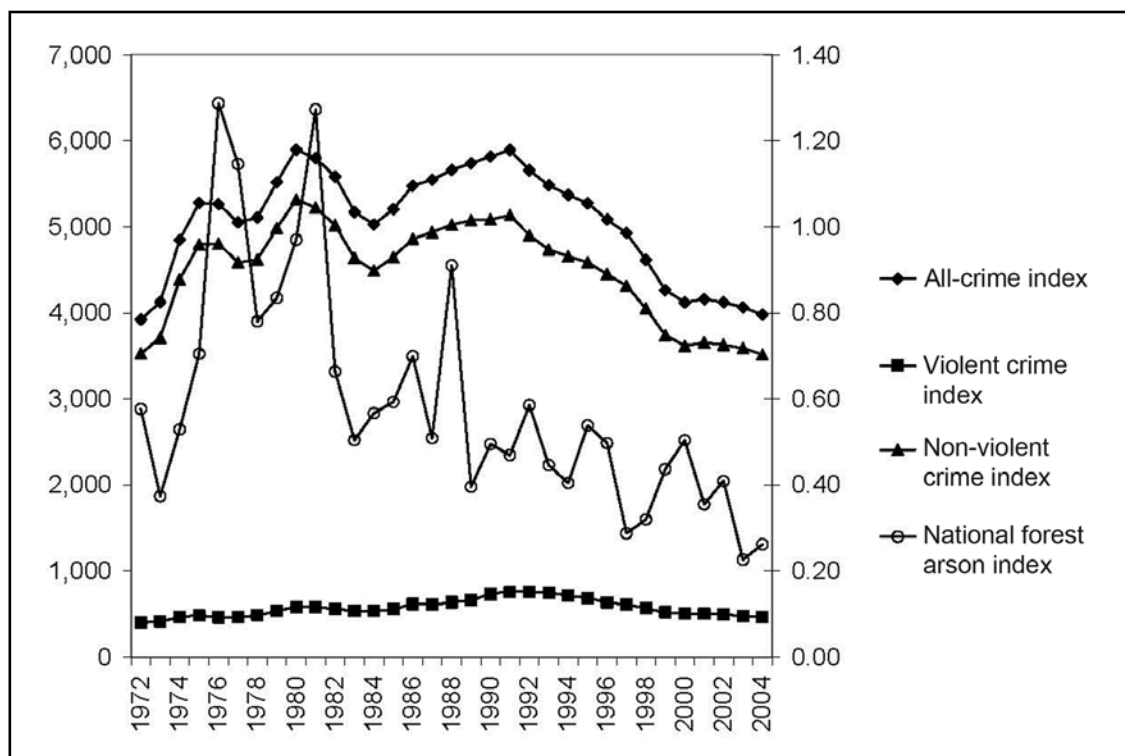


Figure 5—Nationwide index crimes (left-hand scale) and a national forest arson ignition index (right-hand scale), 1972-2004. (Sources: Federal Bureau of Investigation 2005, USDA Forest Service 2007)

function and what Cohen and Felson (1979) labeled “Routine Activity Theory.” Becker’s seminal work conceptualized the decision to commit a crime as:

$$(1) \quad O_i = O_i(\pi_i, f_i, u_i)$$

where O_i is the number of offenses committed, π_i is the probability of being caught and convicted, f_i is the wealth loss experienced by the criminal if caught and convicted, and u_i measures other factors influencing the decision to attempt and success in completion of the crime. From this, Becker (1968) expressed crime commission as generating positive utility (welfare), through either wealth generation or some enhancement of psychic pleasure for the perpetrator. Think of this as, for example, the expected utility generated from successfully igniting an arson wildfire:

$$(2) \quad E[U_i(O_i)] = \pi_i U_i(g_i - c_i - f_i(W_i, w_i)) + (1 - \pi_i) U_i(g_i - c_i)$$

where E is a mathematical expectations operator, U_i is the criminal’s utility function, g_i is the criminal’s psychic and income benefits from the illegal activity (e.g., fire setting), c_i is the production cost for the activity, $f_i(W_i, w_i)$ is the loss

from being caught and convicted of the crime (a positive function of income while employed), W_i is the employment status, and w_i is the wage rate. This theory has been tested by economists and criminologists in analyses of many kinds of crimes, and it has generally held up well. Prestemon and Butry (2005) were the first to successfully test the theory in the case of wildland arson in Florida.

In routine activity theory, crime is committed when a set of necessary conditions coincide: 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 in response to seasonal or economic differences across space and time. Variations in the crime rate across space and time in the United States, then, can be explained by variations in the convergence or the availability of all necessary ingredients. It would be possible to express wildland arson as deriving from routine activity theory by recognizing that the “target” is wildland (or property owned by a target); it would have to

be augmented, however, to accommodate weather and fuel conditions.

A challenge of all theories of crime is applying the results of the research to decisionmaking. One way that criminologists have done this is by developing computer-based tools that are derived from mathematical models of criminal activity. The mathematical models are loosely based on statistical representations of routine activity theory and the economics of crime in the context of Becker (1968). These models can identify crime hotspots, or crime intensity maps, in space and time. Hotspotting crime research is concerned with developing mapping tools for law enforcement and other authorities that can be used to identify crime hotspots and therefore aid in crime control (Bowers and Johnson 2004, Johnson and Bowers 2004). The idea of crime hotspots in space and time is not new. Lottier (1938) pointed out how crime is concentrated in locations across space, which is useful for targeting law enforcement. Boggs (1966) evaluated urban crime patterns and how they tend to be concentrated in space-time dimensions. It seems likely that law enforcement has recognized this kind of clustering for as long as crime has existed. For wildland arson, few efforts have been made to understand the spatial and temporal extent of clustering. However, Butry and Prestemon (2005) developed a preliminary statistical tool for hotspotting of wildland arson, specified at the census tract level and on daily arson ignitions.

Spatio-Temporal Crime Processes—

Crime as a Spatio-Temporal Process

Butry and Prestemon (2005) and Prestemon and Butry (2005) found that wildland arson is a crime that may be particularly suited to hotspotting. Wildland arson, like other crimes, is concentrated in space and in time, due to concentrations of criminals and fuel quantities in space and the concentration of amenable fire-setting weather and dry fuel in time. Genton and others (2006) showed how this clustering in space can last many years, whereas Prestemon and Butry (2005) showed concentrations at the daily time scale, and Butry and Prestemon (2005) showed concentrations at the daily time scale in relatively limited geographical areas.

Wildland Arson as a Spatio-Temporal Process

Prestemon and Butry (2005) found that, for small county aggregates in Florida, wildland arson is concentrated in time, with an elevated risk for more such ignitions for up to 11 days after an initial ignition. This kind of temporal clustering is consistent with models of serial and copycat criminal activity, patterns, and behaviors observed for other crimes (Brandt and Williams 2001, DiTella and Schargrotsky 2004, Surette 2002). Also important are weather, as measured by the Keetch-Byram Drought Index, which indexes fuel conditions on fine temporal scales and captures the effect of fuel moisture on ignition success rates or the cost of igniting a wildfire; historical wildfires and prescribed fire in the location, which reduce fuels and usually reduce arson risk by making fire setting more difficult or costly; and intra-annual patterns of weather, as measured by month dummy variables, which are probably also related to fuel conditions and therefore fire setting cost or success. Other explanatory variables, which fit an economic model of crime, include police per capita, the retail wage rate, and poverty. Daily variations were also found to matter, with arson more common on Saturdays and sometimes holidays.

Butry and Prestemon (2005) evaluated wildland arson patterns in Florida, where ignitions were geolocated to the census tract. This analysis related wildland arson ignitions in a single day to, in addition to the same set of variables used in Prestemon and Butry (2005) at the county level, wildland arson ignitions in previous days in the same and neighboring census tracts in six high-arson tracts in the State. Their analysis found that wildland arson ignitions in the census tract are related positively to ignitions in the same tract in the previous several days and in local (immediately surrounding) and regional (an outer shell of) neighboring tracts in the previous 11 days. They found that a current day's count of ignitions could be explained by local neighbors for up to 11 days, regional neighbors for up to 4 days, and ignitions in the same tract for up to 10 days. Thus, it appears that short-term arson ignition process propagates like a contagion (although the exact pattern in space-time was not identifiable).

In research similar to that reported by Prestemon and Butry (2005), we report here an analysis of arson ignitions

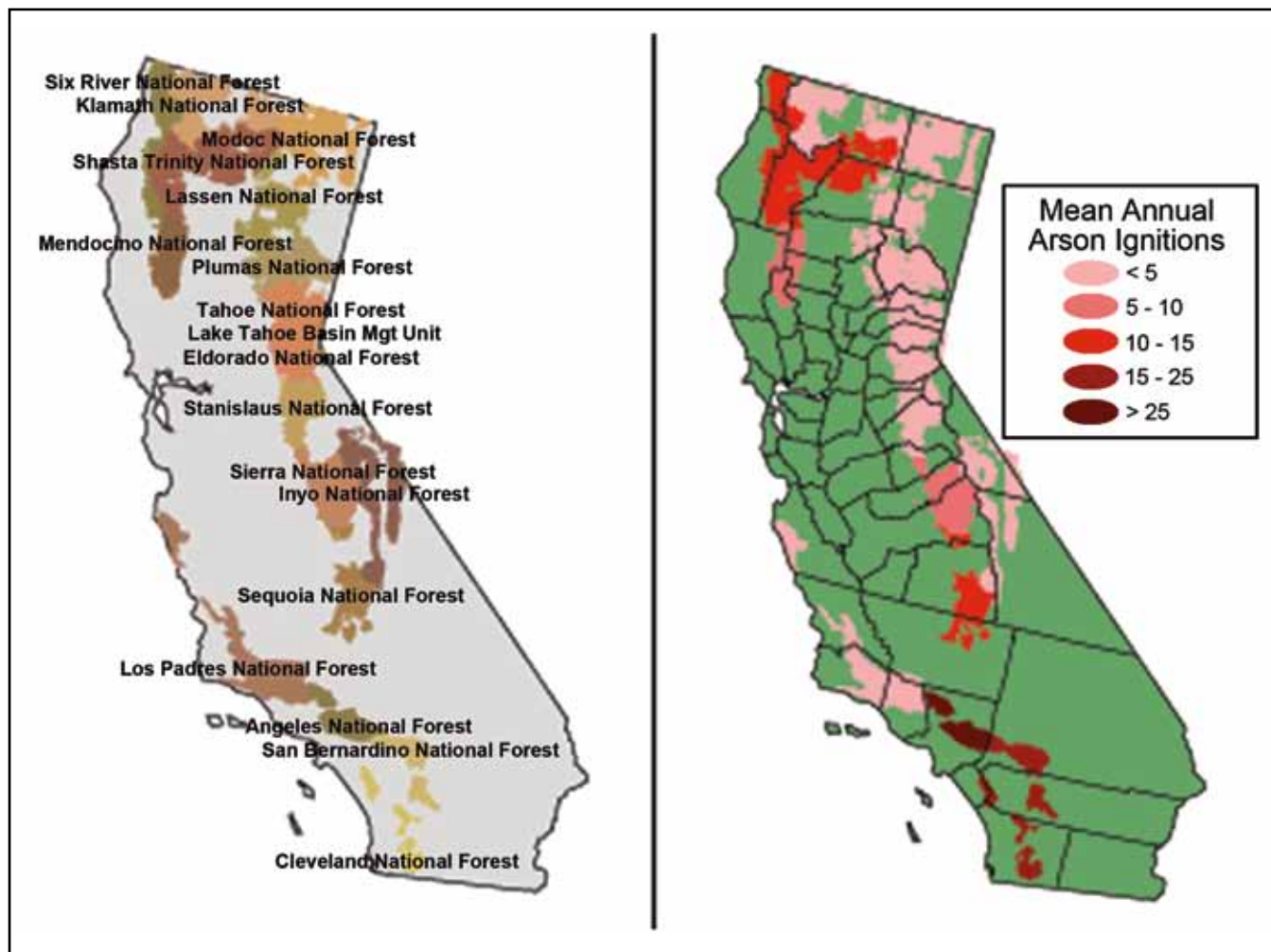


Figure 6—National forests and wildland arson on national forests in California, 1970-2003.

on national forests in California (Figure 6). We estimate daily and annual models of arson fires at the national forest level. Models use wildfire data from the USDA Forest Service National Interagency Fire Management Integrated Database (NIFMID) (USDA Forest Service 2004), unemployment and population data from the California Department of Finance (2005), law enforcement data from the California Department of Justice (2005), and climate and weather data from the National Oceanic and Atmospheric Administration (2005) and the National Climatic Data Center (2004). The models are generally specified as in Prestemon and Butry (2005), capturing temporal autocorrelation and not directly quantifying spatio-temporal patterns, but omit information on wages and poverty. Also, unlike in Prestemon and Butry (2005), the number of sworn full-time

equivalent police officers, a measure of law enforcement effort, is lagged one year to avoid issues of simultaneous determination with crime and to allow for lagged perceptions among potential arsonists on arrest probability (Lochner 2007). As in Prestemon and Butry (2005), a Poisson autoregressive model of order p (Brandt and Williams 2001) is specified for the daily ignitions for the San Bernardino, Sierra, Cleveland, and Angeles National Forests, 1994-2002 (Table 1). An annual panel fixed-effects Poisson model (Greene 2003) is specified using data from all 18 national forests in the State, 1993-2002 (Table 2).

In the daily model, significant variables (at five percent) influencing arson ignitions on the two national forests studied include up to five ignition lags (positively); alternative models that drop these lags (i.e., non-autoregressive

Table 1—Poisson autoregressive models of order p of daily wildland arson fire occurrences for selected high-arson California National Forests, 1994-2002

	San Bernardino N.F.	Sierra N.F.	Cleveland N.F.	Angeles N.F.
Intercept	-1.49	26.97	-1.73	34.65***
Average annual Palmer drought severity index, t	0.93	3.11	-0.65	0.24
Weekday	-0.66***	-0.35	0.16	-0.12
Saturday	-0.74**	1.22**	0.66*	0.40
Sunday	-0.47	-0.54	0.76**	0.15
Holiday	-0.36	1.57**	0.35	0.17
January	-0.20	n.p.	-0.73	0.37
February	1.07	n.p.	-1.18	0.87
March	-0.72	1.97	-0.36	0.20
April	-0.62	1.62	0.22	1.15**
May	1.46*	0.55	0.66	1.55***
June	1.72**	0.96	1.22*	1.57***
July	1.62*	1.72	1.32*	2.04***
August	1.96**	0.96	1.36**	2.16***
September	2.02**	-0.25	1.72***	1.75***
October	1.84**	-0.30	0.87	1.37**
November	0.82	1.26	0.46	0.81
Law Enforcement Officers Per Capita, t-1	-1.90	13.67 *	-17.09*	-1.75
Population, t	n.p.	-0.088	5.6549	-0.0028*
Unemployment, t	4.76*	-6.37**	10.39***	-3.71*
Wildfire, t-1 to t-3	0.02	-1.12	-0.37**	-0.08*
Wildfire, t-4 to t-6	0.01	0.59	-0.14	-0.11
Wildfire, t-6 to t-9	0.06	0.77	0.03	-1.02**
Wildfire, t-10 to t-12	-0.23	0.06	0.02	-1.09**
Wildfire, t-13 to t-15	-0.05	0.75	-0.27**	-0.04
AR(1)	0.29***	0.49**	0.37***	0.26***
AR(2)	0.38***	n.p.	0.60***	0.16***
AR(3)	0.24**	n.p.	n.p.	0.17***
AR(4)	n.p.	n.p.	n.p.	0.21***
AR(5)	n.p.	n.p.	n.p.	0.16***
n	3,089	3,090	3,091	3,103
Likelihood ratio statistic, model all AR terms are zero (df = p)	38.37***	7.47***	70.77***	142.32***
Log likelihood, model	-651.60	-257.29	-566.56	-896.62

Note: ***indicates significance at 1 percent, **at 5 percent, and *at 15 percent; "n.p." indicates that models with this variable included would not converge in maximum likelihood estimation.

alternative versions) explain significantly less variation in daily wildfire at high significance (last row of Table 1). This result is consistent with serial or copycat fire setting activity. The model also found that many month dummy variables are significant, which indicates that arson, like other fire causes, has a seasonal pattern that may be related

to seasonal weather and fuel moisture variations that affect the difficulty or costs of successfully igniting arson fires. Also, for these national forests, the coefficient on the Saturday dummy is significant and positive in two cases (Sierra N.F. and Cleveland N.F.) and the coefficient on the holiday dummy variable is significant and positive in one

Table 2—Fixed-effects panel Poisson model of wildland arson fire occurrences, 18 California National Forests, 1993–2002

Variable	Coefficient	Standard Error
Population, t	-7.89E-07	3.11E-07**
Unemployment percent, t	4.81E-02	3.05E-02*
Per capita sworn law enforcement officers, t-1	-655	195***
Pacific decadal oscillation, t	0.50	0.09***
Palmer drought severity index, January-March, t	-0.21	0.05***
Palmer drought severity index, April-June, t	0.13	0.04***
Palmer drought severity index, July-September, t	9.1E-02	2.8E-02***
Palmer drought severity index, October-December, t	-9.8E-02	2.5E-02***
Nino-3 SST anomaly, t	-0.43	0.07***
Wildfire area, t-1	-2.14E-06	1.35E-06*
Wildfire area, t-2	4.78E-06	1.10E-06***
Wildfire area, t-3	3.47E-06	1.27E-06***
Wildfire area, t-4	-7.21E-06	2.37E-06***
Wildfire area, t-5	-1.03E-06	2.44E-06
Wildfire area, t-6	-2.07E-06	1.49E-06
Wildfire area, t-7	6.16E-07	1.58E-06
Wildfire area, t-8	-3.57E-06	1.66E-06**
Wildfire area, t-9	-9.96E-07	1.46E-06
Wildfire area, t-10	-2.44E-06	1.66E-06*
Wildfire area, t-11	2.58E-06	1.23E-06**
Wildfire area, t-12	4.72E-06	1.42E-06***
Observations	180	
Number of cross-sections (18 national forests)	8	
Number of periods (10 years, 1993-2002)	10	
Log-likelihood model	-521.96	
Likelihood ratio statistic, model versus intercept only (21 d.f.)	1,559.56***	

Note: ***indicates significance at 1 percent, **at 5 percent, and *at 15 percent.

case (Sierra N.F.), both results indicating higher arson fire probability, which, in the context of an economic model of crime, is consistent with lower opportunity costs of time faced by arsonists those days. In contrast, dummy variables for weekdays are not significant, indicating that those days of the week have the same arson probabilities as Mondays. We also find that the average annual Palmer Drought Severity Index is not significantly related to fires. The unemployment rate is negatively related to arson ignitions in two cases (Sierra N.F. and Angeles N.F.), which conflicts with our expectation, but positive in two cases (San Bernardino N.F. and Cleveland N.F.), which does fit with our expectation. A similar conflicting result occurs for law enforcement: higher law enforcement levels are significantly and negatively related to arson rates in one case (the

Cleveland N.F.) but are positively related to arson in another (Sierra N.F.). The latter finding is not expected and could be a consequence of omitted variables related to wages or other justice-related expenditures (e.g., sanction levels). But, the former finding for the Sierra N.F. would be expected based on the opportunity costs of being caught and convicted of setting an arson wildfire. Finally, lagged wildfire area (the running total of the area burned by wildfire in the national forest in previous years) is negatively related to arson wildfire in all cases where statistical significance is found. This is consistent with our expectations, as wildfires can reduce landscape fuels, providing the prospective arsonists with greater difficulty or higher costs of successful arson fire setting. In summary, the majority of these findings are consistent with those of Prestemon and Butry (2005),

supporting an economic model of crime and one that confirms the temporal clustering of arson ignitions on these forests. However, some conflicting results lead us to conclude that further research into the daily arson fire setting process for California National Forests is necessary.

In contrast with our uncertainty about the role of law enforcement and socioeconomic factors in wildland arson, the annual models provide results consistent with expectations (Table 2). Here, law enforcement officers per capita are significantly and negatively related and unemployment is significant and positively related to wildland arson ignitions. The finding of the deterrent (negative) effect of policing on wildland arson is entirely consistent with crime theory, indicating that either (i) arsonists perceive higher opportunity costs of being caught, or (ii) more arsonists are caught and convicted and, hence, are removed from the arson fire setting population.

Biophysical variables also explain significant variation in annual levels of wildland arson ignitions on national forests: lagged wildfire (positively for 2-, 3-, 11-, and 12-year lags and negatively for 1-, 4-, 8-, and 10-year lags); the Pacific decadal oscillation (positively); the Palmer Drought Severity Index (positively for 2- and 3-quarter lags and negatively for 1- and 4-quarter lags); and the Niño-3 SST anomaly (negatively) (Table 2). Population is negatively related to wildland arson risk, a finding that we cannot fully explain, except in the context of omitted variable bias. Nor can we explain the unusual statistical correlations between arson ignitions and longer lags of wildfire activity, so we leave this to future research.

What these results for the California National Forests show is broad consistency with the results found for Florida: wildland arson ignitions are clustered in time (5 days in two high-arson national forests in California, up to 11 days in Florida); law enforcement is generally negatively related to arson rates; climate and weather variables matter, in a more complex intra-annual pattern in California than in Florida; and fuel levels matter, although in a more complicated way in California than in Florida. Left uninvestigated are the influences of other labor market variables, poverty, and other measures of criminal sanctions.

Summary and Conclusions

Law Enforcement Lessons and Programs

Beginning with initial studies by Donoghue and Main (1985) through studies by the authors reported here, it seems clear that law enforcement deployment and other efforts to apprehend and incarcerate arsonists work to reduce wildland arson in the long run in high-arson locations in the United States. As found by Prestemon and Butry (2005) and Butry and Prestemon (2005), wildland arson appears to be clustered in time and space. Law enforcement personnel could use these results to advance hotspotting models for wildland arson or develop tactical responses to reducing the number of such ignitions in such outbreaks or both. Although it may not be clear that reducing wildland arson ignitions results in large-scale and long-run reductions in the amount of area burned on an annual basis, reducing such ignitions could have significant benefits for society, especially in places where arsonists tend to set fires: closer to built-up areas with greater values at risk (Butry and others 2002, Genton and others 2006). The results of more recent research indicate that it might be worthwhile to redirect law enforcement efforts to certain locations during periods of weak labor markets and even higher poverty rates. In this case, however, we caution that careful analysis is needed that would quantify the tradeoffs of redirection away from other policing activities. During certain months of the year and also during droughts, arsonists are more active, so law enforcement could also pay special attention to weather and fire season variations. As well, from a strategic standpoint, authorities could also monitor trends in climate variables or their predictions (Ji and others 1998) as indicators of broad trends in climatic factors that create conditions favorable for fire setting. Special attention to weather and climate is important, as conditions favorable to ignitions may also favor large and intense fires once they are successfully ignited.

Wildland Manager Lessons

Wildland managers can use the same lessons as indicated for law enforcement. There is a distinct degree of seasonality in arson wildfire ignitions, and firefighting resource allocations can conform to this seasonality. Arson ignitions

are correlated with dry weather in ways similar to other ignition types, so wildland managers should be ready for these kinds of fires, even in times when fires are banned (perhaps reducing accidental ignitions) or when lightning storms are not occurring. In time scales longer than days or weeks, managers can also pay attention to forecasts of ocean temperatures (e.g., Ji and others 1998), which might foretell upcoming drought situations that would raise future arson risks and therefore plan firefighting resource allocations accordingly. Finally, wildland managers might be able to reduce landscape fuel levels in ways that reduce wildland arson opportunities.

Future Research and Development Needs

Modern studies of the spatial and temporal patterns of wildland arson are few, and much remains to be investigated. In our view, key needs include understanding how wildland arson fits within the larger picture of structural arson and other crime. The research results reported in this paper are only suggestive of similarities between wildland arson and other crimes, but real connections have not been identified and quantified. Many criminals commit multiple crimes of different types, and it is possible that wildland arsonists do the same. Hence, it may be possible to include information on structural arson or other crimes to improve the accuracy of statistical models of wildland arson.

Another need is to develop hotspotting models of wildland arson that would be applicable in different locations and useful for wildland managers and law enforcement. Before operational models are developed, more work needs to be done to understand the spatio-temporal patterns of wildland arson. To date, these patterns fit with spatio-temporal crime functions found by other criminologists and modelers. A key challenge in development of hotspotting tools is operational usefulness. The hotspotting tools envisioned may require real-time data updating that may not be possible for law enforcement agencies or land management organizations with tight budgets. However, if such tools bring long-term savings for fire and police organizations through reduced firefighting and fire investigation activities and lower property losses, investments would lead to net societal and agency budgetary gains.

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Review of Methods for Developing Probabilistic Risk Assessments. Part 1: Modeling Fire

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Abstract

We describe methodologies currently in use or those under development containing features for estimating fire occurrence risk assessment. We describe two major categories of fire risk assessment tools: those that predict fire under current conditions, assuming that vegetation, climate, and the interactions between them and fire remain relatively similar to their condition during recent history, and those that anticipate changes in fire risk as climate and vegetation communities change through time. Three types of models have proven useful for predicting fire under current conditions: (1) biophysical models that predict fire from vegetation type, fuel load, and climate; (2) statistical models; and (3) fire behavior models. Programs such as LANDFIRE have great promise for using biophysical properties to estimate risk. Statistical models that use historical data to predict fire probabilities if landscape-fire relationships continue to remain relatively unchanged, are gaining interest as more data become available. Fire behavior models are producing accurate predictions of the ways individual fires will move across the landscape. For longer periods, fire risk needs to be evaluated by models that predict the ways vegetation communities will change over time because these changes will alter fire probabilities. We identified models capable of being used to track changes in vegetation and the resulting effect on changes in fire frequency. Risk systems need to be designed to track changes in fire susceptibility as the climate changes, using models such as MAPSS.

Prediction of fire occurrence is just the first part of a complete analysis of risks associated with fire. Fire occurrence risk needs to be combined with models that determine the risk of the effects of fire. Models that predict mortality, fuel consumption, smoke production, and soil heating

caused by prescribed fire or wildfire should be used, as well as those capable of evaluating second order effects, such as changes in site productivity, animal use, insects, and disease. Fire must be looked at in the context of other stresses, such as invasive insects and pathogens, encroaching urbanization, and loss of critical habitat. There are interactions among stresses that play a role in affecting the frequency and intensity of fire, and fire, in turn, can affect the probability of those stresses. Consequently, risk evaluation systems need to be created that can simultaneously estimate the probability of other major stresses influencing ecosystem development.

Keywords: Fire prediction, fire susceptibility, modeling, risk assessment, wildfire.

Introduction

Methodologies are described here that may be useful for estimating fire occurrence risk assessment, including the probability of ignition and the spatial spread and intensity of the fire during its lifetime. Two types of risk need to be assessed: (1) fire risk occurrence (hereafter referred to as fire risk), and (2) risk to the ecosystem as a result of fire (hereafter ecosystem risk) (Finney 2005). For our purposes, fire risk includes the probability of ignition and the spatial spread and intensity of the fire during its lifetime. Ecosystem risk includes all of the consequences to plant and animal populations and to the soil during the recovery period once the fire has concluded. There is significant understanding of the factors that influence fire risk, with many studies analyzing the long-term consequences to ecosystems (Fairbrother and Turnley 2005). Tools to predict the likely distributional frequency of fire risk under combinations of various conditions are the focus of this review.

The two major categories of fire risk assessment tools are (1) those that predict fire under current conditions, assuming that vegetation, climate, and the interactions among them and fire remain relatively similar to their condition during recent history; and (2) those that anticipate changes in fire risk as climate and vegetation communities change through time.

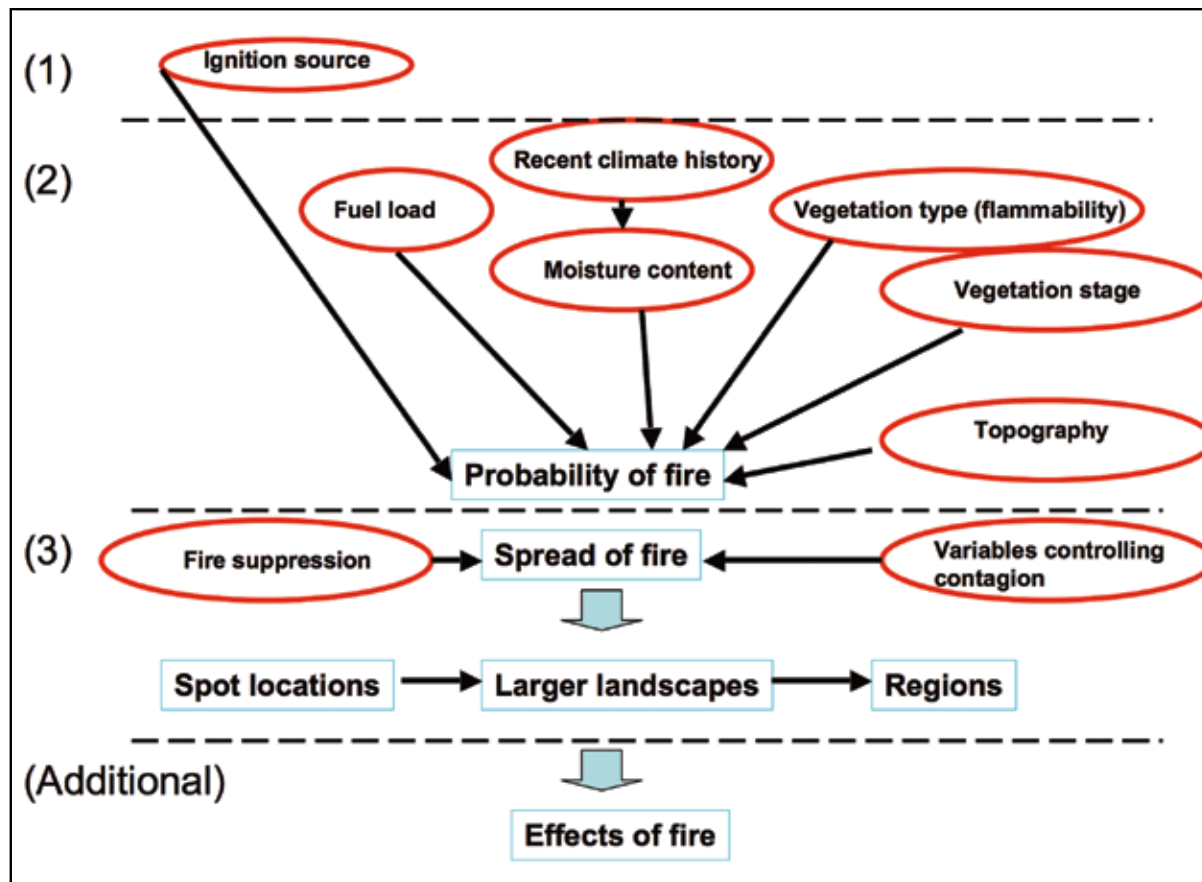


Figure 1—Calculation of fire risk. There are three aspects to predicting fire. Fire is predicted from (1) the probability of ignition; (2) the biophysical influences on fire, such as fuel load, moisture content, flammability of the vegetation, and topography; and (3) the spread of fire once it gets established. Effects of fire can then be calculated to form a complete estimation of the risks related to fire.

Fire Risk under Current Conditions

Many models are available to evaluate fire risk under current conditions, although the majority of these were designed for analysis in specific locations or under specific sets of conditions. One of the initial attempts to implement a more generic assessment of fire risk was the National Fire Danger Rating System (Burgan 1988, Deeming and others 1972). This system relied on expert judgment to evaluate the risk from a set of explanatory variables, principally fuels, topography, and weather. It allowed land managers to estimate fire danger for today or tomorrow for a given rating area. It characterized fire danger by evaluating the approximate probable upper limit of fire behavior in a fire danger rating area during a 24-hour period. A relative rating of the potential growth and behavior of any wildfire was based on

a loose correlation between the date of fire discovery and the eventual size of a fire. Attempts to improve upon the National Fire Danger Rating System fall into one of three categories:

1. Biophysical models that predict fire from vegetation type, fuel load, and climate.
2. Statistical models that rely on relationships extracted from historical data.
3. Fire behavior models that emphasize the role of spatial distributions of wind, topography, and fuel in determining what portions of a landscape will burn.

As an example, the first and third of these can be combined in a fire risk calculation (shown in Figure 1) in which fire is predicted from the biophysical influences on fire and from the spread of fire once it gets established.

Biophysical models combine local weather patterns (temperature, humidity, wind), vegetation (fuel type, moisture level), and topography (elevation, slope) to arrive at an estimate of fire risk. Biophysical models range from deterministic models in which a given set of input variables will always yield the same prediction, to systems that include simulation. Some of these models combine quantitative and qualitative criteria to arrive at a fire danger index. Biophysical models designed to estimate fire risk across large regions tend to base their predictions on fewer variables because the number of variables that have been quantified across large regions is limited.

Statistical models use regression models developed from historical data to estimate probabilities of fire occurrence under various local environmental conditions. Statistical models are valuable for understanding general historical trends. They are used to provide predictions for such factors as the expected number of fires in an area from explanatory variables such as vegetation patterns, fuel moisture conditions, meteorological variables, number of people visiting a forest, and past history of fires. We describe these types of models only briefly because there are currently few published examples of statistical fire models available. However, the increasing availability of large historical data sets on fire frequency will undoubtedly cause a great expansion of studies using this method in the near future.

Fire behavior models attempt to characterize the propagation and spread of fires under various environmental conditions. Fire behavior probabilities are dependent on ignitions occurring off-site and the fuels, topography, weather, and relative fire direction allowing each fire to reach that location. Because these models require input variables that may not be available over large regions such as entire national forests, they tend to be applied to specific watersheds in which these properties have been mapped or quantified. Because there are many possible interactions of weather with spatial landscape features, fire behavior predictions require the use of spatial fire spread simulations.

To provide a sense of how the different types of approaches are applied and which resources each might require, we discuss in detail examples from each category. Biophysical models are first discussed, with particular

attention to two such systems, LANDFIRE and WALTER. Because LANDFIRE is the newest, and, perhaps, most complex approach being pursued, we discuss this approach in depth. WALTER is described as an example of an approach that makes extensive use of expert opinion. An example of the statistical modeling approach relying on large historical data sets is described with a discussion of a probability-based model. Finally, we discuss a suite of fire behavior models—FARSITE, BehavePlus, FlamMap, NEXUS, and Visualized Fire Simulation (VFS).

Fire Risk under Changing Vegetation or Climate

Many models have been constructed with capabilities of projecting changes in vegetation composition over time and the way these changes alter fire risk (Keane and others 2004). Other models are capable of simulating how changes in climate will change biological processes and interactions in ecosystems. Slow changes in vegetation or climate or both can be incorporated into the fire risk calculation using these models. Recent reviews (Keane and others 2003) have concluded that simulation modeling produces the best predictive ability under changing climate and vegetation conditions compared to prediction based on either biophysical properties or statistical prediction from historical data. Over longer time periods, climate and vegetation change is very likely to occur.

We discuss the advantages of these models, building on the extensive evaluations of succession fire models (Keane and others 2004). Further, we attempt to place the models they considered in the context of a wider range of tools. We discuss the four most widely used succession models that contain processes that link vegetation change to fire prediction: SIMPPLLE, MAGIS, VDDT, and TELSA. Many other available models perform well for specific locations for which they were designed. We focused on these four because they have the potential to be easily applied to many different areas in the Western United States.

A final category of simulators includes those that utilize broad concepts of biological processes to project the ways forests are likely to change and where various forest types are likely to be found under future climate conditions. Although this type of model has had a long history

of development in forest ecology, it is only recently that several models have had fire intimately incorporated into the model structure. We describe the MAPSS model as an example of this category.

Biophysical Fire Risk Systems

Biophysical fire risk models traditionally use regional characteristics of weather patterns (temperature, humidity, wind), vegetation (fuel type, moisture level), and topography (elevation, slope) to produce a prediction index of fire risk based on historical correlations among these variables and fire. More recently, models are being developed that substitute spatial simulations for some of these correlations.

LANDFIRE

In response to a need for a national evaluation of the spatial distribution of fire risk, the USDA Forest Service developed a partnership with four other agencies to develop LANDFIRE (<http://www.landfire.gov/index.html>). The goal of LANDFIRE is to identify areas at risk because of the accumulation of hazardous fuel for the purpose of prioritizing hazardous fuel reduction projects and improving hazardous fuel treatment coordination between agencies. The program is designed to produce landscape-scale maps and data describing vegetation, fire, and fuel characteristics across the United States (at a 30-m grid resolution) (Keane and others 2004, Rollins and others 2002, 2004; Schmidt and others 2002). LANDFIRE is providing many of the raw materials that will be necessary to produce an estimate of fire risk. Although it is not yet in general use or publicly available, it is important to discuss it here because of the key role LANDFIRE will play in fire risk assessment over the next decade.

The spatial distribution of potential vegetation, existing vegetation, canopy height, and canopy cover is mapped using gradient-based field inventories coupled with gradient modeling, remote sensing, ecosystem simulation, and statistical analyses. Biophysical gradient maps have been created containing 38 geographical information system (GIS) layers describing the direct and indirect conditions affecting the distribution of vegetation and fire regimes

(Figure 2). The vegetation of the continental United States is divided into approximately 500 biophysical units, based on plant composition and the Ecological Systems categorization (<http://www.natureserve.org/>). Each successional stage of each biophysical type is separately tracked. Fuel models are assigned to each mapping region, producing fuel maps for fire behavior models, canopy fuel projections, and fuel characterization classes. Crown bulk density and height to crown base are calculated at the plot level from tree lists.

Inputs consist of coarse-scale, 1-km² resolution, spatial data layers. These include potential natural vegetation type, current cover type, site characteristics (such as soils, climate, and topography), historical natural fire regime (fire frequency and severity), and Fire Regime Current Condition Class (layer depicting the degree of departure from historical fire regimes). There are three additional databases that are used as input:

1. National Fire Occurrence—Federal and non-Federal fire occurrences from 1986 to 1996.
2. Potential Fire Characteristics—the number of days of high or extreme fire danger calculated from 8 years of historical National Fire Danger Rating System (NFDRS) data.
3. Wildland Fire Risk to Flammable Structures—the potential risk of wildland fire burning flammable structures based on an integration of population density, fuel, and weather spatial data.

Landscape characteristics that are used to determine fire occurrence and behavior include height to crown base, crown bulk density, fuel loadings, cover type, percentage cover, height for each of the forest, shrub, and herbaceous layers, and an estimate of the departure from reference normal fire regime condition class. The interaction between these characteristics and fire probability is estimated by using a suite of computer models, WXFIRE, BIOME-BGC, LANDSUMv4, FARSITE, and HRVStat, discussed in the next section.

LANDFIRE produces three fire regime maps:

1. Simulated historical fire frequency and severity.
2. Fire regime condition class (FRCC).
3. Indices of departure from reference conditions.

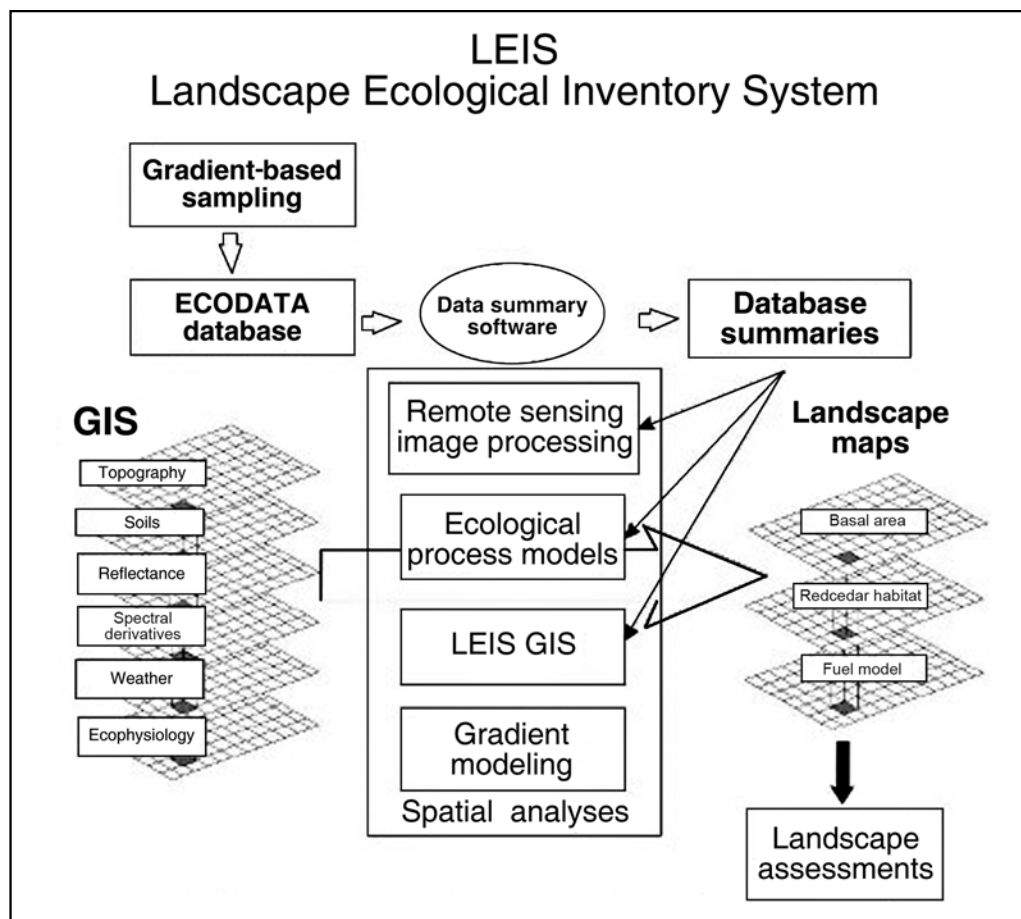


Figure 2—A diagram of the Landscape Ecosystem Inventory System used by LANDFIRE. Boxes in the center represent the steps used to create the ECODATA database, the LEIS GIS, and landscape planning maps. Initial and derived GIS layers (left side) are used as independent variables to predict maps of landscape and ecosystem characteristics (right side) for landscape assessments and ecosystem management (Keane and others 2002).

These tools are used for determining the degree to which current landscape conditions have departed from historical reference condition vegetation, fuel, and disturbance regimes. Figure 3 shows an example map of fire regime condition classes produced by LANDFIRE. An example map of reference fire regimes is shown in Figure 4.

The data layers being produced by LANDFIRE will provide basic information from which a risk assessment can be calculated. However, there are drawbacks to this system. LANDFIRE'S classifications of fire conditions may be too coarse. The calculation of probabilities may require data on a continuous scale. Further, it is not clear whether this data is sufficient to predict the average likelihood of a fire at a

location or the distribution probabilities of fires of different sizes and intensities. The probability of a worst-case scenario would be difficult to estimate from LANDFIRE'S products.

Because the past condition and fire susceptibility had much to do with past climate, it is unclear whether LANDFIRE will correctly predict the relationship between vegetation, fuel loadings, and fire that will be shaped by future climates. LANDFIRE places less emphasis on the importance of the heterogeneity of types of fire and the key differences among these types, such as crown fires vs. ground fires. LANDFIRE focuses on classes that are relevant to management and current vegetation classifications. There may be advantages to having the flexibility to

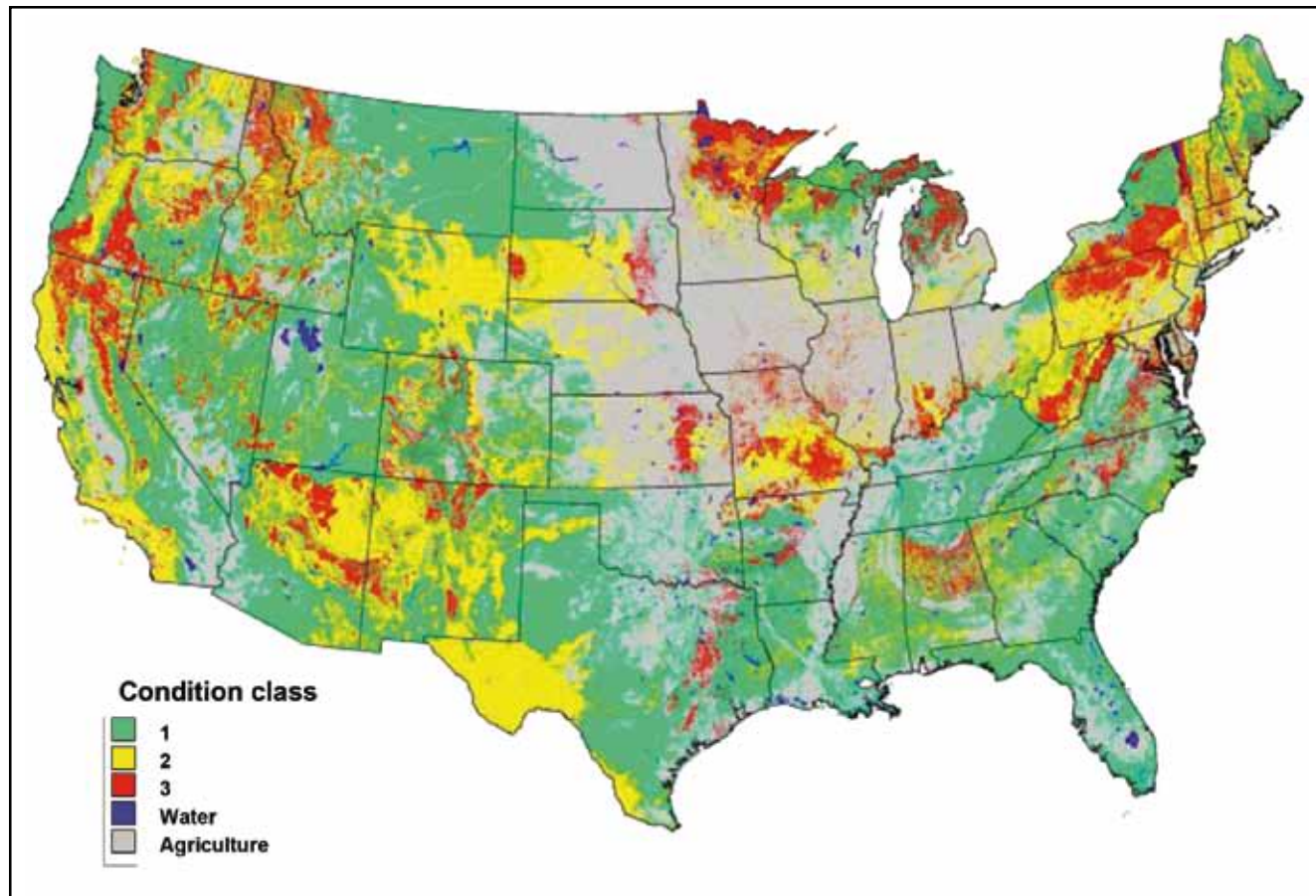


Figure 3—LANDFIRE National Fire Regime Condition Class Map, 2002, Missoula Fire Sciences Laboratory (Hann and others 2004, Hann and Bunnell 2001, Schmidt and others 2002). Conditions 1 (low risk, forest regimes are within their historical range, shown in green), 2 (moderate risk, forest regimes are moderately altered from their historical range, shown in yellow), and 3 (high risk, forest regimes are significantly altered from their historical range, shown in red).

make changes in these classifications. Natural variability in landscape and fire characteristics, and their influence on fire (and the uncertainty with which predictions can be based on these characteristics) is not treated implicitly within the system.

LANDFIRE Models—

WXFIRE (Keane and Holsinger 2006) computes spatially explicit, climate-based biophysical variables at any landscape scale or resolution using daily weather data, topography, and soils parameters, and a diverse set of integrated environmental functions. WXFIRE computes over 50 biophysical attributes such as potential evapo-transpiration for each simulation unit. The user must estimate all input parameters for each simulation unit to create an input file to WXFIRE. WXFIRE then calculates, record by record,

all biophysical attributes by accessing the DAYMET spatial weather database (Thornton and others 1997) (<http://www.daymet.org>) and using the daily weather to compute important climate and ecosystem biophysical variables. The DAYMET database consists of 18 years of daily temperature, humidity, radiation, and precipitation estimates at a 1-km spatial resolution for the contiguous United States. Output from WXFIRE can be used to digitally map those ecosystem characteristics needed by land management including fire regime, fuel load, and vegetation cover type.

BIOME-BGC (Thornton and others 2002) is used to calculate expected forest productivity in response to a given set of environmental conditions. BIOME-BGC is an ecosystem process model that simulates carbon, water, energy, and nitrogen budgets for both vegetation and soil.

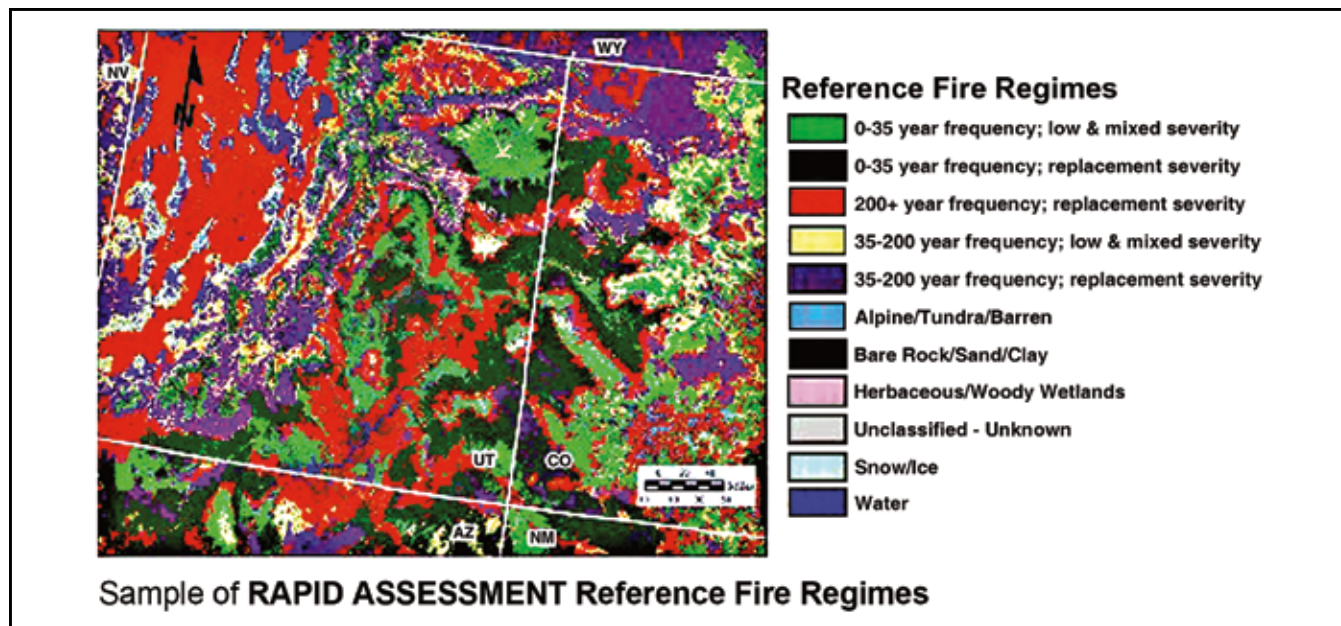


Figure 4—LANDFIRE rapid assessment reference fire regimes. The figure identifies the locations with fire regimes with a 0 to 35-, 35 to 200-, and greater than 200-year frequency and replacement severities ranging from low and mixed to replacement. Also identified are locations of alpine, bare rock, wetland, snow/ice, water, and unclassified landforms. (<http://www.landfire.gov/RA2Image.html>; Updated map <http://www.landfire.gov/RA3Image.html>).

BIOME-BGC is inherently non-spatial and can be run on an area of any size: a single point, a watershed, a continent. BIOME-BGC computes a set of carbon and water budget metrics on a daily time step driven by daily weather data such as gross primary production, net primary production, evapo-transpiration, and runoff. The model requires three types of information as input: site physical characteristics, plant physiological characteristics, and daily weather data. A set of generic plant functional type (PFT) ecophysiological parameter files has been developed from the literature for use with BIOME-BGC (White and others 2000). The PFTs include evergreen needle leaf forest, deciduous broadleaf forest, evergreen broadleaf forest, deciduous needle leaf forest, evergreen shrub, c3grass, and c4grass.

The range and variation of historical landscape dynamics are estimated with LANDSUMv4 (Landscape Succession Model version 4.0) and HRVStat (Historical Range and Variation Statistics) of landscape characteristics (Keane and Holsinger 2006, Keane and others 2002). LANDSUMv4 simulates fire and succession on fine-scale landscapes for land management applications. Species composition and stand structure are assumed to change at a predefined rate,

although disturbance initiation is modeled stochastically, and disturbance effects are based on the current vegetation conditions.

The model FARSITE projects the spread of fire across landscapes, using slope relationships and wind vectors (discussed separately under the fire behavior model heading). FIREHARM (<http://ams.confex.com/ams/pdfpapers/66069.pdf>) is used by LANDFIRE to produce probabilities of a given fire event from long-term weather. It identifies areas of highest risk based on fuel consumption, smoke production, tree mortality, soil heating, crown fire index, and proximity to urban areas. FIREHARM will produce a probability of fire for each category of weather conditions (i.e., the number of days the fire potential is above a certain level). FIREHARM will calculate four fire behavior variables (fireline intensity, spread rate, flame length, crown fire potential), five fire danger variables (spread component, burning index, energy release component, Keetch-Byram drought index, ignition component), and five fire effects variables (smoke, fuel consumption, soil heating, tree mortality, scorch height) for every day in the DAYMET 18-year record for each spatial location. The program will simulate

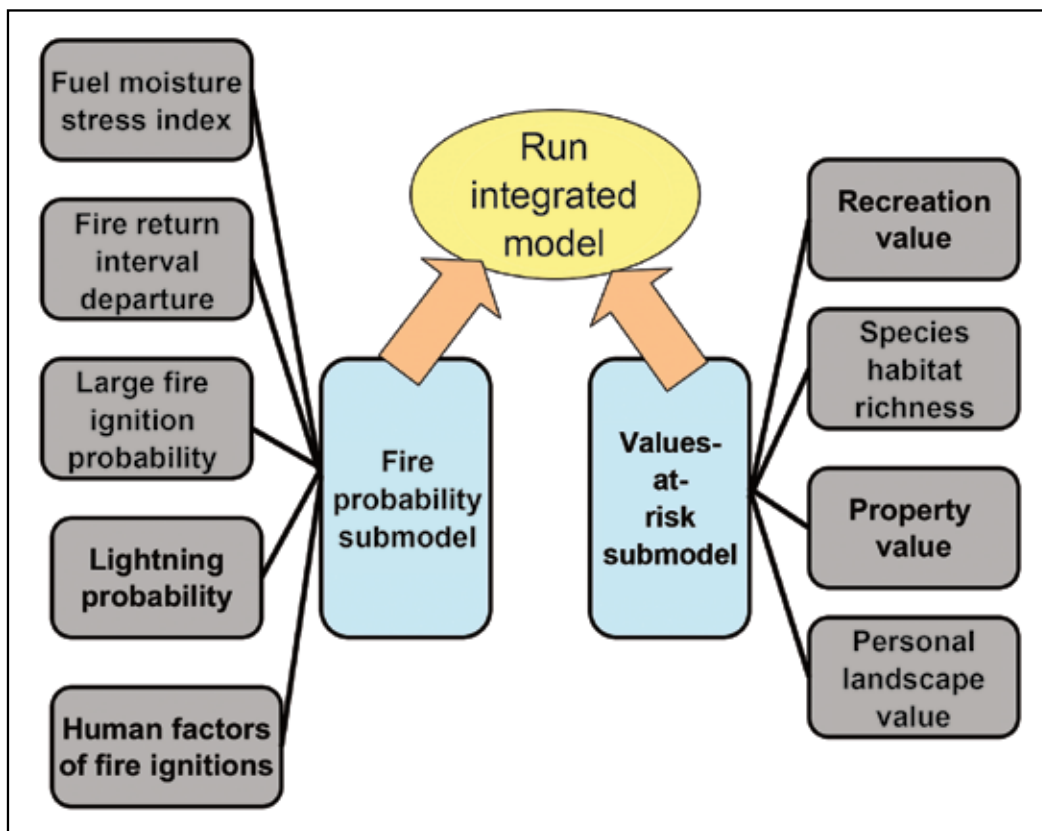


Figure 5—Components of the Fire-Climate-Society Model in Wildfire Alternatives (WALTER), showing the principal processes integrated into the full model analysis. (Adapted from <http://java.arid.arizona.edu/ahp/>).

moisture conditions for each dead fuel component (e.g., duff, litter, downed woody, logs) and live fuel component (e.g., shrubs, herbs, trees) using a complex set of biophysical equations.

Wildfire Alternatives (WALTER) (FCS model)

The Wildfire Alternatives (WALTER; <http://walter.arizona.edu/index.asp>) system for estimating fire risk is being developed at the University of Arizona. It is an interdisciplinary research initiative aimed at improving our understanding of the processes and consequences of interactions among wildfire, climate, and society. WALTER seeks to capitalize on advances in geospatial, analytical, and Web delivery technology to provide access to scientific research activities and findings and educational materials using the decision support tool, Fire-Climate-Society Strategic Fire Model (FCS-1).

FCS-1 is an online, spatially explicit strategic wildfire planning model with an embedded multi-criteria decision process that facilitates the construction of user-designed fire risk assessment maps for different climates (Figure 5). The resulting maps show spatially explicit information about the geographical distribution of fire probability and values at risk for the selected study area.

FCS-1 was developed for the varying vegetation, climate, and topography as well as the unique human dimensions of wildfire found in southeastern Arizona and northern New Mexico. The model currently is made up of five fire probability and four values-at-risk model components. FCS-1 can be run under differing climate and corresponding fuel moisture conditions. Through an analytical hierarchy process, FCS-1 allows users to assign weights to individual model components. The online application

can be used to help understand differing views and build consensus.

Like many expert systems, WALTER has the framework for considering an array of disparate factors in evaluating the ways they may influence fire. However, expert opinion can have uncertain predictive ability. The main value of this type of system is to identify interrelationships that may require management to control fire frequency and pattern.

Expected Net Value Change

A model has been developed for calculating the expected net value change (ENVC) as the product of the probability of a fire at a specific location and the resulting change in financial or ecological value (Finney and others 2007). The model calculates the sum of the product of the probability of the *i*-th fire behavior at a specific location over *N* fires multiplied by the benefits and losses afforded for the *j*-th value of *M* values received from the *i*-th fire behavior. The expected net value change (ENVC) can include financial, ecological, or other values at present day or future discounted values. Assumptions about the effect of wildfire suppression on wildfire probability and value change can also be incorporated into the (ENVC) equation. The estimation of wildfire probabilities for a specific future period is derived from a calculation of the minimum travel time a hypothetical fire would take moving across the landscape, using algorithms from the FARSITE fire behavior model (discussed under the "Fire Behavior Modeling" heading). These models are run in parallel using a set of networked computers and an estimate of the spatial pattern of forest vegetation and fuels.

This model has been applied to a 16 000-ha wildland-urban interface in eastern Oregon simulating 12 fuel management scenarios and four land value schemes. Burn probabilities were estimated by simulating 200 randomly ignited wildfires, and then the average net value change for each fire and pixel on the landscape was estimated. The results indicate that fuel reduction on a relatively minor percentage of the landscape (20 percent) resulted in a 20-percent to 50-percent positive change in ENVC for most of the scenarios simulated.

Probability-Based Fire Statistical Models

Probability-based fire risk models (Brillinger and others 2006) defined risk using three probabilities:

1. The probability of fire occurrence.
2. The conditional probability of a large fire assuming an ignition occurs.
3. The unconditional probability of a large fire.

An illustration of the general approach shared by these types of models is the probability-based fire risk model (Preisler and others 2004), which estimates fire probability by fitting a nonparametric logistic regression to data grouped in cells of 1 km² with a temporal resolution of 1 day. The input used by this model is:

1. The probability of fire occurrence (historical data).
2. The conditional probability of a large fire given ignition (daily values of precipitation, lightning, temperature, windspeed, and humidity).
3. The unconditional probability of a large fire (10-hour lag fuel moisture).

It produces maps of predicted probabilities and estimates of the total number of expected fires in a given region and time period. This method is particularly useful for assessing the utility of explanatory variables, such as fire, weather, and danger indices for predicting fire risk. It has the advantage of basing fire predictions on rigorous use of probability statistics.

Fire Behavior Modeling

Fire behavior models predict the propagation of fire by assuming that the landscape is subdivided into cells, and each cell has a probability of burning that depends on conditions in the cell and in surrounding cells (e.g., Beer 1991). Many of these models use a deterministic version of the elliptical growth model (Green and others 1983, Richards 2000) to simulate spread of forest fires. The input data for these models include a base vegetation map, which can be generated with a vegetation simulator, such as Forest Vegetation Simulator (FVS) (Dixon 2002) and a fuel load simulator, such as Fuels and Fire Extension (<http://www.essa.com/downloads/prognosis/ffe.pdf>). Fire behavior

predictors were initially built for local-scale projections, where initial ignitions and spatial distribution of landscape properties could be precisely specified.

Among the more widely applied fire behavior models is FARSITE, which predicts fire spread and intensity across a specific landscape as a continuous fire over multiple time steps at a user-specified resolution, commonly 30 m. The focus of FARSITE is to simulate fire growth and the changes that occur over time for a specific fire. It uses spatial information on topography and fuels along with weather and wind files to simulate surface fire, crown fire, spotting, postfrontal combustion, and fire acceleration. FARSITE requires spatial landscape information from a GIS to run, including slope, aspect, surface fuel, canopy cover, crown base height, crown bulk density, and stand height. The outputs include a prediction of fire perimeter length and location, spread rate, intensity, flame length, and heat per unit area. Although the focus is on a single fire, a regional prediction could be made by combining output from all fires each year. Fire spread predictions are based on a snapshot of current vegetation structure. In order to simulate fire behavior under future conditions, the changes that will occur in the vegetation structure must be known.

BehavePlus (<http://fire.org/>) predicts fire behavior based on fuels, weather, topography, and wildfire situations. BehavePlus uses a minimum amount of site-specific input data to predict fire behavior for a point in time and space (i.e., spatially explicit data layers are not used). The present version of BehavePlus simulates only surface fire spread, but later versions may include crown fire simulation capability.

FlamMap (<http://fire.org/>) is a fire behavior mapping and analysis program, based in a geographic information system (GIS) that computes potential fire behavior characteristics (spread rate, flame length, fireline intensity, etc.) over an entire FARSITE landscape for constant weather and fuel moisture conditions. FlamMap uses spatial information on topography and fuels to calculate fire behavior characteristics at one point in time, assuming that all cells in the landscape function independently of one another. A map is produced for an area of any modeled value, such

as fuel moisture, fireline intensity, scorch height, or fuel consumption. Comparisons can be made between locations, or the effect of fuel treatment can be examined.

NEXUS (<http://fire.org/>) is an Excel spreadsheet linking surface and crown fire prediction models to compute indices of relative crown fire potential. NEXUS is useful for evaluating alternative treatments for reducing crown fire risk and assessing the potential for crown fire activity. NEXUS includes several visual tools useful in understanding how surface and crown fire models interact.

VFS (<http://fire.org/>) is a graphical user interface-(GUI-) based computer program to simulate and animate fire on heterogeneous landscapes. VFS captures fire spread behaviors based on fuel configuration, wind regime and topographical effects using percolation algorithms such as static percolation, depth first search recursive algorithm, and dynamic percolation with fire front. Users can compare the simulation capability of each method (e.g., burned pattern maps). Furthermore, output from VFS can be linked to GIS and used to cross-validate other fire simulation models. To evaluate the sensitivity of the input parameters of each method, users can specify a range for each parameter in VFS and test the influence of changes in parameters on model predictions. VFS can be used as a parameterization tool for the forest landscape models that incorporate fire spread simulation.

Models to Estimate Effect of Vegetation Change on Fire Risk

Fire risk over long periods cannot be adequately evaluated without projecting the ways that the structure and composition of forest vegetation and fuels will change over time. Fire is very sensitive to vegetation structure and composition (Clark 1993, Swetnam 1997, Swetnam and Baisan 1996). Fire will, in turn, affect the rate and direction of vegetation change (Lenihan and others 1998).

Keane and colleagues classified 44 linked fire succession models, based on their complexity, principal mechanism, and stochasticity in their simulation design (Keane and others 2004). Most of these models fall into the category of fire behavior models because they attempt to simulate, at a fine scale, the dynamics that cause fire to

Table 1—Model selection key for selecting the most appropriate linked fire succession models for fire management and research applications

Criteria	Models
Management application	
Limited computer resources, modeling expertise, and/or input data available	
Fire pattern important	
Support and documentation available	TELSA
Not as above	LANDSUM
Fire pattern NOT important	
Support and documentation available	FFE-FVS
Not as above	SIMPPLLE, FETM
Abundant computer resources, modeling expertise, and/or input data available	
Individual tree or species processes important	
Support and documentation available	None
Not as above	LANDIS, QLAND, FIN-LANDIS
Only stand level characteristics important	
Support and documentation available	LANDMINE, SELES
Not as above	BFOLDS, CAFE, DISPATCH, EMBYR, INTELAND, LADS, LANDSIM, RMLANDS, SAFE- FOREST, SEM-LAND
Research application	
Explore climate, vegetation, and fire dynamics	
Coarse-scale applications	BFOLDS, BIOME-BGC, CENTURY, MC-FIRE, GLOB-FIR
Landscape-scale applications	
Individual tree or species-level processes important	
Fire pattern important	FIRE-BGC, LAMOS, SIERRA
Not as above	DRYADES, ZELIG-L, ZELIG-B
Only stand-level characteristics important	
Fire pattern important	MAQUIS, FIRESCAPE
Not as above	REG-FIRM
Explore fire and vegetation dynamics	
Coarse-scale applications	ALFRESCO, FIREPAT
Landscape-scale applications	
Individual tree-level processes important	
Fire pattern important	FIN-LANDIS, LANDIS
Not as above	None
Only stand level characteristics important	
Fire pattern important	ANTON, CAF' E, DISPATCH, EMBYR, INTELAND, LANDSIM, MAQUIS, MOSAIC, QTIP, RATZ, RMLANDS, SELES, SEM- LAND, SUFF2
Not as above	BANKSIA, FLAP-X, ON-FIRE, SUFF1, SUFF2, VASL

Source: Keane and others 2004.

spread across a landscape. These authors provide a handy table for deciding which models to use based on (1) whether

the purpose is management or research, (2) the importance of predicting fire patterns, (3) whether information is

needed on specific species or just on stand characteristics, and (4) what spatial scale is of interest (Table 1).

Complex models such as Fire-BGC (Keane and others 1989) and LANDIS (Mladenoff 2004, Mladenoff and others 1996) simulate vegetation change as a complex function of either the development of nutrient cycling conditions or the driving force of individual life history characteristics. The most complex of these are the gap models such as ZELIG-SP (Miller and Urban 1999) that predict successional development by simulating the dynamics of each individual tree on representative plots in the forest. Simpler models, such as SIMPPLLE (Chew and others 2004), represent vegetation changes as a predictable succession of stages following a resetting disturbance. We have selected a few of these models that are in common use for further discussion below.

SIMPPLLE and MAGIS

SIMPPLLE (SIMulating Vegetative Patterns and Processes at Landscape scaLEs) is a stochastic non-spatial simulation model for projecting vegetative change over time in the presence of natural processes, either with or without management treatments. It models interaction of various natural processes on a landscape. Because it is stochastic, multiple simulations are run to generate a record of the frequency of natural disturbances for each polygon in the landscape. These frequencies represent an estimate of the risk of these natural processes occurring over a given period of time and are used to develop a risk index for each section of the landscape.

MAGIS (Multiple-resource Analysis and Geographic Information System (<http://www.fs.fed.us/rm/econ/magis/>)) is a spatial decision-support system for using the risk index to find optimal management practices spatially and temporally for a landscape. MAGIS uses optimization to select the spatial arrangement and timing of treatments that fit user-determined objectives and constraints. A variety of resource effects, management targets, and economic costs or benefits can be used to specify the objective and constraints for scheduling both vegetation treatments and road activities.

With these two programs, the user can evaluate a variety of management alternatives. However, their projections

are dependent on the estimate of current fuel distributions across the landscape. When projecting into the future, they will be increasingly in error as these fuel loads change.

Vegetation Disturbance Dynamics Tool

Vegetation Disturbance Dynamics Tool (VDDT) (<http://www.essa.com/downloads/vddt/>) was developed to support the Interior Columbia River Basin Assessment. This nonspatial tool uses a state-and-transition matrix approach to predict changes in vegetative composition and structure using disturbance probabilities and successional pathways, including infrequent large-scale disturbances such as stand-replacement fires.

VDDT models typically apply to potential vegetation types. For each of these types, succession classes are defined according to the cover type and structural stage. In the absence of disturbance, vegetation community assemblage changes from one succession class to the next. Both natural and man-caused disturbances that affect vegetation can be examined. In VDDT, disturbances are defined for each succession class according to type (e.g., wildland fire, harvest, etc.), succession class destination, probability of occurrence, and the relative ages for which each probability applies. For each year of the simulation, VDDT determines whether each landscape unit is subjected to a disturbance.

Tool for Exploratory Landscape Scenario Analyses

Tool for Exploratory Landscape Scenario Analyses (TELSA) (<http://www.essa.com/downloads/telsa/index.htm>) is a spatially explicit extension to VDDT that simulates forest succession, natural disturbances, and forest management activities. It is designed to simulate up to 250 000-ha landscape units. TELSAs can be used to simulate multiple scenarios, each characterized by different assumptions about management actions and natural disturbances. Because wildfires and other natural disturbance events that affect vegetation dynamics are inherently unpredictable, the model can use multiple stochastic simulations of each scenario to provide estimates of the mean, range, and variability of the selected performance indicators. Unlike

many other strategic planning models of landscape dynamics, TELSA takes into account natural disturbances so users can explore how their proposed management strategies will interact with vegetation succession and disturbances to alter landscape composition and structure. This model has been used to define the transition times between various succession classes (combinations of species composition and structural stage), the probabilities and impacts of disturbance by insects, fire or other agents, and the impacts of landscape management actions on structure and composition.

Models to Estimate the Effect of Climate Change on Fire Risk

Biophysical process models can be used to estimate the effect of vegetation change on fire risk. The representation of basic processes in biophysical process models allows them to project the consequences of known relationships under future conditions. These models are capable of examining the effects of long-term changes in conditions such as climate change. An example of these model types, MAPSS (<http://www.fs.fed.us/pnw/corvallis/mdr/mapss/>) predicts vegetation distributions from either the availability of water in relation to transpirational demands or the availability of energy for growth (Neilson and Wullstein 1983, Neilson and others 1989, Stephenson 1990, Woodward and Williams 1987).

MAPSS and its recent derivative, MAPSS-CENTURY1 (MC1), simulate life-form mixtures and vegetation types, fire disturbance, and ecosystem fluxes of carbon, nitrogen, and water (Lenihan and others 2003). MC1 is routinely implemented on spatial data grids of varying resolution (i.e., grid cell sizes ranging from 900 m² to 2500 km²) where the model is run separately for each grid cell (i.e., there is no exchange of information across cells) (Bachelet and others 2000, 2001; Daly and others 2000). MAPSS has been implemented at a 10-km resolution over the continental United States and at a 0.5-degree resolution globally (Neilson 1993, 1995; Neilson and Marks 1994). It has also been implemented at the watershed scale (MAPSS-W, 200-m resolution) through integration with a distributed catchment hydrology model (Daly and others 1994, Wigmosta and others 1994).

Using climate data at a monthly time step, the model calculates the leaf area index (LAI) of three generic life form groups—trees, shrubs, and grasses—in competition for both light and water given a site water balance consistent with observed runoff (Neilson 1995). Water in the surface soil layer is apportioned to the two life forms in relation to their relative LAIs and stomatal conductance.

MAPSS is used to develop midterm forecasts of fire risk by using spatially distributed, high-resolution climate data and potential future climate forecasts from climate models. An example of these predictions is given in Figure 6. The PRISM model (<http://www.ocs.orst.edu/prism>) is used to produce high-resolution data grids of observed fire weather extending back to 1895 and interpolations of weather station data that are sensitive to topography. Fire risk forecasts, including fire occurrence, area burned, and fire behavior are generated from the historical and forecast weather data (Bachelet and others 2000, Lenihan and others 2003) (<http://www.fs.fed.us/pnw/corvallis/mdr/mapss/pubs.html>). A fire event is triggered in any given cell on any day if one of three thresholds is exceeded. A threshold of the 12-month standardized precipitation index (SPI) is used as an indicator of moderate to severe drought to control the inter-annual timing of fire events. A threshold of the 1,000-hour fuel moisture content of dead fuels is used as an indicator of extreme fire potential to control the seasonal timing of fire events. A threshold of fine fuel flammability is used as an indicator of the sustainability of fire starts to control the timing of fire events at the daily time step. There is no constraint on fire occurrence by the availability of an ignition source, such as lightning or human-caused ignition. Once a fire event is triggered, the MC1 fire module determines the fraction of each cell burned, which is dependent on the current vegetation type, the current drought condition, and the number of years since fire.

Potential fire behavior is also influenced by estimates of the mass, vertical structure, and moisture content of several live and dead fuel size-classes. The consumption of aboveground biomass, carbon, and nitrogen stocks are simulated as a function of the moisture content of each live and dead fuel size-class and the vertical structure of the canopy. The more rapidly growing grasses are assumed to

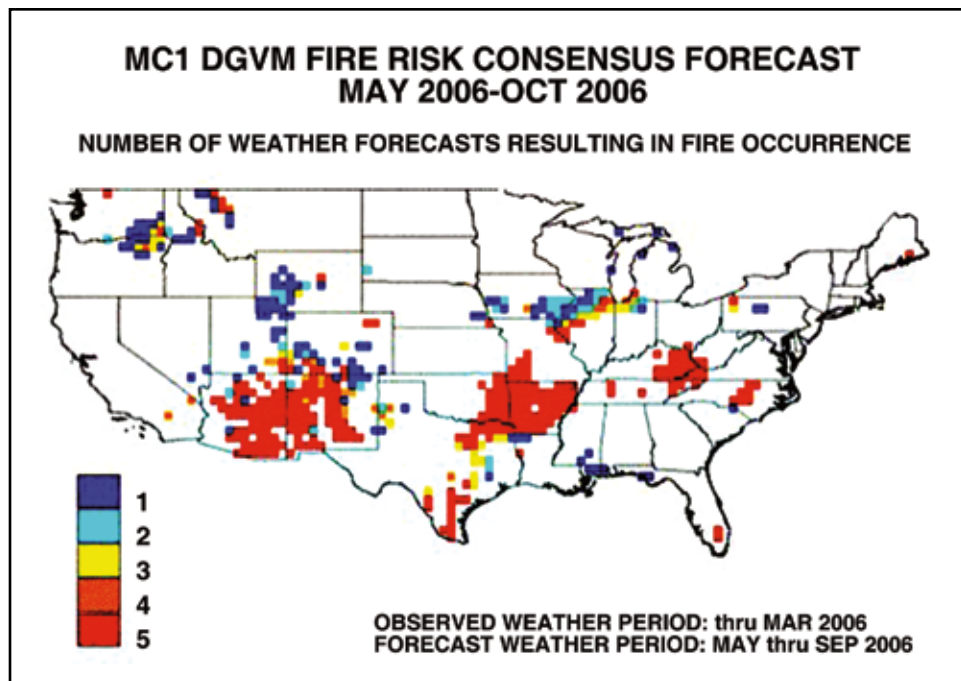


Figure 6—MAPSS simulation of fire probability MC1 DGVM fire risk consensus forecast, May to October 2006. The map shows the locations where from one (blue) to five (red) weather forecasts resulted in a fire occurrence (<http://www.fs.fed.us/pnw/corvallis/mdr/mapss/GRAPHICS/CONSENSUS>).

gain an advantage over woody life forms in the competition for water and nutrients, promoting even greater grass production which, in turn, produces a more flammable fuel-bed and more frequent fire. MAPSS is capable of producing predictions of future vegetation change and fire risk over a large geographical area.

Conclusions Concerning the Use of Fire Modeling Systems

Much effort has gone into creating a capability of predicting fires throughout the region, both in their likely location and frequency. We described two major categories of fire risk assessment tools: those that predict fire under current conditions, assuming that vegetation, climate, and the interactions between them and fire remain relatively similar to their condition during recent history; and those that anticipate changes in fire risk as climate and vegetation communities change through time. Three types of models have proven useful for predicting fire under current conditions:

1. Biophysical models that predict fire from vegetation type, fuel load, and climate.
2. Statistical models that use historical data to predict fire probabilities if landscape-fire relationships continue to remain relatively unchanged.
3. Fire behavior models that produce predictions of the ways individual fires will move across the landscape.

Programs such as LANDFIRE have great promise for using biophysical properties to estimate risk. The LANDFIRE program is creating base data sets of fuel loadings, biophysical variables, and vegetation composition. Risk assessments conducted with LANDFIRE tools will depend heavily on well-tested models such as FARSITE. Since the intention is to implement the LANDFIRE project nationally at a fine-scale resolution, its data sets could provide the framework around which to build other risk evaluations. However, the data sets produced by LANDFIRE could be made even more valuable if they contained information on

the probabilities of fires of different sizes, intensities, and heterogeneity of fire types at any given location.

For longer periods, fire risk must be evaluated by models that predict the ways vegetation communities will change over time because these changes will alter fire probabilities. Risk systems must be designed to track changes in fire susceptibility as climate changes, using models such as MAPSS. LANDFIRE is not currently designed to track these changes, so it is unclear whether it will correctly predict the relationship between vegetation, fuel loadings, and fire that will be shaped by future climates. MAPSS would have a much higher likelihood of being able to track these changes in relationships.

Prediction of fire occurrence is just the first part of a complete analysis of fire risk. Fire occurrence risk must be combined with models that determine the risk of the effects of fire. For example, one such effects model is FOFEM (a First Order Fire Effects Model) (<http://fire.org/>), which predicts mortality, fuel consumption, smoke production, and soil heating caused by prescribed fire or wildfire. It uses an algorithm key that selects different functions for different geographic areas and cover types. It can be used for setting acceptable upper and lower fuel moistures for conducting prescribed burns, determining the number of acres that may be burned on a given day without exceeding particulate emission limits, assessing effects of wildfire, developing timber salvage guidelines following wildfire, and comparing expected outcomes of alternative actions. There are second-order effects, such as changes in site productivity, animal use, insects, and disease that need to be evaluated by other models.

Fire must be looked at in the context of other stresses, such as invasive insects and pathogens, encroaching urbanization, and loss of critical habitat. There are interactions among stresses that play a large role in affecting the frequency and intensity of fire, and fire, in turn, can affect the probability of those stresses. Fire probability can increase in stands that have experienced large amounts of tree mortality caused by native pest infestations. Because these precursor stresses have received less attention than fire, the uncertainty for predicting their probability is much

higher. The effects of fire are dramatic, but its role in shaping the future of forest systems may be equaled or exceeded by other stresses. Consequently, it is vitally important that risk evaluation systems be created that can simultaneously estimate the probability of all the major stresses influencing forest and grassland development.

In the development of fire modeling systems, data sets containing fine-scale grids of key data on the landscape have been established, with most containing detailed information on variables such as fuel loads, vegetation, and climate trends. These variables are likely to be useful in evaluations of many other kinds of risks. LANDFIRE'S classifications of fire conditions may be too coarse and inflexible to be useable to assess risks across a number of stresses at a given location. Vegetation mapping systems being developed (Kerns and Ohmann 2004, Ohmann and Gregory 2002, Wimberly and Ohmann 2004) may provide a much more flexible solution to this problem. Although there is not yet a movement toward a standard set of spatial data where models require identical inputs, such standardization could certainly aid risk comparisons. However, in order to compare fire risks to those from other threats, a flexible classification of forest types may be needed that is not just centered on fire risk characteristics.

The great proliferation of fire modeling systems in different portions of the regions suggests that each has specific strengths in simulating fires in the area for which the model was originally designed. However, maintaining so many different types of models in the fire modeling toolbox will inevitably prove unwieldy and confusing to potential users. Consequently, an effort toward consolidation (or choice of the best models) is likely to occur. The ultimate question is whether any of the tools discussed above could provide an initial framework on which evaluations of risks to other threats could be added. The best models would be those that facilitate comparison between fire risk and its associated ecosystem risk and the risk from other threats. It was not possible in this manuscript to evaluate which fire models offer a better route for considering multiple interacting stresses. This question must remain a primary one to consider in the choice and use of models.

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Fire Case Studies

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Managing Wildland Fire Risk in Florida

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Abstract

Florida's Wildland Fire Risk Assessment (FRA), which was completed in 2002, is a statewide effort to develop a comprehensive suite of standardized spatial data layers developed to support implementation of a statewide fuels management strategy. By maintaining focus on fire and fuel dynamics for use with scientifically credible local to statewide applications, the FRA builds on a statewide surface fuels map, fire history data from many agencies, and weather data collected over a period of 20 years. Change detection is currently being utilized to update the statewide surface fuels layer. The process used in the FRA builds on a process first applied in the Lake Tahoe Basin Land Management Unit. Subsequently, the methods used in the FRA have recently been applied to 13 Southern States in the Southern Wildfire Risk Assessment.

Introduction

Florida possesses a unique set of characteristics that make much of the state highly susceptible to wildfire. The state is blessed with an abundance of wildlands. The state has also experienced an influx of new residents into these wildlands, creating an intermingling of urban settlement within wildlands and increasing need for wildfire protection services.

Florida's weather is conducive to starting and spreading numerous and sometimes large wildfires. Florida's rate

of lightning strikes is unequalled in the Nation. Lightning, coupled with extended periods of drought, sets the stage for catastrophic fire episodes. Whereas lightning accounts for a large proportion of wildfire ignitions, human-caused fires are increasing as the population rises.

Florida's wildland vegetation evolved in a fire ecosystem. The vegetation is adapted to burn periodically. Fine fuels, which are easily ignited and spread fire rapidly, are abundant throughout Florida. The lack of managed fire in much of the wildlands has promoted an accumulation of these fuels that can burn with high intensity and be difficult to control.

To reduce the risk to life and property loss from wildfire, communities and fire management organizations are encouraged to actively manage wildland fire risk. Managing wildland fire risk can be challenging as fuels frequently change across the landscape and through time. Fire behavior can be affected by changes in land development policies, fuels, weather conditions, and topography. In addition, many social, technical, and institutional barriers to proactive fire-risk management and planning frequently exist.

Florida's fire managers face a complex problem of managing wildfire risk that is compounded by increasing fire intensities owing to accumulation of vegetative materials, continued residential growth into wildland fire-prone areas, and increasing firefighting costs. To address this problem, the Florida Division of Forestry (FDOF) initiated a process to assess fire risk and the values to be protected. The process developed (the Florida Fire Risk Assessment, or FRA) provides managers with a strategic view of the state to improve public safety and protect them from property losses like those experienced in 1985, 1989, 1998, and 1999.

The purpose of the FRA is to identify the potential for wildfires within the State of Florida and prioritize areas where mitigation options may be desirable. The FRA can also be used to locate areas within the state where inter-agency planning may be of value to effectively manage wildland fire risk. The results can be used to complete a more detailed analysis at the local level and communicate wildland fire management issues to the public.

The objective of this case study is to present the risk assessment methodology and results from the Southern

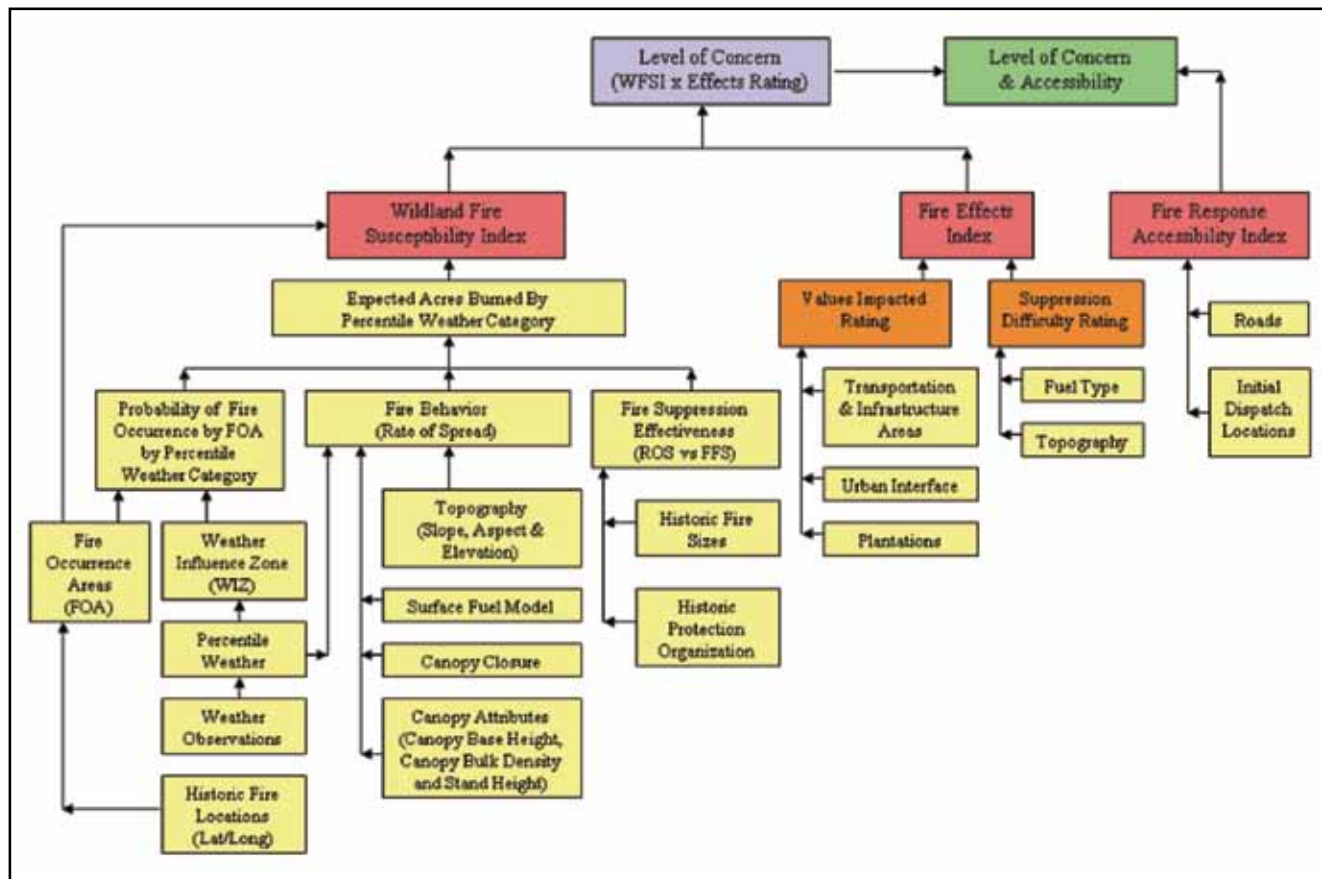


Figure 1—Southern Wildlife Risk Assessment (SWRA) model is shown above.

Wildfire Risk Assessment (SWRA) (Buckley and others 2006) for the State of Florida. We start by describing the methods involved in defining our index of wildland fire risk, which we call Level of Concern, based on its major components, wildland fire susceptibility and fire effects. We then describe the construction of a fire response accessibility index. Finally, we discuss how, in Florida, the fire risk and fire response accessibility indices are being used and applied to improve fire protection decisions.

Wildland Fire Risk

Webster's dictionary defines risk as "the possibility of suffering harm or loss." As one can see, there needs to be both a likelihood and effect of an action or event before one can incur a risk. Two primary indices were assigned to each 30- by 30-m cell in all 13 Southern States including Florida. These are the Level of Concern (LOC) Index and the Fire Response Accessibility (FRA) Index (Figure 1).

Within the risk assessment, the Level of Concern is the best measure of wildland fire risk. The Level of Concern Index is calculated from the likelihood of an acre burning, called the Wildland Fire Susceptibility Index (WFSI) and the expected effects of the fire (Fire Effects Index, or FEI). The FRA Index is a measure of the initial attack response time to a cell from existing initial dispatch locations for fire protection resources. Taken as a pair, these two indices (LOC and FRA) define a cell's accessibility and its vulnerability to wildland fire occurrence and effects.

Wildland Fire Susceptibility Index

As used in the Florida and Southern Wildfire Risk Assessment, the Wildland Fire Susceptibility Index is a value developed to represent an index related to the probability of an acre burning. The determination of an acre burning integrates the probability of an ignition and expected final

fire size, the latter being affected by rates of fire spread in four weather categories and fire suppression effectiveness.

Fire Occurrence (Fire Occurrence Areas)—

The first task to determine the WFSI is to determine the probability of an acre igniting. A fire occurrence area (FOA) is an area where the probability of each acre igniting is the same. The historical fire ignition locations for a defined period of time are used. Pictorially, if one were to locate the point location for historic ignitions on a map of an FOA, the points would appear with randomly dispersed densities different from adjacent FOAs.

A grid illustrating the probability of a wildfire igniting was developed using ArcMap by analyzing the location of historic ignitions from 1997 to 2003. Fire occurrence rates were described as the number of fire ignitions per 1,000 acres per year. A surface grid with fires per 1,000 acres per year was generated using a spatial filtering calculation available in ArcMap. The FOAs were developed to identify areas where the probability of a fire igniting was similar. Hence, within an FOA, the probability of each acre igniting is the same.

An example of a FOA map for Flagler County in Florida is shown in Figure 2.

Weather Influence Zones—

To determine an estimate of fire spread upon fire ignition using a fire behavior model, environmental conditions are needed so that fuel moisture and windspeed values can be used in the fire behavior models. To determine these environmental conditions, areas of uniform weather conditions were defined and the weather conditions within each area determined. A weather influence zone (WIZ) is an area where the weather on any given day is uniform. A fire weather meteorologist developed 20 weather influence zones in Florida, and these are displayed in Figure 3.

Development of Percentile Weather Values—

Within each WIZ, daily weather data are gathered for a defined period of time. These data were gathered from land management agency weather stations (National Fire Danger Rating System [NFDRS]) and from National Oceanographic and Atmospheric Administration (NOAA)-maintained weather stations. A computer program

developed by research meteorologist Dr. Scott Goodrick (Forestry Sciences Laboratory, 320 Green Street, Athens, GA 30602-2044) was used to change weather observations from NOAA stations to NFDRS standards. Another program developed by Dr. Goodrick was used to georeference the weather observations from the weather stations within a WIZ to the geographical center of the WIZ. Hence, one weather data set was developed with a weather observation for each day during the defined period for each WIZ. From this weather data set, percentile weather was developed for each WIZ.

The weather observation data set was checked for errors and then imported into the USDA Forest Service's FireFamilyPlus program. The NFDRS index spread component (SC) was calculated for each day. The fire season was set for each WIZ and the SC calculated using the NFDRS fuel model G. Fuel model G is used, as it contains fuel loading in all of the dead (1-h, 10-h, 100-h and 1,000-h) and live (herbaceous and woody) fuel categories. This allows for the influence in the spread component calculation of the fuel moisture values from all of the fuel categories. In addition, climate class 3 (subhumid/humid) and slope class most applicable to the WIZ were used.

The spread component was then divided into four commutative percentile categories: low (0-15 percent), moderate (16-90 percent), high (91-97 percent), and extreme (98-100 percent). The median SC was determined for each category. The environmental values for 1-h, 10-h, 100-h timelag fuel moisture, live herbaceous fuel moisture, live woody fuel moisture, and the 20-foot, 10-minute average windspeed were determined as the average of the respective values on days when the SC was equal to the median SC. This allows for the determination of four percentile weather categories with the percentage of occurrence of each category and with environmental values to define the weather conditions within each category.

Probability of a Fire Occurrence Within Each FOA by Percentile Weather Category—

We allow for the possibility that the higher percentile weather categories may be relatively more conducive to generating fire ignitions from ignition-initiating sources. That is, if 15 percent of the days during the fire season are

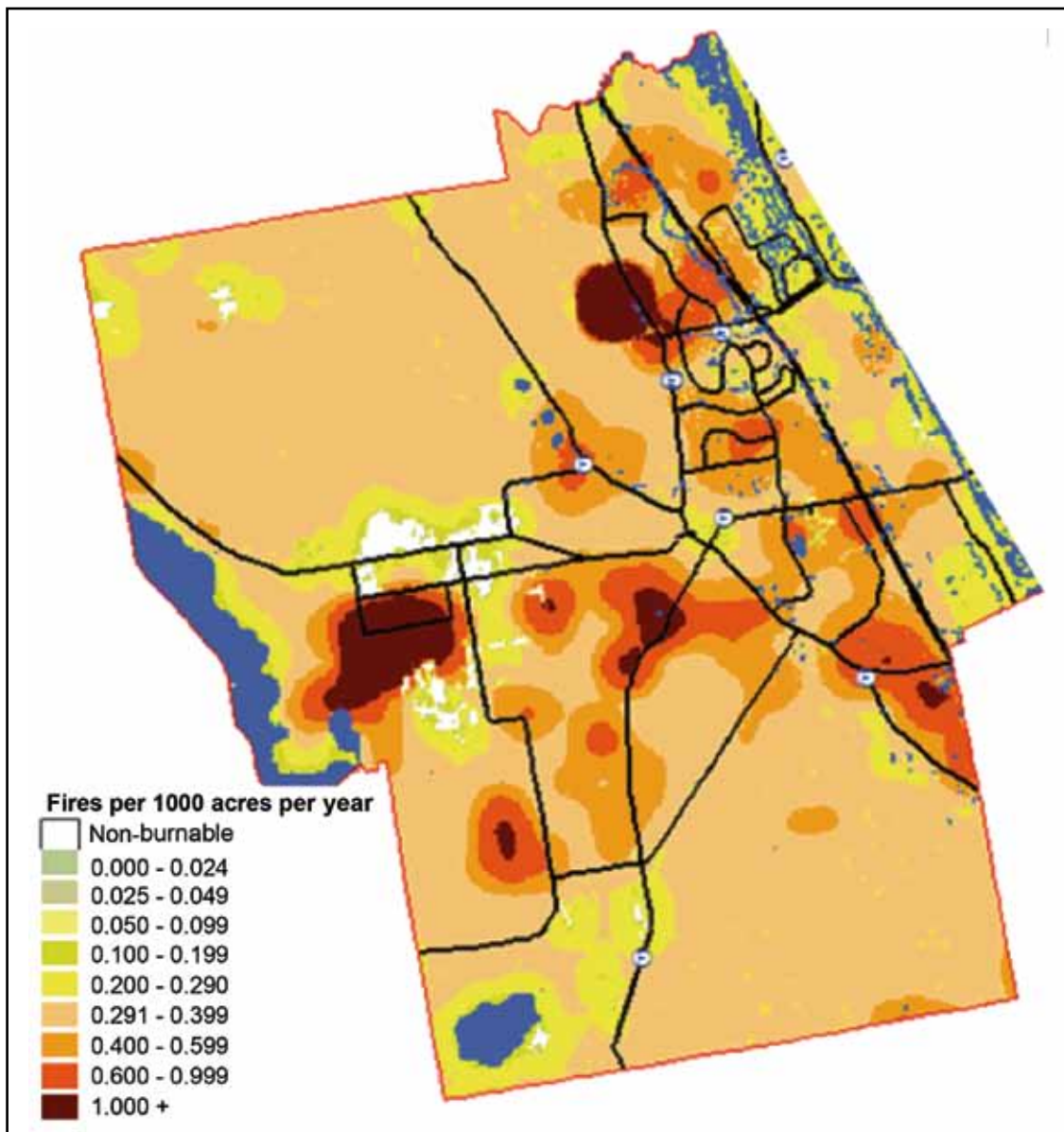


Figure 2—Flagler County, fire occurrence areas are detailed above.

in the low percentile weather category, one cannot assume that 15 percent of the fires during the fire season will occur on the days in this percentile weather category. Four percentile weather categories were developed: low, moderate, high, and extreme. The percentage of days within each is 15 percent, 75 percent, 7 percent, and 3 percent, respectively.

Each fire within the fire occurrence database for all agencies within a weather influence zone has a fire start date. Each historical fire was assigned a spread component based on the fire's start date from the results of the

FireFamilyPlus runs. The four percentile weather categories were also developed using the same assumptions for SC, and the four categories have SC ranges. Hence, a correlation is made assigning each historical fire to one of the four percentile weather categories. From these assignments, the proportion of fires that occurred in each percentile weather category by WIZ was determined. For Florida, 14.2 percent, 74.1 percent, 8.1 percent, and 3.5 percent of the fires started within the low, moderate, high, and extreme categories, respectively.

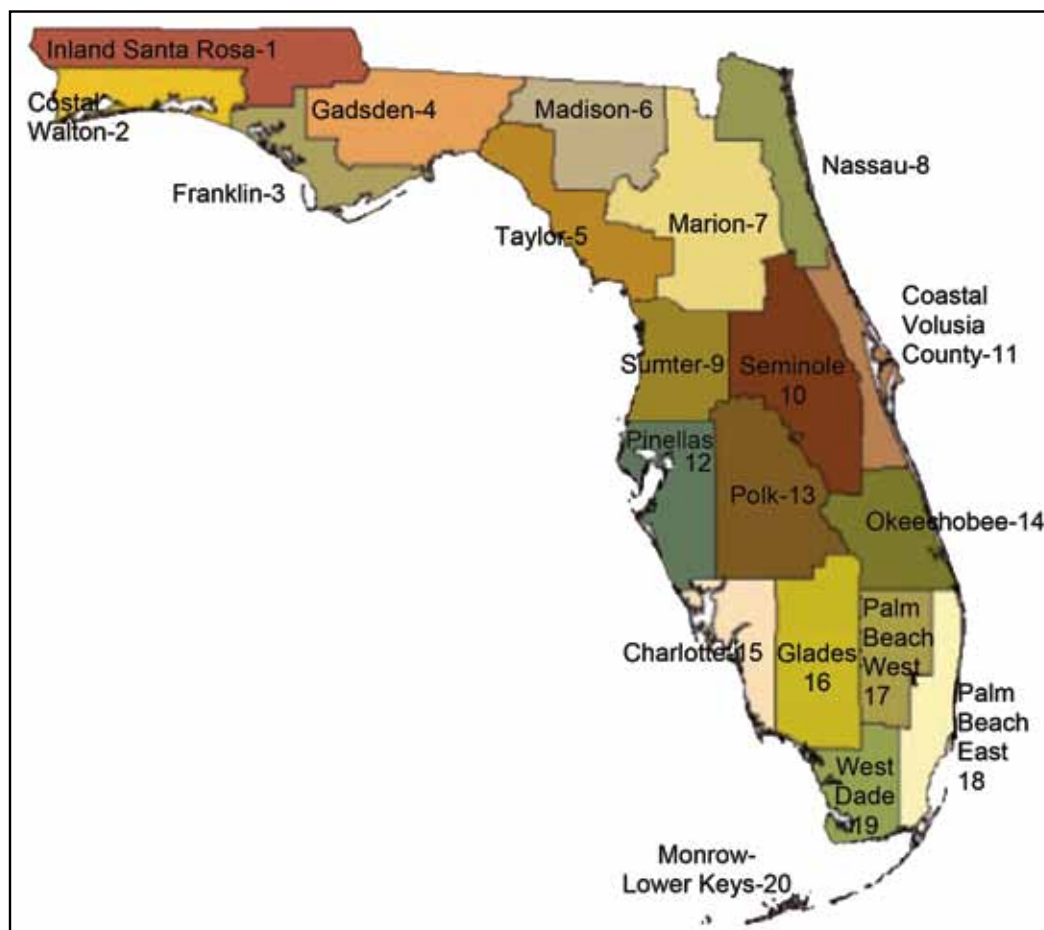


Figure 3—Weather influence zones are illustrated in Florida.

The probability of a fire within an FOA for each percentile weather category is the product of the total fire occurrence rate in the FOA by the proportion of fires within each percentile weather category.

Fire Behavior Prediction Inputs

Predicting fire behavior requires knowledge of fuels, weather, and topography. The previous section provides information on how the environmental conditions (weather) can be determined. The topographic conditions required are knowledge of slope steepness, aspect, and elevation. Aerial and surface fuel data required include canopy cover and the surface fuel model. If aerial fuel attributes are provided, then the occurrence and behavior of a crown (canopy) fire can be modeled. The aerial attributes needed are canopy base height (CBH), canopy bulk density, and stand height.

Data layers for the state were developed for slope, aspect, and elevation for U.S. Geological Survey Digital Elevation Model information. Fuel models for Florida were developed in 2002 for the FRA using a process where actual satellite imagery was correlated with the surface fuel model. These fuel models were also used in the SWRA, and a statewide fuel model map is displayed in Figure 4. An SWRA and FRA design requirement was to classify each acre of burnable land using the fuel models (Anderson 1982) in the Fire Behavior Prediction System (FBPS). A data layer defining the percentage canopy cover was developed using satellite imagery for Florida. For the FRA and SWRA, none of the canopy fuel layers (Canopy Ceiling Height, Canopy Base Height, and Canopy Bulk Density) were developed or used. All fire behavior predictions were based on surface fuel models.

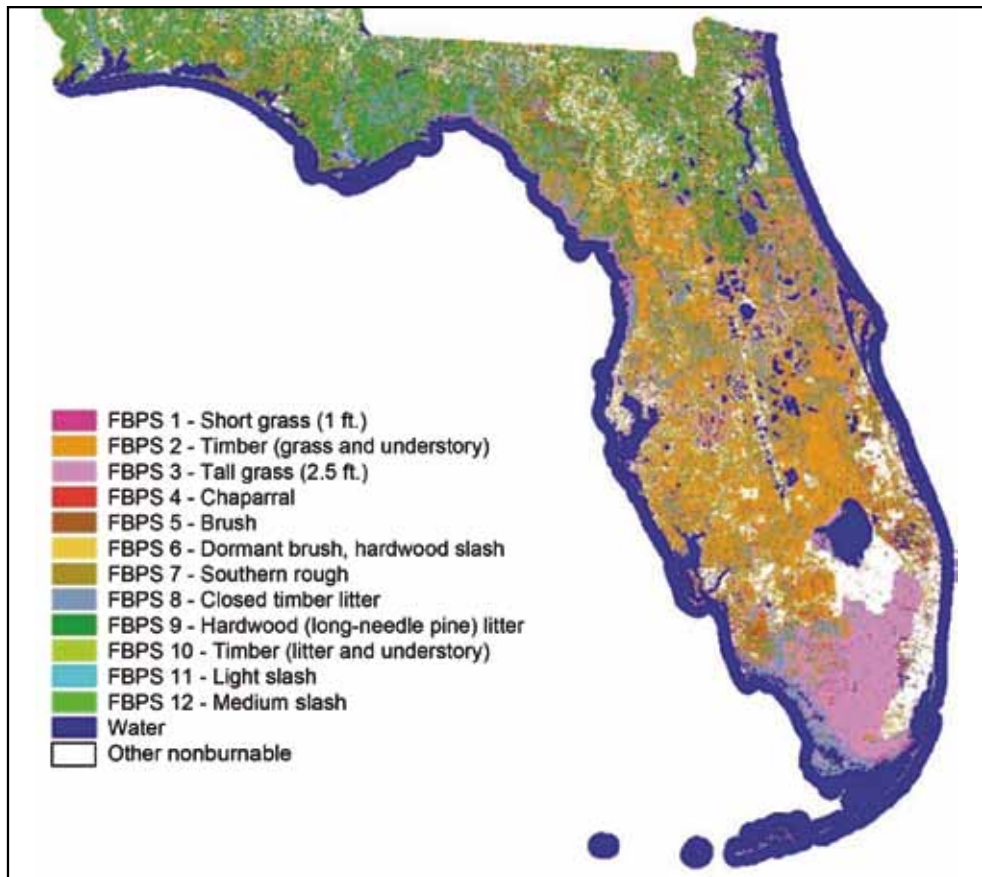


Figure 4—Surface fuel models are displayed in Florida.

Fire Behavior Outputs

Fire behavior outputs are a key component of the model used to estimate the WFSI. Potential fire behavior can be evaluated using a fire behavior prediction program, much like FARSITE (Fire Area Simulator) (Finney 1998) and FlamMap (Finney 2006). For the FRA, the FlamMap program was used.

The fire behavior program uses topographic information, fuel characteristics, and weather to calculate rate of spread, flame length, fire type, and other characteristics of fire behavior. Fire behavior prediction can also be done using a fire behavior dynamic link language (dll) program, which provides a more flexible and customizable method of calculating the required fire behavior outputs needed for the risk assessment model. The fb3.dll used in the SFRA has the advantage of providing tight integration capabilities with geographic information systems (GIS) and other programs.

The main fire behavior variable calculated by the fire behavior prediction programs such as the fb3.dll for the calculation of the WFSI is fire spread rate. This variable was developed because it can be used to estimate a fire's expected size.

For further analysis and display, it is worthy to note that additional fire behavior outputs such as fire intensities and flame length are available outputs of the fb3.dll program. The FlamMap program and the fb3.dll calculate the behavior of a fire occurring in each 30- by 30-m cell under defined weather conditions. Fire behavior is described independently for each individual cell.

Fire Suppression Effectiveness—Rate of Spread vs. Final Fire Size Relationships

For a cell, the FOA designation provides an estimate of the cell igniting. To calculate the WFSI, the expected size of a fire needs to be determined. To do this, it is necessary

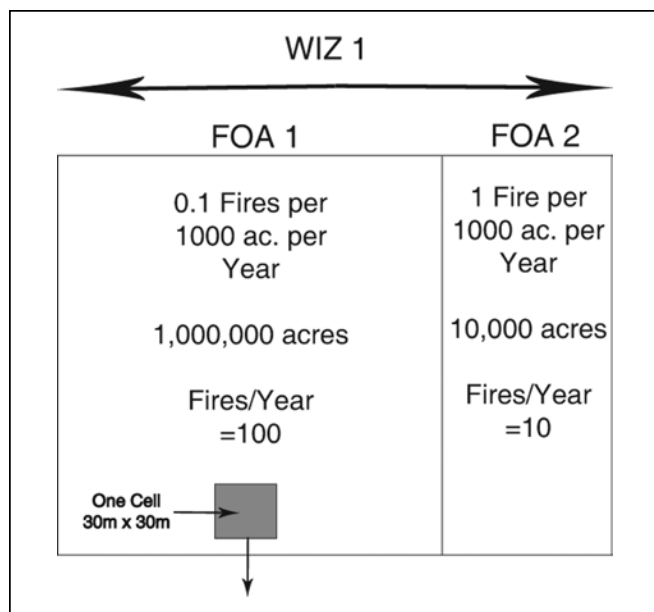


Figure 5—Example WIZ and FOA are detailed.

to develop relationships between fire spread rates and the expected final fire size. The inputs to this relationship are the expected fire behavior, which depends on fuels, weather, and topography and a measure of suppression effectiveness of fire protection forces.

For each weather influence zone, a relationship between the rate of spread and final fire size is developed using historical fire report data. This relationship can also be determined from the outputs of preparedness staffing modeling. Development using historical fire reports data requires the creation of several fire size classes where the time from fire start to fire containment can be estimated using fire report data. For all weather influence zones, the time from fire start to fire containment for the benchmark fire sizes of 0.5, 2, 10, 50, 100, 500, and 1,000 acres was determined. Additional fire sizes greater than 1,000 acres are used when fires of these sizes occurred historically within a WIZ.

The average fire rate of spread for each benchmark fire size is estimated by using the double ellipse area model developed by Fons (1946) as documented by Anderson (1983). The model calculates fire size (Area) as: $\text{Area} = K \times D^2$ where K is a constant dependent solely on midflame wind speed, and D is the distance the fire has traveled from

its point of origin (D = rate of spread times containment time). A relationship between the fire size and average rate of spread values for the benchmark fire sizes is developed using multivariate regression using a power series equation form ($Y = A + BX^C + DX^E$ where X = rate of spread, Y is the expected fire size and A through E are the regression coefficients). In some cases, a fourth-order polynomial equation form was utilized. In some WIZes, the constant term A was changed so that a 0.5-acre fire was expected when the rate of spread was 1 chain per hour (1.1 feet per minute). In addition, for each WIZ a maximum fire size was assigned.

Example of the Calculation of the Cellular Value for the Probability of an Acre Burning

The cellular value for the probability of an acre burning (CPAB) is calculated for each percentile weather category for each 30- by 30-m cell on burnable acres within the State of Florida. The four values from the four percentile weather categories are summed to obtain the total cellular value for the probability of an acre burning for a cell. The calculation is done for cells within an FOA and WIZ intersection. When the calculation is done for a cell, it is assumed that all cells in the FOA and WIZ intersection have the attributes of the cell. In essence, one is asking, “What would be the expected probability of an acre burning if all cells in the FOA and WIZ intersection were the same as the selected cell?”

To assist in the understanding of the calculation, an example is presented. Assume that the calculation is being done for a cell in FOA 1, WIZ 1 (Figure 5). The data flow is shown via the example in Table 1.

For the example, assume that the fire occurrence rate in FOA 1 is 0.1 fire per 1,000 acres per year and assume there are 1,000,000 acres in the FOA 1, WIZ 1 intersection (Figure 5).

Note there are 100 fires per year. Row 1 of Table 1 gives the percentage of fires that have historically occurred within each of the percentile weather categories. Multiplying the proportion of fires in each percentile weather category by the total number of fires in the FOA 1 / WIZ 1 intersection (100 fires) allows for determination of the number of fires in

Table 1—Example calculation of the cellular Wildland Fire Susceptility Index

Row	Item	Percentile weather				Total
		Low	Moderate	High	Extreme	
1	Percentage of fires	10%	80%	8%	2%	100%
2	Number of fires	10	80	8	2	100
3	Rate of spread (chains/hr)	2	5	12	24	N/A
4	Final fire size (acres)	1	6	98	900	N/A
5	Annual acres burned	10	480	784	1800	3074
6	Cellular probability of an acre burning	0.00001	0.00048	0.000784	0.00180	0.003074

each percentile weather category, Row 2 of Table 1.

The fire program (FlamMap for the FRA and the fb3.dll for the SFRA) has calculated a rate of spread for each percentile weather category (Row 3, Table 1) and a rate of spread versus expected final fire size relationship (Row 4, Table 1) has been determined. This allows for the determination of the expected final fire size within each percentile weather category.

Multiplying the number of fires per year in each percentile weather category by the expected final fire size yields the annual expected acres burned for each percentile weather category (Row 5, Table 1). Dividing the annual expected acres burned for each percentile weather category by the total acres within the FOA 1, WIZ 1 intersection (1,000,000 acres) yields the CPAB within each percentile weather category (Row 6, Table 1). The CPAB for the cell is the sum of the four percentile weather category CPAB values (Figure 6).

To consider the flammability of cells in the area of a given cell, a roving window (eight cells in radius) is drawn around each cell, and the average WFSI for all of the cells within that roving window is determined resulting in the roving window probability of an acre burning value (Figure 7). This allows for integration of the nearby CPAB values to reflect the flammability of the cells around a given cell.

Fire Effects

The Fire Effects Index comprises two input ratings: values impacted and suppression difficulty. The purpose of the index is to identify those areas that have important values at risk to wildland fire or are costly to suppress, or both.

Values Impacted

Several important values potentially impacted by wildfire were combined into an index for inclusion in the SWRA.

These values were also used in the FRA and include:

- Transportation and infrastructure areas
- Urban interface
- Plantations (natural and planted)

The transportation infrastructure effects were created from calculating a 300-m buffer around level 1, 2, and 3 roads and a 500-m buffer around elementary and secondary schools, airports, and hospitals.

The wildland urban interface (WUI) was downloaded from the SILVIS Lab at the University of Wisconsin - Madison (USDA FS 2001). The WUI is composed of both interface and intermix communities. In both interface and intermix communities, housing must meet or exceed a minimum density of 1 structure per 40 acres (16 ha). Intermix communities are places where housing and vegetation intermingle. Intermix areas have continuous wildland vegetation cover of more than 50 percent and more than 1 house per 16 ha. Interface communities are areas with housing in the vicinity of contiguous vegetation. Interface areas have more than 1 house per 40 acres, have less than 50 percent vegetation, and are within 1.5 miles of an area (made up of one or more contiguous census blocks) over 1,325 acres (500 ha) that is more than 75 percent vegetated. The minimum size limit ensures that areas surrounding small urban parks are not classified as interface WUI.

The plantation data were obtained from each individual state. This information was supplemented with a crosswalk from Gap Analysis Program data where available.

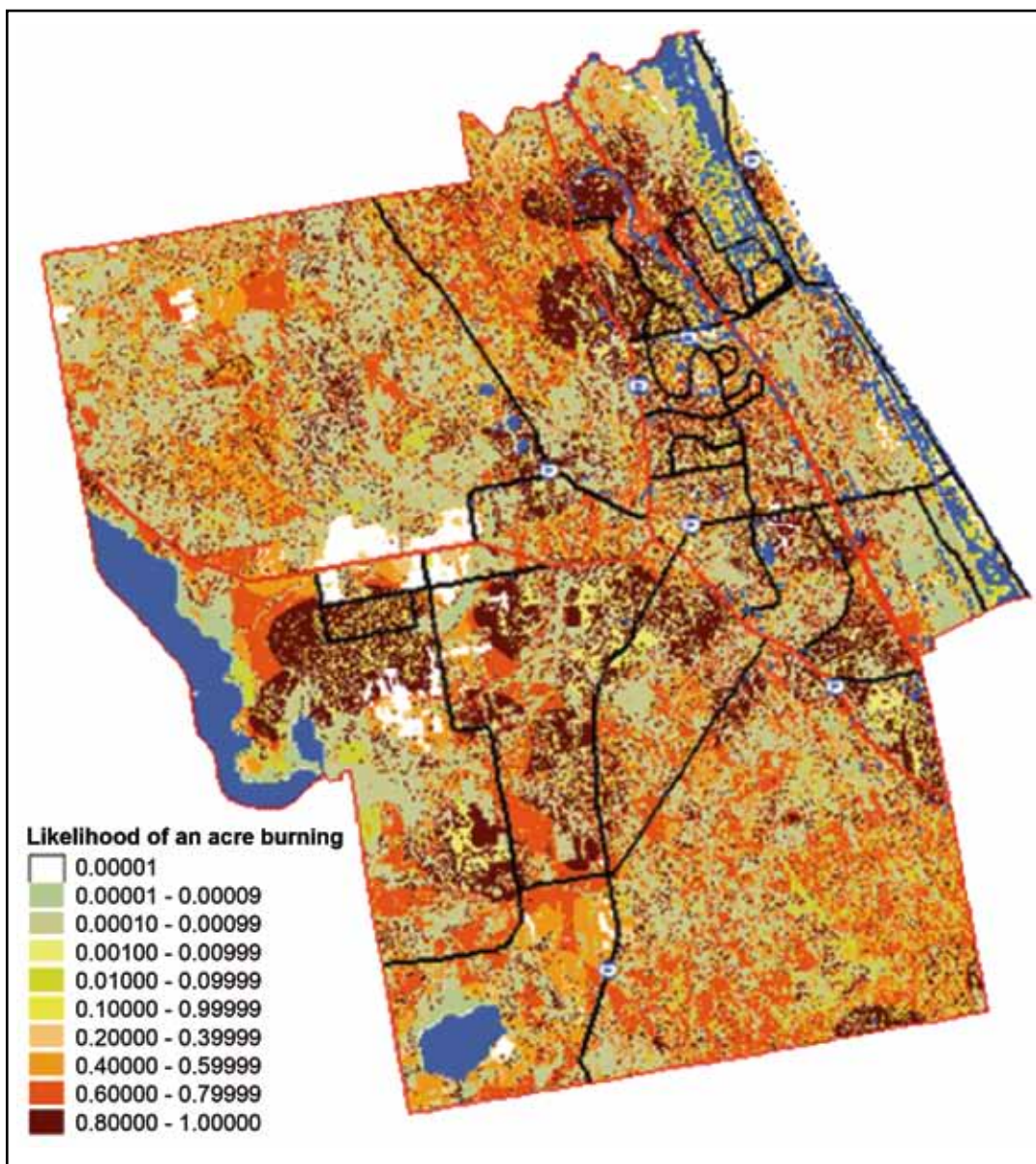


Figure 6—Flagler County, Cellular Wildland Fire Susceptibility Index.

Each value-impacted input was assigned an impact score by state fire managers using a matrix to assign a value of 1 to 4 (1 being low effect, 4 for high) for each flame length vs. fire size scenario. To arrive at a score for a value impacted, the individual values in the matrix were summed, i.e., 33. The Values Impacted Rating was determined by summing the values impacted scores for a cell $\times 100$ and dividing that total by the maximum possible score to normalize the result to a value between 1 and 100.

Suppression Difficulty Elements

The suppression difficulty elements are fuel type, topography, and soil type. A fuels layer was used to assign each cell in the state a fuel type of grass, shrub, timber litter, or slash. A topography multiplier was assigned to each of the following slope classes by state fire managers: slope class 1 is 0 to 25 percent; slope class 2 is 26 to 40 percent; slope class 3 is 41 to 55 percent; slope class 4 is 56 to 75 percent, and slope class 5 is 76+ percent. Organic/peat (muck) soils

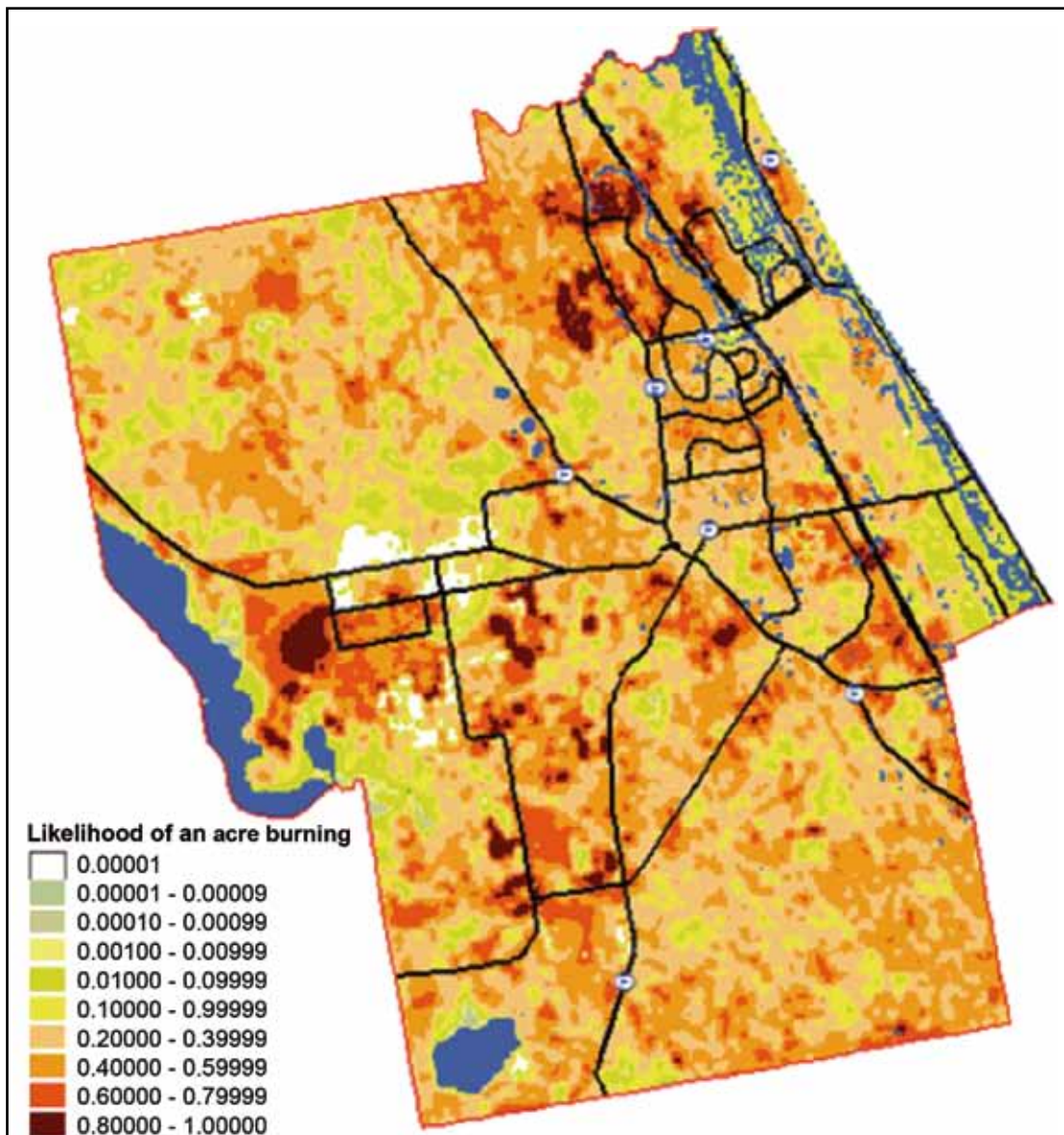


Figure 7—Flagler County, roving window Wildland Fire Susceptibility Index.

were extracted from SURGO data by the states. These soils constitute areas of concern for firefighting efforts, as fires within these areas tend to be expensive and difficult to extinguish.

In arriving at the Suppression Difficulty Rating, suppression costs are evaluated by fuel type and topography. Each burnable cell in the state was assigned a suppression score using a matrix process similar to the one used for the values impacted score. The grass, shrub, timber litter, and

slash fuel type scores were based on professional judgment of the state fire managers. The increased difficulty of suppression based on slope was also made. The suppression difficulty score for organic/peat soils was assigned to be 60, which is 1.25 times the maximum score of 48. This higher value is to reflect the increased suppression difficulty in this situation. The Suppression Difficulty Rating for each cell was calculated by multiplying the fuel type score and the topography multiplier by the product of the maximum fuel type score and the maximum slope multiplier.

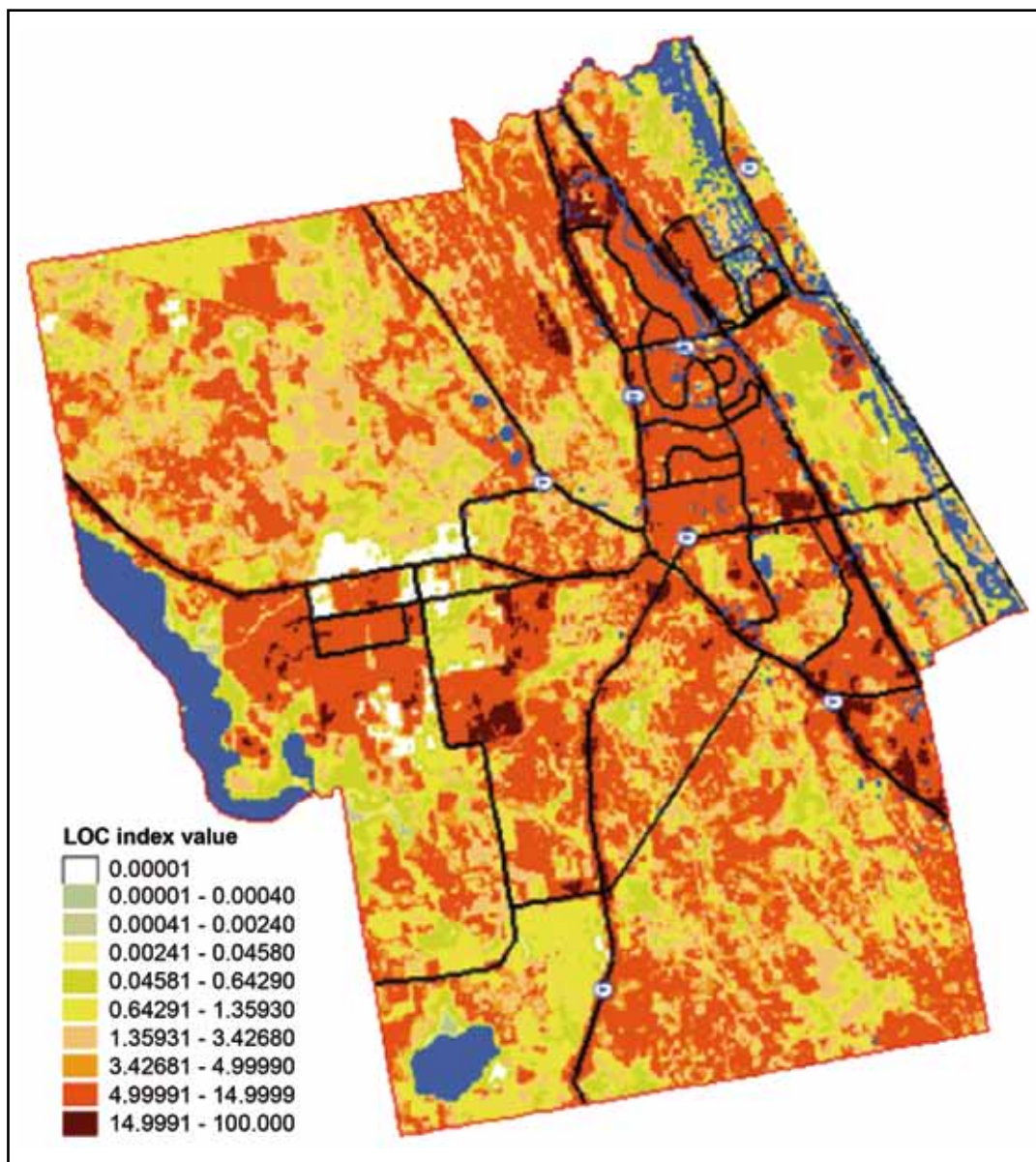


Figure 8—Flagler County, Level of Concern (LOC).

Fire Effects Index

The Fire Effects Index was calculated as the sum of the Values Impacted Rating times 0.68 and the Suppression Difficulty Rating times 0.32. The final Fire Effects Index can range from 0 to 100.

Level of Concern Index

The Level of Concern Index is calculated as the Wildland Fire Susceptibility Index times the Fire Effects Index. The

WFSI is a value between 0 and 1. The Fire Effects Index is a value between 0 and 100. Hence, the LOC is a value between 0 and 100.

The output values were assigned to 10 LOC categories ranging from low concern to high concern. The LOC output can be used to prioritize areas for further analysis. An example LOC map for Flagler County in Florida is shown in Figure 8.

The LOC results can be used to complete a more detailed analysis at the local level and communicate wildland fire management concerns. The LOC results can be used to:

- Identify areas where mitigation options may be of value.
- Allow agencies to work together and better define priorities.
- Develop a refined analysis of a complex landscape and fire situations using GIS.
- Increase communication with local residents to address community priorities and needs.

Fire Response Accessibility Index

The Fire Response Accessibility Index (FRAI) is a relative measure of how long it would take initial attack resources to drive from their resource location to each cell. The Fire Response Accessibility Index is calculated based on the distance from resource locations. The speed traveled on roads was estimated based on the class of road. Travel off of roads was assumed to be at 5 mph. Water was coded as “NO DATA,” meaning that travel across water could not be done unless there was a road crossing. A cost distance analysis was run allowing Arc/Info to assign an approximate time to reach each cell.

The Fire Response Accessibility Index allows users to identify areas of low accessibility from their resources. Coupled with the Levels of Concern data, this information will highlight areas where accessibility is low and the level of concern is high, providing valuable information for those concerned with the impacts of wildland fire. An example FRAI map for Flagler County in Florida is shown in Figure 9.

Uses and Application in Florida

During the initial development phase of the Florida Risk Assessment, the development team outlined the following project objectives:

- Rapidly identify areas that may require additional tactical planning.
- Allow agencies to work together to better define priorities and improve emergency response.

- Develop refined analysis of a complex landscape and fire situations using GIS.
- Increase communication with local residents to address community priorities and needs.
- Plan for fire protection resource needs.
- Identify fire protection resource allocation based on potentially severe fire problems.

Although it is generally believed that the goals and objectives were met, one point concerning the assessment should be emphasized. The FRA has many parts, and some of these parts have been used to support other state agency critical applications such as the Fire Management Information System. The success of the FRA extended beyond the original goals and objectives.

The Division of Forestry has produced two products to convey the Wildland Fire Risk in Florida. The first is the stand-alone desktop application that requires the following software: ArcView 3.x, Spatial Analyst, and FlamMap (Version 1) as well as the data for the specific areas of concern. This application permits the user to view the published data and make changes to both the fire occurrence and fuels to alter the relative risk in the area. The purpose of the modifications to fire occurrence or to fuels is to determine the effect a changing prevention effort or fuels management effort might have on the overall wildfire risk in a particular area.

The second application is Web based and can be found at: http://www.fl-dof.com/wildfire/wf_fras.html. This tool allows anyone with Web access to view the four primary published results data layers for Florida. These include the Wildland Fire Susceptibility Index, which is an indicator of the potential for wildfire in that area; the Fire Occurrence Areas, which is a map of the probability of an acre igniting based on the fire history in an area; the Surface Fuel Model Layer, which maps the surface fuels across the state; and the Level of Concern, which is a combination of all of the above as well as the suppression costs and environmental effects to give the user the general associated risk from wildland fire for a particular area. This tool has been very popular with homeowners and the media.

As an example of the interest in the FRA by the media, the following example from Brevard County is provided.

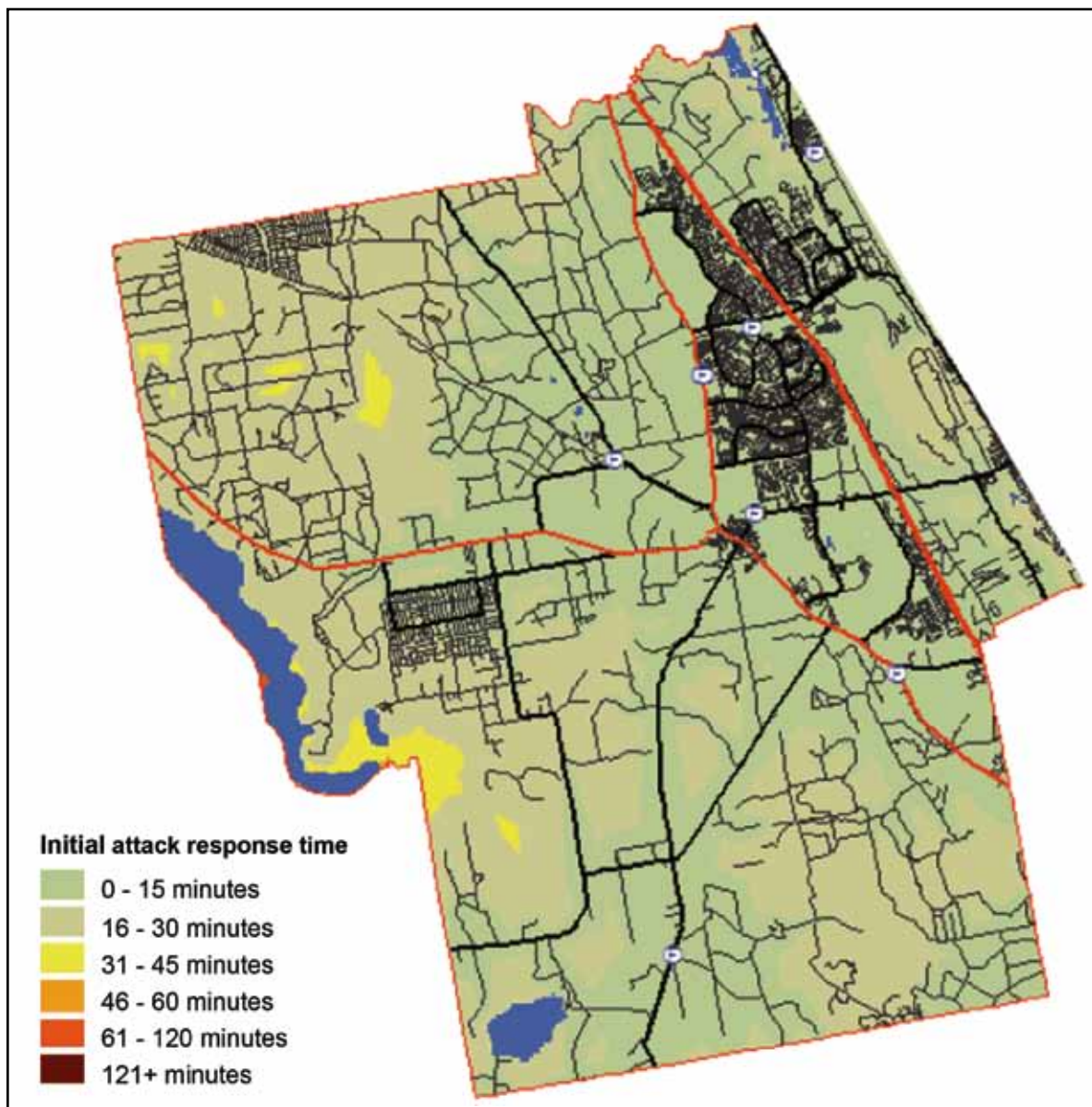


Figure 9—Flagler County, Fire Response Accessibility Index.

In May 2006, the *Florida Today* newspaper published an article about the Florida Wildland Fire Risk Assessment and highlighted some particular points the paper felt the general public should know. The paper detailed information about certain areas in Brevard County, which was impacted by the 2006 fire season. The following is a list of bullets that were included in the article:

- Thousands of homes in Brevard County lie within zones state foresters deem the most dangerous places for wildfires but often the least practical to burn or clear.
- Many single-family houses (5,000+) in Brevard were built on land considered at highest risk for wildfires.
- Thousands more mobile homes and other structures fall within the same danger zones.
- In Brevard County, 78,669 acres are in the high-risk zone, or about 12 percent of the county. Most of it, 49,545 acres, is in unincorporated areas, such

as those surrounding Lake Washington, west of Melbourne, and Lake Poinsett, west of Cocoa.

- West Melbourne, Melbourne, Melbourne Village, and Rockledge had the highest percentage of high-risk land, West Melbourne being the worst with 67 percent of its 3,648 acres within the highest risk area.
- Trees and brush border most homes, making few Brevardians immune from the wildfire threat.
- State fire managers focus most of their preseason prevention where forest hugs neighborhoods and important infrastructure. So they hope people such as Dyan Hilton, who lives in Poinsett Trailer Park—west of Cocoa and across from a huge wildfire danger zone to the south—take steps to keep the flames away.

In addition to the media, county and municipal governments are using the FRA as part of the county or local Comprehensive Planning Program that requires that all risk be considered as part of the planning for new development.

Fire departments and county planners are closely monitoring how communities structure access as well as do vegetation/fuel management in the initial phases of community development. It is emphasized that developers and homeowners should accept each party's responsibility in the protection of the property. When drought conditions occur and the fire weather gets to the point that fires begin to impact neighborhoods, firefighters have difficulty providing structure protection at every location within a subdivision. Planning for the future can prevent many of the problems experienced in recent years.

Florida has staffed four fuels mitigation teams that exclusively work in urban-interface areas. The FRA is the primary designator as to where these teams plan their efforts. The FRA paints a bull's-eye across the Florida landscape for everyone managing land and fire today, placing the priorities where they need to be based on fuels, climate, and historical fire activity.

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Air Pollution Increases Forest Susceptibility to Wildfires: A Case Study for the San Bernardino Mountains in Southern California

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Abstract

Many factors increase susceptibility of forests to wildfire. Among them are increases in human population, changes in land use, fire suppression, and frequent droughts. These factors have been exacerbating forest susceptibility to wildfires over the last century in southern California. Here we report on the significant role that air pollution has on increasing forest susceptibility to wildfires, as unfolded in the San Bernardino Mountains from 1999 to 2003. Air pollution, specifically ozone (O_3), and wet and dry deposition of nitrogenous compounds from fossil fuel combustion, has significantly increased since industrialization of the region after WWII. Ozone and elevated nitrogen deposition cause specific changes in forest tree carbon, nitrogen, and water balance that enhance individual tree susceptibility to drought and bark beetle attack, and these changes contribute to whole ecosystem susceptibility to wildfire. For example, elevated O_3 and N deposition increase leaf turnover rates and leaf and branch litter, and decrease decomposability of litter. Uncharacteristically, deep litter layers develop in mixed conifer forests affected by air pollutants. Elevated O_3 and N deposition decrease the proportion of whole tree biomass in foliage and roots, the latter effect increasing tree susceptibility to drought and beetle attack. Because both foliar and root masses are compromised, carbohydrates are stored in the bole over winter. Elevated O_3 increases drought stress by significantly reducing plant control of water loss. The resulting increase in canopy transpiration, combined with [O_3 + N deposition]-induced decreases

in root mass significantly increase tree susceptibility to drought stress, and when additionally combined with increased bole carbohydrates, perhaps all contribute to success of bark beetle attack. Phenomenological and experimental evidence is presented to support the role of these factors contributing to the susceptibility of forests to wildfire in southern California.

Keywords: Bark beetle, fire suppression, forest densification, N deposition, O_3 exposure.

Introduction

Many factors combine to increase forest susceptibility to wildfire in southern California, and most of these were set in motion decades ago. These factors include a rapid increase in human population and resource use; a shift from timber production to recreational forest use; fire suppression with subsequent forest densification; periodic, extreme drought; and bark beetle outbreaks. The contribution of air pollution to forest susceptibility to wildfire has not been studied extensively. In this paper, we will link air pollution to increasing forest densification, litter build up, drought stress, tree susceptibility to successful bark beetle attack, tree mortality, and increased forest susceptibility to wildfire (Figure 1). A case study will be presented for the San Bernardino Mountain Range in the Transverse Range north and east of Los Angeles, California. We will focus on pollutant effects on ponderosa pine, which dominates the mixed conifer forest in the western part of the range.

In the late 19th century, gold and other valuable minerals were discovered in the San Bernardino Mountains, and the population rapidly increased (Minnich 1988). The forest was logged for buildings, mine shaft support, and for fuel. In 1899, a severe drought occurred, water was limiting, and a premium was placed on reservoir development (Lake Gregory, Arrowhead, Big Bear). As the reservoirs were established, they became magnets for recreation use in the 1920s. With the shift from resource utilization to recreation, incursions of fire from the chaparral into the forest were suppressed, and forest density increased through the 1940s.

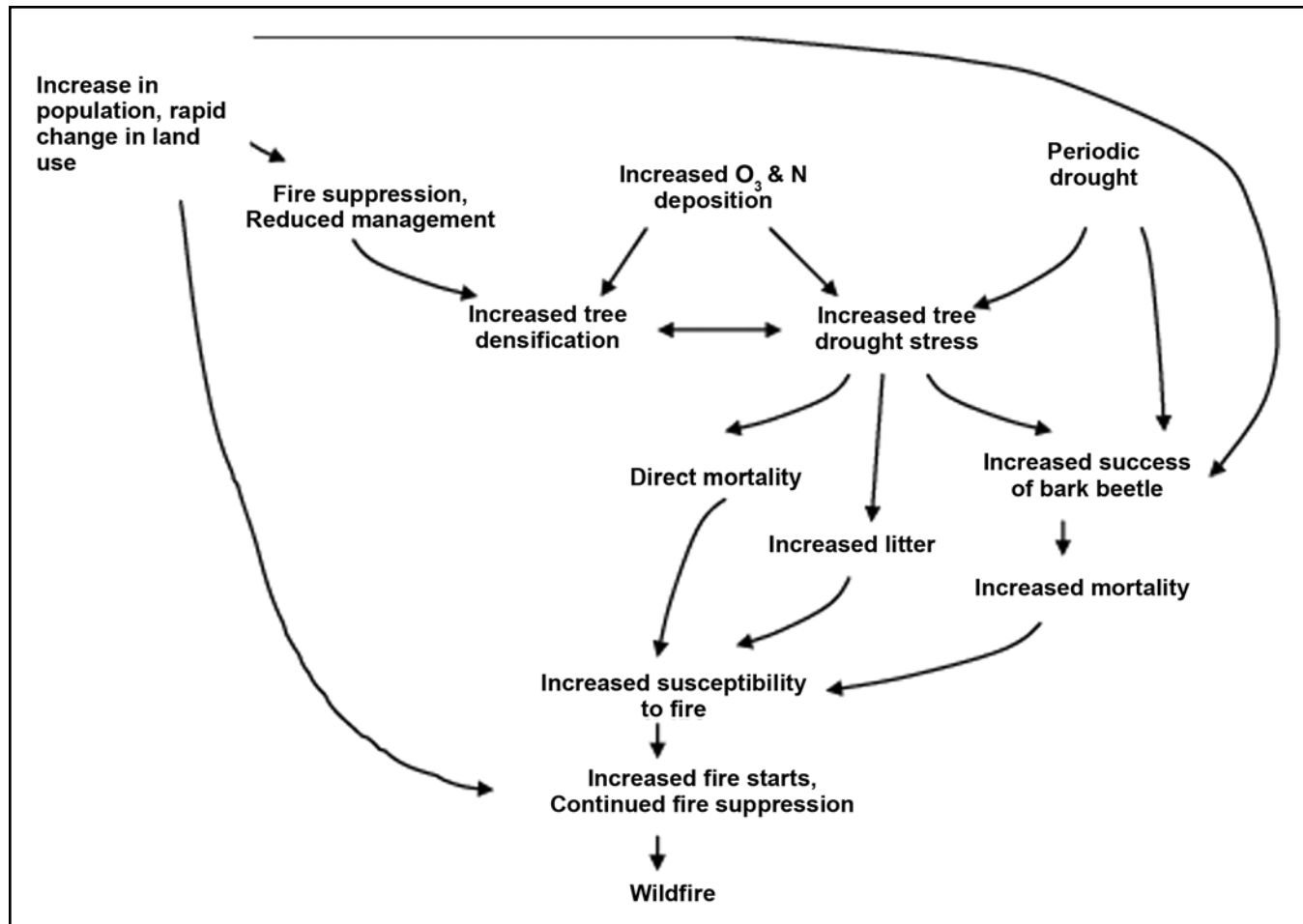


Figure 1—Many factors contributing to forest susceptibility to wildfire were set in motion decades before ignition. However, increased O₃ exposure and nitrogen (N) deposition significantly contribute to forest susceptibility to wildfire by increasing tree drought stress (with subsequent increase in successful bark beetle attack) and promoting formation of deeper, more recalcitrant litter layers.

In the 1950s, the Forest Service made an attempt to thin the forests, but, for aesthetic reasons, the mountain communities strongly opposed both branch trimming and stand thinning. As a consequence, the forest continued to increase in density, and trees grew increasingly closer to structures. In the 1980s, the community councils drew up “Forest Plans” that included branch trimming and thinning of trees within 30 m of valued structures (Asher and Forrest 1982). However, these recommendations were not followed or enforced. The region was, and is, highly susceptible to wildfire.

Air Pollution Effects: O₃ and N Deposition

The primary source of air pollution is fossil fuel combustion from trucks, cars, trains, ships, and industry (South Coast Air Quality Management District 1997). Fossil fuel

combustion emits nitrogen oxides, which are converted to other nitrogen oxides and ozone (O₃) in the presence of high energy UV light. Both nitrogen oxides and O₃ are strong oxidizing agents and cause damage to cells. Ozone is transported long distances. Nitrogen oxides are not transported as far as O₃, but at moderate O₃ levels, dry and wet deposition of nitrogen to plant communities is significant (6 to 9 kg/ha per year) (Fenn and others 1996) and accumulates through time. The effects of O₃, N deposition, and periodic drought are evaluated here as contributing factors to forest susceptibility to wildfires.

Ozone is primarily deposited on surfaces (such as surfaces of leaves, branches, bark, soil, and litter), and there decomposes. Bauer and others (2000) estimated that approximately one-third of the O₃ is taken into the plant

via stomata. When plants take up CO_2 , they also take up O_3 . Once O_3 enters the leaf, it may be decomposed in a thin film of water (apoplastic water) that surrounds cells in the substomatal chamber, or it may pass across the cell membrane to the chloroplast where it degrades photosynthetic pigments. The decomposition of O_3 in the apoplastic water requires the regeneration of oxidized ascorbate with glutathione in the cytosol and energy (De Kok and Tausz 2001). As it passes across membranes, the acidity changes, and membrane permeability is altered such that ions that should be retained by the cell now leak out (K^+), others influx along a chemical gradient (Zhang and others 2001), and other mechanisms of ion transport are blocked (e.g., Ca^{++} channeling) (McAinsh and others 2002). When strong oxides degrade photosynthetic pigments in the chloroplast, the pigments must be reconstructed into functional arrays, which requires energy.

The first measurable effect of O_3 on plants is a decrease in the efficiency of photosynthesis or the carbon-capturing mechanism. Because O_3 damages tissue, there is a metabolic cost in energy and constituent building materials that increases respiration and decreases the total carbon stored by the plant. Carbon-carbon links store energy in plants for later use. Lower total carbon stored and greater requirements for N (to build more photosynthetic pigments) result in retranslocation of materials out of older needles. Older branches are excised in O_3 -exposed trees because the net carbon balance is lower. Presumably, when net carbon balance drops below zero, the branch is excised (Sprugel and others 1991).

At the whole plant (tree) level, O_3 exposure results in premature senescence of needles (within-whorl loss of needles, fewer needle-age classes) in ponderosa pine (*Pinus ponderosa* Dougl. ex Laws) (Grulke and Balduman 1999) and lower branch abscission (Miller and others 1996b). Greater O_3 exposure and N deposition cause more needle and branch loss. The western end of the San Bernardino Mountains, the part of the mountain range closest to Los Angeles, has the highest N deposition and O_3 exposure (30 to 40 kg N/ha per year, 80 parts per billion (ppb) O_3 per hour, averaged over 24 hours for the 6-month growing season). Premature needle senescence is so extreme that 95

percent of the total canopy biomass in whole-tree harvests is in current year foliage (Grulke and Balduman 1999). At an atmospherically clean site (low N deposition, and 38 ppb O_3 per hour, averaged over 24 hours for the 6-month growing season) near Lassen Volcanic National Park, canopy foliar biomass was evenly distributed across four to five needle-age classes. In response to pollutant deposition, increased needle and branch loss significantly contribute to increased litter inputs to the mixed conifer ecosystem. Needles produced in high O_3 exposure environments have higher lignin content. Greater lignin content reduces decomposability (Fenn and Dunn 1989), further exacerbating litter layer buildup. For example, in the western end of the San Bernardino Mountain Range, where trees are most affected by transported air pollutants, litter depth averaged 25 cm. In the eastern end of the range with significantly lower long-term pollutant exposure, litter depth averaged 0 to 3 cm (N.E. Grulke, field observations). Significant litter buildup in mixed conifer forest was predicted by the simulation model BGC for both high N deposition and O_3 exposure (Arbaugh and others 1999) based on field leaf turnover rates (Miller and others 1996a).

Perhaps because of the increased repair costs for aboveground tissues, less biomass is retained in roots. From the western to the central San Bernardino Mountains, both higher O_3 concentration and greater N deposition contributed to significantly lower fine and medium root mass at 10-, 30-, and 50-cm depths. Root mass at a cleaner site was 6 to 14 times as great as that at a moderate pollution site (6 to 9 kg N/ha per year, and 62 ppb O_3 per hour, averaged over 24 hours for the 6-month growing season) (Grulke and others 1998).

Ponderosa pine had lower leaf mass, lower root mass, and lower carbohydrate content in both leaf and root tissue in areas with higher pollution exposure. In general, over the winter, carbohydrates for spring growth are stored in the roots. However, because of the lower leaf and root mass, overwintering carbohydrate is stored in tree boles (Grulke and others 2001). Although the theory is untested, increased carbohydrate storage in tree boles may enhance fecundity of bark beetles.

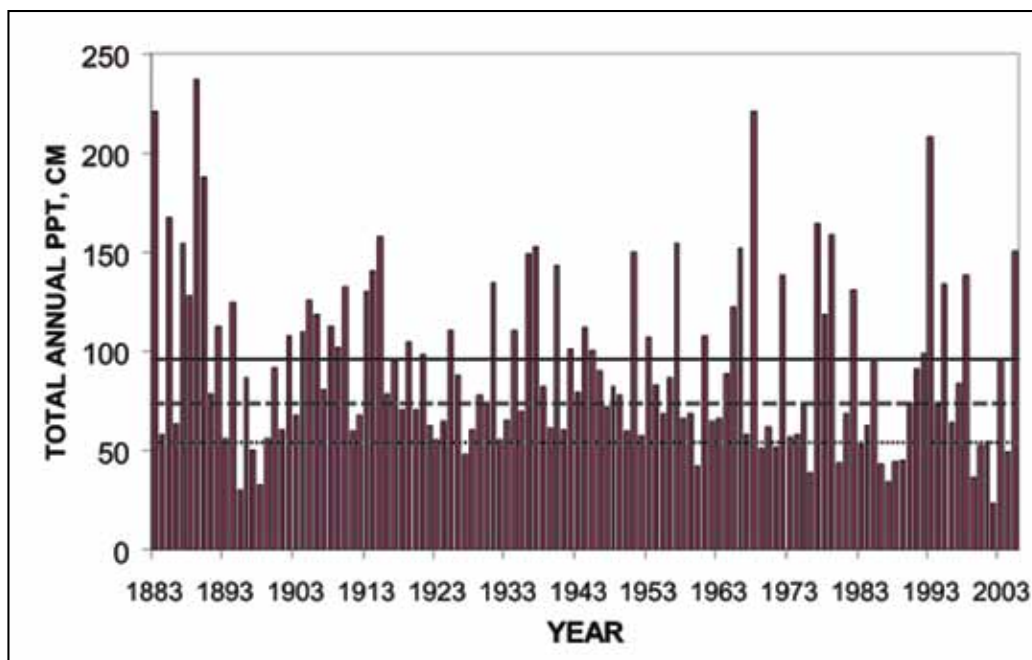


Figure 2—Long-term record of precipitation at Big Bear Dam, California. Total annual precipitation (y-axis in cm) from October 1 to September 30 was accumulated and plotted. The overall average is denoted by a solid line (96 cm), with the 80 percent of average (below which moderate drought stress is incurred) and the 60 percent of average (below which severe drought stress is incurred) denoted by dashed and dotted lines respectively.

At the scale of the stand, both N deposition and moderately high to high O₃ concentrations increased stand density (Arbaugh and others 1999, Miller and others 1989), especially on north-facing slopes and in microsites with more water availability or lower evapotranspiration (topographic lows). Individual tree growth further increased after O₃ concentrations had been decreased in the early 1990s because of heightened regulatory controls (see Tingey and others 2004). Across seven sites varying in both O₃ exposure and N deposition, three of the four sites with the highest pollutant load (Cedar Pines Park, Dogwood Campground, and Camp Angeles) had the highest tree mortality rate (primarily ponderosa pine) (Arbaugh and others 1999).

Effects of Periodic Drought

Although there is an increase in evapotranspiration from west to east, weather that results in precipitation in the San Bernardino Mountains is generally a regional phenomenon. We have used the longest record of precipitation for the range, collected over the last 120 years at Big Bear Dam

(San Bernardino Water Management District), to identify the level of drought stress experienced by ponderosa pine from year to year. Moderate drought stress is defined physiologically as reduced cell turgor that generally results in reduced stomatal conductance (reduced water loss from the leaf), and lower cellular water potential, which allows the tissue to hold onto the water that is in the leaf more tenaciously (Levitt 1980). In 1994, a year of 80 percent of the average precipitation (preceded by an above-average precipitation year), ponderosa pine experienced moderate drought stress from mid-July through the end of the growing season (Grulke 1999). Severe drought stress is also accompanied by reduced cell turgor, reduced stomatal conductance, and reduced cell-water potential. The water potential is lowered sufficiently that cell solutes are concentrated enough to disrupt enzymatic function, and cell turgor is reduced enough and for a long enough duration that cell elongation growth is limited. Needles produced in years of severe drought stress are shorter. In 1996, a year of 60 percent of the average precipitation (preceded by an above-average precipitation year), ponderosa pine experienced

severe drought stress from the end of June through the end of the growing season (Grulke 1999). Over the period of the long-term precipitation record, roughly 15 percent of the years had low enough total annual precipitation to result in moderate drought stress; 30 percent of the years had low enough total annual precipitation to result in severe drought stress. Using this rough index of the level of physiological stress, ponderosa pine experienced drought stress 45 percent of the years since 1883 when precipitation records were initiated (Figure 2).

Where O_3 exposure and nitrogen deposition reduce root biomass, trees are predisposed to drought stress. In general, low to moderate O_3 exposures (<60 ppb hourly O_3 , averaged over 24 hours for the 6-month growing season) reduce water loss from trees. Ozone reduces photosynthetic rates, less CO_2 is required, and stomatal apertures are reduced to conserve water. However, under concentrations that are moderately high or higher, O_3 exposure modifies stomatal behavior in ways that increase drought stress.

For example, sugar maple (*Acer saccharum* Marsh.) was exposed to O_3 concentrations of 70 ppb during daylight hours (Tjoelker and others 1995). Early in the growing season and experiment, neither the net photosynthetic rate nor stomatal conductance was affected by the treatment. By midseason, there was a significant decrease in water-use efficiency—at the same level of carbon gain, seedlings growing in chronic O_3 exposure had twice the level of water use as had control seedlings grown in charcoal-filtered air. By late season, both net photosynthesis and stomatal conductance were suppressed in plants grown in chronic O_3 exposure. In a field study of sensitive and tolerant genotypes of Jeffrey pine (*Pinus jeffreyi*; Grev. & Balf.) exposed to the same ambient O_3 levels (~68 ppb O_3 averaged over 24 hours, for the 6-month growing season in Sequoia National Park), sensitive genotypes had lower water loss under moist, favorable conditions and higher water loss under dry, unfavorable conditions (Patterson and Rundel 1989). Under favorable conditions, Jeffrey pine had less water loss, but because the stomatal apertures were smaller, there was also less photosynthetic carbon gain. Under unfavorable conditions (most of the day in the Sierra Nevada), sensitive

Jeffrey pine had higher water loss, which would result in greater desiccation.

Although physiologists often report plant response under steady state (stable) conditions, the light environment in the forest is often dynamic. Understanding stomatal responses under rapidly changing environmental conditions with concurrent O_3 exposure can perhaps better explain why trees exposed to moderately high and higher concentrations of O_3 lose more water. In typical forest environments, foliage on a primary branch on the southern aspect of an open-grown tree receives flecky light two-thirds of the time (Grulke 2000). For example, the cutleaf coneflower (*Rudbeckia laciniata* L. var. *digitata* Mill. Fiori) is one of the most sensitive native plants to ambient O_3 concentrations in Great Smoky Mountains National Park. It persists in forest gaps and on forest-meadow margins, both with flecky light environments. Tolerant plants of cutleaf coneflower had normal responses to experimentally manipulated change in light from low to high and back down to low levels. However, O_3 -sensitive plants had either no stomatal response or a muted stomatal response to changes in light level. The level of water loss from leaves with no or muted response to changing light level was high—they did not conserve water when light was low, and this failure to conserve water would contribute to desiccation. When humidity was lowered slowly, O_3 -sensitive plants closed their stomata at much lower relative humidities than did O_3 -tolerant plants, and this also contributed to greater desiccation (Grulke and others 2007a). In a similar experiment with California black oak saplings (*Quercus kelloggii* Newb.) exposed to anthropogenic high O_3 in a natural stand, stomatal closure in response to abruptly reduced light level was slower in plants without additional N amendment, and N amendment partially mitigated the desiccating effects of high O_3 exposure (Grulke and others 2005).

Moderate to high O_3 exposure can also cause stomata to remain partially open at night. In experimental O_3 exposures, this was first observed in Norway spruce (*Picea abies* (L.) Karst), (Weiser and Havranek 1993) and in birch (*Betula pendula* Roth), (Matyssek and others 1995). Nighttime water loss rates were 25 percent as great as

Table 1—Historical record of drought years and bark beetle epidemics in the San Bernardino Mountains

Drought years	Aver. percent ppt	Beetle epidemic?
1898-1900	50	Not known
1923-1925	66	Not known
1927-1930	70	Not known
1948-1951	79	Yes ¹
1959-1961	63	Yes ¹
1970-1972	59	no
1976-1977	61	Yes ¹
1988-1990	45	Yes ¹
1999-2002	45	Yes ²

Note: Shown are the years of moderate or severe drought, the average percentage of total average annual precipitation (based on the 120-year record), and whether a bark beetle epidemic occurred after the sequence of drought years.

¹ California Pest Reports, 1949 to present.

² N. Grulke, personal observation.

full daytime rates for Norway spruce, and 50 percent as great for birch. This was corroborated in ponderosa pine across the San Bernardino Mountain pollution gradient, with both higher O₃ and NO₂ and HNO₃ exposure. In the San Bernardino Mountains, nighttime water losses were 10 percent of full daytime rates (Grulke and others 2005). Because these studies were largely phenomenological, a new gas exchange system was designed and built to directly test known O₃ concentrations on single leaves. Chronic, moderate O₃ exposure (70 ppb O₃ for 8 hours per day for 1 month) significantly increased nighttime foliar water loss in California black oak and blue oak (*Quercus douglasii* Hook. & Arn.) (Grulke and others 2007b). Nighttime water losses were attributable directly to O₃ exposure and were 30 percent and 20 percent, respectively, of daytime rates in these species. Moderately high (or higher) O₃ exposure increases foliar water loss and increases tree susceptibility to drought stress.

Susceptibility to Successful Bark Beetle Attack

Air pollution exposure (O₃ and N deposition) increases tree susceptibility to drought stress, and drought stress increases tree susceptibility to successful beetle attack. Dunning

(1928) was one of the first to report a relationship between drought conditions and increased levels of tree mortality caused by western bark beetles. In the west, bark beetles reach epidemic proportions after 2 to 3 years of drought. The correlation between beetle attacks and climate can be diffuse because bark beetles may delay or prolong the exact time of tree mortality. However, mortality tends to increase in multiyear droughts, particularly in highly dense stands or those with pre-existing damage or stress. Figure 2 indicates that sequences of 2 to 3 years of drought have occurred nine times (Table 1) in the last 120 years. We can document five bark beetle epidemics associated with those sequences.

Beetles are opportunists that attack trees in a weakened state. With only a few exceptions, either the host tree is killed by the colonizing bark beetles or the host resistance of the tree kills the attacking adults. To kill a tree, large numbers of bark beetles must successfully colonize it in a relatively short period of time (Paine and others 1984, 1997). However, fewer beetles may be sufficient to kill a compromised tree (Paine and others 1984). The bark beetles most commonly responsible for tree mortality in the western San Bernardino Mountains are western pine beetle (*Dendroctonus brevicomis*) and mountain pine beetle (*D. ponderosae*). Western pine beetle can produce up to four generations in a year in southern California, where the Mediterranean climate is conducive to rapid population expansion when an abundance of susceptible host material is available for colonization.

Eggs are laid in the inner bark and the larvae excavate galleries. Pupation occurs in either the inner or outer bark, depending on the species of beetle. Adults emerge from the larval host tree and search for susceptible hosts. Healthy pines and firs respond by exuding pitch, which either pitches out the adults or blocks their progress. Resin production impedes bark beetle attack both physically and chemically. Oleoresin pressure, caused by turgidity of cells lining the resin ducts, forces preformed resin to the site of injury or invasion and results in a flushing action. The cell turgidity is derived from the transpirational stream, so if the tree is under moisture stress, the cells become increasingly flaccid, the resin pressure is reduced, and the effectiveness

of the preformed resistance is compromised (Vite 1961). In weak trees with reduced resin pressure, the adults are able to initiate colonization and produce specific pheromones that attract other colonizing adults. Pheromone production ceases when the host tree ceases resin flow (i.e., when the tree dies) (Raffa and Berryman 1983).

Severe drought and other stresses also reduce the photosynthetic capacity of trees and the levels of carbohydrates used for growth, defense, and tissue repair. This can have significant impact on the ability of the tree to induce an effective response to invasion (Paine and Stephen 1987a, 1987b). Drought-stressed trees are also known to have elevated levels of free, translocatable proteins (Lei and others 2006), which are produced to generally increase cell osmoticum. We conjecture that increased bole carbohydrate content as a result of O₃ exposure + N deposition + drought and elevation of protein levels in response to drought enhance beetle fecundity. Pollutant-exposed trees may thus be primed for successful bark beetle attack.

The forest had been recently thinned early in the late 19th century by commercial logging, so we would not expect to observe an epidemic beetle infestation in immature, low-density stands despite the drought stress experienced in the late 1920s (Minnich and others 1995). Human population in the Los Angeles Basin significantly increased after World War II, but air pollution levels were not quantified (or reconstructable) until 1963. From 1963 through 1980, peak 1-hour O₃ concentrations averaged 250 to 425 ppb (Lee and others 2003). From 1980 on, peak 1-hour O₃ concentrations were still high (>250 ppb), but cumulative O₃ exposures over the growing season began to decline. Through strong regulatory controls, O₃ concentrations declined further to tens of occurrences to only isolated occurrences of hourly concentrations exceeding 95 ppb from the mid-1990s to present in the mountains. Throughout this time, N deposition continued to accumulate, and drought stress was exacerbated by chronic, if not acute, O₃ exposure.

The most extreme drought recorded in 250 years was experienced in the hydrologic year 2002 (10/1/01 through 09/30/02) after 3 years of chronic drought. Results of 3 years of chronic drought (1999–2001) and extreme drought

(2002) are shown in a sequence of imagery taken at 5 km above the forest canopy at the most polluted site in the western San Bernardino Mountains (Figure 3). After the chronic drought, bark beetles attacked, and there were clusters of tree mortality. However, the average stand tree mortality rate was near 0 percent. After the chronic and severe drought, tree mortality (primarily ponderosa pine) from both bark beetle and drought increased to approximately 5 percent at the stand level. In the spring following the wet winter, bark beetle populations reached epidemic proportions, and 42 percent of the stand had died (ponderosa pine, white fir [*Abies concolor* (Gord. & Glend.) ex Hildebr.], and sugar pine). The stand was at high risk for an intense fire with high litter layers, high numbers of standing dead trees, and exacerbated drought stress. In autumn of that year, the Old Fire swept through the stand. Interestingly, not all of the red trees (standing dead trees with needles retained) were consumed in a crown fire because the highly dense understory was not in contact with the lower branches of the 100+ year-old trees—the effects of O₃ exposure, N deposition, and drought had promoted lower branch abscission (Miller and others 1996b), so that the lowest branches were attached at heights of 60 ft or greater. Trees are still dying from bark beetle at this site (as of 7/06), but the rate of change is now statistically undetectable.

Conclusions

The role of air pollutants in increasing tree susceptibility to drought, successful bark beetle attack, tree mortality, and the susceptibility of forests to wildfire have not been formally supported. Air pollutants, specifically strong oxidants and nitrogen deposition, contribute to increased litter accumulation and increased tree susceptibility to drought stress. It is well known that drought-stressed trees are more susceptible to bark beetle attack. The combination of chronic drought in 1999–2001 and acute drought in 2002 resulted in a bark beetle epidemic in the western San Bernardino Mountains. We contend that the severity of tree mortality in the western San Bernardino Mountains was significantly exacerbated by the higher air pollutant deposition in this region.

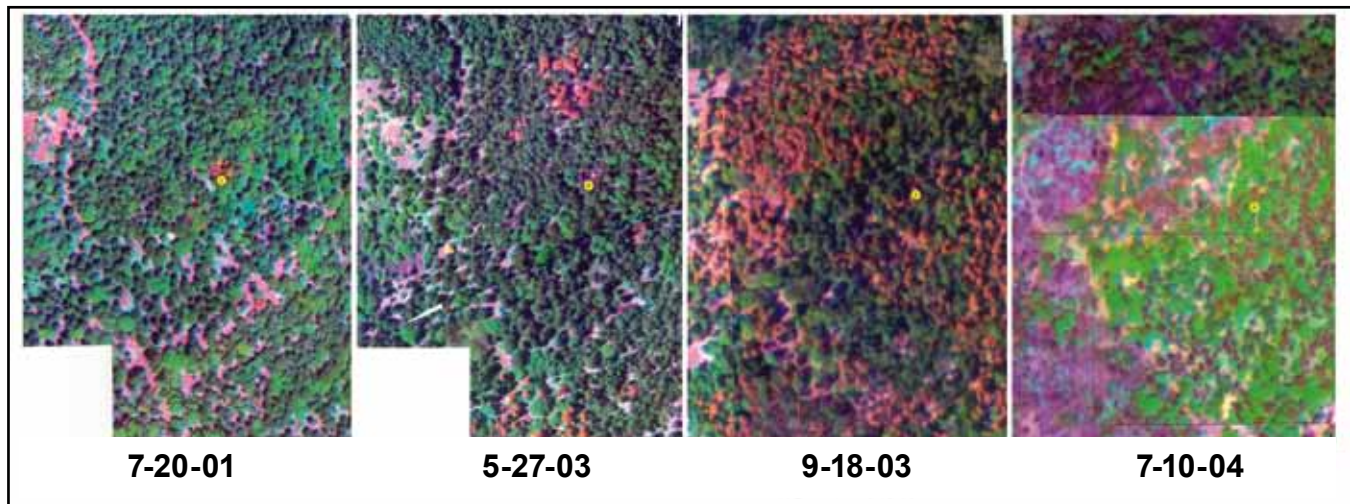


Figure 3—Remote imagery of Cedar Pines Park, San Bernardino National Forest, CA. This forest stand is the most affected by air pollution deposition in the USA and is only second (to forests surrounding Mexico City) to the worst deposition in North America (Miller and McBride 1999). The sequence of remote imagery was constructed from red, near-infrared, and thermal wavelengths at 5 km above the forest canopy. The yellow dot denotes the same location in each image. The forest stand is a mix of ponderosa pine, California black oak, white fir, incense cedar, and sugar pine. A dirt road and bare soil or dead herbaceous vegetation are indicated in fuchsia. On 7-20-01, the third year of a chronic drought, the first sign of bark beetle attack occurred on the site near the yellow dot (copper-colored trees). On 5-27-03, after 3 years of chronic drought and an acute drought (2002), additional points of bark beetle infection were observed, including some possible drought-induced mortality (more scattered, individual dead ponderosa pine near the bottom of the image). On 9-18-03, after the drought years plus the wet year (2003), tree mortality continued to accelerate, primarily in ponderosa pine, white fir, and some sugar pine. On 7-10-04, tree mortality was further increased (purple and fuchsia-colored areas) after the Old Fire swept through the area on 10-12-03. At the stand level, the tree mortality for the first three dates, respectively, was 0 percent, 5 percent, and 42 percent. Observed mortality (estimated from proportion of pixels) declined to 32 percent in the 7-10-04 image from needle loss on standing dead trees.

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Evaluating Wildland Fire Danger and Prioritizing Vegetation and Fuels Treatments¹

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Abstract

We present a prototype decision support system for evaluating wildland fire danger and prioritizing subwatersheds for vegetation and fuels treatment. We demonstrate the use of the system with an example from the Rocky Mountain region in the State of Utah, which represents a planning area of about 4.8 million ha and encompasses 575 complete subwatersheds. In a logic model, we evaluate fire danger as a function of three primary topics: fire hazard, fire behavior, and ignition risk. Each primary topic has secondary topics under which data are evaluated. The logic model shows the state of each evaluated landscape with respect to fire danger. In a decision model, we place summarized fire danger conditions of each evaluated landscape in the context of the amount of associated wildland-urban interface (WUI). The logic and decision models are executed in EMDS, a decision-support system that operates in ArcGIS. We show that a decision criterion such as relationship to WUI can significantly influence the outcome of a decision to determine treatment priorities. For example, we show that subwatersheds that were in relatively poor condition with respect to fire hazard, behavior, and ignition risk may not be the best candidates for treatment. Additional strategic or logistical factors such as proximity to population centers, presence of endangered species, slope steepness, and road access all might be taken into account in selection

of specific watersheds within a management area for treatment. Thus, the ecological status of each ecosystem can be placed in one or more social and human values contexts to further inform decisionmaking. The application introduced here can be readily expanded to support strategic planning at national and regional scales and tactical planning at local scales.

Keywords: Decision support, EMDS, fire behavior, fire danger, fire hazard, forest restoration, ignition risk, landscape evaluation, monitoring, NetWeaver, wildland-urban interface.

Introduction

Wildland fuels have accumulated in many western forests of the United States for at least the past 70 years owing to 20th century settlement and management activities (Agee 1998, Hessburg and Agee 2003), and, to some extent, changing climatic conditions (Burkett and others 2005, Schoennagel and others 2004). As demonstrated by recent wildland fires, added fuels are fostering more intense wildfires that are more difficult to contain and control. Consequently, valuable property and natural resources have been destroyed, costs of fire management have escalated, fire-dependent forest ecosystems have deteriorated, and risks to human life and property continue to escalate (U.S. GAO 2002, 2003, 2004).

Historically, fires of varying size, frequency, and intensity maintained spatial patterns of forest vegetation, as well as temporal variation in those patterns (Agee 2003, Hessburg and others 2005, Schoennagel and others 2004, Turner 1989). In fact, many agents interacted to shape vegetation patterns and their spatio-temporal variation, including forest insect outbreaks, forest diseases, fires, weather and climatic events, and intentional aboriginal burning (Hessburg and others 2005, Whitlock and Knox 2002). Their interactions resulted in characteristic landscape patterns and caused variation in forest structural attributes, species composition, and habitats that resonated with the dominant disturbance

¹This document also appears in 2007 Forest Ecology and Management. 247(1–3): 1–17.

processes. Patterns of forest vegetation were directly linked with the processes that created and maintained them (Hessburg and others 2005, Pickett and White 1985, Turner and others 2001).

Circumstances are quite different today. Patterns and processes are still tightly linked, but not as before. Human influences have created anomalous vegetation patterns, and these patterns support fire, insect, and disease processes that display uncharacteristic duration, spatial extent, and intensity (Ferry and others 1995, Hessburg and others 2005, Kolb and others 1998). For example, 20th century fire suppression and prevention programs significantly reduced fire frequency in many dry mixed coniferous forests. Contemporary wildland fires are now larger and more intense on average than those of the prior 2 or 3 centuries (Healthy Forests Restoration Act 2003; (U.S. GAO 2002, 2003, 2004, and references therein). In short, settlement and management activities have altered spatial patterns of forest structure, composition, snags, and down wood at patch to province scales. As a result, significant changes in fire frequency, severity, and spatial extent are linked to changes in forest vegetation patterns at patch to province scales (Agee 1998, 2003, Ferry and others 1995, Hessburg and Agee 2003).

Here, we present a decision-support system for evaluating wildland fire danger and prioritizing subwatersheds for vegetation and fuels treatment. In our descriptions, we adopt the nomenclature of the National Wildfire Coordinating Group (NWCG 1996, 2005) and Hardy (2005). The decision-support system consists of a logic model and a decision model. In the logic model, we evaluate danger as a function of three primary topics: fire hazard, fire behavior, and risk of ignition. Each primary topic has secondary topics under which data are evaluated. The logic model shows the state of each evaluated landscape with respect to fire danger. In the decision model, we place the fire danger summary conditions of each evaluated landscape in the context of the amount of associated wildland-urban interface (WUI). The logic and decision models are executed in EMDS (Reynolds and others 2003), a decision-support system that operates in ArcGIS. We show that a decision criterion such as relationship to WUI can significantly

influence the outcome of a decision to determine treatment priorities. We demonstrate use of the system with an example from the Rocky Mountain region in the State of Utah, which represents a planning area of about 4.8 million ha and encompasses 575 complete subwatersheds. We discuss considerations for extending the application to support strategic planning at national and regional scales and tactical planning at local scales.

This decision-support system is comparable in some aspects to the National Fire Danger Rating System (NFDRS) (Burgan 1988, Deeming and others 1977), but there are important differences and advances, too. For example, the NFDRS summarizes fire danger information pertaining to fire hazard, fire behavior, and ignition risk, the primary topics of fire danger, at a regional scale using annual weather and forest conditions information. The fire danger variables computed by FIREHARM and used in this application reflect a broader set, are computed at a stand or patch scale, and summarized to subwatersheds, and the variables are computed as probabilities of exceeding a severe fire threshold using 18 years rather than a single year of data.

Methods

Study Area

We selected one map zone as a proving ground for our modeling approach, but these methods could be applied to any and all United States map zones. Map zones were developed in the United States by the Earth Resources Observation and Science (EROS) Data Center (<http://www.nationalmap.gov>). They are broad biophysical land units represented by similar surface landforms, land cover conditions, and natural resources; there are 66 in the continental United States (Figure 1). Map zone 16 falls almost entirely within the State of Utah. Within this study area, we evaluated wildland fire danger for the 575 subwatersheds that were entirely contained within map zone 16 (Figure 2). The average size of subwatersheds was 8,300 ha, and size ranged from 2,800 to 18,000 ha. For reference, a subwatershed represents the 6th level in the watershed hierarchy of the U.S. Geological Survey (Seaber and others 1987).

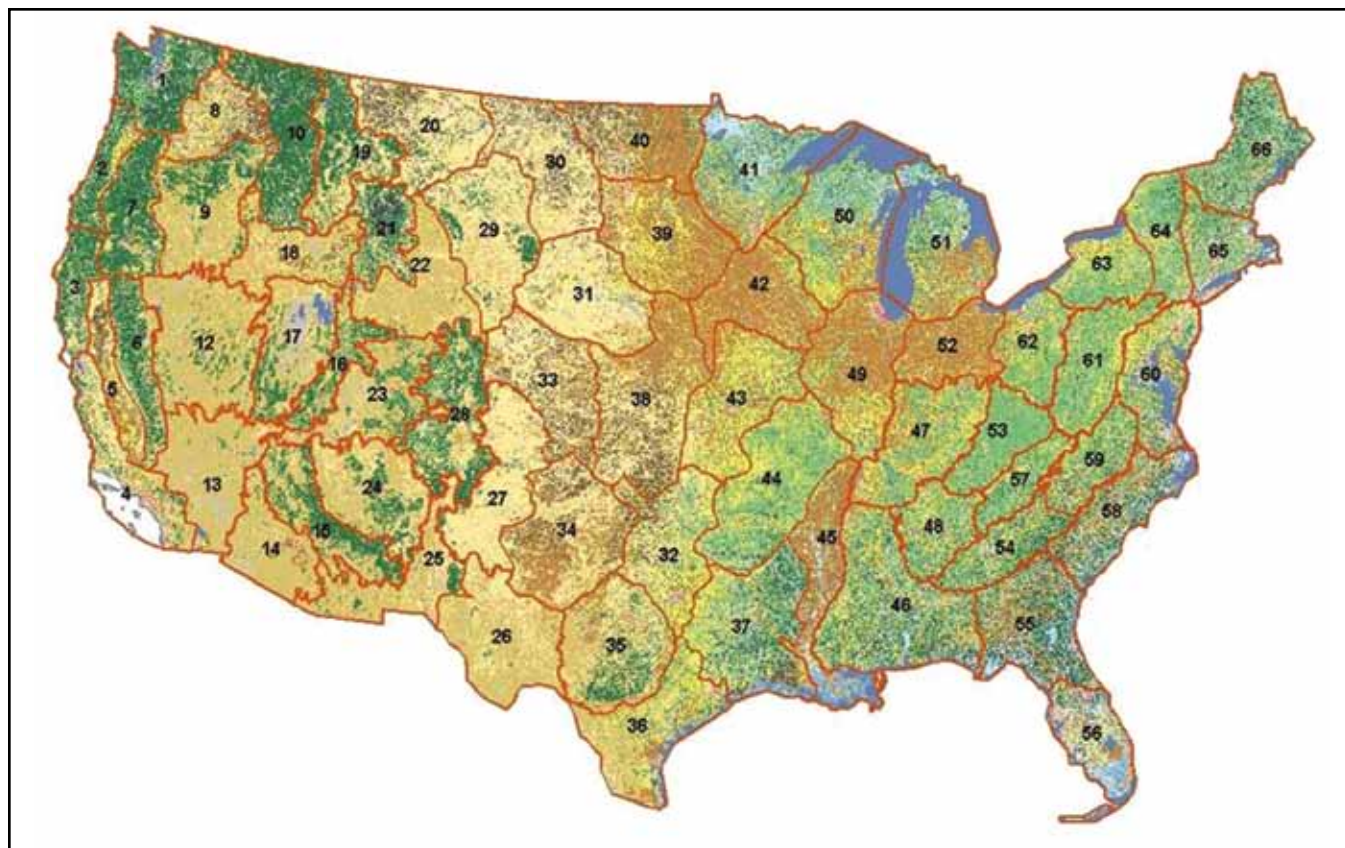


Figure 1—Map zones of the United States from the Earth Surface Resources and Science (EROS) Data Center. There are 66 map zones in the continental United States. The study area is map zone 16.

Data Sources

Most spatial data used in this study came from the LANDFIRE prototype project mapping effort. The LANDFIRE project created spatial data layers of topography, biophysical environments, vegetation, and fuels at 30-m resolution for two map zones in the Rocky Mountains (map zones 16 and 19). All layers were available at the <http://www.landfire.gov> Web site.

The fuels layers used in this study included two surface fuel classifications: (1) the 13 fire behavior fuel models (FBFM) of Albini (1976), defined by Anderson (1982), and mapped using methods described by Keane and others (1998, 2000, 2007); and (2) the default fuel characterization classes defined in the Fuel Characterization Classification System (FCCS) described by Sandberg and others (2001) (<http://www.fs.fed.us/pnw/fera>) and mapped using methods described by Keane and others (2007). The FBFMs, which do not represent actual surface fuels, provided an indication

of the expected surface fire behavior whereas the FCCS classes indicated the characteristics of the actual surface fuelbed, information useful for fire effects simulation (Ottmar and others 2004). In the next update of our fire danger model, we will incorporate the expanded set of 40 recently derived fire behavior fuel models of Scott and Burgan (2005). Note that when we refer to “fire behavior” we are referring to the physical characteristics of the combustion process (Rothermel 1972). When we refer to “fire effects” we are referring to the direct and indirect consequences of the combustion process (DeBano and others 1998).

The canopy fuels layers used were the LANDFIRE canopy bulk density and canopy base-height layers. Canopy bulk density (CBD) represents the mass of available canopy fuel per unit volume of canopy in a stand (Scott and Reinhardt 2002), and it is defined as the dry weight of available canopy fuel per unit volume of the canopy including the spaces between the tree crowns (Scott and



Figure 2—Subwatersheds of map zone 16 in State of Utah, U.S.A. The average size of subwatersheds was about 8,274 ha (min 5,000 ha, max 10,000 ha). A subwatershed represents the 6th level in the established U.S. Geological Survey watershed hierarchy (Seaber and others 1987).

Reinhardt 2001). Canopy base height (CBH) represents the level above the ground at which there is enough aerial fuel to carry the fire into the canopy, and it is defined as the height from the ground to the bottom of the live canopy (Scott and Reinhardt 2001) but may also include dense, dead crown material that can carry a fire. These two map layers were developed for the forested lands of map zone 16 using a predictive landscape modeling approach that integrated remotely sensed data, biophysical gradients, and field reference data (Keane and others 2007). The canopy fuel characteristics were calculated for numerous plots distributed throughout the map zone using the FUELCALC model (Scott and Reinhardt 2001), and each plot was described from a set of predictor variables computed and mapped

specifically for the LANDFIRE project. The predictor variables were related to CBD and CBH using a classification and regression tree (CART) approach.

Fire behavior was simulated with these surface and canopy fuel layers assuming 90th percentile weather conditions using the FIREHARM (Keane and others 2004) program to estimate surface fire spread rate, flame length, and fireline intensity based on the Rothermel (1972) fire spread model and crown fire intensity and spread based on the Rothermel (1991) and the Scott (1999) crown fire algorithms. FIREHARM is a computer program that calculates four fire behavior variables (fireline intensity, spread rate, flame length, crown fire potential), five fire danger variables (spread component, burning index, energy release component, Keetch-Byram drought index [Burgan 1993], ignition component), and five fire effects variables (smoke emissions, fuel consumption, soil heating, tree mortality, scorch height) for each day across an 18-year climate record (6,574 days), and for every polygon in a user-specified landscape. Daily values across the 18-year period can be used to estimate probabilities that fire behavior, fire danger, or fire effects variables may exceed important thresholds. These probabilities can be mapped onto the landscape in a geographic information system (GIS), and maps can be used to prioritize, plan, and implement fuel or fire treatments.

In addition, LANDFIRE provided a fire regime condition class (FRCC) digital map created by simulating historical landscape conditions and comparing these simulations with current vegetation conditions derived from satellite images. FRCC is an ordinal index with three categories that describe how far the current landscape has departed from presettlement-era conditions (Hann 2004) (see <http://www.frcc.gov> for complete details).

Several other data layers were used to derive ignition risk. Relative plant greenness was estimated from an AVHRR image from June 1, 2004 (Burgan and Hartford 1993). These data were obtained from the USDA Forest Service, Rocky Mountain Research Station, Missoula Fire Sciences Laboratory. The effects of long-term drought were estimated from Palmer Drought Severity Index data obtained from the National Climate Data Center. Available PDSI data represented a span of 20 years (1971-1990), and

data were derived from a 2.5-degree continental scale grid of PDSI reconstructed by Cook and others (2004). Lightning strike data were obtained from the National Lightning Detection Network (Vaisala 2010).

Broad Outline

We evaluate relative fire danger in individual subwatersheds of an entire map zone. We show how evidence for fire danger can be modeled as a logic-based discourse in a decision-support system to support national, regional, and local landscape analysis and planning. Results of evaluations are expressed in terms of evidence for low wildfire danger in each subwatershed. This information is used subsequently in a decision model to prioritize subwatersheds for treatment, considering additional logistical information.

Implementation Steps

Under the fire hazard topic (Table 1), we estimated for each elementary topic (lowest level in the model where data are evaluated) the percentage area and degree of aggregation of observations exceeding a specified threshold value using spatial data layers provided by the LANDFIRE project and a spatial analysis program (FRAGSTATS, McGarigal and others 2002, Table 2). For each elementary topic under fire behavior and ignition risk, we estimated the probability that conditions within a given watershed exceeded a specified threshold value based on spatial layers of fire spread rate and intensity generated by the FIREHARM model using the Rothermel (1972) spread model. We constructed a logic model within the EMDS modeling system to show how all elementary topics contributed to an evaluation of fire danger. We evaluated evidence for low wildfire danger within watersheds of a map zone to provide an ecological basis for determining treatment priority. A decision analysis was then run in a separate but related decision model to incorporate ecological and logistical considerations for planning fuels treatment across the study area.

Logic Model Design

We graphically designed the logic model for evaluating the relative danger of wildland fire (hereafter, fire danger) with the NetWeaver Developer (Rules of Thumb, Inc., North

East, PA) modeling system. Note that the use of trade or firm names in this publication is for reader information and does not imply endorsement by the U.S. Department of Agriculture of any product or service. We present the formal logic specification both as a topic outline for readability and compactness (Table 1) and as a dendrogram (Figure 3). Each topic in a NetWeaver model represents a topic for which a premise or proposition is evaluated. For example, the overall fire danger topic, representing the top level in the model, evaluates the proposition that wildland fire danger is low (Table 1, Figure 3). All other propositions in the model similarly take the null form; i.e., the test for all topics is always for a low condition.

The complete evaluation of fire danger depends on three primary topics—fire hazard, fire behavior, and ignition risk—each of which incrementally contribute to the evaluation of fire danger, as indicated by the union operator (Table 1). Moreover, because the union operator specifies that premises incrementally contribute to the proposition of their parent topic, low strength of evidence for one topic can be compensated by strong evidence from others. Notice that if the fire danger topic is thought of as testing a conclusion, then the three topics on which it depends can be thought of as its logical premises. Similarly, each of the three topics under fire danger has its own logic specification that includes a set of secondary topics or premises. The full logic structure (Table 1), considered in its entirety, constitutes what we referred to earlier as the logical discourse. Note that this logic model represents one of many possible logical configurations, and the current configuration is readily adapted. Any of the primary and secondary topics may be modified, and topics may be added or removed with relative ease. Likewise, thresholds of elementary topics (discussed below) can be modified to fit customized or evolving evaluations as a function of adaptation and learning.

Primary Topic—Fire Hazard—

Evaluation of fire hazard (Table 1, Figure 3) depends on the union of topics addressing surface fuels, canopy fuels, and fire regime condition class, each of which depends on two additional elementary topics that directly evaluate data (Tables 1, 2). Evaluation of each elementary topic under

Table 1—Logic outline for evaluation of wildfire danger^a

Model topic	Primary topic ^b	Secondary topic	Elementary topic	Proposition ^c (stated in the null form)	Data inputs ^d
Fire danger (union ^e)	Fire hazard (union)	Surface fuels (union)		Danger of severe wildfire is low	
				Fuel conditions do not support severe wildfire	
			Fire behavior fuel model	Condition of surface fuels is not conducive to severe wildfire	
			Fuel characterization class	Expected fire behavior is not severe	FBFMarea, FBFMaggregation
		Canopy fuels (union)		Observed fuel load classes are not conducive to severe wildfire	FCCarea, FCCaggregation
Fire behavior (union)	Fire regime condition class	Canopy bulk density	Canopy bulk density	Condition of canopy fuels is not conducive to severe wildfire	
			Canopy base height	Canopy bulk density is not conducive to severe wildfire	CBDarea, CBDaggregation
				Canopy base height is not conducive to severe wildfire	CBHarea, CBHaggregation
				Fire regime condition class is not conducive to severe wildfire	FRCCarea, FRCCaggregation
				Expected fire behavior associated with wildfire is relatively benign or low impact	
Ignition risk (union)			Spread rate	Likelihood of high spread rate of surface fire is low	Spread rate
			Flame length	Likelihood of high flame length is low	Flame length
			Fireline intensity	Likelihood of high fire line intensity is low	Fireline intensity
			Crown fire potential	Likelihood of high crown fire spread potential is low	Crownfire potential
				Likelihood of wildfire ignition is low	
			Palmer drought severity index	Likelihood of long-term drought is low	Palmer index
			Keetch-Byram drought index	Likelihood of short-term drought is low	Keetch-Byram index
			AVHRR NDVI ^f	Relative plant greenness for the subwatershed is high	AVHRR-NDVI
			Lightning strike	Relative lightning strikes in the subwatershed are low	Lightning strike

^a The logic outline specifies how data related to wildfire danger (Table 2) are interpreted in NetWeaver[®], a logic modeling system.^b The level of each primary, secondary, and elementary topic in the outline is indicated. The overall topic of the model is wildfire danger. Evaluation of overall fire danger depends directly on the evaluation of the primary topics—fire hazard, fire behavior, and ignition risk. Terms in parentheses following a topic indicate the logic operator used to evaluate the propositions under a topic. For example, fire danger is evaluated as the union of hazard, behavior, and ignition risk. Subtopics shown under “Elementary topics evaluated” indicate the elementary topics occurring at lowest level in the logic model where data are evaluated (Table 2).^c Each proposition evaluates a set of premises (see footnote b) or data relative to a specific landscape unit. For this analysis, subwatersheds were the landscape units.^d Definitions of data items are presented in Table 2.^e The union operator treats the premises of a proposition as factors that incrementally contribute to the proposition.^f Advanced very high resolution radiometer (AVHRR) is used in producing normalized difference vegetation index (NDVI).

Table 2—Definition of data inputs evaluated by elementary topic, data source, and reference conditions for each datum^a

Datum	Definition	Data source	Reference conditions ^b	
			No evidence	Full
AVHRR-NDVI	AVHRR-NDVI ^d relative greenness value on June 1, 2004	Missoula Fire Lab ^e	0.00	1.00
CBDaggregation ^f	Aggregation index for canopy bulk density > 0.15 kg/m ³	LANDFIRE ^g (derived)	93.02	75.97
CBDarea	Likelihood ^c of canopy bulk density > 0.15 kg/m ³	LANDFIRE	0.79	0.29
CBHaggregation	Aggregation index for canopy base height < 3.1 m	LANDFIRE (derived)	72.99	36.92
CBHarea	Likelihood of canopy base height < 3.1 m	LANDFIRE	0.38	0.04
crownfirePotential	Likelihood of index for crown fire potential > 7	FIREHARM ^e (derived)	1.00	0.89
FBFMaggregation	Aggregation index for fire behavior fuel model > 9 ^h	LANDFIRE (derived)	35.83	3.05
FBFMarca	Likelihood of value for fire behavior fuel model > 9	LANDFIRE	1.00	0.02
FirelineIntensity	Likelihood of fireline intensity > 400 kW/m	FIREHARM	0.97	0.59
FlameLength	Likelihood of flame length > 1.2 m FIREHARM		0.92	0.09
FCCaggregation	Aggregation index for fuel loading > 56 Mg/ha ⁱ	FCCS ^j (derived)	89.73	33.00
FCCarea	Likelihood of fuel loading > 56 Mg/ha	FCCS	0.80	0.03
FRCCaggregation	Aggregation index for fire regime condition class ^j	FIREHARM (derived)	99.50	97.76
FRCCarea	Likelihood of fire regime condition class > 2	FIREHARM	0.28	0.01
Keetch-byramIndex	Likelihood of a Keetch-Byram drought index value > 400	FIREHARM	0.84	0.46
LightningStrike	Probability of cloud-to-ground lightning strike indexed by the maximum value ^j	NLDN ^m	1.00	0.00
PalmerIndex	Likelihood of summer Palmer drought severity index value < -2	NCDC ^o	37.00	0.00
SpreadRate	Likelihood of a wildfire spread rate > 8.0 kph	FIREHARM	1.00	0.89

^a Data items in this table correspond to data listed for elementary topics evaluated in Table 1. Each datum represents an observation for a subwatershed, the unit of analysis in this study.

^b Reference conditions for no evidence and full evidence define critical values for which the fuzzy membership function of the associated elementary topic (Table 1) indicate no support and full support, respectively, for the proposition. The range of the reference conditions is the median 80-percent range of data for the variable of interest. An observed value for the associated datum that falls in the open interval defined by the two reference conditions maps to partial support for the proposition based on linear interpolation. Data with the suffix, Area, are not evaluated with respect to reference conditions; however, they are compared to minimum and maximum conditions within conditional tests to determine the logic for evaluation of elementary topics (see text for additional explanation).

^c Each likelihood was estimated as the proportion of raster grid cells in the subwatershed area that exceeded the specified threshold for the attribute. All likelihoods were estimated from 30-m resolution data, except those for LightningStrike, and PalmerIndex, which were estimated from available 1-km resolution data.

^d The normalized difference vegetation index (NDVI), obtained from NOAA-11, AVHRR satellite image, represents relative greenness, and, in this usage, the effect of apparent moisture level on vegetation drying or curing. For further details, see Borgan and Hartford 1993, White and others 1997, and <http://www.fs.fed.us/land/wfas/wfas11.html>.

^e Obtained from the USDA Forest Service, Rocky Mountain Research Station, Missoula Fire Laboratory, Missoula, MT, rbartlette@fs.fed.us.

^f An aggregation index was computed with FRAGSTATS (McGarigal and others 1995) for each attribute of hazard (see also Table 1) by reclassifying data in the 30-m resolution raster grid for the attribute to 0 (attribute ≤ threshold) or 1 (attribute > threshold).

^g LANDFIRE (<http://www.landfire.gov/>) is a multipartner wildland fire, ecosystem, and fuel mapping project, one of whose partners is the Missoula Fire Sciences Laboratory of the USDA Forest Service, Rocky Mountain Research Station, Missoula, MT, who provided the data. Data sources labeled "LANDFIRE" indicate base data layers provided by the LANDFIRE project. Data sources labeled "FIREHARM" indicate data derived from base LANDFIRE layers by the FIREHARM model (Keane and others 2004) of the LANDFIRE project. With the exception of the data source for crownfirePotential, data sources labeled (derived) indicate an aggregation statistic that we derived from the LANDFIRE base layers with the FRAGSTATS (McGarigal and others 1995) spatial analysis package. In the case of crownfirePotential, (derived) indicates a composite index that we developed from FIREHARM crown fire ignition and crown fire spread outputs.

^h Fire behavior fuel models represent 13 distinct distributions of fuel loadings found among surface fuel components (live and dead), fuel size classes, and fuel types. The fuel models are described by the most common fire carrying fuel type (grass, brush, timber litter, or slash), fuel loading, and surface area-to-volume ratio by size class and component, fuelbed depth, and moisture of extinction. Further detail about the original fire behavior fuel models can be found in Albini 1976, Anderson 1982, and Rothermel 1972 and 1983.

Table 2—Definition of data inputs evaluated by elementary topic, data source, and reference conditions for each datum^a (continued)

^j Fuel Characteristic Class System (Sandberg and others 2001, http://www.fs.fed.us/pnw/fera/nfp/haze/FCCS-lower48.zip)
^k Fire regime condition class is a qualitative measure of departure from historical vegetation and fire regime conditions (Schmidt and others 2002).
^k In contrast to the Palmer drought severity index, the Keetch-Byram drought index represents the short-term effects of precipitation and temperature on duff, litter, and soil drying in the top 20 cm. An index value of 400 corresponds to a deficit of 10 cm of water in the top 20 cm.
^l The lightning strike probability is based on actual strikes triangulated and recorded over 15 years (1990 to 2004, Schmidt and others 2002).
^m Data were obtained from the National Lightning Detection Network (NLDN, http://ghrc.msfc.nasa.gov/).
ⁿ The Palmer drought severity index is used to characterize effects of long-term drought. An index value of -2 corresponds to moderate drought conditions. Continuous maps of PDSI for the continental United States were interpolated by Cook and others (2004) based on their reconstructions of drought at grid points on a 2.5-degree grid of the continent.
^o Web site for the National Climate Data Center (NCDC), NOAA. (http://www.ncdc.noaa.gov/paleo/newpdsi.html).

hazard involved two class metrics computed by the FRAG-STATS program: (1) the proportion of subwatershed area exceeding a specified threshold value, and (2) an index that shows the degree of spatial aggregation of observed values exceeding the threshold value. Threshold values were based on the fire literature, and, where literature values were lacking, were based on our judgment. Use of the metrics to evaluate the elementary topic for canopy bulk density (CBD) is presented below as an example; methods for evaluation of each of the other elementary topics under hazard are analogous.

Within the elementary topic for CBD, the logic first tests the value of CBDarea; the percentage of the subwatershed area with CBD exceeding a threshold value of 0.15 kg/m³ (Table 2):

- If CBDarea is < 0.29, (i.e., < 29 percent of the subwatershed area exhibits CBD values > 0.15 kg/m³), then evidence for low CBD is fully satisfied, else
- If CBDarea is > 0.79, (i.e., > 79 percent of the subwatershed area exhibits CBD values > 0.15 kg/m³), then there is no evidence for low CBD, else
- Evidence for low CBD is evaluated as a function of CBDaggregation.

The value 0.29 represents the lower bound of the median 80-percent range for the set of all CBDarea data in map zone 16. The value 0.79 represents the upper bound of the median 80-percent range (Table 2). If the last condition above was satisfied, then we tested the observed value for CBDaggregation against a fuzzy membership function (Figure 4). This was done to determine the strength of evidence for a low degree of aggregation of high CBD values (i.e., values of CBD exceeding the threshold value of 0.15 kg/m³) relative to a set of reference conditions that defined the median 80-percent range of the CBDaggregation data from the set of all subwatersheds (Table 2). Notice that each elementary topic (Table 2) is similarly evaluated against the median 80-percent range of its associated datum, hence our characterization of fire danger as relative.

- If CBDaggregation is ≤ 76, (i.e., ≤ 76 percent of the maximum value of aggregation), then evidence for low aggregation of high CBD values is fully satisfied, else

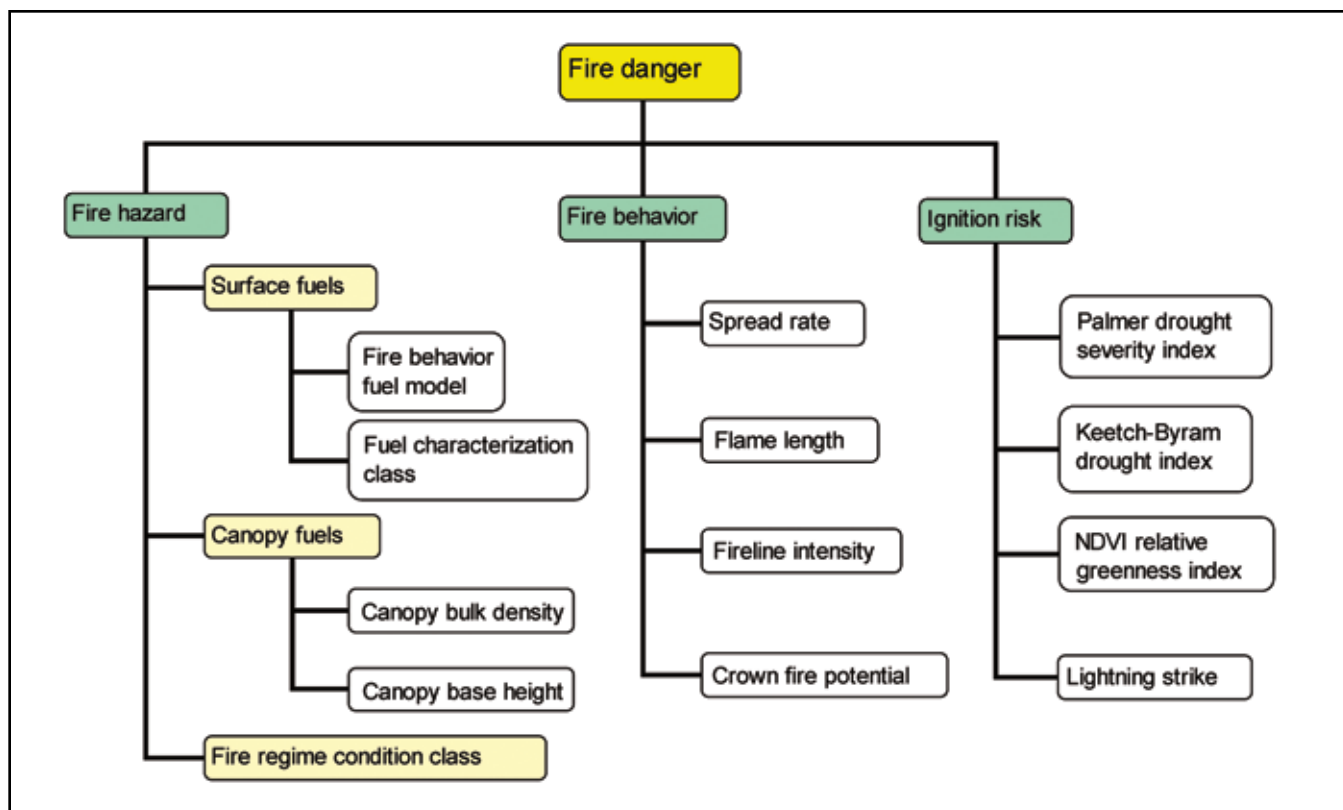


Figure 3—Dendrogram showing how the overall fire hazard topic is organized and evaluated. The complete evaluation of fire hazard is made up of three parts—evaluation of fire hazard, fire behavior, and ignition risk, which are primary topics. Under each of these three primary topics are secondary and elementary topics. Under fire hazard are the topics surface fuels, canopy fuels, and fire regime. Under behavior are the elementary topics spread rate, flame length, fireline intensity, and crown fire potential. Under ignition risk are the secondary topics fire weather and ignition potential.

- If CBDaggregation is ≥ 93 , (i.e., ≥ 93 percent of the maximum value of aggregation), then there is no evidence for low aggregation of high CBD values, else
- Observed values of CBDaggregation fall within the open interval (76, 93), and evaluate to partial support for the proposition, based on a linear interpolation between 76 and 93. The open interval (76, 93) represents the median 80-percent range of the data.

Primary Topic—Fire Behavior—

Evaluation of fire behavior depends on the union of topics addressing spread rate, flame length, fireline intensity, and crown fire potential (Table 1), each of which is an elementary topic that directly evaluates data (Tables 1, 2). The spread rate topic evaluates the proposition that likelihood of spread rate of surface fire > 8.0 kph within the subwatershed is low. The flame length topic evaluates the proposition

that likelihood of flame length > 1.2 m within the watershed is low. The fireline intensity topic evaluates the proposition that likelihood of fireline intensity > 400 kW/m within the watershed is low. The crown fire potential topic evaluates the proposition that likelihood of crown fire spread potential > 7 within the watershed is low. This last metric is an index based on crown fire ignition and crown fire spread potentials (Keane and others 2004) and represents the ratio of crown fire behavior to surface fire behavior based on Rothermel (1972, 1991) surface and crown fire algorithms.

None of the fire behavior elementary topics are entirely independent of the other topics; rather, one or more of these topics is used in the calculation of the others. For example, flame length influences the spread rate calculation, and fireline intensity influences flame length. In fact, fireline intensity is double weighted in our model because of the

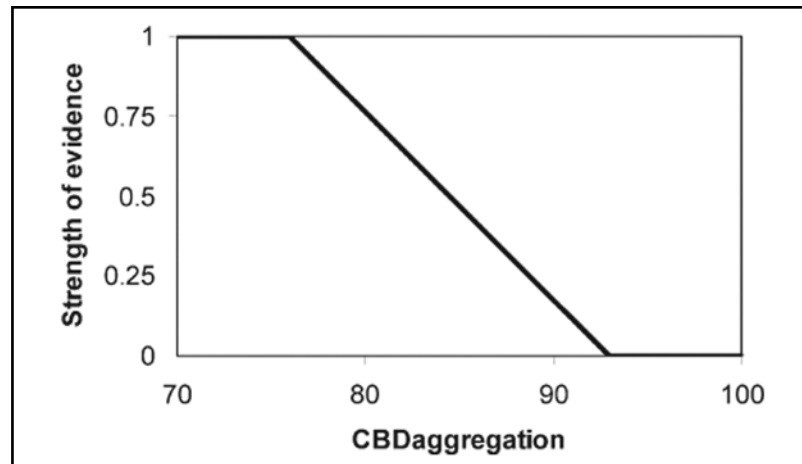


Figure 4—The fuzzy membership function used to evaluate strength of evidence for the proposition of low canopy bulk density. The proposition is fully satisfied when the observed value of CBDaggregation ≤ 76 , and there is no evidence for the proposition if CBDaggregation ≥ 93 (Table 2). Observed values of CBDaggregation that fall within the open interval (76, 93) evaluate to partial support for the proposition, based on linear interpolation between 76 and 93.

equivalence of flame length and fireline intensity (Chandler and others 1983). We used both in the model because intensity relates best to fire effects, and flame length is easily observed and often asked for. Each selected elementary topic is used here to provide a more comprehensive picture of expected fire behavior. Whereas complete independence among the topics would be desirable, there is no set of fire behavior attributes with such independence, and there is also no independent set that provides a comprehensive picture of expected fire behavior.

Primary Topic—Ignition Risk—

Evaluation of ignition risk depends on the union of four elementary topics—Palmer drought severity index (Palmer 1965), the Keetch-Byram drought index (Keane and others 2004), the advanced very high resolution radiometer normalized difference vegetation index (AVHRR-NDVI) relative greenness index (Keane and others 2004), and lightning strike probability (Tables 1, 2). First, the probability of a summer Palmer drought severity index value < -2 is evaluated. A value of -2 corresponds to moderate drought in the Palmer rating system. This elementary topic is included because it allows consideration of the effects of long-term drought on vegetation and fuels. Second, the probability of a Keetch-Byram drought index (KBDI)

value > 400 is evaluated. The topic considers the short-term effects of precipitation and temperature on duff, litter, and soil moisture in the top 20 cm. An index value of 400 corresponds to a deficit of 10 cm of water in the top 20 cm; Burgan (1993) suggested that severe fire behavior often occurs when the KBDI exceeds this value.

The AVHRR-NDVI relative greenness value on Julian day 152 (June 1, 2004) is then considered as a topic that indirectly represents fuel condition by incorporating vegetation drying or curing in a measure of relative greenness. June 1 is used to represent the height of the growing season in the study area; the greenest values indicate lesser chance for fire ignition. Future versions of this modeling system would include dates to capture the span of the fire season of each unique map zone.

Finally, lightning strike probability is evaluated, which we base on actual strikes triangulated and recorded over 15 years (1990 to 2004). The probability of human-caused ignitions is also important but omitted in this implementation. We constructed a logic module for evaluating the likelihood of human-caused ignitions, but it is not implemented in this version because wall-to-wall human ignition density data were unavailable for map zone 16. In a future version, we will incorporate a direct evaluation based on recorded

human-ignition densities, or an indirect measure of likelihood involving road density maps and maps of human congregation sites.

Priorities for Fuels Treatment

A decision model for determining priorities of subwatersheds for fuels treatment was graphically designed with Criterium DecisionPlus (InfoHarvest, Inc., Seattle, WA), which uses both the analytic hierarchy process (AHP, Saaty 1992) and the Simple Multi-Attribute Rating Technique (SMART, Kamenetsky 1982) to support planning activities such as priority setting, alternative selection, and resource allocation. We used a decision model structure that was nearly identical to that of the logic model (Figure 3). In the context of decision models based on the AHP, the concept of topics is replaced by criteria. Thus, in the decision model for fuels treatment, the first level of the model contained the three criteria, fire hazard, wildfire behavior, and ignition risk. However, for purposes of setting treatment priorities for subwatersheds, we also added a fourth criterion, percentage of subwatershed area classified as wildland-urban interface (WUI), to illustrate expanding the scope of analysis to include additional logistical factors that can influence decisions about priorities. Note that numerous other criteria and subcriteria could be included to account for other logistical considerations that might influence decisions about treatment priorities.

Weights for each criterion at the first level of the decision model were derived from the standard pair-wise comparison procedure of the AHP (Saaty 1992) in which a decisionmaker is asked to judge the relative importance of one criterion versus each of the others. We provided the judgments on relative importance for our example application. Weights for sets of subcriteria under each criterion (the second level of the decision model) were derived in the same manner. For purposes of subsequent discussion, criteria at the lowest level of an AHP model are commonly referred to as attributes of a decision alternative, and these attributes correspond to the elementary topics of the logic model (Table 1).

A SMART utility function was specified for each attribute of a subwatershed, and this function represented

the mirror image of the fuzzy membership function of its corresponding elementary topic; i.e., the fuzzy parameters defining no support and full support (Table 2) were now used to define utility values of 1 (full utility) and 0 (no utility), respectively, on the SMART utility scale of [0, 1]. Note, however, that the WUI criterion is both a primary (first level) criterion of the decision model and an attribute of a subwatershed for which there is no corresponding elementary topic in the logic model. In this case, the critical values corresponding to full and no utility were separately specified as 67 and 0 percent, respectively, and represent the maximum and minimum of observed WUI percentages.

Analysis

Fire danger evaluation (Table 1) for all subwatersheds in the study area was performed with the NetWeaver logic engine (Miller and Saunders 2002) in EMDS (Reynolds and others 2003). Continuous data related to recent burns in map zone 16 were not available and were not implemented in this version of the fire danger model. This component should be added as data become available. Priority setting for fuels treatments among subwatersheds was performed with Priority Analyst, an engine for running Criterium DecisionPlus models in EMDS.

Results

We describe results in terms of the strength of evidence in support of the overarching proposition of low fire danger or of subordinate propositions under fire danger. Recall that all propositions take the null form; for example, low strength of evidence based on the underlying evaluation implies that the proposition of low fire danger has poor support.

Fire Danger

There were pronounced differences in fire danger between subwatersheds in the northern and southern portions of the study area (Figure 5). Support for the proposition of low fire danger was generally moderate in the north and low in the south, which also contained small pockets of very low support. Dangerous wildfire conditions were largely driven by conditions conducive to severe fire behavior. Figure 6 shows the partial products of the entire evaluation process;

from viewing this composite, it is possible to see the various contributions to overall fire danger. We summarize the results of the partial products immediately below.

Fire Hazard—

Throughout much of the northern half of map zone 16, evaluation of fire hazard showed moderate to full support for the proposition of low fire hazard. The outstanding exception was the northern peninsula of subwatersheds extending to the east, where most of the subwatersheds showed low support for the proposition (Figure 6). Likewise, in much of the northern half of the map zone, evaluation of fire regime condition class showed moderate to full support for the proposition of low departure of vegetation and fuel conditions from historical ranges. The southern half was mixed in its support but with a considerable number of subwatersheds showing low, very low, and no support.

The canopy fuels evaluation was composed of the partial evaluations of canopy bulk density and canopy base height. In general, the canopy fuels evaluation showed subwatersheds displaying conditions favorable to severe wildfire in both the northern and southern portions of the map zone. Evaluation of canopy base height showed conditions conducive to severe wildfire in the northern peninsula of subwatersheds extending to the east and especially in the southern subwatersheds. Evaluation of canopy bulk density showed conditions favorable to severe wildfire throughout the map zone, but most especially in the northern peninsula of subwatersheds extending to the east.

The surface fuels evaluation was composed of the partial evaluations of fire behavior fuel model and fuel loading. In general, the surface fuels evaluation showed subwatersheds displaying conditions favoring severe wildfire in both the northern and southern portions of the map zone, but most especially in the northern peninsula of subwatersheds extending to the east (Figure 6). Here, fuels were dominated by shrub types with grassland-savanna fuel types also common. Evaluation of fire behavior fuel model showed that with the exception of the northernmost peninsula of subwatersheds extending to the east, the northern half of the map zone showed moderate to full support for the proposition that expected fire behavior would be low. In the subwatersheds of the southeastern portion of the

map zone, the evaluation suggested that expected wildfire behavior would be severe. The evaluation of fuel characterization class showed highly mixed results throughout the map zone, with the exception of the northernmost peninsula of subwatersheds extending to the east where surface fuels were conducive to severe wildfire.

Fire Behavior—

The fire behavior evaluation consisted of the partial product evaluations of fire spread rate, flame length, fireline intensity, and crown fire potential (Table 1, Figure 6). Throughout the map zone, there was low to very low support for the proposition that expected wildfire behavior would be low.

The evaluation of wildfire spread rate showed that expected spread rate of surface fires would be high under 90th percentile conditions especially in the central and northern sectors. In the flame length evaluation, the likelihood of high flame length was high in the southern half of the map zone and in the southernmost peninsula of subwatersheds extending to the east in the northern sector. The evaluation of fireline intensity produced results similar to those of the flame length evaluation, and crown fire potential results were similar to those of the spread rate evaluation (Figure 6).

Ignition Risk—

The ignition risk evaluation consisted of the partial product evaluations of the Palmer drought severity index, the Keetch-Byram drought index, NDVI-relative greenness, and the relative number of cloud-to-ground lightning strikes. Throughout the southern half of the map zone, there was low support for the proposition that likelihood of wildfire ignition is low. In general, higher overall ignition risk was driven by the tendency for more severe annual summer drought and lower relative greenness in the southern portion of map zone 16, and moderate to full support for relatively fewer lightning strikes in the northern and central sectors of the map zone.

Priorities for Fuels Treatment

The map for fuels treatment priorities (Figure 7a) took into account most of the same factors as used to produce the map for fire danger and its components (Figure 6) but with

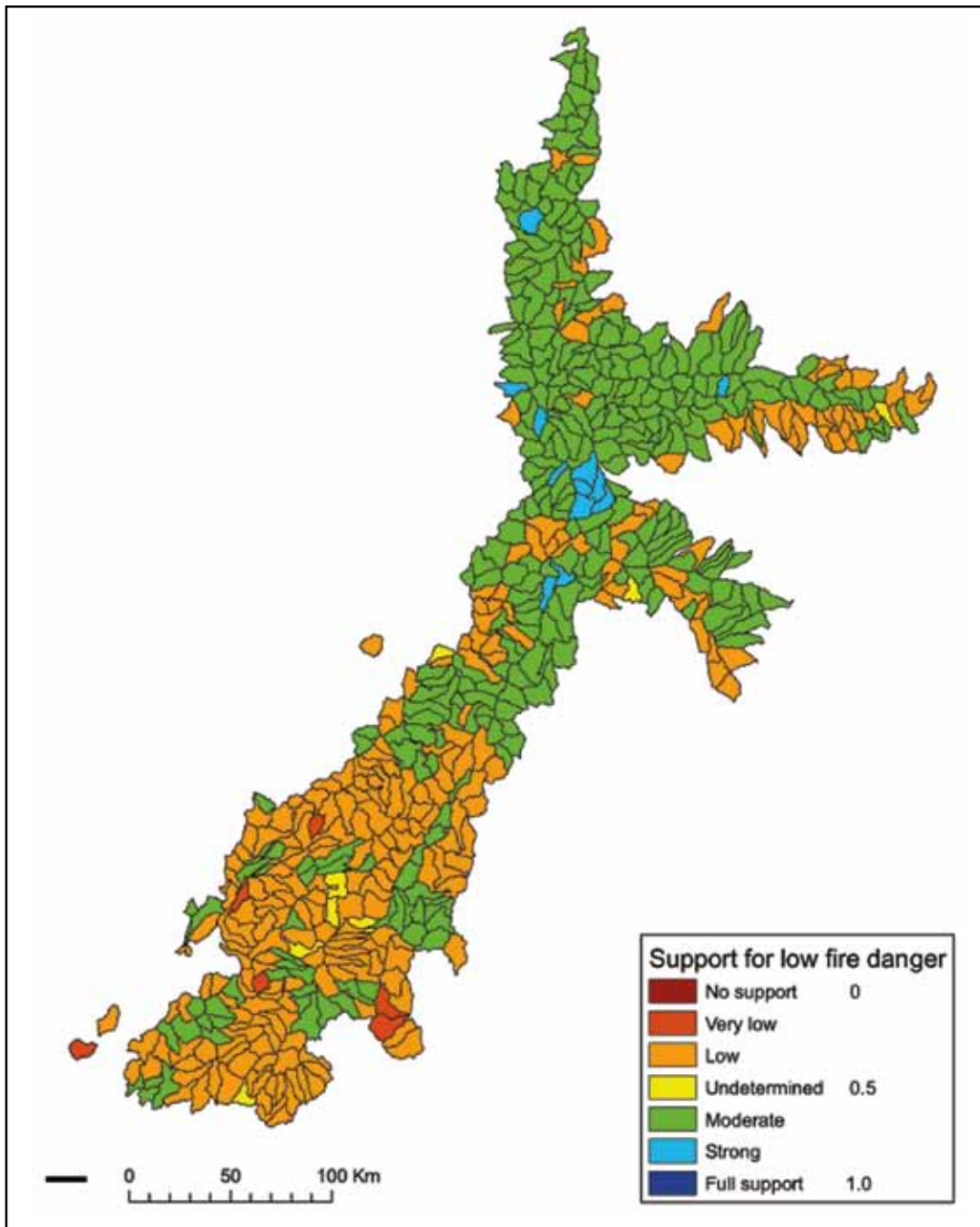


Figure 5—Evaluation of fire danger for map zone 16. Full support denotes complete support for the proposition of low fire danger.

weighting of criteria and subcriteria by a fire ecologist and also considering the influence of wildland-urban interface (Figure 7b). Ideally, when developing operational decision models for management, derivation of weights would be performed by a panel of managers and scientists. Indeed,

we emphasize the importance of such collaborative development in our conclusions. Here, for illustration purposes, and considering a simple decision model in which three of the four decision criteria are more technical in nature, development of weights by a fire ecologist seemed appropriate.

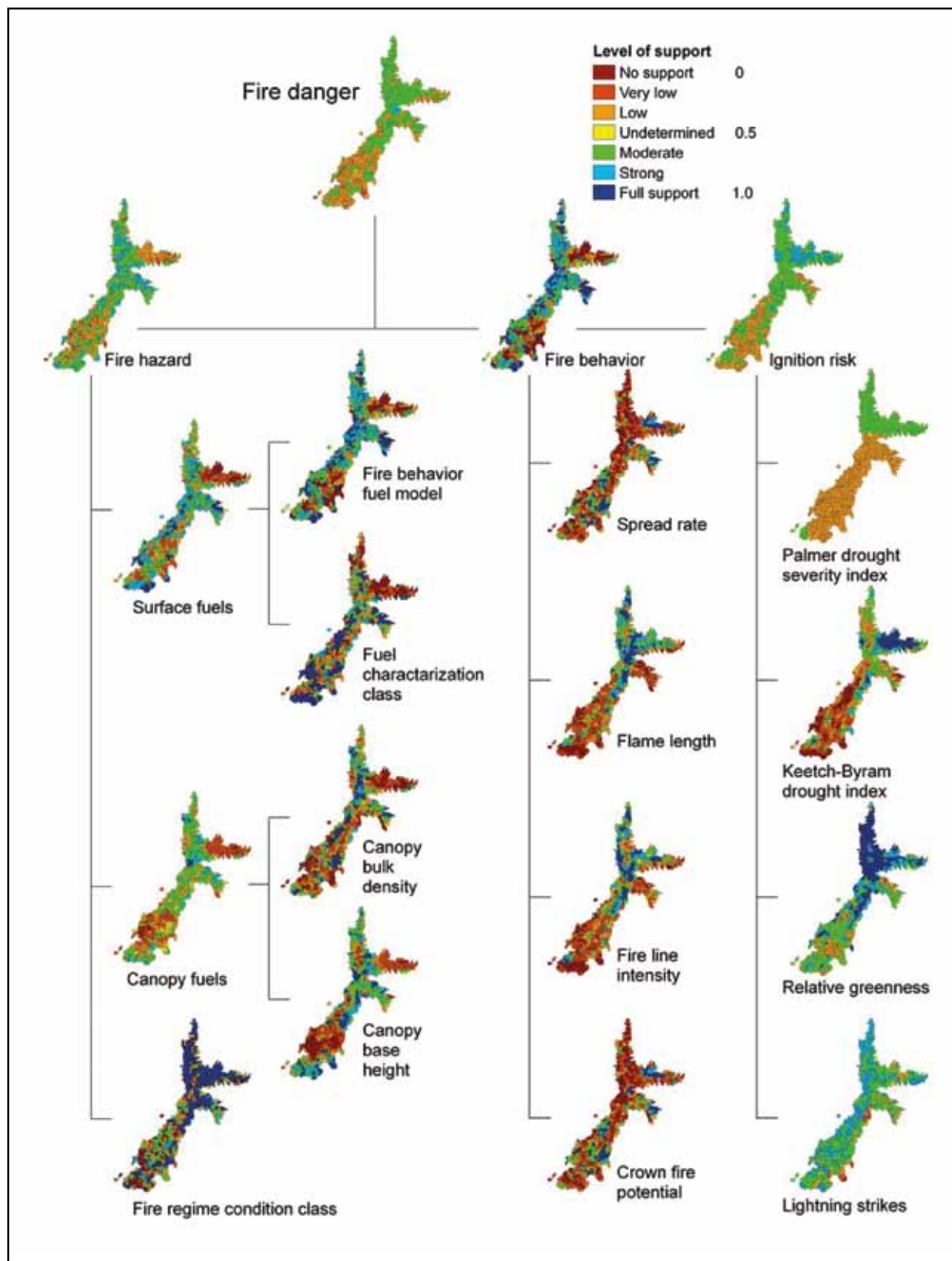


Figure 6—Composite of all partial product evaluations leading to the full evaluation of fire danger (Figure 5) for map zone 16.

The majority of subwatersheds with a priority rating of high or very high occurred in the southern two-thirds of the map zone (Figure 7a). The map of treatment priorities (Figure 7a) was strongly conditioned by the presence of wildland-urban interface in a subwatershed because of the emphasis placed on this criterion in the decision model. Normalized weights on primary criteria, derived from the pair-wise comparison process, were: wildland-urban interface, 0.50; fire behavior, 0.27; fire hazard, 0.15; and ignition risk, 0.08. A more detailed view of a small region in Figure 7 (Figure 8) shows the correspondence between wildland-urban interface and decision scores for fuels treatment for subwatersheds. Indicated subwatersheds with wildland-urban interface ≥ 10 percent (Figure 8b) were classified as very high priority (Figure 8a). Model output from the Priority Analyst (Figure 9) shows how the four primary decision criteria contribute to the overall decision score for a sampling of 10 subwatersheds. The three highest ranked subwatersheds (Figure 9) are also labeled in Figure 8b. Notice that the three highest ranked cases could be distinguished from the next seven cases by the level of influence of the wildland-urban interface. Furthermore, although the relative contribution of fire behavior was fairly consistent across the top 10 cases, the contributions of fire hazard and ignition risk were relatively low among the top three.

Discussion

The relative nature of our evaluation of fire danger has at least three important implications. First, the observed data value for each elementary topic in the logic model and for each attribute in the decision model was evaluated against reference conditions that were defined by the data themselves (Table 2). As a result, basic evaluations at the lowest level of each model were relatively objective. A second consequence of defining reference conditions in this manner was that the models were maximally sensitive to the data, thus assuring a high level of discrimination among outcomes over the set of subwatersheds in map zone 16. Finally, this method of deriving reference conditions means that the values used depended on the spatial extent of the assessment area. For example, reference conditions appropriate to an assessment of the entire Southwestern United

States would be at least somewhat broader than those for map zone 16 alone.

Evaluation outcomes and their underlying premises are affected by the scale of input data, whether they are at a relatively fine (e.g., 30- to 90-m pixels) patch scale or, in the case of the PDSI data used here, the continental scale. For map zone 16, evaluating the likelihood that a subwatershed experienced drought in the past 20 years was derived from a 2.5-degree continental-scale grid of reconstructed PDSI (Figure 10). Although there was wide variation in the probability of experiencing a long-term drought ($\text{PDSI} < -2$) for the continental United States (0 to 37 percent, Figure 10), map zone 16 exhibited a relatively narrow range of probabilities from 14 to 23 percent; or about 25 percent of the continental-scale variation. Thus, one might be concerned that the contribution of long-term drought to the evaluation of ignition risk at the scale of a map zone may be neutral, as if adding a constant. This was not the case. Figures 11a and 11b illustrate the influence of including continental-scale drought data in the map zone evaluation of fire danger. Differences can be seen among subwatersheds within evaluations of fire danger (Figure 11a) and ignition risk (Figure 11b) when comparing the same evaluations with and without PDSI. For map zone 16, PDSI does provide information on long-term drought that is beneficial to managers.

In addition to considering the scale of input data, the contributions of topics at each level to overall fire danger should be considered when interpreting an evaluation. For example, 10 subwatersheds that share a similar overall result for evaluation of fire danger (i.e., moderate support, 0.56, for the proposition of low fire danger) are shown in Figure 12, but they differed by evaluation result at the primary topic and lower levels. Use of the union operator in the design of the knowledge base made it possible for relatively high fire hazard within a subwatershed to be offset by relatively low predicted fire behavior in the event of a wildfire (e.g., see subwatershed 224, Figure 12). Similarly, subwatershed 339 (Figure 12) displayed evidence for low fire behavior but high ignition risk. An important strength of the logic model is that the full range of variability is expressed among subwatersheds at the level of an elementary topic, and each elementary topic contributes to evaluations of

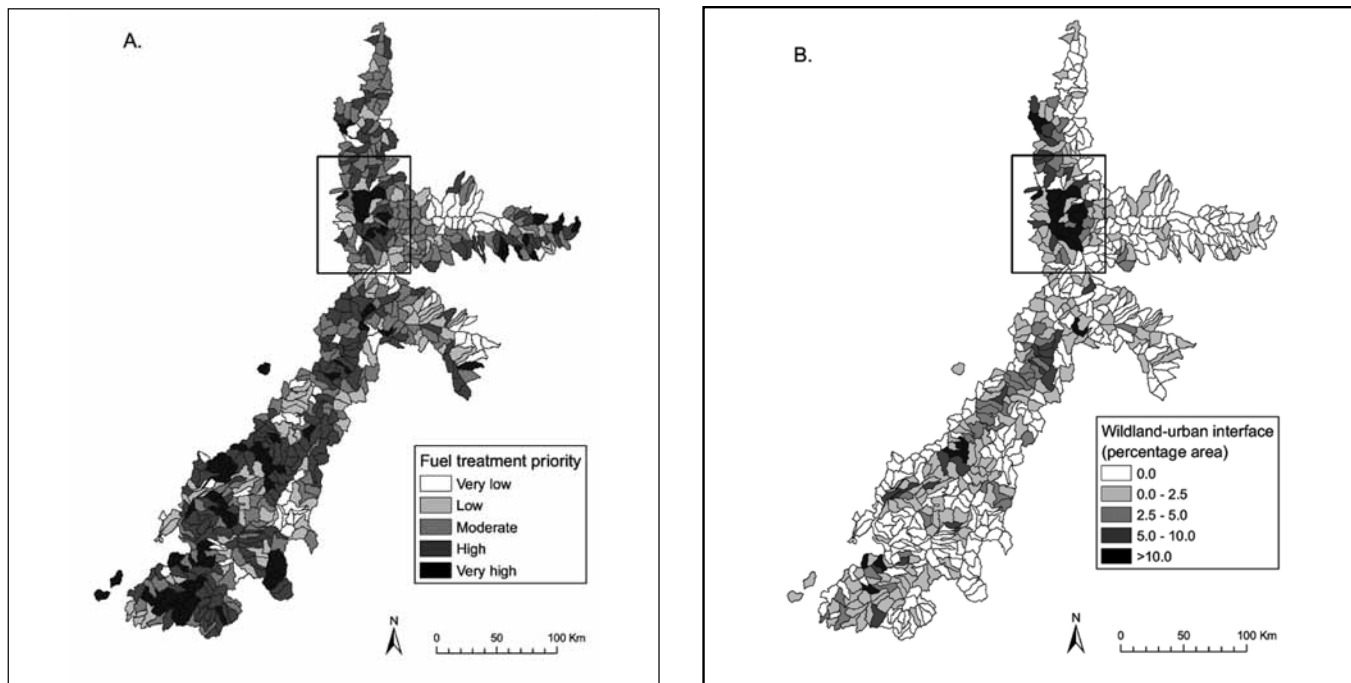


Figure 7—Priorities for fuels treatment in subwatersheds of map zone 16. (A) Priorities of subwatersheds. This map, which reflects the influence of both weighting decision criteria and consideration of proximity to the wildland-urban interface, should be compared with Figure 5. (B) Percentage of wildland-urban interface in each subwatershed. Both maps are symbolized using a natural breaks algorithm in ArcMap to define the classes in the legend. Bounding boxes in A and B indicate corresponding detailed views in Figure 8.

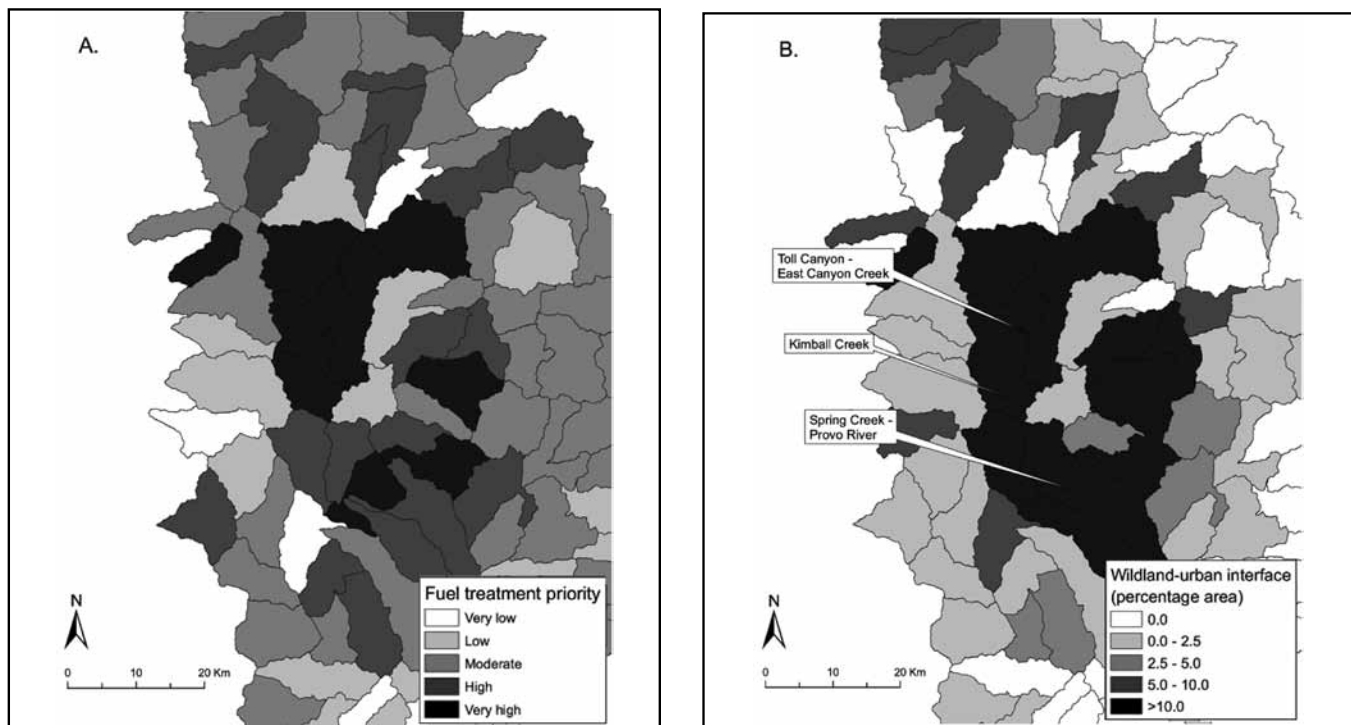


Figure 8—Detailed views of example subregions from bounding box in Figure 7, (A) priorities and (B) percentage of wildland-urban interface. Both maps are symbolized using a natural breaks algorithm in ArcMap to define the classes in the legend.

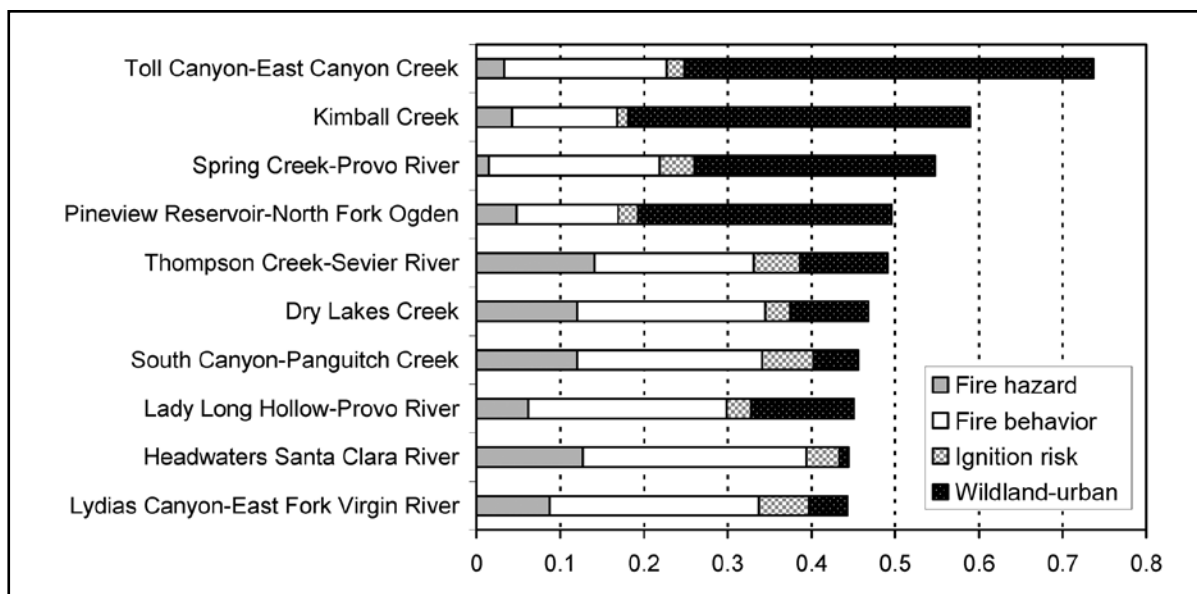


Figure 9—Contributions of primary decision criteria to decision scores for priority of fuels treatment in subwatersheds of map zone 16.

secondary and primary topics within a subwatershed and among subwatersheds. Thus, it is important to keep in mind that variability of support for a subwatershed at the elementary topic level in the hierarchy should be considered when interpreting a primary or secondary topic level evaluation result for any subwatershed and among subwatersheds.

The present study illustrates application of EMDS for evaluating wildland fire danger and prioritizing vegetation and forest fuels treatments at the spatial extent of a U.S. Geological Survey (USGS) map zone. With the national LANDFIRE mapping effort (<http://www.landfire.gov>) complete for the continental United States (CONUS), it is technically feasible to conduct an analysis of fire danger for all subwatersheds in the CONUS in the same manner as we have illustrated here. Moreover, it is a relatively simple matter, given such a base analysis, to summarize such watershed-scale evaluations to various intermediate broader scales such as States, geographic regions, forest boundaries, or forest planning zones as a basic input to broad-scale planning and resource allocation.

At the other extreme, the present study provides a starting point for finer scale planning. We have examined the evidence for fire danger in subwatersheds of map zone 16, but this information, by itself, is not necessarily sufficient

for fuels treatment planning. As shown above, subwatersheds that exhibit a similar moderate level of fire danger do not necessarily share the same evaluation results for primary topics (Figure 12). Thus, variability of support for propositions within a subwatershed at the level in the logic model where data are evaluated should be considered when interpreting an evaluation result among subwatersheds at the level of the primary or secondary topics.

To that end, subwatersheds in the worst condition with respect to fuels may not be the best candidates for fuels treatment. In particular, additional strategic or logistical factors such as proximity to population centers, presence of endangered species, slope steepness, and road access all might be taken into account in selection of specific watersheds within a management area for fuel treatment. Such an approach was illustrated by Reynolds and Hessburg (2005) using the Priority Analyst component of EMDS, which uses a decision engine for such purposes. In that study, they considered the compositional and structural integrity of forests along with contemporary fire risks, and the technical and economic feasibility of restoration. Carefully designed decision models can not only assist with a more circumspect approach to selection of individual treatment units, but can also show which of several treatment options may be most

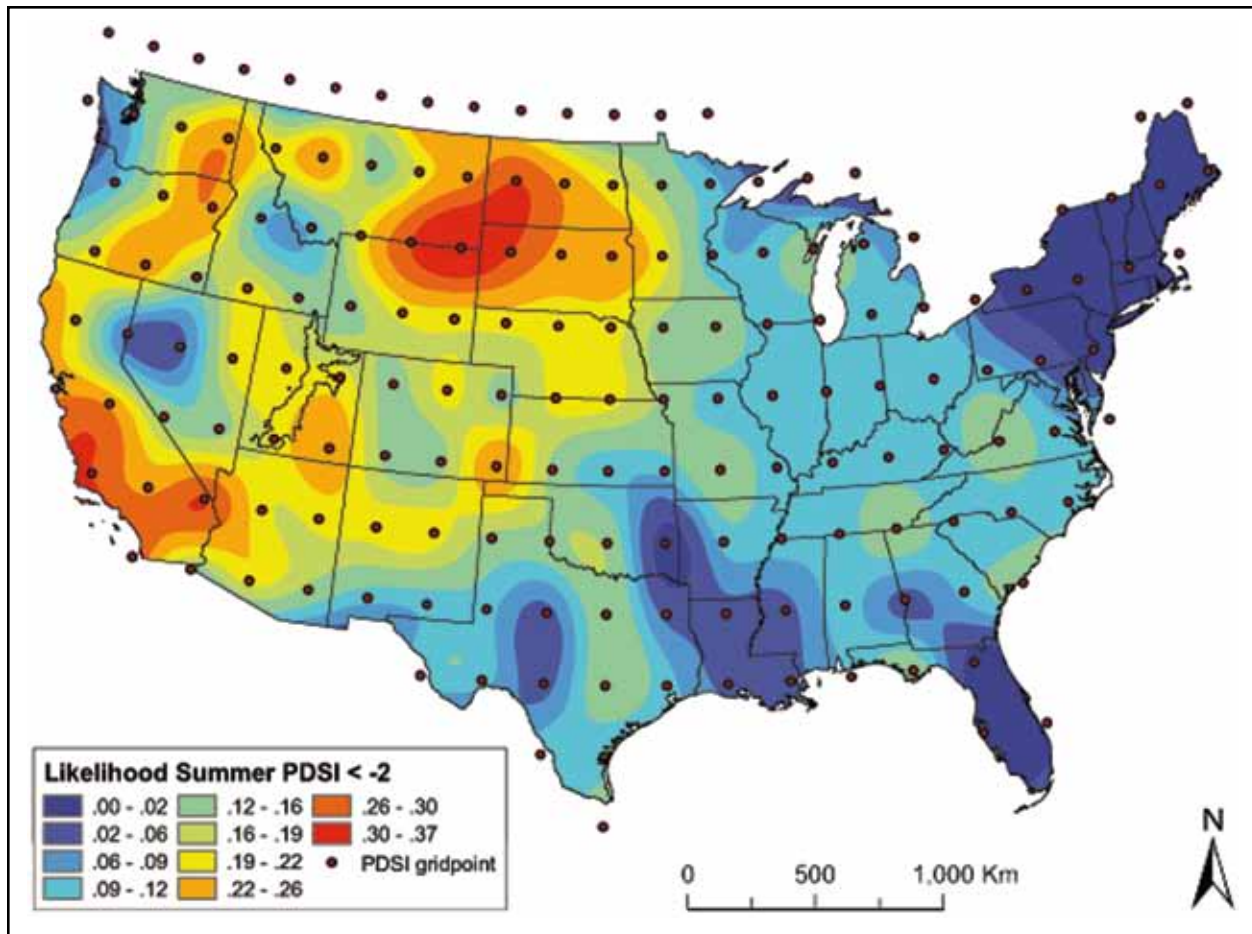


Figure 10—Grid points of the Palmer Drought Severity Index (PDSI) and drought map for the average of 20 years (1971-1990).

suitable in a given unit, thus also providing support for the tactical level of planning.

Similarly, evaluation of treatment priorities related to fire danger is not necessarily limited to fuel and fire characteristics; it can also incorporate human impacts and social or economic, or other value considerations. One such consideration, when evaluating the context of fire danger, may be the pattern of wildland-urban interface in the study area (Figure 7b). Readers might fairly ask, “Given that the structures of the logic model for danger evaluation and the decision model for treatment priorities are so similar in this example, why bother with two separate models?” First, and perhaps most obviously, the two models produce very different interpretations of the data (compare fire danger in Figure 5 with treatment priority in Figure 7a). The logic model is a relatively objective interpretation of fire danger,

given that parameters used to interpret observations (Table 2) were derived from field data, and given that the logic is presented in a relatively pure form insofar as all topics (with the exception of fireline intensity and flame length) are equally weighted. Although weights can easily be applied to topics in a logic model, they also add an additional level of subjectivity that is more effectively managed within the context of decision models, such as those based on the analytic hierarchy process, for example, that are more specifically designed to deal with such issues (Reynolds and others 2003). Logic models also offer the opportunity to synthesize and summarize potentially complex information, thus simplifying the structure of a decision model. In this study, for example, the decision model used summarized information about the topics under fire hazard that would otherwise have been difficult to adequately represent in an intrinsically

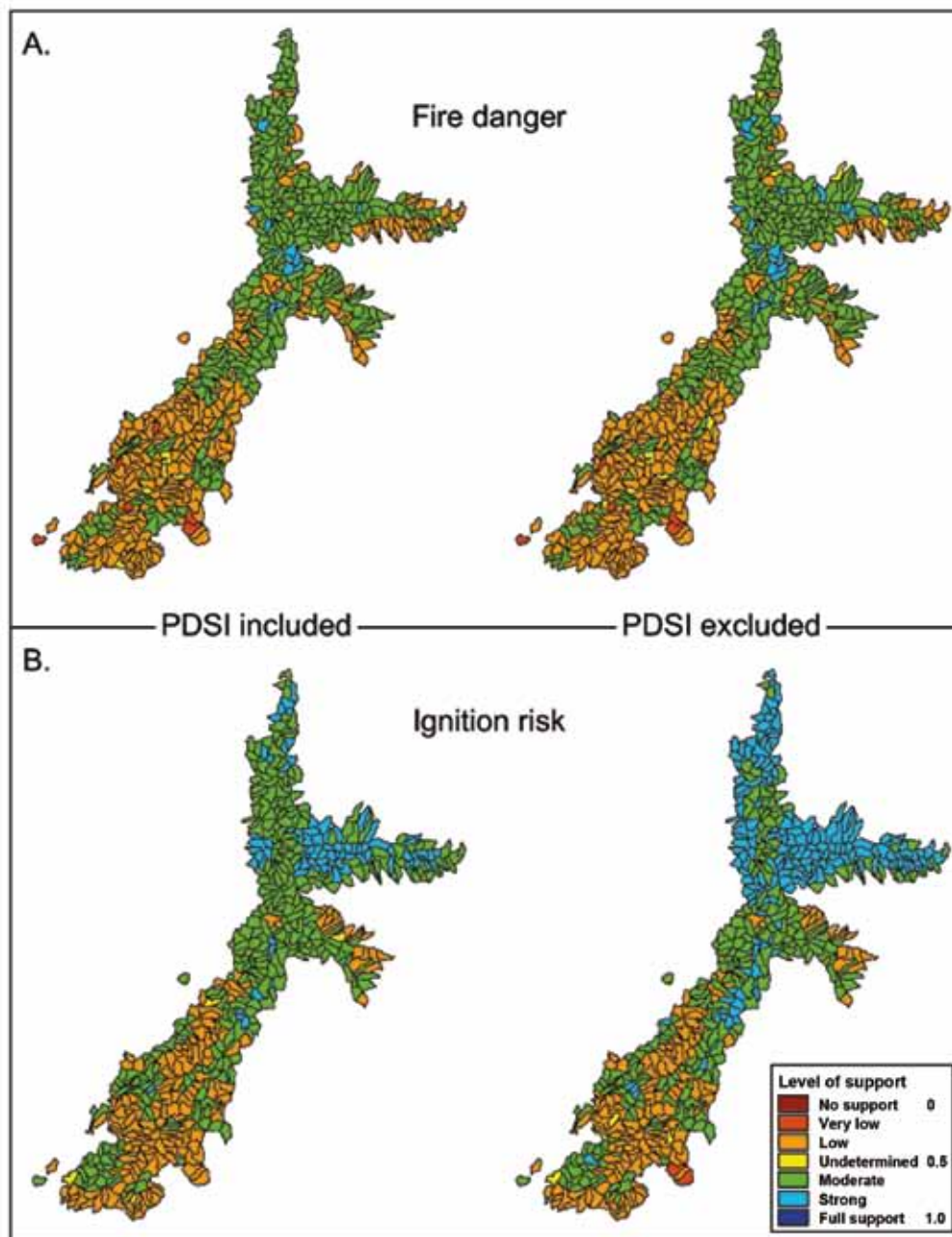


Figure 11—Comparison of (A) overall fire danger and (B) ignition risk evaluations with and without the Palmer Drought Severity Index (PDSI) elementary topic evaluation.

linear decision model (see, for example, the description of the CBD topic in “Primary Topic—Fire Hazard”).

Finally, the two types of models are very complementary in the sense that the logic model focuses on the question, “What have I got?”, whereas the decision model focuses on the question, “Now that I know what I have, what should I do about it?” Notice that logistical issues are

not pertinent to the first question, but they may be extremely important for the second. An important consequence of separating the overall modeling problem into these two complementary phases is that each phase is rendered conceptually simpler. The logic model evaluates and keeps separate the status of the components of each ecological system under evaluation; in this case, the components of

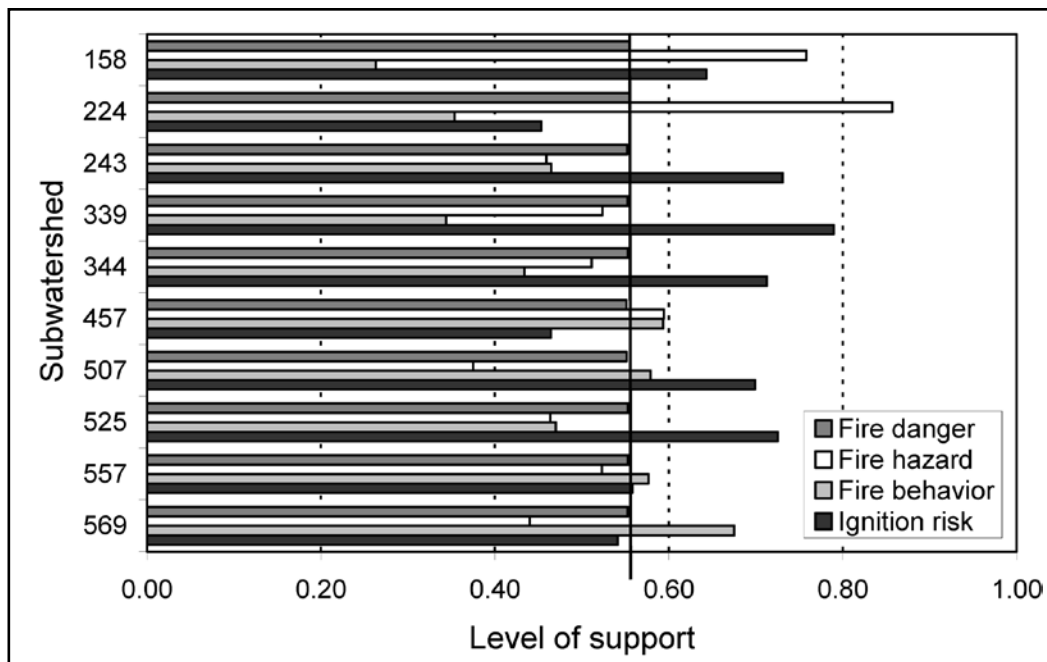


Figure 12—Comparison of 10 subwatersheds in map zone 16, each of which displayed moderate support (strength of evidence=0.56 in the interval [0,1]) for the proposition of low fire danger. Note that level of support varies considerably by primary topic (fire vulnerability, fire behavior, ignition risk).

wildland fire danger of each subwatershed in the map zone. The decision model takes the ecological status of each ecosystem and places it in one or more social contexts that are designed to further inform decisionmaking. The decisions will be based only partially on the ecological status information. They will also be based on social context and human values, in this case, proximity to and amount of wildland-urban interface, which captures a measure of the potential risk of fire damage to people and their structures. After priorities have been derived by the decision model concerning what to do about the existing fire danger conditions, the decisionmaker can look back at the decision and see the relative contributions of the ecological states and their social context(s) to the overall decision. This transparent model design and structure aids in decision explanation, and it allows decisionmakers to consider, in the sense of scenario planning, the effects of alternative weightings of important decision criteria.

As George Box (1979) noted, “All models are wrong; some are useful.” Thus, as with any model intended to support significant management decisions, our model of fire danger requires both verification and validation because all

models are necessarily simplifications of reality. The present model has, in fact, been substantially verified in the sense that it performs as expected based on our own analyses and has been vetted in several meetings over the past year involving substantial numbers of prominent fire managers and fire scientists who agree that the representation of fire danger is reasonable. In contrast to verification, validation is a more rigorous process in which model accuracy is objectively evaluated by comparing predicted and actual outcomes, ideally with statistical procedures. Readers unfamiliar with logic-based models may wonder if validation is even possible. However, models based on logic are no better or worse in this respect than their probabilistic counterparts. Although a detailed discussion of this assertion is beyond the scope of this report, it may be sufficient to note that metrics expressing strength of evidence have commonly been treated as subjective probabilities (Zadeh 1968). Finally, model validation was not feasible within the temporal scope of our study. Realistically, even a preliminary validation in this context would require 5 to 10 years. If the model for fire danger were to be adopted as a tool to support strategic planning for fuels treatment, then we certainly recommend

that explicit provisions for validation be an integral part of any ongoing assessment process designed to support it.

Conclusions

Given the widespread increase in danger of wildland fire throughout the Western United States over the past 70 years or more, the sustainability of western forest ecosystems is clearly at stake. Decision-support systems such as EMDS can play a role in assisting with restoration to improve or maintain their sustainability. Issues surrounding decisions about fuels management are complex and often require abstraction, but logic and decision models are well suited to representing the inherent complexities and abstractness of the problem, thus rendering the analytical problem more manageable. This particular application of EMDS also is an example of how decision-support systems can not only be used as tools for technical specialists and decisionmakers, but as tools for communicating clearly and effectively with the general public who understandably have a strong interest in the topic of wildfire and want to understand and be involved in any proposed solution. Both logic and decision models are good at explaining themselves in relatively intuitive terms, and thus provide a basis for an effective public dialog.

Finally, there is an important interdependency between science, policy, and decision-support systems such as EMDS. Although logic models are sometimes used for prediction, they are fundamentally concerned with interpretation (Reynolds and others 2003). In other words, what does the information mean? Meaning can be highly normative or highly subjective, and usually falls somewhere in between the two extremes. As a result, virtually all interpretation embeds some degree of subjectivity; that is, to some degree, values and policy are inextricable aspects of logic and decision models. The practical implication is that successful application of most decision-support systems to real-world situations ultimately depends on a close collaboration between the scientific community that brings its facts to the table and the policymakers that need to reach decisions based on that information and additional social and economic considerations. Decision-support systems provide a conspicuous advantage in this context—detailed

documentation of a decisionmaking process. With ongoing monitoring and evaluation, lessons learned can be readily incorporated into decision models providing increasing effectiveness to decisionmaking and an explicit vehicle for adapting management.

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Digital Aerial Sketchmapping and Downlink Communications: A New Tool for Fire Managers

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Abstract

Aerial sketchmapping is the geolocating of features that are seen on the ground below an aircraft and the subsequent recording of those features. Traditional aerial sketchmapping methods required hand-sketching on hardcopy maps or photos and the translation of that information to a digital file. In 1996, the U.S. Department of Agriculture (USDA) Forest Service embarked on a project to develop a digital aerial sketchmapping system (D-ASM) to replace or augment the traditional (manual) methodology. Advances in microprocessor speed and Personal Computer (PC) system performance made possible the use of PCs to aid in aerial sketchmapping. The USDA Forest Service Remote Sensing Applications Center (RSAC) and the Forest Health Technology Enterprise Team worked with a software vendor to develop a product that would meet the needs of aerial surveyors. Several hardware and software options were investigated.

The Remote Internet Protocol Communications System (RIPCom) culminated from a successful 2-year collaboration between National Aeronautics and Space Administration (NASA)-Goddard and RSAC to develop a cost-effective, multitask communications solution for the Forest Service based on NASA expertise. The RIPCom consists of commercial off-the-shelf components that were each chosen because of their desirable performance characteristics. The 2003-2004 field tests demonstrated that the RIPCom would deliver the required data throughput (1 Mbps) up to a range of 20 mi. A contractor constructed the second generation RIPCom system incorporating the

lessons learned from flight testing and operational deployments.

The planned integration of the D-ASM and RIPCom systems will enable firefighting experts to collect information about an incident (fire perimeter and hotspot locations) on the D-ASM and rapidly disseminate this information, via wireless devices, to the incident's geographic information system (GIS) technician or directly to firefighters on the ground. Potential system users within the wildfire community include Air Attack, Helitack, Situation Unit Leaders, Operations Chiefs, and Incident Commanders.

Keywords: GIS, global positioning system, pen-tablet, Remote Internet Protocol Communications System (RIPCom), wildland fire monitoring and suppression.

Introduction

Aerial Sketchmapping

Aerial sketchmapping for Forest Health Protection has been conducted since the 1940s. Typically, sketchmapping surveyors fly the area to be surveyed in a high-winged, high-performance aircraft at elevations of 1,000 to 3,000 ft above the ground. The surveyor tracks the plane's location on a hardcopy map (typically several maps that are trimmed, edge-joined, and taped together) or aerial photographs, then sketches areas of the forest that have been damaged during the past year (Figure 1). The sketched features (points and polygons) are then registered and digitized into a geographic information system (GIS). Finally, the resulting GIS data are used to create maps and reports at forest, regional and national levels.

Although skilled surveyors have successfully employed manual sketchmapping techniques for years, there is great potential for error in the process. Some examples of the sources of error are:

- Not knowing the aircraft's location on the map.
- Reinking map data upon completion of the day's flying.
- Inaccurate registration, line following, and attribute capture when digitizing data into the GIS.

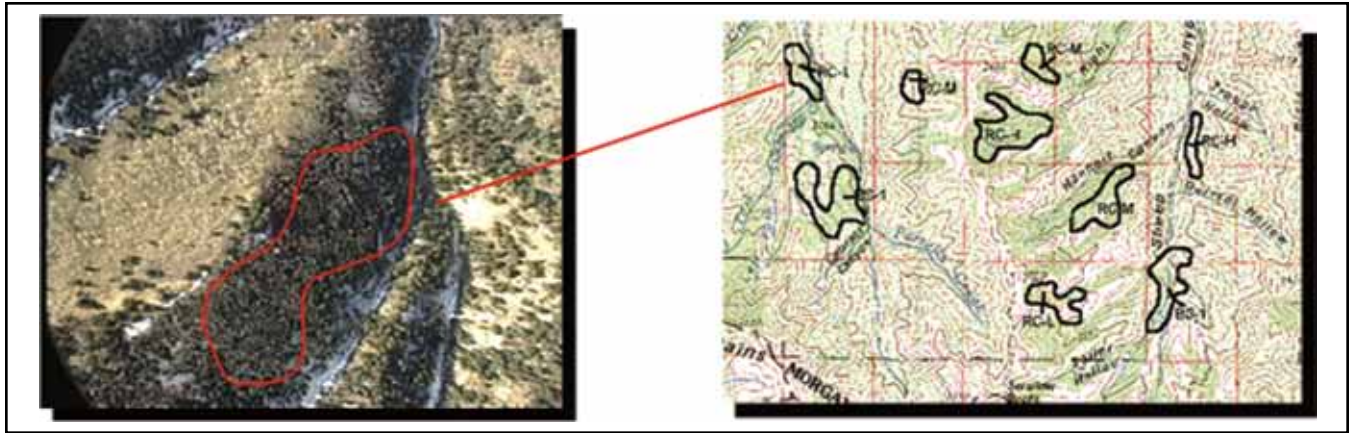


Figure 1—The principle of aerial sketchmapping—observing ground features and annotating those features on a paper or digital map—is displayed in the aerial photograph and accompanying map.

In addition to the potential for error, some of the processes used in manual sketchmapping require additional time to complete. Examples of those processes that can be eliminated with a digital system are:

- The reinking of map data upon completion of the day's flying.
- The management of numerous taped and folded maps in the cockpit.
- The taping and folding of maps prior to the mission, the cutting apart of those maps prior to digitizing, and the retaping of the maps after digitizing, but prior to proof-checking the digitizing results.
- The digitizing process itself.

The positional accuracy of sketched features is largely determined by the ability of the sketchmapper to keep track of the aircraft's position on the map and to correctly relate features seen on the ground to the map. This can be especially difficult in unfamiliar or flat terrain or both. At typical airspeeds of 100 to 140 miles per hour (mph), a sketchmapper has approximately 15 sec/mi^2 to accurately locate, sketch, and attribute features (assuming a visual mapping distance of 2 mi from the aircraft). Accuracy is a relative term, and the end use of the data must be considered. If the data collected during a sketchmap mission are for general monitoring purposes, then more error can be tolerated than if field crews need to locate the affected stands using the sketchmapped data.

To be favorably compared against manual sketchmapping, a competent digital aerial sketchmapping system should address the problems listed above and should meet the following list of requirements, which were compiled for early system development:

1. The map display must be linked to a global positioning system (GPS) receiver so that an icon on the display represents the current position of the aircraft on the movable digital map.
2. The map display must update quickly at ground speeds of 130 mph and reorient when turning, if desired by the surveyor, based on the GPS position and heading of the aircraft.
3. The screen must be viewable under a variety of lighting conditions including full sunlight and must also display, at minimum, full (256) color.
4. The viewable screen size must be at least 10 inches measured diagonally.
5. The software must have the capability to:
 - Digitize points, lines, and polygons – including nested and overlapping polygons.
 - Attribute digitized features (points, lines, and polygons) quickly and easily.
 - Allow editing of feature attributes quickly and easily.

- Allow the user to predefine the feature attributes to be collected during that day's mission.
 - Collect the GPS log file of the flight.
 - Allow the user to zoom quickly to different map scales.
 - Retrieve and display common raster and vector data types as background maps quickly and seamlessly (including the U.S. Geological Survey 1:100,000-scale digital raster graphics DRG aerial surveys currently use in hardcopy form).
 - Export files in a format that is easily imported into ESRI Arc/Info software, thus eliminating the intermediate digitizing step
 - Save data automatically to the computer's disk.
6. The hardware should be operational in moderately harsh conditions (32 to 120 °F, high humidity, dusty conditions).
 7. The primary input device must be a touch screen with stylus.

After a great deal of testing and evaluation, the Forest Service successfully met these requirements, and two systems emerged and are in common use today; a two-screen system and a pen-tablet system. Advantages of the Digital Aerial Sketchmapping System (D-ASM) include automatic tracking of the aircraft's position on a map base through a link to a GPS receiver and a significant reduction in the time spent moving data into a GIS. The D-ASM greatly speeds data collection and processing and improves the positional accuracy of the data being collected. Now a mature system, the D-ASM has been widely embraced by the sketchmapping community with over 90 systems in service with Federal and State forest health agencies.

Following initial system development, a D-ASM was shown to wildfire air attack crews who liked the concept, but wanted the system to be smaller and for it to include a means to get the information down to the incident command in real time. Two technological developments solved these requirements: commercially available touchscreen

pen-tablet computers and the National Aeronautics and Space Administration (NASA) developed long-range high-speed data link named RIPCom. or Remote Internet Protocol Communication.

The Forest Service is now working to further expand the utility of the D-ASM by adding live downlink capabilities to the existing system.

D-ASM Mapping Software

For the D-ASM, the Forest Service selected the consumer off-the-shelf (COTS) mapping software that most closely met the system requirements, and the software provider agreed to make necessary modifications to their product to meet our sketchmapping needs.

With this software, feature type keys (point, line, and polygon) are used to display the aircraft's position on the on-screen map, and a user-defined keypad is used to attribute features. The observer selects the type of feature to be sketched, sketches the feature on the screen, and then assigns attributes to the feature using the keypad. The screen remains frozen while the feature is being sketched and is updated after the enter key is pressed. When the aircraft icon advances to the edge of the window, the map display updates and moves the aircraft icon to the center of the window. Upon completion of the sketchmapping mission, a translation step converts the sketched features into ESRI shapefiles. These shapefiles are then copied to an external disk for transfer to a GIS.

Aerial surveyors often want to display large shapefiles over their raster map data; these shapefiles may be the previous seasons' survey data, administration boundaries, or section lines. Fire managers may want to see the previous day's fire perimeters shown on top of raster aerial imagery or digitized raster maps. The software can display many different types of geospatial data as the background map and thus can accommodate a variety of user preferences.

The software provider continues to enhance the software by developing modifications such as editing of polygons, multiple user-defined keypad sets for attributing features, display of feature attributes, and editing of feature attributes.



Figure 2—Forest Service digital sketchmapping systems are displayed. At left is the two-screen system with external global positioning system (GPS) receiver. At right is the pen-tablet sketchmapping table with built-in GPS receiver (the windshield-mounted antenna is shown in the lower right).

D-ASM Mapping Hardware

Two computing options have been tested and are in use in Forest Service sketchmapping systems: a two-screen laptop-based system with a separate touchscreen, and a pen-tablet sketchmapping table (Figure 2). Because of the potential for bright sunlight on the screen, the visibility of the map image on the touchscreen display is critical. We evaluated several touchscreens, which varied in screen size, brightness, and touchscreen technology. Resistive touchscreens have a membrane layer installed over a fine wire mesh grid in the screen; when this membrane is pressed on to the grid, a signal is sent to the computer, and the cursor jumps to that spot on the screen. Virtually any object can be used on the screen—a finger, an eraser, stylus, etc. This can be a disadvantage for sketchmapping, because any inadvertent touch to the screen may result in undesired actions. Many touchscreens have a rather small screen size (10.4 diagonal), which limits the amount of map area visible on the screen.

A supplier manufactured several screens to Forest Service specifications for early testing. These screens have 800 by 600 resolution, 12.1-in diagonal screen size, 1,500 nits brightness, and capacitive touchscreens. This type of touchscreen is activated by completion of an electrical circuit using a tethered stylus. There is no inadvertent activation of the resistive touchscreen because the screen can be activated only with the tethered pen. It should be noted that this system is quickly becoming obsolete and will no longer

be supported by the Forest Health Technology Enterprise Team (FHTET).

Pen-tablet Personal Computers (PCs) were also tested as an alternative platform for the D-ASM. The advantages of the pen tablet for use in sketchmapping include a smaller, simpler hardware profile and fewer cables. The disadvantages include a smaller screen area and decreased screen visibility in full sunlight.

We have tested many GPS receivers, and it seems that any receiver capable of producing an National Marine Electronics Association 0813 output string will be compatible with the software. Expensive, survey-grade receivers are not necessary; various recreation-grade units are adequate for the task. The GPS signal enters the PC via a COM port Universal Serial Bus ([USB] or Serial) connection. Newer USB GPS units have the advantage of drawing power from the laptop PC. Wireless GPS receivers provide an attractive alternative, and testing of such receivers is ongoing.

RIPCom Background

The RSAC has been working with NASA Goddard Space Flight Center (GSFC) on adapting technology NASA has developed for unmanned aerial vehicles (UAV) to Forest Service applications. The RIPCom system that is being developed by NASA-GSFC is of particular interest to the Forest Service because it can be used to download fire imagery during flight, which will save the Forest Service

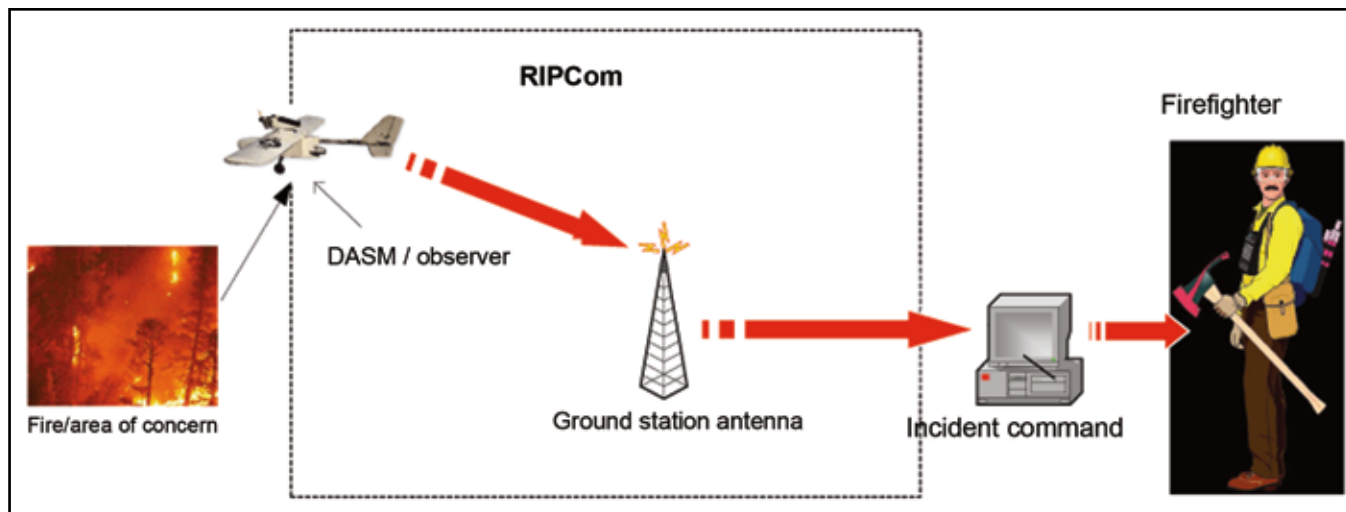


Figure 3—RIPCom concept is shown in detail.

time and money while enhancing safety. RIPCom is a wireless communication system that makes any RIPCom-equipped aircraft look like a network node in the sky. RIPCom provides an Ethernet to Radio Frequency (RF) connection solution for real-time data transmission, and its design allows the end points of the communication system to become nodes on a network with assigned IP address. RIPCom's versatility makes it valuable for any system that requires a high-speed, digital wireless network.

The NASA-GSFC has modified the original RIPCom design so that it meets Forest Service requirements. These include the ability to dump data at much faster rates and at greater ranges.

Specific Forest Service requirements include:

- Providing a wireless network connection at distance of at least 10 mi.
- Supplying a data rate of at least 1 Mbps.
- Providing a long-distance wireless network that can also maintain a connection with a moving target.
- Design for aircraft that is 10,000 ft above-ground level.
- Data acquired at the ground station from an aircraft operating at 300 mph.

The RIPCom consists of COTS components from different vendors chosen because of their unique performance characteristics to meet Forest Service goals. Numerous

wireless network technologies currently exist that are candidates for RIPCom, and many factors such as frequency, RF power output, size, and cost were considered for the Forest Service system design. The aircraft and ground station nodes were designed individually owing to the unique conditions of each environment. Figures 3 and 4 illustrate the major components of the RIPCom concept and details of the system's ground station and aircraft components.

The major consideration for the airborne components is the antenna, which must be very aerodynamic while still providing a workable radiation pattern for this implementation. In addition, the antenna must be placed so that shadowing by the wings or other aircraft components is minimized. A 2.4 GHz, 5dB gain blade antenna (Figure 5) has been installed on the aircraft's fuselage just aft of the forward landing gear bay, which gives the antenna an excellent field of view.

RIPCom's ground station is designed so that it is able to provide bi-directional communication with a lower gain antenna 10 mi away. The ground station accomplishes this by utilizing a three-sector, omni-directional antenna. The resulting system is able to provide a point-to-point connection for high-speed data transfer at an affordable cost (\$7,500 for the ground station, less tripod, and \$3,200 for aircraft station, less installation). Because RIPCom is platform independent, (it does not matter which operating system is used), the only requirement is that ground station

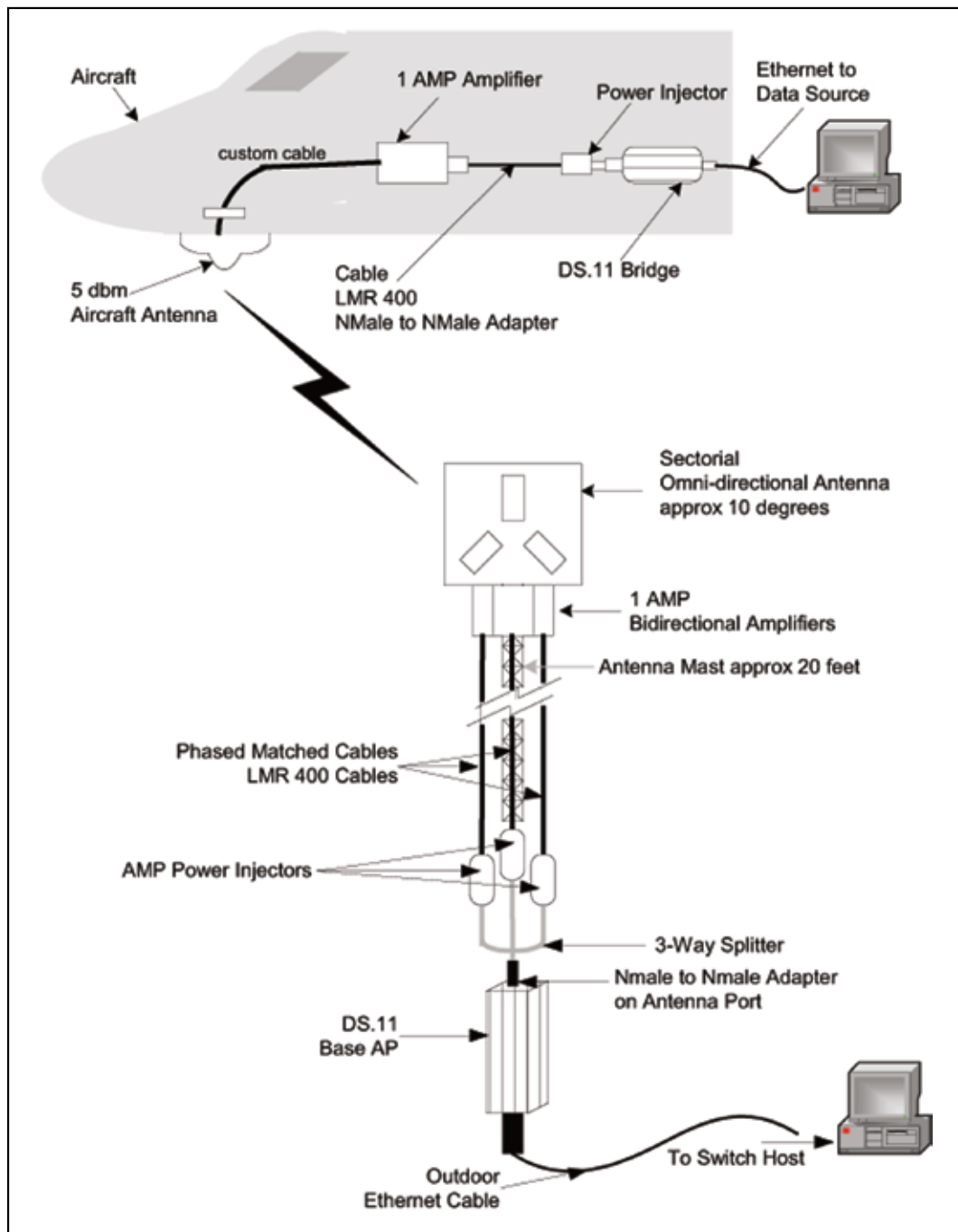


Figure 4— Detailed diagram of Forest Service RIPCom solution is shown.

and aircraft have a Transmission Control Protocol/Internet Protocol connection.

Flight Tests

Initial flight-testing of the RIPCom system took place near Boise, Idaho, the week of April 14-19, 2003. Tests were

conducted for 6-, 8-, 9-, 10-, 11-, 12-, and 15-mi orbits. During the flight test, several issues were evaluated, checked, and recorded:

- Determine the strongest channel (One is able to choose among 11 channels.).



Figure 5— View of the aircraft gain blade antenna that is attached to the aircraft's fuselage.

- Ping each network device for each throughput setting (11, 5.5, 2 Mbps), and transmit a 36 Mb and 77 Mb file from the plane to the ground station.
- Determine the time it took to transmit the data.
- Characterize the location and duration of any data drops during data transmission.

Other tests included having the pilots fly elliptical orbits, flying directly over or to the sides of the ground station to characterize the radiation pattern of the ground station and to determine if there was any Doppler interference in the data transmissions.

The RIPCom exceeded expectations during flight-testing as it had with the ground tests. Performance was very good. Data rates of 2 to 3 Mbps were achieved at a range of 30 mi. The system was robust; File Transfer Protocol transfers could be completed without errors even if the transmission was interrupted (up to 45 sec). We also found that the jet could fly directly to and from the ground station at 300 mph without the data transmissions being affected by the Doppler Effect.

RIPCom 2

The April 2003 flight tests were an unqualified success based on the National Infrared Operations requirement for long-range broadband real-time transmission of high-resolution imagery acceptable by the Infrared Interpreter. Upon later assessment, it was decided that the current state of technology could not fulfill both the performance and updated portability (single suitcase as carry-on luggage) requirements.

The NASA/GFSC and the Forest Service went back to work to develop a self-contained unit that could be easily transported by one person. It was a conscious decision to sacrifice both range and data bandwidth in order to meet the portability requirement. This collaborative effort resulted in a second generation RIPCom unit that utilizes a 36-in omni-directional antenna and a single radio. The portable approach yields a much lighter package that can be transported in two pelican cases. One case contains the modem, radio, and laptop computer. The second contains the antenna, mast, and cables (Figure 6). The bottom line for the end-user is that the portable RIPCom ground station is capable of 1 Mbps data rate at a range of 8 to 10 miles.

Summary

By integrating two successful technologies, the Forest Service is working to create a new tool that will allow fire managers to relay critical fire information to dispersed field units in real time. The Remote Sensing Applications Center will be working with NASA GSFC and wildland fire personnel to integrate the RIPCom sketchmapping system into wildland-fire operations during the 2007 fire season. The ultimate goal is to increase firefighter safety and to improve overall operational effectiveness in wildland fire monitoring and suppression.



Figure 6— (Left) Case 1, (Center) Case 1 with computer tray removed, (Right) Case 2.

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Assessing Risks to Multiple Resources Affected by Wildfire and Forest Management Using an Integrated Probabilistic Framework

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Abstract

The tradeoffs that surround forest management are inherently complex, often involving multiple temporal and spatial scales. For example, conflicts may result when fuel treatments are designed to mediate long-term fuel hazards, but activities could impair sensitive aquatic habitat or degrade wildlife habitat in the short term. This complexity makes it hard for managers to describe and communicate the conditional nature of risk and to justify planned activities to stakeholders. In addition, our understanding of how proposed activities will affect resources of concern is often limited owing to informational shortcomings and imprecise models. To be robust and transparent, a risk assessment framework needs to reveal these limitations while quantifying the probable outcomes of project effects to multiple resources of concern. In this analysis, we describe the effects of fuel treatments using such a planning framework called CRAFT (Comparative Risk Assessment Framework and Tools). CRAFT provides a platform from which diverse ancillary models and other relevant information can be transparently integrated and evaluated.

We conducted our case study in the southwestern Klamath Mountains of California. As is typical of most montane forests of California, this area has experienced decades of fire suppression, and severe effects from wildfire are a concern. Working with managers, we identified a range of measurable objectives involving the wildland urban interface, fire behavior, fire effects, and sensitive wildlife. We then developed a conceptual model describing

how components of the system interrelate. From this, we developed a probabilistic framework, using Bayesian belief networks, in which we employed existing fire models to address how expected fire behavior varies across different burning scenarios. Our framework provides decisionmakers and stakeholders with insights into the condition probability that management alternatives will be successful.

Keywords: Bayesian belief network, comparative risk analysis, CRAFT, fire effects, objectives, tradeoffs.

Introduction

Forest management decisions are often difficult because ecosystems are inherently complex, and the system's response to management is uncertain. Management tradeoffs typically involve very different objectives that can be difficult to compare or model across spatial and temporal scales. In addition, future conditions are often dependent on stochastic variation in the system that can be difficult to predict. One way for managers to address this complexity is to consider outcomes as conditioned on a range of influential factors that are assessed in terms of how likely they are to occur. This approach is more comprehensive than the single-scenario analyses that are commonly used in forest planning today, and it provides key information for decisionmakers and stakeholders.

Traditional risk assessment approaches were developed to reduce the likelihood of catastrophe such as engineering failures, insurance-related loss, or environmental contamination. These approaches may provide a poor model for forest risk assessments because management activities and nonactivities may influence a wide array of values across space and time. When something could happen that is unambiguously bad, such as a nuclear plant meltdown, the failure of a critical aircraft part, or a toxic spill, prevention is a clear and high-priority objective. In forest management, disturbances may be both a threat to the system and a critical requirement for the long-term viability of the system. For example, frequent fires may reduce surface

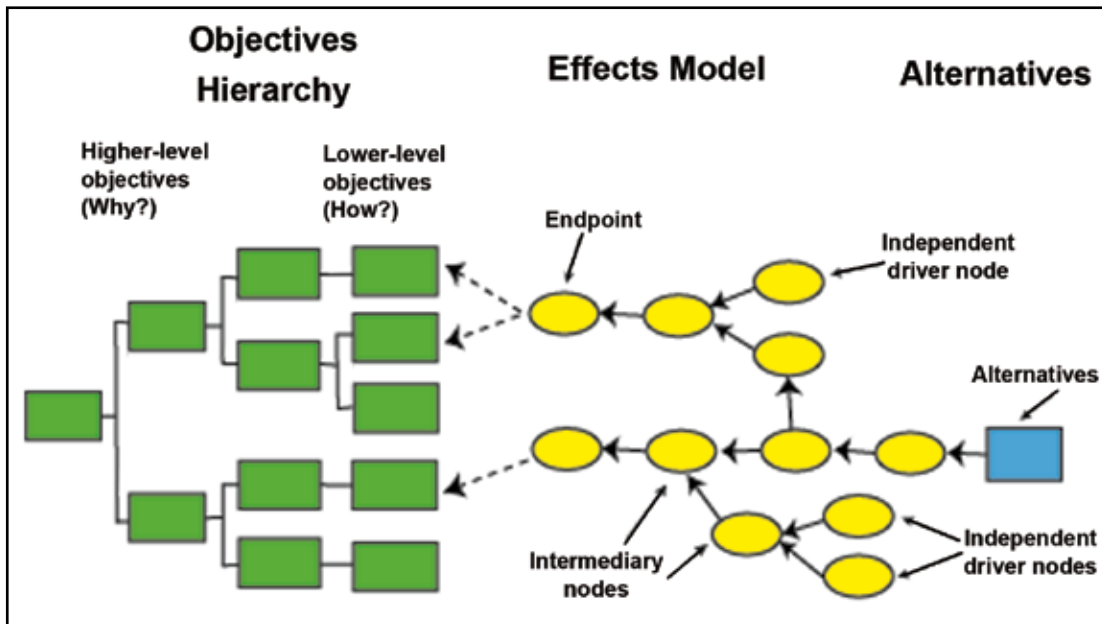


Figure 1—Overview of the Comparative Risk Assessment Framework and Tools process showing how increasingly specific objectives (in green) from left to right tier to risk assessment endpoints. The effects of different management alternatives (in blue) and scenarios are modeled using a probabilistic belief network (in yellow).

fuels and increase the resilience of old trees, yet they may also spread invasive species, reduce air quality, and threaten homes in the surrounding wildland-urban interface. Rather than decide how to prevent fire, forest managers increasingly must decide where, when, and how to conduct fire and fuel management to balance competing tradeoffs and diverse stakeholder values. Unlike risks associated with an unambiguous catastrophe, the risks and tradeoffs associated with forest management are better viewed comparatively. In this analysis, we introduce a comparative risk assessments framework and tools called CRAFT that was developed to address the tradeoffs associated with forest management decisions. A more detailed discussion of CRAFT can be found at: http://www.fs.fed.us/psw/topics/fire_science/craft/. [Date accessed unknown].

Study Area

We selected a 3000-km² area located in the Klamath Mountains of northwestern California centered on the town of Hayfork. The majority of the landscape is under Federal management and includes portions of the Shasta-Trinity and Six Rivers National Forests. Private inholdings are common

and contribute to wildland-urban interface management problems that are characteristic of the West. These include issues related to wildland fire, air quality, and biodiversity management. Both national forests fall under the Northwest Forest Plan, a 1994 regional plan designed to maintain sustainable forest products and to sustain species. As part of that plan, areas were allocated for different purposes. The primary land allocations present in the study area include late successional reserves that were set aside for old-growth species, including the northern spotted owl, and the Hayfork Adaptive Management Area set aside for experimentation and learning.

The vegetation of the study area consists of a diverse array of conifers, hardwoods, shrublands, and grassy meadows. Douglas fir (*Pseudotsuga menziesii* (Mirb.) Franco) forests dominate a large portion of the area, but forests of ponderosa pine (*Pinus ponderosa* Dougl. ex Laws.) occur on dry sites, and Jeffrey pine (*Pinus jeffreyi* Gred. & Baif.) and incense-cedar (*Calocedrus decurrens* Torr.) dominate forests on ultramafic soils. Mixed-conifer forests are found where pine, Douglas fir, white fir (*Abies concolor* (Gord. & Glend.) Lindl. ex Hildebr.), and incense-cedar co-occur and typically include various species of hardwoods in

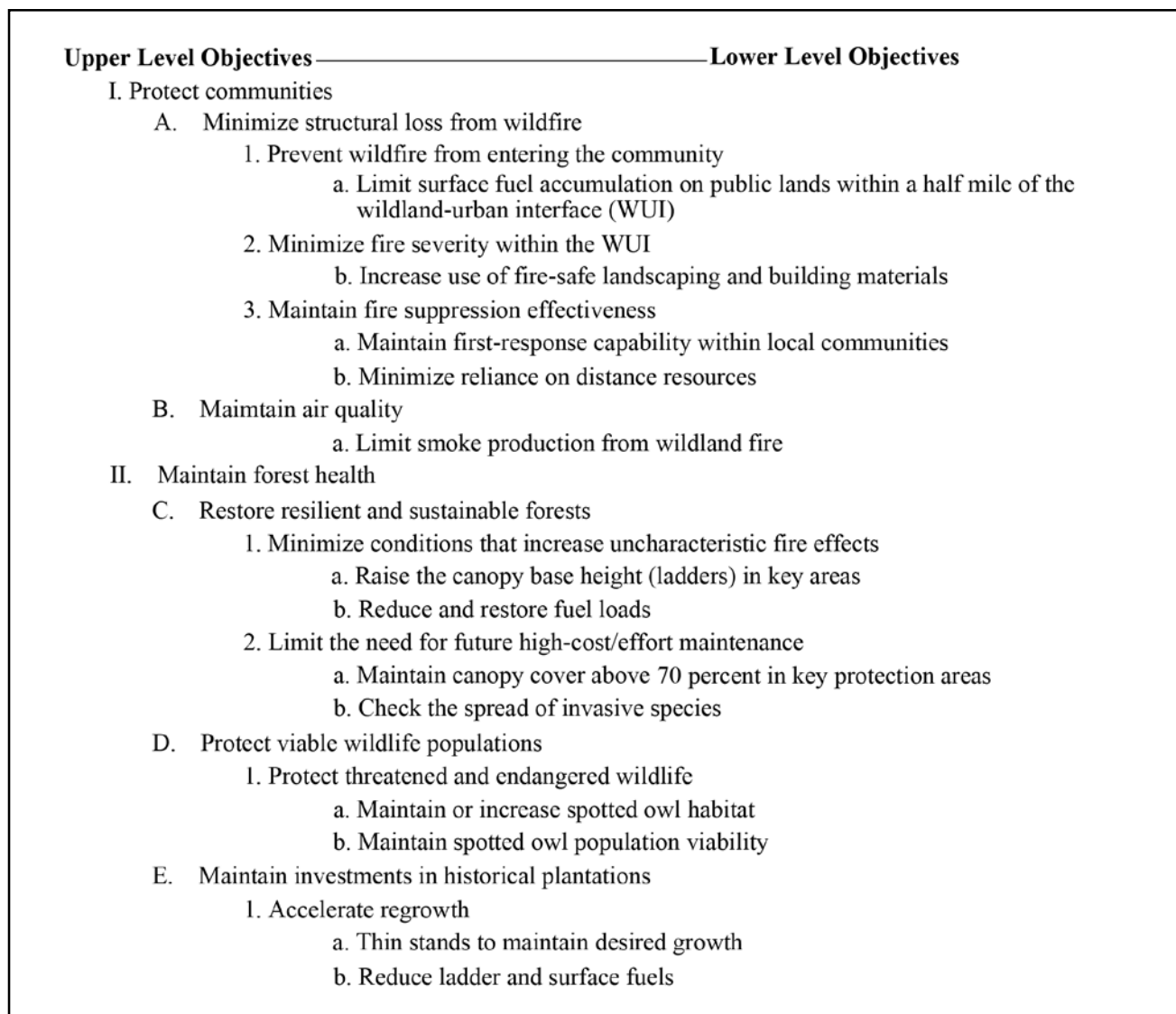


Figure 2—A partial objectives hierarchy for the Hayfork study area. Upper level objectives (left) suggest general goals, whereas lower level objectives (right) can be formulated into specific risk assessment endpoints, monitoring endpoints, or specific management activities.

the canopy or understory. The most common hardwoods include tanoak (*Lithocarpus densiflorus* (Hook. & Arn. Rehd.), canyon live oak (*Quercus chrysolepis* Liebm.), black oak (*Quercus kelloggii* (Newb.) and chinquapin (*Chrysolepis chrysophylla* (Hook.) Hjelmqvist). Surface and live fuel composition and structure have been altered from their historical condition by 20th-century fire suppression and logging. Wildfires occur on a regular basis, but extensive, long-burning wildfires have only occurred in 1987 and 1999.

The CRAFT Process

The CRAFT planning process leads managers through four stages involving (1) objective setting and problem conceptualization, (2) alternatives design, (3) probabilistic modeling of effects, and (4) synthesis. In this analysis, we emphasize steps 1 and 3 to show how objectives can be incorporated within a single modeling framework and then considered in terms of their uncertainties. Figure 1 provides a graphical overview of the CRAFT process.

Specifying and Structuring Objectives

A critical first step in CRAFT is to fully understand the problems at hand. It is easy to sidestep this step by jumping to alternatives or even modeling, but a thoughtful, interdisciplinary exploration of the objectives and the problems is needed before specific modeling tools and data sets are considered. Existing information and familiar models may be ill suited for the specific problems at hand.

In CRAFT, problem formulation is addressed using two planning tools: (1) an objectives hierarchy that helps planners and stakeholders focus on specific measurable objectives rather than on overarching goals, and (2) by a cause-and-effect model that provides a transparent documentation of how the system is thought to operate. In addition to documenting values and beliefs, these tools serve to identify potential tradeoffs for the comparative assessment of risk.

Objectives hierarchies have been used for a variety of planning purposes, but they are rarely used by public agencies in the United States (Clemen and Reilly 2001, Keeney 1992). An objectives hierarchy simply separates intangible objectives (the upper levels) from detailed, measurable objectives (the lower levels) and structures them within a hierarchy. Specific lower level objectives suggest how upper level objectives might be achieved. This structure ensures that planners have formulated explicit lower level objectives that will be translated into risk assessment endpoints. Detailed objectives may also serve as appropriate monitoring endpoints as part of a broader forest management strategy. We show a partial Objectives Hierarchy for our study area in Figure 2.

From an agency perspective, an objectives hierarchy simplifies risk assessments by providing focus, context, and clarity. Upper level objectives often correspond to general forest mandates, whereas lower level objectives may relate to the extension of statutes to specific instances or locations. For example, although maintaining biodiversity is mandated by a number of statutes, the specific requirements for maintaining viable habitat and populations for individual species often reflect regulatory policy.

From a stakeholder perspective, objectives hierarchies can be powerful tools for involvement. Stakeholder values

differ in terms of their specificity, and, by using an objectives hierarchy, stakeholders are more likely to see how their values fit with those of the agency. Comprehensive objectives hierarchies can also suggest opportunities for stakeholder-agency collaboration. As examples, the protection of homes from wildfire and invasive species control both require agency and stakeholder involvement to ultimately be successful. Perhaps, most valuably, an objectives hierarchy can provide a common vision for what a managed ecosystem could look like. Stakeholders are more likely to remain engaged when they can imagine desired future conditions. Once future objectives are clarified, specific management choices and tradeoffs can be considered through a separate process.

Conceptualizing Cause and Effect

Following the identification of measurable objectives in CRAFT, cause-and-effect models are developed that identify the factors that are known to influence relevant lower level objectives of concern. A conceptual model consists of a bubble-arrow diagram and supportive text that explains the relationship in detail (U.S. Environmental Protection Agency 1998). Conceptual models help managers identify the key reasons why a particular management objective may or may not be met. Causes and effects can be established across spatial and temporal scales so that the interconnectedness of the ecosystem is apparent. This step is designed to be conceptually inclusive and unrestrained by the availability of local data or existing supportive models that have been used in the past. Later on, when a quantitative effects model is being developed, ancillary models, data, and expert opinion are more likely to be used appropriately when a clear conceptual model exists. Cause-and-effect relationships may be based on existing research, ancillary models, or expert opinion.

From an agency perspective, this task transparently documents the factors that were considered beyond those that are carried forward in the formal risk modeling step that follows. Conceptual models also provide a transparent venue for group involvement by describing the problem in a way that both interdisciplinary team members and stakeholders can dispute and ultimately come to agreement.

At some point, the cause-and-effect model will likely suggest specific management activities. In CRAFT, as with National Environmental Policy Act, these activities are then combined into formal management alternatives.

Effects Modeling

A formal effects model results from careful restructuring of lower level objectives and the conceptual model. Two key decisions are (1) the selection of suitable risk assessment endpoints, and (2) how cause and effect will be analyzed and integrated.

Risk assessment endpoints tier to lower level objectives, and they must be place and time specific. For example, a well-focused wildland-urban interface endpoint may be to prevent any wildfire from entering within 1.0 mi of the town of Hayfork indefinitely. This specificity allows management alternatives to be tailored to real places. Further, spatial and temporal needs will influence the structural type of effects model used.

The effects modeling platform can incorporate an assortment of ancillary models, available data, and expert opinion. In CRAFT, that integrative platform is a Bayesian belief network that reveals assumptions, uncertainties, and likelihoods. Belief networks have been used for a range of natural resource planning issues (e.g., Lee and Irwin 2005, Marcot and others 2001). Belief networks have strict structural limitations that require a careful translation from the conceptual model. Relationships between parent and child nodes in the network can be calculated using multiple runs of deterministic ancillary models, analyses of existing data sets, or expert opinion. Each set of relationships provides a scenario that is defined by the model assumptions. The relative importance of each scenario in the model is then conditioned on the specified probability distributions of the nodes that have no parent nodes (i.e., the driver nodes). Importantly, the model can provide information even if no distributions are specified. Decisionmakers can readily explore the sensitivity of outcomes to different assumptions of the driver nodes.

The Hayfork Effects Model—

During the conceptual modeling phase, wildfire behavior was identified as a key node that was critical for wide

ranging objectives of concern. To illustrate how variability in fire behavior can be implemented in a belief network, we defined a number of vegetation-related variables and varied two others—fire weather and wind direction.

Fire behavior can be modeled by a wide array of models, but relatively few are useful for characterizing how fire behavior is likely to vary across the landscape. Landscape modeling is especially important in mountainous terrain such as our Klamath study area because patterns of vegetation and fuel, fire behavior, and winds are strongly influenced by topography. We used FlamMap (Finney 2006) to describe spatial patterns of fire behavior. FlamMap is not a dynamic fire spread model, and we did not calculate fire behavior using real-time fire weather or results that are conditional on spread from discrete ignition points. We essentially burned the entire landscape under fixed scenarios to limit the number of permutations and to transparently represent the assumptions used in our belief network.

The spatial information used in FlamMap includes slope, aspect, canopy cover, and surface fuel model. To model crown fire, we also used stand height, lower canopy height, and canopy bulk density. We derived topographic parameters (i.e., elevation, slope, and aspect) from a 30-m digital elevation model. Canopy cover was based on a 30-m, satellite-derived geographic information system layer from 2001 (<http://www.seamless.usgs.gov> [Date accessed unknown]). Other vegetation attributes were directly or indirectly derived from a U.S. Forest Service Region 5 existing vegetation layer from the mid-1990s that provided compositional and coarse age structural data in polygons of variable size. Stand height was calculated by using the dominant tree species in the classified polygons and height equations used by the Forest Vegetation Simulator (FVS, Dixon 2002). Canopy bulk density was calculated using published information for individual trees and was then multiplied by percent canopy cover. Canopy base height includes ladder fuels that are typically low across the study area owing to fire suppression (Taylor and Skinner 2003). Canopy base heights were assigned a value of 0.5 m in mature conifer stands and 0.2 m in young conifer plantations with a 0.8 live crown ratio for hardwoods. Fuel models were assigned by recasting the original 13 fuel models used

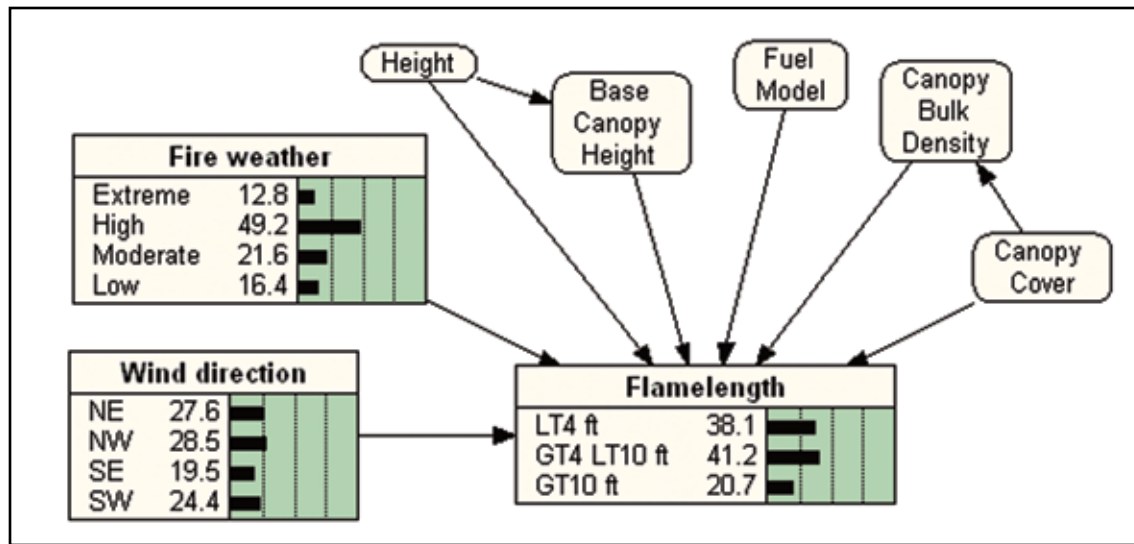


Figure 3—The probabilistic effects model used to characterize fire behavior (flamelenlength). For simplicity, only fire weather and wind direction were varied in this example. The probabilities for states of these two nodes were derived from retrospective analyses of the area burned during long-duration fires in northwest California, whereas the values in the node flamelenlength represent iterative calculations of the ancillary model, FlamMap.

in the Shasta-Trinity National Forest fire management plan for different vegetation types based on the new 40 fuel models (Scott and Burgan 2005). Although there are substantial uncertainties associated with most of these vegetation-related factors across the landscape, we fixed all of the foregoing parameters to focus on fire weather and wind.

Fire weather was modeled with four fuel moisture scenarios (i.e., 99th, 95th, 90th, and 80th percentiles) using two decades of fire weather collected at the Hayfork Ranger Station. Percentile conditions reflect the percentage of the time that weather conditions are likely to occur based on historical patterns. Data were manipulated using the software Fire Family Plus (Bradshaw and McCormick 2000). In FlamMap, fuel moistures were conditioned across topographic positions and aspects for 7 days using a daily range of temperature and humidity typical of the percentile conditions. Gridded windspeeds were paired with fuel moisture scenarios based on the speeds modeled on Hayfork Bally, the highest peak north of the weather station. Four fire weather scenarios were defined based on the fuel moisture conditions that characterized extreme, high, moderate, and low fire weather during the fire season (i.e., the 99th, 95th, 90th, and 80th percentile conditions). Gridded wind maps were assigned to these moisture conditions using

mountain-top windspeeds of 32, 24, 12, and 6 miles per hour (mph), respectively.

Wind is an important driver of fire behavior in mountainous terrain, and our study area is no exception. Extensive fire runs have been observed during east-wind events that are typically associated with frontal systems. Topography can greatly reduce wind speed and direction, but locally, this effect is often contingent on the regional wind direction. To address the effect of regional wind direction, we generated estimates of local winds using a beta version of WindWizard (developed by the U.S. Forest Service, Rocky Mountain Research Station, by B. Butler and J. Forthorfer). Whereas downbursts, fire-generated winds, and winds related to land use and gravity are ignored in this model, gridded wind reduces windspeeds according to topographic position consistent with observations. Gridded wind from four synoptic directions (i.e., northwest [NE], southeast [SE], southwest [SW], and northwest [NW]) was used in FlamMap to provide a better estimate of local wind direction and speed than simply modeling a uniform speed and direction across all topographic positions.

Every node in a belief network has a probability table associated with it, but values are derived differently, depending on where the node sits in the network. Nodes that

Table 1—Conditional probabilities of low (< 1.2 m), moderate (1.2 to 3.1 m), and high (> 3.1 m) across combinations of four fire weather and four wind direction scenarios for the study area

Fire weather	Wind direction	High	Moderate	Low
Extreme	NE	0.397	0.320	0.281
Extreme	NW	0.381	0.328	0.291
Extreme	SE	0.384	0.354	0.262
Extreme	SW	0.362	0.368	0.270
High	NE	0.243	0.403	0.353
High	NW	0.224	0.413	0.363
High	SE	0.242	0.421	0.337
High	SW	0.226	0.424	0.349
Moderate	NE	0.135	0.444	0.421
Moderate	NW	0.127	0.449	0.422
Moderate	SE	0.135	0.448	0.417
Moderate	SW	0.129	0.443	0.428
Low	NE	0.090	0.418	0.492
Low	NW	0.089	0.411	0.498
Low	SE	0.089	0.416	0.495
Low	SW	0.088	0.410	0.502
Joint probabilities		0.207	0.412	0.381

Joint probabilities reflect the relative likelihoods of these different scenarios occurring based on the priors shown in Figure 3.

are influenced by other nodes have conditional probability tables that show how probabilities change in response to changes in the parent nodes. These relationships are often calculated using ancillary models. Nodes that vary independently of other nodes must have their probabilities (or priors) specified. In our belief network, the probability distributions for fire weather and wind direction were based on analyses of daily fire progression maps of two recent and nearby long-duration fires—the Biscuit Fire and the Big Bar Complex. Fuel moistures calculated in Fire Family Plus were associated with daily area burned using midday fire weather data, and the fire spread direction of a stratified sample of points was used to approximate wind direction in terms of area burned.

Results of Effects Modeling—

Retrospective analysis of fire weather during the Big Bar long-duration Fire indicates that nearly half of the area burned when Fire weather was high (defined as fuel moistures ranging from the 93th to 96th percentile conditions during the May 1 through October 31 fire season). Roughly 22 percent of the area burned when conditions

were moderate (86th to 92th percentile), 16 percent burned when conditions were low (less than 86th percentile), and only 13 percent of the area burned when fuel moistures were extreme (97th to 100th percentile). These values were used as priors in the fire weather node of the belief network (Figure 3).

Analysis of fire spread during the Big Bar and Biscuit Fires showed that sites burned under a range of wind conditions, but winds from the NE and NW were associated with a greater area burned than were winds from the SE. An average of these two long-duration fires was used to define priors in the wind direction node of the belief network (Figure 3).

We calculated the conditional probabilities of fire behavior (i.e., flamelength) with multiple runs of FlamMap using different scenarios of fire weather and wind direction. For each scenario, we determined the area with modeled flamelengths less than 1.2 m (4 ft), 1.2 to 3.1 m (4 to 10 ft), and greater than 3.1 m (10 ft). These conditional probabilities are shown in Table 1. Wind direction has no appreciable effect on the overall pattern of expected flamelength as it varies by only a few percentages across wind scenarios.

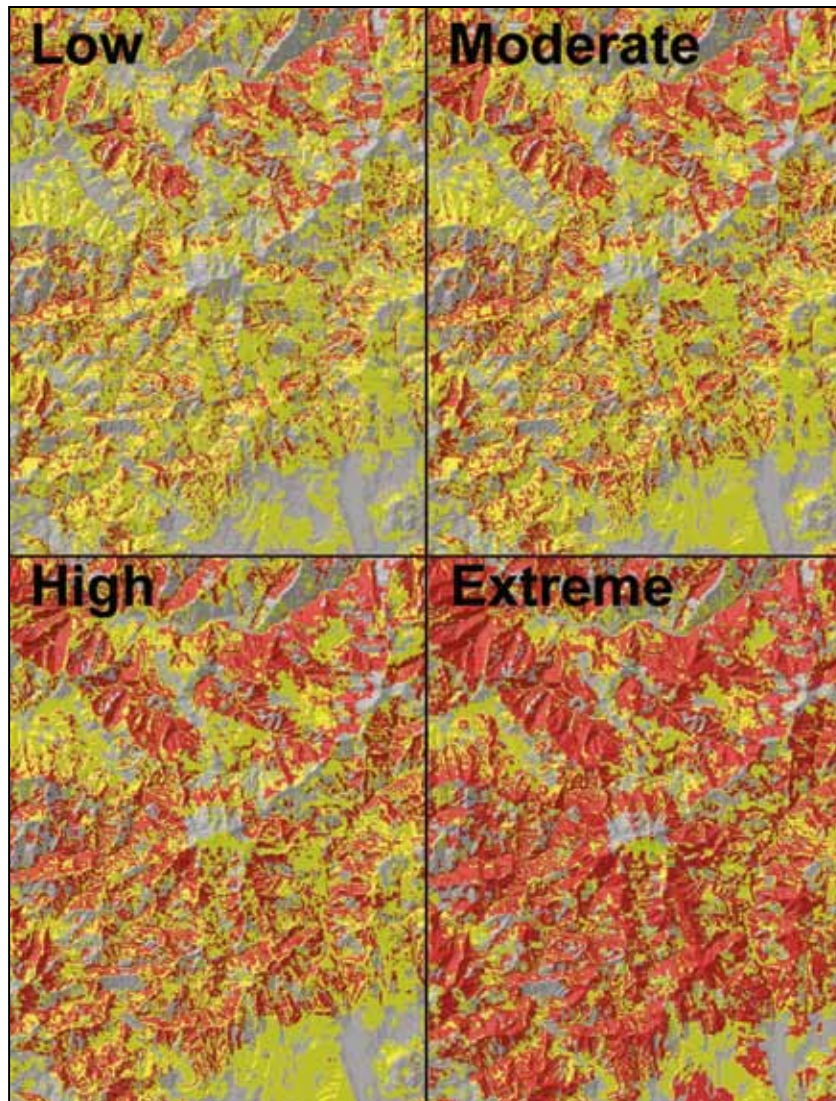


Figure 4—Maps showing conditional fire behavior (flamelenh) according to four fire weather scenarios for an 18- by 23-km portion of the study area. Flamelenhs were classified as low (shown in grey; less than 1.2 m), moderate (in yellow; 1.2 to 3.1 m), and high (in red; greater than 3.1 m). The values shown on each map were calculated with a weighted average of the four wind directions shown in Figure 2.

In contrast, fire weather has a substantive effect on flamelenh (Figure 4). When fire weather is low, half the area is expected to have flamelenhs that are low. These flamelenhs are likely to result in minimal mortality to mature trees while reducing surface fuels and fuel ladders. During extreme conditions, high flamelenhs dominate the landscape, and fires are likely to cause widespread mortality. Based on the joint probability distribution (Table 1), low- or moderate-severity flamelenhs are likely to occur,

presuming that future fire weather and winds are consistent with those of prior long-duration fire events. About 21 percent of the landscape can be expected to burn with flamelenhs greater than 3.1 m during low to moderate fire weather compared to 38 percent that is likely during extreme fire weather.

We then compared the effects of fire weather and wind on the flamelenhs of different portions of the study area. For example, the expected flamelenhs in 43 northern

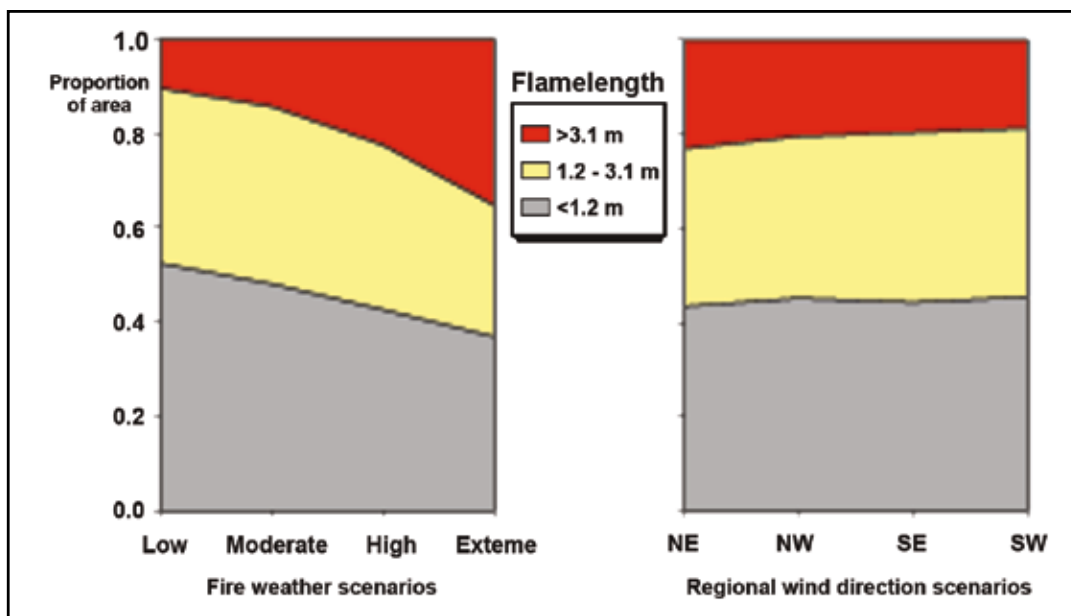


Figure 5—Change in fire risk for northern spotted owl habitat areas across fire weather and wind direction scenarios. High flamelengths shown in red are much more likely to be lethal to nesting trees than are low flamelengths. Note that under any modeled scenario, the habitat is likely to experience a range of flamelengths, as is typical of the region's mixed-severity fire regime.

spotted owl (*Strix occidentalis*) core areas is shown in Figure 5. Each core area is 60 ha (100 ac). Consistent with the overall pattern, the area expected to burn under different fire weather scenarios increases with the severity of fire weather, but flamelengths show little variability across the four modeled wind directions.

Summary

The simple-effects model shown here demonstrates how an important driver of forest outcomes—namely fire behavior—can be modeled in terms of conditional probability. This process transparently shows how assumptions of expected results may or may not matter. By varying vegetation and fuel inputs with proposed fuel treatment options, management outcomes can be considered within this same framework. Moreover, the uncertainties associated with the effects of different fire behaviors can be addressed with an expanded belief network.

The range of fire behavior that occurs during a single scenario in our model shows that the project area has a mixed-severity fire regime. This is consistent with retrospective analyses of past fires (Odion and others 2004,

Taylor and Skinner 2003). Large fires burn under a range of fire weather and wind scenarios, and this leads to further variation in fire behavior. Owing to the combined influence of these spatial and temporal patterns, the fire regimes of the Klamath Mountains are truly complex. This complexity is likely to have contributed to the high biodiversity that characterizes the Klamaths, but it makes multiresource management decisions more challenging.

To be meaningful, comparative risk assessments must be capable of accommodating this ecological complexity. The CRAFT process allows managers to incorporate diverse objectives and sophisticated effects modeling to deal with these complexities and their associated uncertainties.

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Probabilistic Risk Models for Multiple Disturbances: An Example of Forest Insects and Wildfires

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Abstract

Building probabilistic risk models for highly random forest disturbances like wildfire and forest insect outbreaks is a challenging. Modeling the interactions among natural disturbances is even more difficult. In the case of wildfire and forest insects, we looked at the probability of a large fire given an insect outbreak and also the incidence of insect outbreaks following wildfire. We developed and used a probabilistic model framework for estimating (1) the probability that a wildfire, at a given location and time, reaches a given size class under the conditions at the site—including history of insect outbreaks; and (2) the probability of an insect infestation at a given location and year under the conditions at the site—including history of fire occurrence and size. The study used historical data (1980 through 2004) on fire occurrence and forest insect outbreaks collected in Oregon and Washington. Spatial data on insect activity was obtained from aerial sketch maps created by the Forest Service Forest Health Protection program. Federal wildfire data obtained from the Desert Research Institute included information on the date, location, and size of the fire. Average monthly temperature and Palmer Drought Severity Indices were obtained from the National Climatic Data Center's climate division data set Web page. The methods employed provide an objective tool for modeling complex hybrid processes and estimating associated probability maps.

Keywords: Forest threats, multinomial regression, multiple stressors, nonparametric regression, spatial regression, spline functions.

Introduction

Wildfire and insect infestations are two major disturbances of forest lands in the United States. Historically, insect infestations and wildfires have had a dominant influence on successional processes in forests of the Western United States (Agee 2003). Fire suppression over the past 100 years has resulted in larger, more severe wildfires and insect outbreaks (Hessburg and others 1994). In 2005, over 0.14 million hectares (0.34 million ac) of Federal lands in Oregon and Washington were affected by wildfires (<http://www.nifc.gov>) and approximately 0.8 million hectares (2.1 million acres) sustained damage from insects such as bark beetles and defoliators (<http://www.fs.fed.us/r6/nr/fid>). In response to concerns over the size and severity of wildfires and insect outbreaks, Federal land management agencies have adopted forest management strategies that call for restoration activities that include reintroduction of natural and prescribed fire over wide areas of the Western United States. These activities have demonstrated beneficial effects in terms of moderating wildfire, but the potential effects on insect dynamics and insect-caused tree mortality are less clear.

A reciprocal and synergistic association has frequently been described between fire and insects. For example, tree mortality resulting from insect outbreaks has been seen as setting the stage for subsequent wildfires (Geiszler and others 1980, Parker and Stipe 1993). Conversely, wounding and mortality from fire can create focus trees, which act as magnets for bark beetles (McCullough and others 1998). In separate studies, McHugh and others (2003) and Cunningham and others (2005) followed tree mortality and beetle attacks for 3 years after wildfire and found bark beetles more likely to attack trees with fire injury. Similar results were found following prescribed fire (Bradley and Tueller 2001, Wallin and others 2003). However, others have found that beetles did not preferentially attack trees with fire-injured boles, but attack success was higher in injured trees when beetle population levels were low (Elkin and Reid 2004). Additionally, Sanchez-Martinez and Wagner (2002)

found low population levels of bark beetles regardless of stand treatment history, including stand replacement by wildfire. These results confirm that the association between insects and fire is a complex one, particularly when evaluated over time and at a large scale.

Numerous studies have begun to examine functional and numerical interactions between insects and fire at the tree and stand level, but few quantitative studies have been carried out that consider the spatiotemporal dynamics of wildfire and insect outbreaks at the landscape scale (Barclay and others 2005, Bebi and others 2003, Fleming and others 2002, Kulakowski and others 2003, Lynch and Moorcroft 2004, Veblen and others 1991). Existing analytical techniques for spatiotemporal analysis of multiple interacting disturbances are not well developed, and large-scale data sets suitable for studying interactions are few (Lynch and Moorcroft 2004).

The present study describes a framework for estimating probabilities, at a landscape level, of various forest disturbances as a tool for quantifying associations between multiple interacting disturbances. The framework builds on related work by the authors and collaborators on estimating wildfire probabilities in relation to weather and fire danger indices and spatial location (Brillinger and others 2003, Preisler and Westerling 2007, Preisler and others 2004).

Methods

Data

Our study area was Oregon and Washington national forest lands. We obtained historical Federal wildfire occurrence data (1980 through 2004) from Desert Research Institute (DRI), <http://www.dri.edu>. The DRI fire occurrence data are based on the National Interagency Fire Management Integrated Database (NIFMID) at the USDA Forest Service National Information Technology Center in Kansas City, Missouri. The data included information on the date, location, and size of the fire. The DRI version of the data had been subjected to a quality-control procedure in which each fire-occurrence record was flagged as usable or otherwise (Brown and others 2002). Fire locations are the latitude and longitude of the fire when first discovered.

We obtained average monthly temperature and Palmer Drought Severity Indices (PDSI) from U.S. Climate Divisions (NCDC 1994). The values at the climate-division level were projected onto a 1-km² grid to provide a monthly climate record for each grid cell. Variables at the climate division level may not be good estimates for local weather conditions, but they were used here to demonstrate how weather variables can be included in the model.

Spatial data on insect activity was obtained from aerial sketch maps created by the Forest Service Forest Health Protection program (<http://www.fs.fed.us/r6/nr/fid>). In particular, we obtained data on (1) total number of trees (lodgepole pine, *Pinus contorta* (Dougl. ex Loud.), and ponderosa pine, *P. ponderosa* (Dougl. ex Laws.) killed by two bark beetle species (mountain pine beetle, *Dendroctonus ponderosae* Hopkins, and western pine beetle, *D. brevicornis* LeConte) per year (1980 through 2004) in each square milometer grid of Forest Service lands (Region 6); and (2) total area defoliated by western spruce budworm (*Choristoneura occidentalis* Freeman) in each square kilometer of Forest Service land per year.

In this study we did not use information on fire boundary. Consequently, the models described below are used for estimating associations between histories of fire sizes and insect infestations within 1 km of each other. Additional predictors (e.g., distance to nearest infestation or fire) may improve estimation results. In this study, we are mainly concerned with describing the statistical framework given a set of predictors.

Probability Framework

We are interested in obtaining estimates of disturbance probabilities in the presence of multiple stressors. For example, we would like to quantify:

Prob {damage from disturbance 1 given the size of damage from disturbance 2 and values of other predictors}

Two specific disturbances, bark beetle infestation and fire size, were analyzed in the present case study. Fires of all sizes were used in the analysis.

Bark Beetle Infestation—

Let Y be a random variable such that $Y_i = 1$ if an infestation is present in a square kilometer grid at location ($utmx_i$, $utmy_i$) and year yr_i and $Y_i = 0$ otherwise. The random variable Y is assumed to follow a Bernoulli distribution with probability of response

$$\pi_i = \Pr[Y_i = 1 | \mathbf{X}_i] = e^{\theta(\mathbf{X}_i)} / [1 + e^{\theta(\mathbf{X}_i)}] \quad [1]$$

where $\theta(\mathbf{X}_i) = \beta_0 + g(utmx_i, utmy_i) + \sum g_k(X_{ik})$

and where the matrix $\mathbf{X}$ of predictors includes history of fire and insect infestation in previous years and within 1 km of infestation. The functions $g_k(X_k)$ in equation [1] are transformations of the variable X_k using linear-basis splines (Hastie and others 2001). For the spatial component, $g(utmx, utmy)$, we utilized the two-dimensional version of the basis function, specifically, the thin plate spline function. All estimations were done within the statistical package R (R Development Core Team 2004). The required modules for fitting thin plate splines within R were downloaded from the Web (Geophysical Statistics Project 2002). By including variables such as location and history of insect infestation in the model, we were able to study the effect of fire (occurrence and size), on the probability of an infestation, in the presence of confounding factors (e.g., presence of infestations in previous years). The spatial component in the model can be viewed as a surrogate for other predictors (e.g., elevation, fuel type, and other variables) that do not change substantially over time and are not part of the model for a variety of reasons (Hobert and others 1997). Because the number of cells with no insect damage was very large (2.4 million), only a random sample of insect-free locations was included in the analysis. The final probabilities were then adjusted accordingly (Maddala 1992, Preisler and others 2004).

Results of fitting equation [1] were expressed as maps of estimated probabilities (and estimated standard errors) that are spatially explicit on 1-km² grid cells. The maps may be used to forecast probabilities of occurrence of an insect infestation for a succeeding year, given the history of fire and insect infestation up to that year. Outputs can be appraised directly by comparing observed frequencies of occurrences with predicted probabilities using cross-validation.

Using this model, we were able to quantify the interactions between different disturbances by studying the significance and shape of the relationships between the probability of an infestation and the occurrence and size of fire in previous years. In particular, we produced estimates of the odds of an infestation as a function of fire size relative to the odds when no fire is present. Odds were defined as $\pi/(1 - \pi)$, where π is the probability of a response of interest, in this case, bark beetle attack.

Fire Size

To study the association between insect disturbances and the risk of wildfire becoming large at a given location, we divided fires into three size classes. Consider the ordinal random variable Y where $Y_i = 1$ if a fire at location ($utmx_i$, $utmy_i$) and time t_i burns an area less than or equal to 1 ha; $Y_i = 2$ if area burned is 1100 ha; and $Y_i = 3$ if area burned is greater than 100 ha. Next, assume that the random variable, Y , follows a multinomial distribution with probability of response at level k ($k = 1, 2, 3$), given by $\pi_{ik} = \Pr[Y_i = k | \mathbf{X}]$ and $\pi_{i1} + \pi_{i2} + \pi_{i3} = 1$. The list of predictors, $\mathbf{X}$, included average monthly temperature and PDSI, size of bark beetle infestation (trees killed per square kilometer), and size of spruce budworm infestation (area defoliated per square kilometer) in the previous 6 years. We obtained estimates for π_{ik} by fitting a complementary log-log function to the conditional probabilities of response. Specifically, we used the model

$$p_{ik} = \Pr[Y_i = k | Y_i > k-1, \mathbf{X}_i] = 1 - \exp[-\exp[\beta_0 + g(utmx_i, utmy_i) + \sum g_j(X_{ij})]] \quad [2]$$

where the functions $g(utmx_i, utmy_i)$ and $g(X)$ are as defined in (“Bark Beetle Infestation”). We obtained estimates of the multinomial probabilities from the conditional probabilities using the relationships $\pi_{i1} = p_{i1}$; $\pi_{i2} = (1 - p_{i1})p_{i2}$; and $\pi_{i3} = (1 - p_{i1} - p_{i2})$. As in section (“Bark Beetle Infestation”) the model goodness-of-fit was appraised by comparing observed and predicted frequencies of fires in the three different size classes.

We produced maps of estimated probabilities displaying the spatial patterns of fires of different size classes. We also studied association between insect infestations and fire size by producing graphs of estimated odds of a fire of a

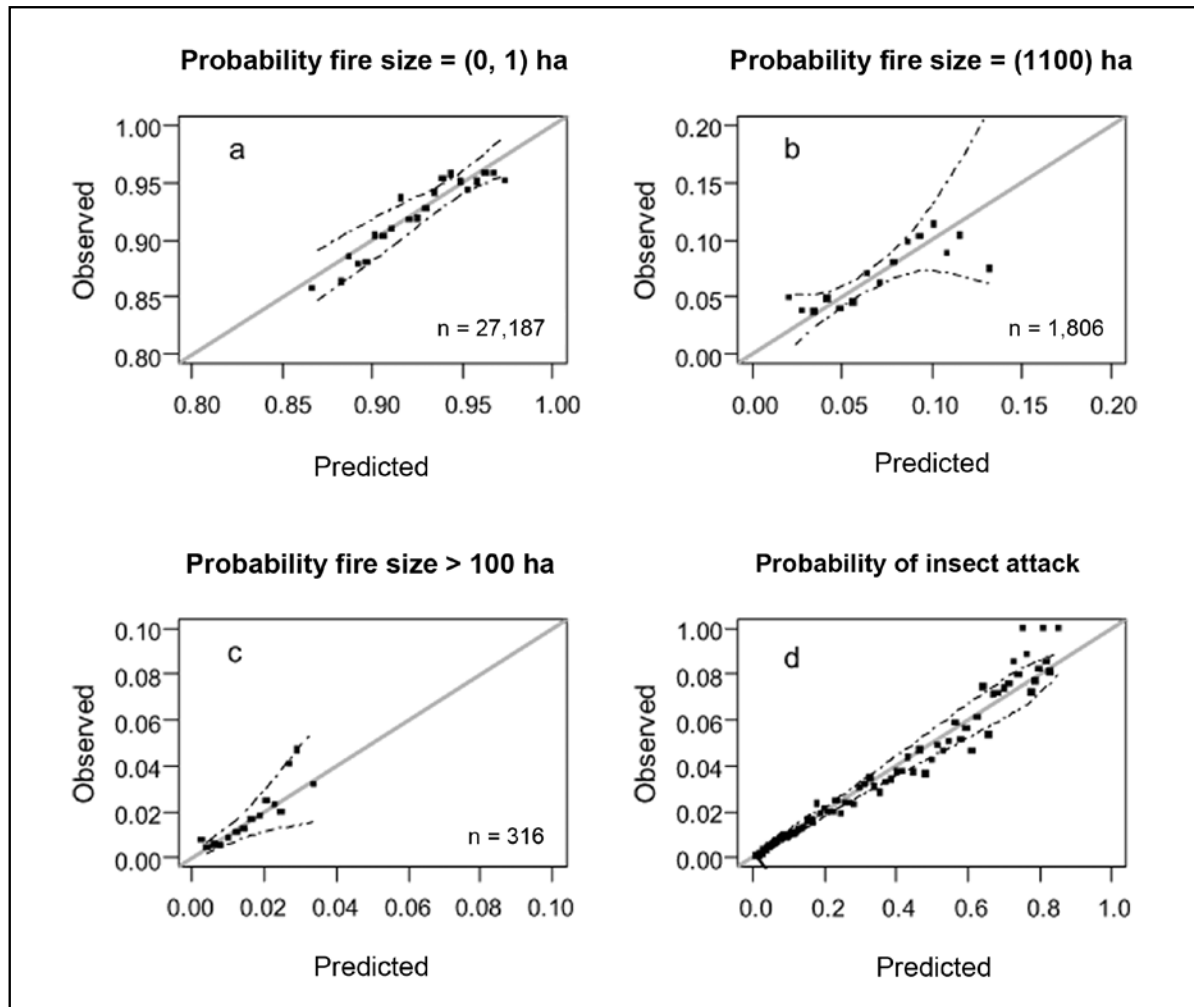


Figure 1—Observed versus predicted multinomial probabilities of a fire reaching size class (a) (0 to 1] ha, (b) (1 to 100] ha, (c) >100 ha. The value n in each panel is the total number of fires in the corresponding size class during the study period. (d) Observed versus predicted probability of a bark beetle occurrence in a given 1-km² grid cell.

given size class—plotted against infestation size—relative to the odds when no infestation is present.

Results and Discussion

The models in equations [1] and [2] appeared to give reasonable fits to 1982 to 2004 data on bark beetle attacks and wildfire sizes on Forest Service lands in Region 6 (Figure 1). For example, consider all cells over the years when the model in equation [1] predicted a 40-percent probability of a beetle attack (bottom right panel, Figure 1), the actual percentage of time an attack was observed was 37 percent, which is well within the 95-percent limits (36 to 40 percent) of the predicted probability.

Bark Beetle Infestation

The probability of beetle infestation was significantly influenced by spatial location, size of infestation in previous year, and size of 1-year-old fire within 1 km (P -value < 0.05). The estimated probabilities of bark beetle outbreaks, evaluated after controlling for all other significant predictors, demonstrated a spatial pattern with increasing probabilities of outbreak as one moves away from the coastal areas and into the eastern regions of Washington and Oregon (Figure 2). This pattern of declining forest health in the dry coniferous forests of eastern Oregon and Washington has been noted by others in recent assessments

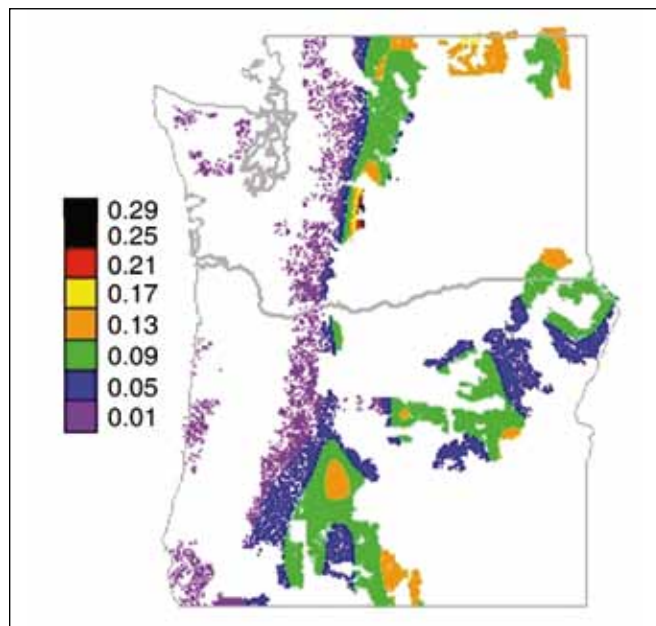


Figure 2—Estimated spatial pattern of beetle attacks on national forests in Washington and Oregon.

and syntheses (Gast and others 1991, Hayes and others 2001, Hessburg and others 1994, Jaindl and Quigley 1996).

One of the most important predictors when evaluating a bark beetle outbreak appears to be size of the beetle infestation in the previous year. The odds of a beetle outbreak seem to increase as the number of trees killed per square kilometer by bark beetles in the previous year increases (Figure 3). Although this outcome is probably expected, given the outbreak dynamics of insect populations in general (e.g., Barbosa and Schultz 1987), it is important to know that it was included in the model and, consequently, is accounted for in a reasonable fashion. The other significant predictor was the size of fire in the previous year. The odds of an insect outbreak appeared to increase as the size of a fire increases from 0 (no fire) to ~750 ha, after which the odds appear to decrease. The standard errors are large, especially for large fire sizes. This result is consistent with the expectation that as fire increases in size, the number of trees with significant crown damage and bole or root scorch increases, and, therefore, susceptibility to bark beetle attack may increase. However, very large fires may completely eliminate susceptible hosts in the area, resulting in less observable beetle damage when large fires are nearby.

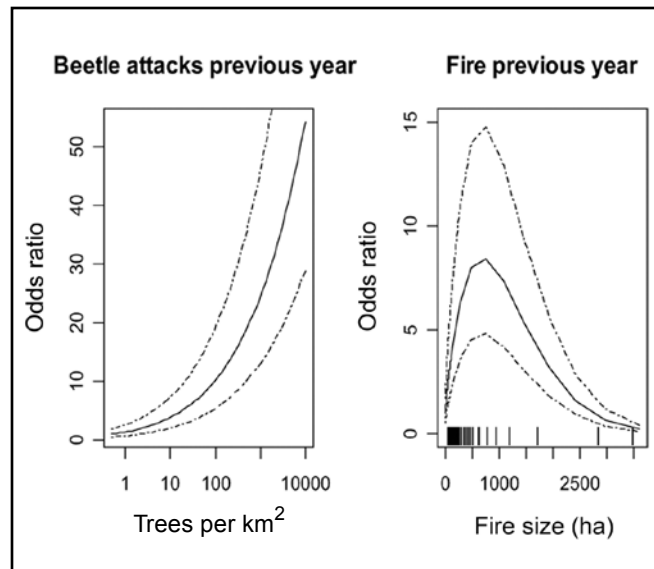


Figure 3—Estimated odds ratio (solid curves) and 95-percent confidence bounds (dashed curves) describing the increase in odds of a beetle attack (a) as the number of trees killed by beetles in the previous year increases from 0 to 10,000 trees/km² (left panel) and (b) as the size of a fire in the previous year, and within 1 km of the infestation, increases from 0 to > 2,500 ha (right panel). The hatched marks in the bottom of the right panel indicate distribution of observed fire sizes. Note that only four fires of size > 1,000 ha were observed in the sampled data.

Fire Size

The following predictors were found to have significant effects (P -value $\ll 0.05$) on the multinomial probabilities of a fire reaching one of three size classes: spatial location; size of beetle infestations in previous 3 years; size of spruce budworm infestations 4 to 6 year ago; average monthly temperature; and average monthly PDSI.

The estimated spatial pattern of fire sizes (Figure 4) seems to indicate northern Washington and eastern Oregon as some of the regions with the highest probabilities of a fire getting large after controlling for all other predictors in the model. Again, these results appear to be consistent with observed changes over the last 60 to 100 years in fire frequency and size in interior forests of the Pacific Northwest (Hessburg and Agee 2003).

According to the present data, beetle infestations that are 3 to 6 years old do not seem to be significantly associated with fire size; however, the odds of a fire getting large seem to increase significantly when the total size of beetle infestation in the last 3 years is greater than 5,000 trees

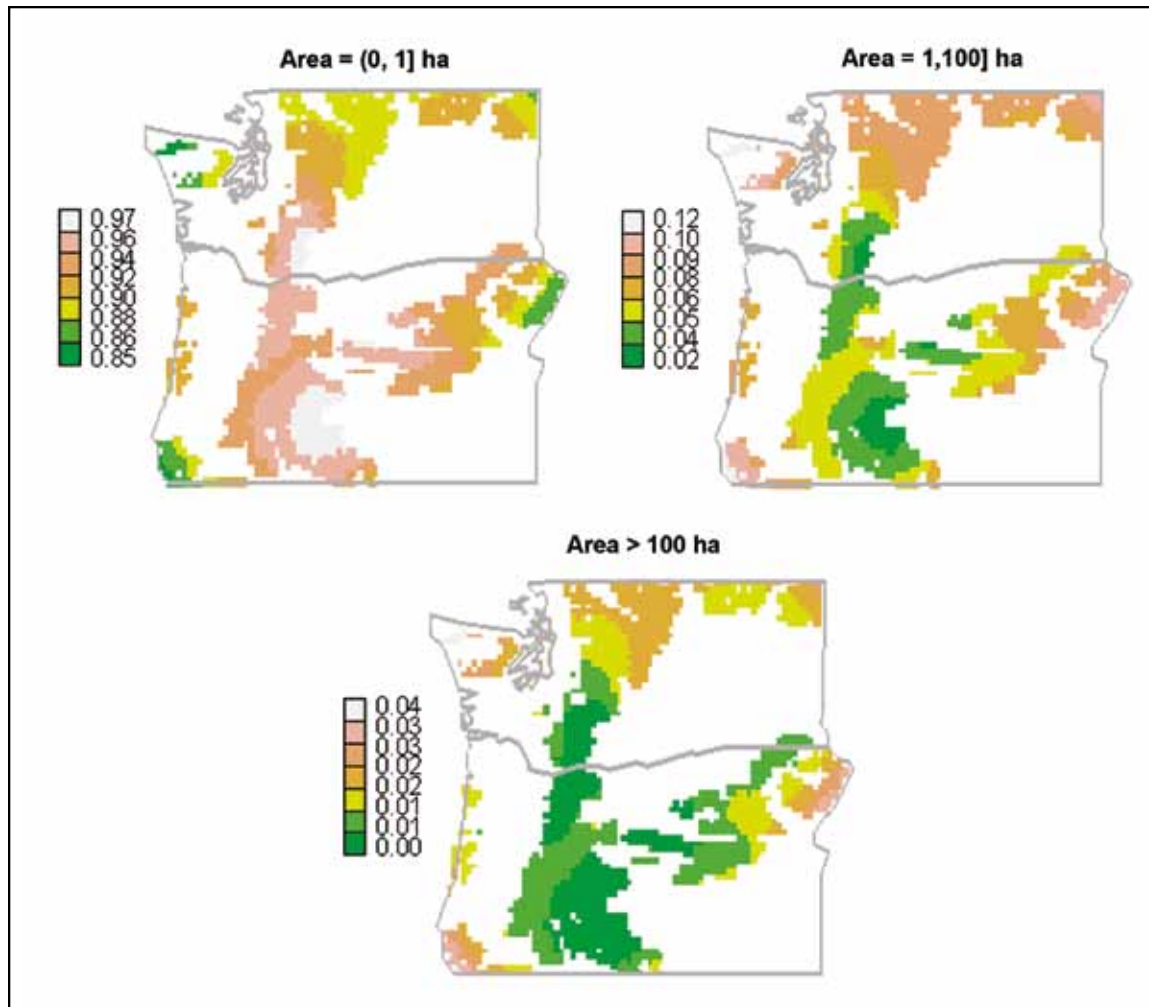


Figure 4—Probability of fire in one of three size classes on national forests in Washington and Oregon, 1970-2004.

killed per square kilometer (Figure 5, top panels). Canopies with residual dead needles and limbs in recently killed trees may contribute to crown conditions that significantly influence fire behavior. Following a simulated outbreak, Ager and others (in press) found that beetle-caused tree mortality increased surface fuel loadings and the potential for stands to sustain crown fire in the simulation. A different association was seen between fire size and size of spruce budworm infestation. No significant association was apparent between recent defoliations (area defoliated in previous 3 years) and sizes of fires nearby. However, a significant decrease was seen in the odds of a fire getting large as the size of older defoliations (total area defoliated 3 to 6 years ago) increased from a total of 100 to 300 ha (Figure

5, bottom panels). Lack of foliage following defoliation 4 to 6 years prior may actually reduce the risk of crown fire. Although the decrease in odds appears to continue as the size of defoliated area increases past 300 ha, the standard error for this range of defoliation is too large to warrant a precise conclusion.

Conclusions

Multiple disturbances, such as wildfire-insect outbreak interactions, are not well understood at provincial scales. Lack of large-scale data sets and easily available analytical techniques to address multiple disturbances have contributed to a limited number of quantitative studies. These studies have had confounding results. Here, we have

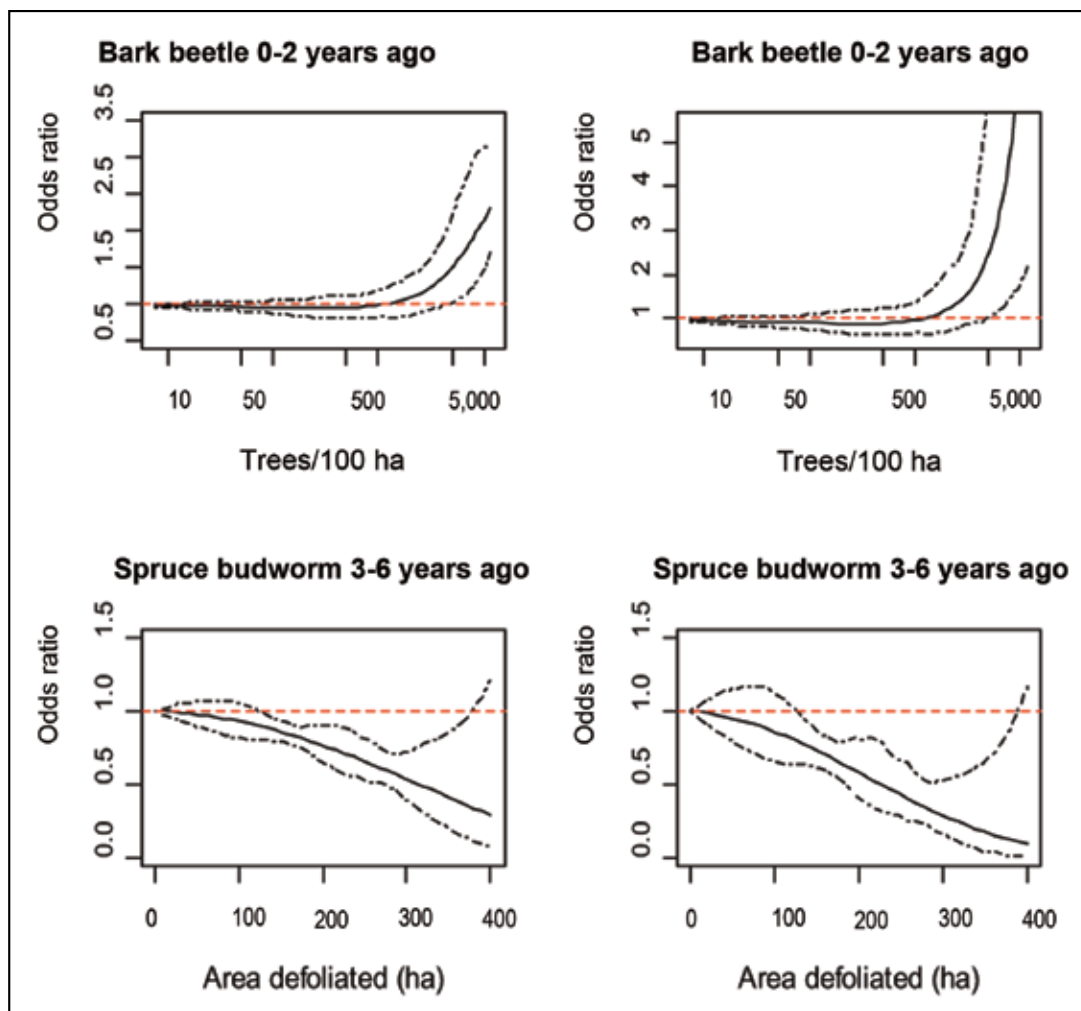


Figure 5—Estimated odds ratio (solid lines) and 95-percent confidence bounds (dashed lines) of fire reaching size class 2 or 3 plotted against values of two predictors (bark beetle and spruce budworm). The odds ratio is relative to when there is no insect infestation. The red dashed line indicates an odds ratio of one. When the red line is within the 95-percent bounds, the odds are not significantly different from the odds when no insect infestation is present.

described a probabilistic model framework that was used in a case study of spatiotemporal interactions between wildfire and insect outbreaks in the Pacific Northwest. To create the model, we used historical spatial data (1980 to 2004) on fire occurrence and insect outbreaks on forested lands of Oregon and Washington along with average monthly temperature and PDSI.

Results of our analyses seem to imply an increasing risk of new bark beetle infestations with increasing size of nearby fires (within 1 km) up to fires of ~750 ha. The latter was indicated by the estimated odds relative to locations with no fire damage. Also, there appears to be a decrease in

risk of a fire getting large at locations with 3- to 6-year-old spruce budworm damage of size greater than 100 ha.

The methods we employed provide an objective tool for modeling complex processes with time lags in the predictors and estimating associated probability maps for competing risks (multiple disturbances). The framework is especially amenable to quantifying uncertainties from multiple sources (stochastic disturbance, sampling and measurement errors, approximate models, and missing predictor variables), which are essential for model appraisal purposes. The analyses can be expanded to other areas using the rich, spatially explicit, historical data set available for wildfire

occurrence across all of the United States (NIFMID). It also has application for forest insects and diseases through Forest Health Protection annual aerial survey data. Although we only examined three insect species (mountain pine beetle, western pine beetle, and western spruce budworm), other stressors could be considered, including nonnative invasive species and stress level owing to pollution. Risk assessment maps can be produced by multiplying the estimated probabilities by a loss function.

Acknowledgments

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Establishing a Nationwide Baseline of Historical Burn-Severity Data to Support Monitoring of Trends in Wildfire Effects and National Fire Policies

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Abstract

There is a need to provide agency leaders, elected officials, and the general public with summary information regarding the effects of large wildfires. Recently, the Wildland Fire Leadership Council (WFLC), which implements and coordinates National Fire Plan (NFP) and Federal Wildland Fire Management Policies adopted a strategy to monitor the effectiveness and effects of the National Fire Plan and the Healthy Forests Restoration Act. One component of this strategy is to assess the environmental impacts of large wildland fires and identify the trends of burn severity on all lands across the United States.

To that end, WFLC has sponsored a 6-year project, Monitoring Trends in Burn Severity (MTBS), which requires the U.S. Department of Agriculture, Forest Service (USDA-FS) and the U.S. Geological Survey (USGS) to map and assess the burn severity for all large current and historical fires. Using Landsat data and the differenced Normalized Burn Ratio (dNBR) algorithm, the USGS/EROS Data Center and USDA-FS/ Remote Sensing Applications Center will map burn severity of all fires occurring from 1984 to 2010. Only fires that are greater than 500 ac in the East, and 1,000 ac in the West will be included. We anticipate mapping a total of more than 9,000 historical fires and fires that occur during the course of the study.

The MTBS project will generate burn-severity data, maps, and reports, which will be available for use at local, State, and national levels to evaluate trends in burn severity and help develop and assess the effectiveness of land management decisions. Additionally, the information developed will provide a baseline from which to monitor the recovery and health of fire-affected landscapes over time. Spatial and tabular data quantifying burn severity will augment existing information used to estimate risk associated with a range of current and future resource threats. For example, fire severity data along with associated biophysical characteristics provide an analytical basis for assessing risk from invasive species as well as native insects and pathogens. All data and results will be distributed to the public via a Web interface.

Keywords: Burn severity, Landsat, monitoring, spatial, temporal, trends, wildland fire.

Introduction

Consistent geospatial information characterizing effects of large wildland fires does not exist for lands within the United States. Changing trends in fire frequency, severity, and size have resulted in the need to acquire data and develop information that can establish a baseline for trend analysis and begin to look at recent historical shifts in these fire characteristics. Furthermore, there is a need to understand the impacts of fire and resource management policies on fire occurrence and severity (Stephens and Ruth 2005). These needs are recognized across agencies and at various levels within land management organizations. Moreover, the general public is increasingly exposed to information suggesting that increases in uncharacteristic fire are due in part to past land management practices. It can be assumed that public interest in current and future fire policy will increase.

The Wildland Fire Leadership Council (WFLC), a national-level interagency body with responsibility for implementing and coordinating the National Fire Plan (NFP) and Federal Wildland Fire Management Policies (<http://www.forestsandrangelands.gov/>), has adopted a strategy to monitor the effectiveness and effects of the NFP

and the Healthy Forests Restoration Act. One component of this strategy is to assess the environmental impacts of large wildland fires and identify the trends of fire severity on all lands across the United States (WFLC 2004 Monitoring Proposal, Module 2.1). In 2004, the Government Accountability Office recommended that the Forest Service and Bureau of Land Management develop and implement comprehensive assessments of fire severity to provide consistent summary information characterizing the environmental effects of wildland fires and to meet the requirements of WFLC.

Beyond the needs of WFLC, it is widely recognized that nationally consistent and current data are necessary to address issues of ecosystem health and sustainability. The National Report on Sustainable Forests (USDA Forest Service 2004) describes criteria and indicators that are an important framework for sustainable management. Whereas the report does not specifically address the need for fire occurrence and effects data, it is easy to see both the direct and indirect relationships that exist between these criteria and indicators and the understanding of spatial pattern, magnitude, and frequency of fire in ecosystems across the United States. Specifically, the criterion addressing the maintenance of forest ecosystem health and vitality and the criterion addressing maintenance of forest contribution to global carbon cycles are sensitive to the spatial and temporal variability of fire effects and the gradient of change in forest ecosystems.

Project Background

In 2006, WFLC sponsored a 6-year project to map the fire severity and perimeters on large fires (>500 ac in the East and 1,000 ac in the West) in the United States across all ownerships for the period of 1984 through 2010. The project is referred to as the Monitoring Trends in Burn Severity (MTBS) project and is implemented jointly by the U.S. Geological Survey (USGS), National Center for Earth Resources Observation and Science (EROS), and the USDA Forest Service, Remote Sensing Applications Center (RSAC). This work is an extension of the existing cooperation between these two national centers that has provided rapid-response burn-severity mapping products to Forest

Service and Department of the Interior Burn Area Emergency Response teams.

The primary objective of this project is to provide for a national analysis of trends in fire severity for the NFP. Due to severe periodic droughts, increased fuel loads, and a higher frequency of uncharacteristic fires in recent years (since 2000), it is essential for the trend analysis to span a significant period of time to better account for variability in factors potentially affecting fire severity (e.g., climate). Secondary objectives include providing geographic and fire-specific data for use at regional and subregional scales to support resource and risk assessments, resource management planning, project planning and implementation, monitoring, and research activities. Sufficiently fine spatial and thematic resolution is necessary to support the wide range of operational and research-related information needs at larger scales.

This project will serve four primary user groups with one set of data and information:

- National policymakers such as WFLC, that require information about long-term trends in burn severity and recent burn severity within vegetation types, fuel models, condition classes, and treatment accomplishments.
- Field managers that benefit from geographic information system ready maps and data for informing and supporting prefire and postfire management decisions and monitoring.
- Project managers for existing databases such as LANDFIRE and the National Land Cover Data set that benefit from burn-severity data produced at comparable spatial scales and resolution for validation and updating of geospatial data sets.
- Academic and agency research entities interested in fire-severity data over significant geographic and temporal extents.

Burn Severity Definition

Terminology commonly used when discussing fire behavior and fire effects is often inconsistently and interchangeably applied. Inconsistent definitions associated with risk,

hazard, and severity confound our ability to characterize and communicate postfire effects and their implications to resource and cultural values (Hardy 2005). Data and information developed by this project are intended to primarily characterize fire effects in aboveground biomass. Despite significant variation in published definitions of burn severity and fire severity (Lentile and others 2006), postfire effects in aboveground biomass have been described using both terms. The term burn severity as it applies to this project is best represented by the definition for fire severity in the National Wildfire Coordination Group Glossary of Wildland Fire Terminology, stated as the “Degree to which a site has been altered or disrupted by fire; loosely, a product of fire intensity and residence time” (NWCG 2005). The following additional statements have been adopted to further clarify the nature of the products developed by this project:

- Burn severity is a composite of first and second-order fire effects on biomass.
- Occurs on a gradient or scale (ordinal).
- Occurs within a fire perimeter.
- Occurs within landcover strata.
- Longer term effects are complicated by variables that this project is not characterizing.
- Severity is mapable.
- Remote sensing provides a measurement framework.

Project Scope

The project has been divided into geographic mapping zones representing broadly similar ecological conditions. The mapping zones illustrated in Figure 1 were created from aggregations of National Land Cover Data set (NLCD) mapping zones originally derived from Bailey’s ecological sections (Homer and others 2004). The primary purpose of the mapping zones is to provide ecologically meaningful processing areas that are also efficient production units. Secondary consideration was given to significant administrative boundaries where they correlated closely with ecological unit edges. We recognize that application of the products will occur at a variety of ecological and administrative extents, and the analysis and summarization of data

for the primary sponsors may have limited utility at larger scales. However, use at larger scales is both appropriate and technically feasible owing to the spatial and thematic resolution inherent in the products. Furthermore, the spatially aggregate nature of fires and the ability to easily identify discreet events within the product sets allow for analysis ranging in extent from a single fire to multiple fires spanning space or time, or both. This is not to say that all analysis scales will be supported by these data. Indeed, as with all geospatial data, there are limits to effective application, and users will be encouraged to consider their analysis objectives and information needs relative to the spatial and thematic characteristics of the products.

Burn-severity mapping is being conducted in two time phases. Fires occurring in 2004 through 2010 are considered current and will be mapped and reported annually for the entire project extent. Historical fires occurring from 1984 through 2003 will be mapped, analyzed, and reported by mapping zone through the duration of the project. Mapping zones have been prioritized based on fire frequency, acres affected, and data availability. Figure 1 also illustrates processing schedules for historical fires by mapping zone.

The historical range of this project (1984 through 2010) was determined on the basis of availability of remotely sensed data necessary to consistently characterize the extent and severity of individual fires. A longer historical period would afford the ability to more precisely analyze environmental and policy-based influences on wildland fire that are significant over decades and centuries. Recent land and fire management activities may be the most directly relatable influences on fire effects as depicted by these products. Beyond the scope of this project, but a valuable extension, would be the use of other, historically extensive sources of remotely sensed data to extend the fire effects record generated by MTBS, albeit with compromises in consistency.

Products

Products for the MTBS project fall into three categories: unclassified (input) data, geospatial layers and maps (raster and vector), and summary analysis.

Unclassified or input data are comprised of Landsat Thematic Mapper (TM) and Enhanced Thematic Mapper



Figure 1—Monitoring Trends in Burn Severity mapping zones. Historical fires are scheduled to be mapped during the fiscal years illustrated within each zone.

(ETM) images that form the basis for measuring spectral response of individual fires. Methods used to process and classify these data will be discussed in more detail under section 2. These data have been processed by USGS EROS through the National Landsat Archive Production System (NLAPS) (Huang and others 2002) and are representative of the level and format of Landsat data typically delivered to the scientific and operational communities.

A series of geospatial layers make up the intermediate and final products characterizing postfire spectral response, burn severity, and fire perimeters. The following are the principal outputs:

- 30-m resolution Normalized Burn Ratio (NBR) calculated from Landsat (prefire and postfire);
- 30 m-resolution, continuous differenced NBR (relative and absolute).
- 30-m resolution, arithmetic partitions of dNBR.
- Vector format fire perimeter based on dNBR.
- Metadata for geospatial data.

Analysis outputs are necessarily limited in scope to achieve the primary objectives of the project. In depth trend analysis, correlations to other factors, including climate change and management practices, and implications for other resources fall outside the scope of this project. Formats and resolution of the geospatial products are designed to allow flexibility for application to a wide range of analysis objectives that pertain to burn severity. Indeed, it is expected (and desired) that these data will be used in broad- and moderate-scale research and management activities

where a consistent data record of postfire effects is valuable. Primary analysis products delivered by this project include:

- Tabular data summarizing acres burned by severity class.
- Tabular data summarizing acres burned by severity classes and vegetation cover types (where available).
- Tabular data summarizing acres burned by severity classes and presence or absence of fuel treatments (where available).

Methods

Methods selection for this project was fundamentally driven by two requirements: (1) The need to develop consistent information across all lands within the project extent, and (2) the need to develop consistent information spanning a significant historical period. Based on these requirements, remotely sensed data were considered to be the only cost-effective and spatially resolved source to consistently delineate and measure the response of thousands of individual fires across a continental extent and multidecade timeframe. A significant body of literature exists evaluating the effectiveness of various scales of remotely sensed data to characterize fire severity (Brewer and others 2005, Chuvieco and Congalton 1988, Diaz-Delgado and others 2003, Fernandez and others 1997, Justice and others 1993, Kasischke and French 1995, Key 2005, Milne 1986, Patterson and Yool 1998, Pereira 1999, Roy and Landmann 2005, Sa and others 2003, Smith and others 2005, Sunar and Ozkan 2001, Wagtendonk and others 2004, White and others 1996). Scientific and operational precedent exists for the use of a remote sensing-based approach.

The Landsat TM and ETM data provide the longest record of relatively high spatial and spectral resolution data for mapping fire severity. Not only does this enable the mapping of historical fire severity, it also facilitates the use of multitemporal approaches for characterizing postfire effects. Landsat data have been shown to be responsive to relative changes in aboveground biomass as a result of fire (Epting and others 2005, Kushla and others 1998, Lopez-Garcia and Caselles 1991, Miller and Yool 2002). More specifically, multitemporal change detection approaches

based on prefire and postfire Landsat data have proven to be a cost-effective and relatively accurate means of mapping fire severity (Brewer and others 2005). Furthermore, the availability and low cost of Landsat data were additional factors supporting their use for a project of this geographic and temporal extent.

Multitemporal approaches applying image ratios and image differencing techniques to Landsat data have been developed for a variety of assessment objectives. Imagery is commonly transformed mathematically into indices by ratioing a spectral component(s) or band with another spectral component(s) for each pixel. The transformation of Landsat data into vegetation indices (e.g., Normalized Difference Vegetation Index) to strengthen the relationship between spectral response and vegetation characteristics has been widely used, and a number of indices exist (Lyon and others 1998). Lopez-Garcia and Caselles (1991) published the first index specifically derived to enhance the relationship between Landsat spectral response and burned vegetation. This Normalized Difference index was subsequently adapted and operationally implemented by Key and Benson (2002) and was used to develop historical fire-severity data and atlases on numerous national parks. The approach has been named the Normalized Burn Ratio (NBR) and, combined with multitemporal differencing, has been utilized in fire-severity mapping efforts by the USGS and the USDA Forest Service since 2002. The NBR is calculated as $(TM4 - TM7)/(TM4 + TM7)$ where TM4 represents the near infrared spectral range and TM7 represents the short wave infrared spectral range. A differenced NBR image (dNBR) is created by subtracting the prefire NBR image from a postfire NBR image.

The dNBR data have been used operationally for both rapid response and longer term assessment and monitoring (Bobbe and others 2003, Gmellin and Brewer 2002, Key and Benson 2002). Rapid response needs require the use of immediate postfire imagery to show first order fire effects on vegetation and soils and to prioritize rehabilitation resources. Longer term assessments have relied on image data typically acquired during the growing season following the fire in order to include delayed first order effects (e.g., delayed tree mortality) and dominant second order

Table 1—Brief description of database elements contained within the Monitoring Trends in Burn Severity (MTBS) fire history database

ID	Fire name		Agency	Year	Start date
Unique MTBS ID that includes source ID	Incident name from the source database		Reporting agency from the source database	Year occurred from the source database	Incident start day/month/year from the source database
Reported acres	Long	Lat	Path	Row	Disposition
	Incident acres from the source database	Longitude	Latitude	Landsat path	Landsat row

effects that are ecologically significant (e.g., initial site response and early secondary mortality agents). Extended assessments are intended to provide a more comprehensive ecological indication of fire severity. Both initial and extended assessments have uncertainty associated with the dNBR-based fire-severity characterizations. Prefire vegetation conditions and postfire management activity influence the nature and magnitude of this uncertainty. The sensitivity of a given set of analysis objectives to the uncertainties associated with initial and extended assessment dNBR data should be considered when using these data.

Based on the scientific foundation in the literature and operational precedent, the dNBR approach was selected to characterize fire severity and delineate fire perimeters for this project. Extended assessments will be conducted on forest and shrub ecosystems, and initial assessments will be conducted on grasslands and other specific vegetation communities known to recover from fire within a single growing season. A simple production model was developed around this approach to ensure timely and consistent products. We recognize that application of a simple production model across the ecological extent and variation covered by the project may itself be a source of uncertainty. Indeed, limitations associated with this approach to characterizing postfire effects are not yet known in many ecosystems, and we expect the MTBS project to expand our understanding of how consistently and precisely Landsat data can map burn severity. The MTBS product suite is a reflection of the need for a range of data to suit specific analysis objectives.

The following steps outline the process of identifying fire locations through summarization of the results:

- Fire history database compilation
 - Data acquisition
 - Data standardization and aggregation
- Image data selection and preprocessing
 - Scene selection
 - Preprocessing
 - Delivery and archiving
- Fire-severity interpretation and perimeter delineation
 - Normalized Burn Ratio calculation and differencing
 - Interpretation and thresholding into severity classes
 - dNBR partitioning
 - dNBR fire perimeter delineation
- Stratification and summarization of severity information.

Fire History Database Compilation

Existing fire history and location databases were compiled into a single, standardized project database that formed the basis for image scene selection. Fire history sources were from two general origins—Federal agency databases and State databases. In some cases, State and Federal agencies have collaborated in developing and maintaining a single database for State and Federal incidents. Federal agency data are aggregated into the Incident Command System database known as the ICS 209 and are maintained by the

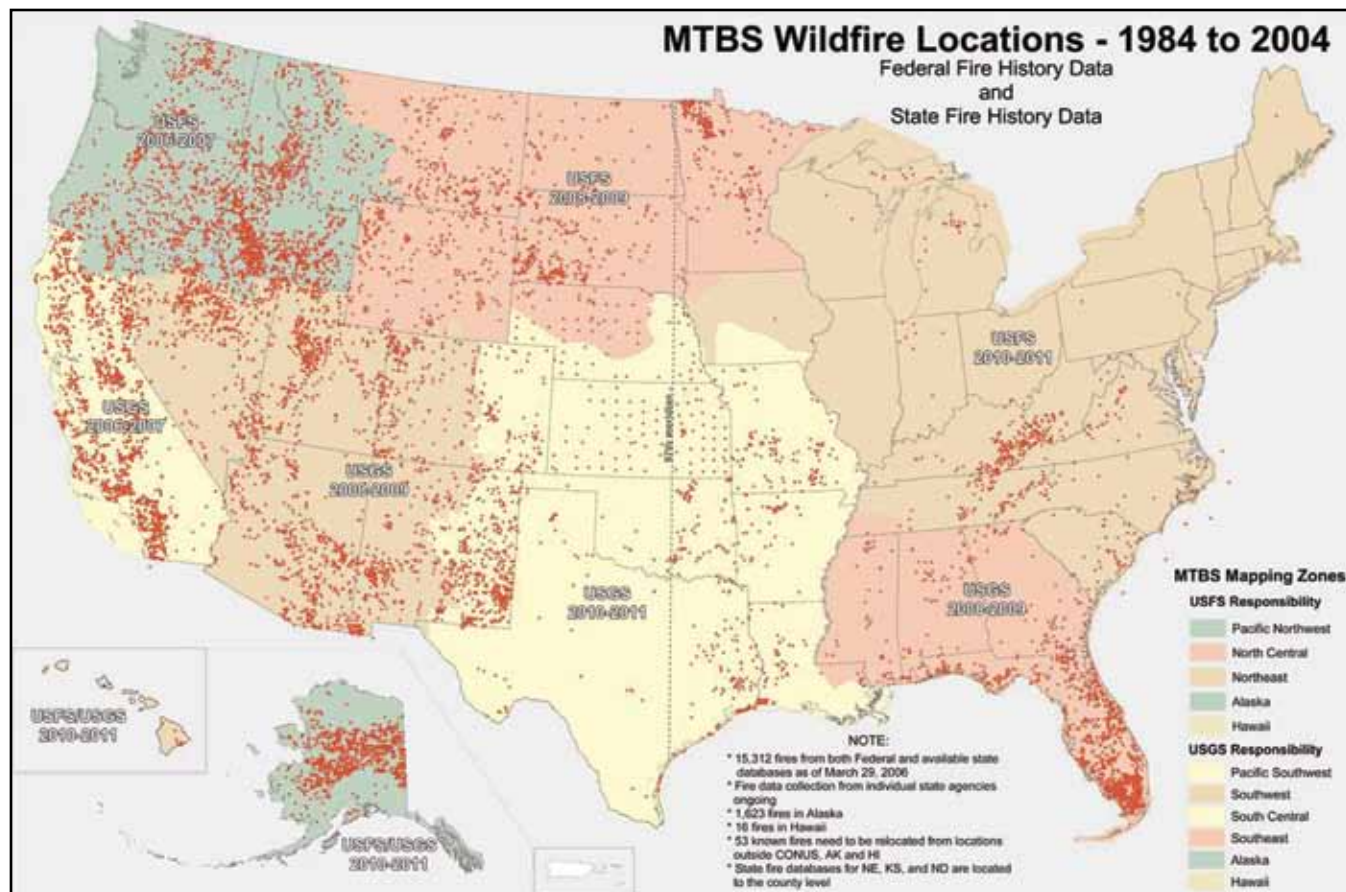


Figure 2—Wildland fire locations across the project extent. Patterns of fire frequency are clearly visible. Suspected omissions of fire occurrences as well as coarse spatial precision are also visible in portions of the project, particularly the Midwestern, Southern, and Northeastern United States. MTBS refers to “Monitoring Trends in Burn Severity.”

National Interagency Fire Center (NIFC) in Boise, Idaho (<http://www.nifc.gov/>). ICS 209 data make up the bulk records in the MTBS project database. States were solicited for State-maintained data where there was uncertainty of their inclusion in the ICS 209 data.

Whereas there is some level of standardization within ICS 209, Federal land management agencies have varying or no standards for content, geospatial accuracy, and nomenclature. Duplicate records and name changes are common because the same incidents are often reported by multiple agencies. Further standardization and correction, where possible, were performed as part of the compilation of an MTBS project database. For the purposes of this project, standardization was accomplished by selecting data elements common to the source databases and not through record editing or manipulation of the source data except

in the case of geospatial coordinates. Where a record was grossly and obviously incorrect, and a correction could be made confidently, coordinates were adjusted. Table 1 lists the elements that comprise the MTBS fire history database. Records for these elements were harvested from the ICS 209, and State data and source links were included to ensure that data could be traced to their databases of origin. Figure 2 depicts the spatial distribution and relative frequency of fire occurrences across the project extent. Likely omissions and coarse spatial precision are noticeable, particularly in the Central and Eastern portions of the conterminous United States.

Image Scene Selection and Data Preprocessing

Scene selection is driven by the MTBS fire history database. Scenes are selected using the Global Visualization Image

Selection (GLOVIS) browser developed by USGS-EROS (<http://glovis.usgs.gov/>). Enhancements were made to GLOVIS specifically to facilitate the magnitude of scene-selection effort required for this project. These enhancements, available to all GLOVIS users, include the ability to incorporate ESRI ARCGIS shape files in the viewer to aid scene selection and scene-specific Advanced Very High Resolution Radiometer greenness graphs for determining peak periods of photosynthetic activity, or peak-of-green periods. The fire history shapefile, specific to an MTBS mapping zone, is loaded into the viewer, and analysts use fire locations to guide scene selection for each fire. Prefire and postfire images are selected for each incident. Scenes selected for fires that will be processed as an extended assessment are based on peak-of-green condition or as close as cloud-free data are obtainable. Limitations in data availability due to atmospheric conditions will naturally compromise selections for fires in areas prone to summer and fall cloud and smoke obscurity. Northern latitudes will also be subject to a shorter period of optimal scene selection owing to undesirable sun angles in the fall.

Selected scenes are ordered and processed according to existing USGS-EROS protocols. Image data are geometrically (including terrain correction) and radiometrically corrected through the NLAPS process. Image data are delivered to EROS and RSAC analysts to be processed into fire-severity information. Landsat image data acquired for this project will become part of the national image archive and will be available at NLCD archive costs. Current estimates expect an increase in available archive data of more than 7,000 scenes. The existing USGS Multi-Resolution Land Characteristics image archive (<http://www.mrlc.gov/>), available through several existing Web portals (e.g., GLOVIS), will be the primary repository for these data.

Fire Severity and Perimeter Mapping

The NBR index is calculated for prefire and postfire images as described in “Methods.” Prefire and postfire images are inspected for coregistration accuracy and corrected if spatial differences are excessive and extensive (> 30 m). NBR images are differenced for each fire-scene pair to generate the dNBR. A relativized dNBR (RdNBR) is also calculated

based on the work of Miller and Thode (2007) to evaluate potential limitations of dNBR to characterize fire severity on low-biomass sites and potentially enhance interfire comparability of the results at larger scales. The RdNBR data have been shown to have stronger correlations to Composite Burn Index (CBI) plot data in some low-biomass western ecosystems (Miller and Thode 2007, Thode 2005). Figure 3 illustrates the sequence of data layers generated.

Ecological Severity Thresholding—

Processing the Landsat image data to dNBR is a straightforward series of objective calculations requiring limited analyst interaction and relying principally on automated production sequences. Subsequent to dNBR derivation, the process of developing fire severity and perimeter maps becomes much more dependent on analyst interpretation. The dNBR data are calculated as signed 16-bit with a maximum digital number (DN) range of -32,767 to +32,767. However, the practical range of DN values representing fire-related change and no change is typically within -2,000 to +2,000. Values further away from zero represent greater change as a result of both first and second-order fire effects (within the fire perimeter). Negative values indicate a positive vegetation response (growth) and positive values indicate a negative vegetation response (mortality). Figure 4 illustrates a dNBR image for the Cerro Grande Fire (2003), and Figure 5 depicts the associated data range. The analyst evaluates the dNBR data range and determines where significant thresholds exist in the data to discriminate between severity classes. Interpretations are conducted on the dNBR data aided by raw prefire and postfire imagery, plot data, and analyst experience with fire behavior and effects in a given ecological setting. The CBI data (Key and Benson 2006) have been the most commonly collected ground-based data to estimate postfire effects. Correlations between CBI and dNBR have been used to demonstrate the sensitivity of dNBR to postfire effects and to establish numerical thresholds in dNBR data that discriminate severity categories (Cocke and others 2005, Key 2005). Where CBI and similar plot data have been collected, and plot-dNBR relationships published, analysts will guide their interpretations based on these relationships. Limited extrapolation of

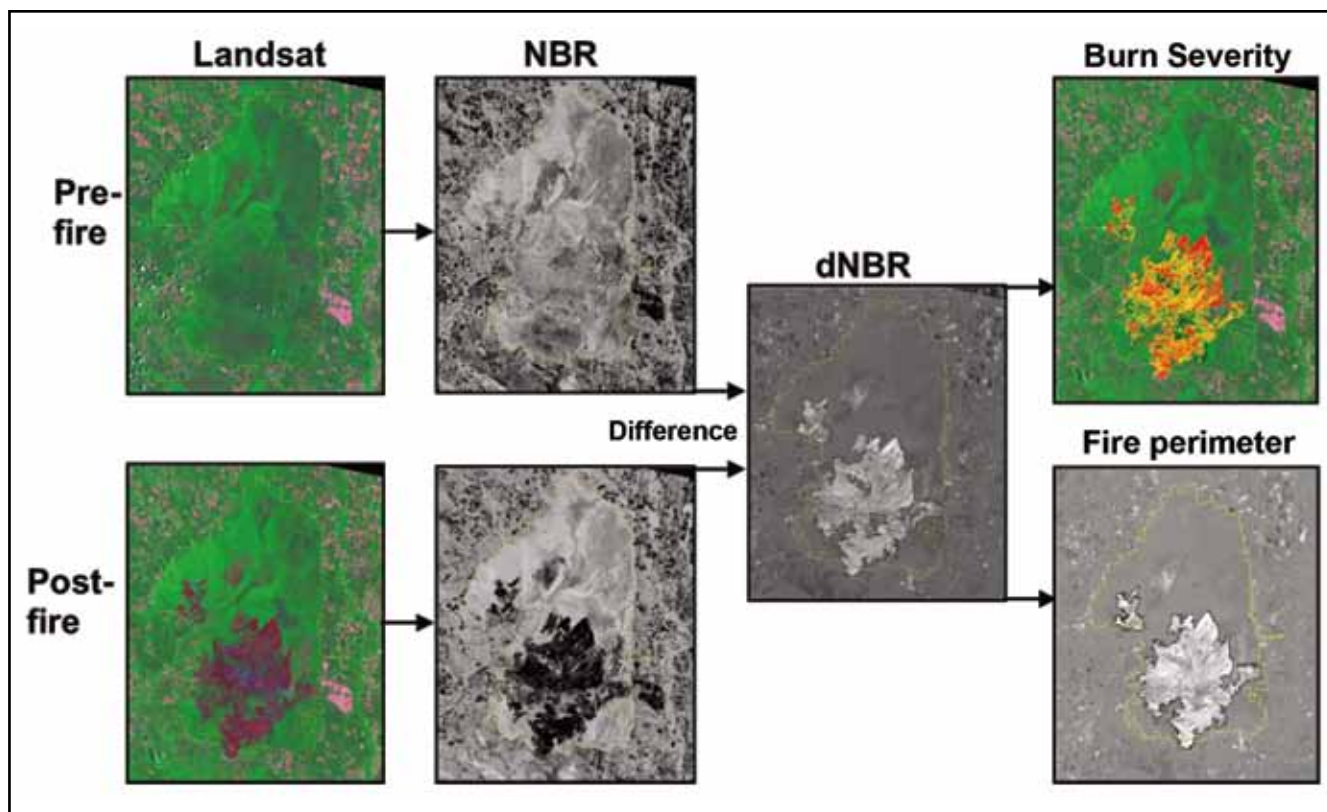


Figure 3—The sequence of layers produced between Landsat images and a fire perimeter. NBR = Normalized Burn Ratio. dNBR = differenced Normalized Burn Ratio.

plot-based thresholds beyond their geographic bounds but within ecologically similar conditions will be examined.

Thresholding dNBR data into thematic class values results in an intuitive map depicting a manageable number of ecologically significant classes (typically 4 to 7 class values). Within this project, the thematic raster data will characterize severity in five discrete classes—unburned/unchanged, low severity, moderate severity, high severity, and increased postfire response. These classes will serve as a means to easily summarize severity across broad scales and provide a coarse look at effect gradients within fires. However, there are uncertainties in this approach stemming from analyst subjectivity and limited or no plot data to guide threshold selection. Large-scale analysis may best be conducted on the continuous dNBR data, which provide the greatest range of data quantifying postfire change. Although not a direct measure of fire severity, dNBR data have been shown to correlate to field-based estimations of fire severity (Hudak 2006, Key 2005).

Ecological significance is issue dependent, and one set of thresholds cannot be expected to apply equally well to all analysis objectives and management issues. Other severity classifications such as described by Stephens and Ruth (2005) may be used as the basis for thresholding but must be considered for the appropriateness of their application to dNBR data. Fire-severity classifications that are based on fire effects not readily discernible on Landsat data (e.g., subsurface biomass combustion or soil chemistry changes) should not be applied to these data.

dNBR Partitioning—

In addition to ecological thresholds as a means of discriminating severity classes, dNBR will be arithmetically partitioned into discrete classes to facilitate objective and flexible pattern-and-trend analysis. Arithmetic partitioning is not intended to provide information on the ecological severity of fires at large spatial scales or limited temporal extents. Methods for partitioning dNBR have yet to be

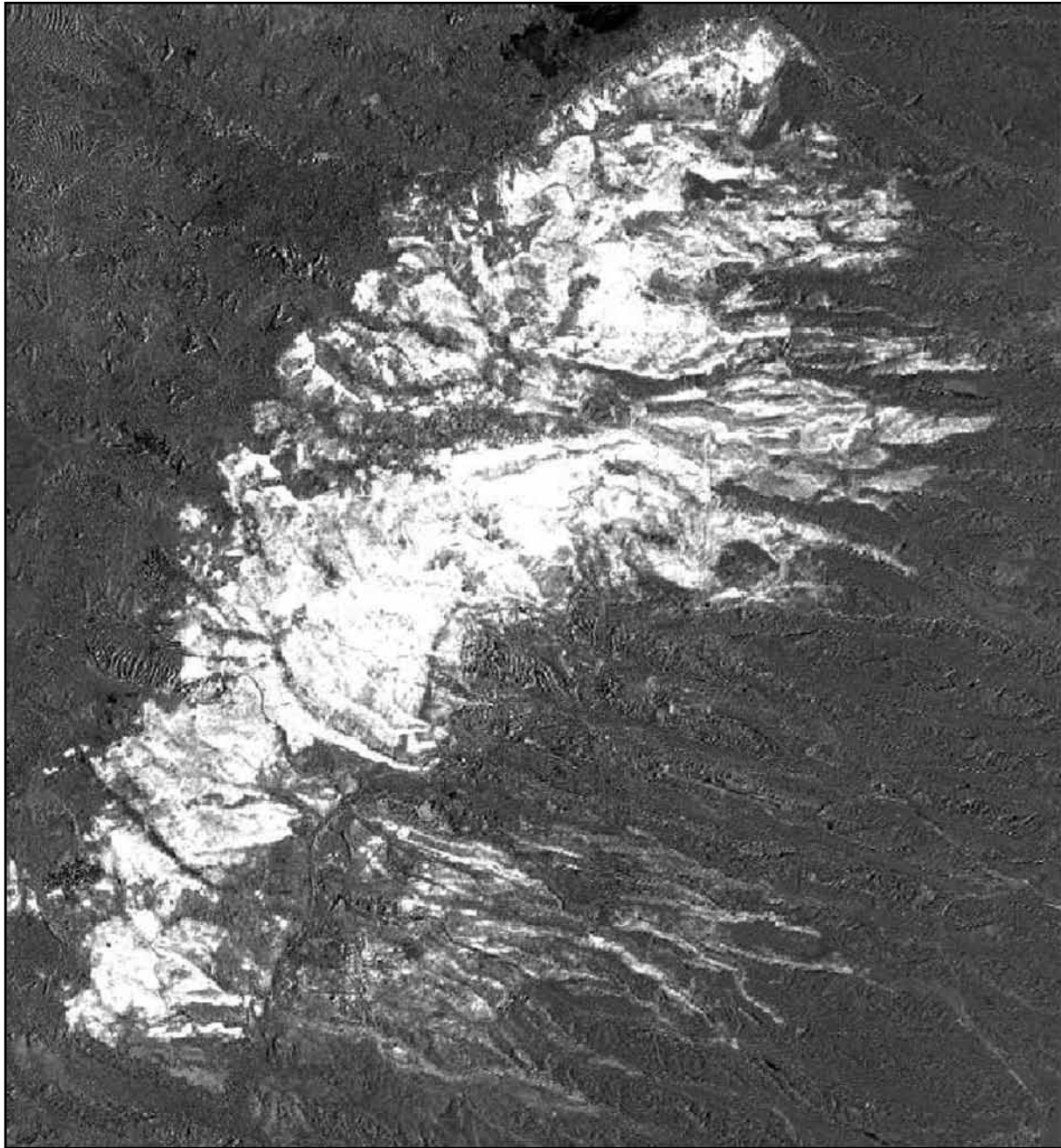


Figure 4—A dNBR image for the Cerro Grande Fire (2003) where dNBR refers to differenced Normalized Burn Ratio. Lighter areas represent higher positive values corresponding to greater degrees of change (i.e., higher fire severity).

determined, and the algorithm(s) and subsequent grain of partitioning will depend on a given technique's ability to reveal meaningful patterns in fire severity over time. Gmellin and Brewer (2002) used a simple equal-interval calculation to establish objective burn severity classes between observed unburned and high-severity conditions

in the Northern region of the Forest Service. Brewer and others (2005) used the same approach in a methods comparison study that concluded dNBR to be the most effective approach of those evaluated for mapping fire severity. The relative ease and quickness of arithmetically partitioning dNBR data will allow for rapid evaluation of meaningful

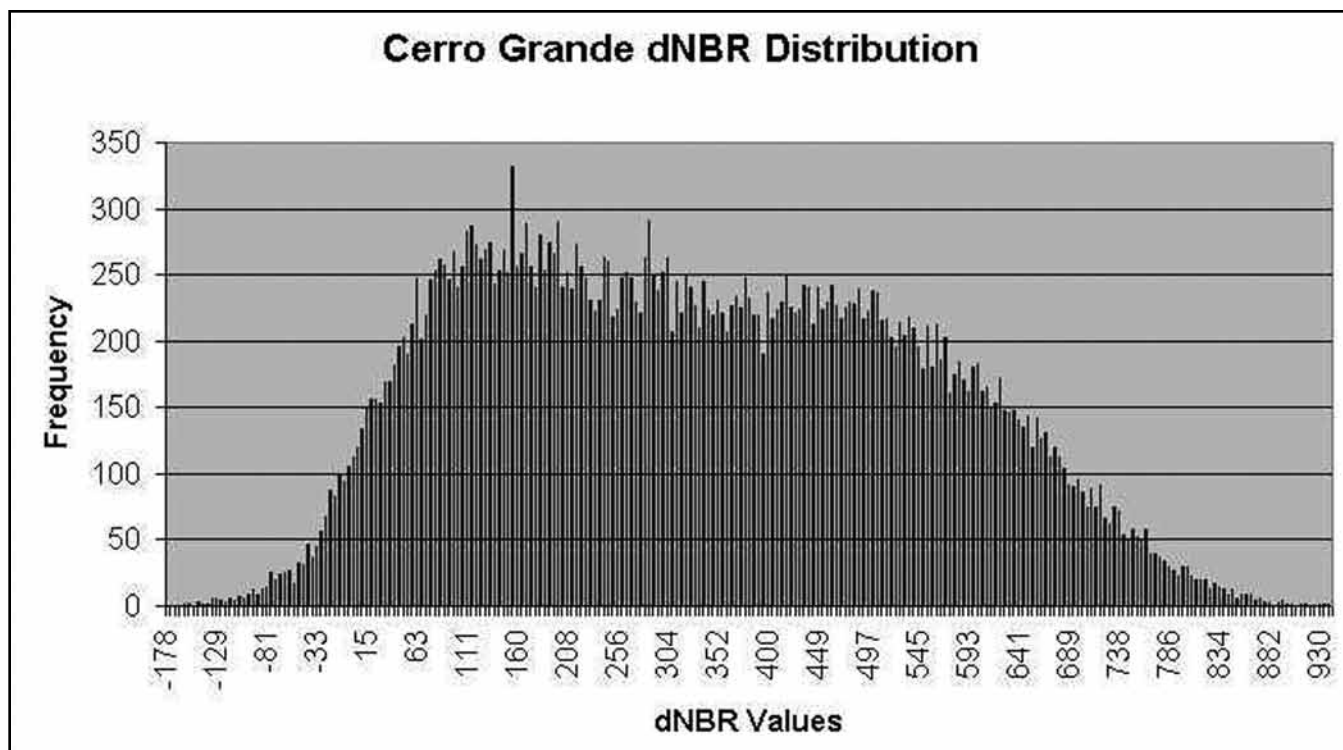


Figure 5—A graphical depiction of the dNBR data range associated with Cerro Grande Fire where dNBR refers to differenced Normalized Burn Ratio. Positive values represent a gradient of increasing fire severity whereas negative values represent increased spectral response usually associated with higher photosynthetic activity in postfire vegetation over prefire vegetation. Values near zero represent little or no change.

spatial and temporal scales in the context of fire severity trends. Moreover, dNBR data can be efficiently analyzed and classified to suit the fire severity information needs of a specific management issue.

Perimeter Delineation—

Fire perimeters will be generated by on-screen interpretation and delineation of dNBR images. Analysts will digitize perimeters around dNBR values reflecting fire-induced change. To ensure consistency and high spatial precision, digitization will be performed at on-screen display scales between 1:24,000 and 1:50,000. Incident perimeters, where available, will be used in an ancillary fashion to inform the analyst. This can be particularly useful in identifying unburned islands within a perimeter or isolated, disjunct spots outside the main perimeter. Owing to limited and variable availability, as well as inconsistent spatial precision, incident perimeters were not considered appropriate as a source for MTBS project perimeters.

Data Summarization

Tabular data will be generated from statistical summaries of the fire severity class layers. Reporting units will vary in extent depending on the needs of WFLC and other multiagency user groups but, at a minimum, summary data will be produced for each State as well as at a national extent. Three sets of tabular data are currently specified in the MTBS product suite and are listed in “Products.” Total acres burned by severity class are the most directly extractable information summary from the spatial data.

Summarizing acres burned by severity class and vegetation cover types requires consistent geospatial vegetation data of similar resolution to provide the most meaningful stratifications. Existing vegetation types currently being mapped by the LANDFIRE program (<http://www.landfire.gov/index.php>) will offer the most spatially extensive and nationally consistent data by which severity classes can be summarized. Since Landsat imagery is the spatial basis for both MTBS and LANDFIRE data, uncertainties that may

result from summarizing severity classes within vegetation cover types mapped at a significantly different spatial scale should be minimized. In conjunction with the information needs of WFLC, the accuracy of LANDFIRE data will need to be evaluated to determine the most appropriate thematic scale for reporting.

Summarizing acres burned by severity class in relation to fuel treatment activities will be dependent on the availability of spatial data depicting the location and extent of individual treatments. Currently, the National Fire Plan Operations Reporting System (NFPORS) database is the primary standardized Federal database containing such data in digital format. As described on the Web site (<http://www.nfpors.gov/>), “NFPORS is an interagency system designed to assist field personnel in managing and reporting accomplishments for work conducted under the National Fire Plan.” It is expected that non-Federal fuel treatment data will not be available, and spatial data on Federal lands will be sporadic. Tabular data generated under these criteria will only have applicability to specific administrative and geographic extents.

It is recognized that these tabular data may have limited utility at finer spatial scales and for addressing research and management information needs not considered within the scope of this project. A composite database containing additional ecological and administrative spatial units, including 4th-field hydrologic units (Seaber and others 1987) and Federal ownership, will be available to enable user-specified summarization of MTBS data at larger scales. The production and distribution of both continuous and thematic spatial data sets described in “Fire Severity and Perimeter Mapping” are considered the primary geospatial data legacy available to scientific and operational interests outside this project.

Data Distribution

All spatial and tabular data will be distributed through Web-based interfaces. Data portals developed and maintained by USGS and the USDA Forest Service will be primary access points for the data and associated reports as they are completed and become available. Access to MTBS data-distribution nodes will be through the project Web

site (<http://www.mtbs.gov>). Additional distribution nodes may be developed in partnership with other Federal and academic institutions.

A technology transfer phase of the project will be initiated subsequent to completion of the first historical data sets. The intent of this effort will not only be to educate potential users about the structure and content of the data but also to explore applications of the data at multiple scales. It is expected that independent studies that utilize MTBS data will reveal utility and limitations that will be important to guiding operational use. The technology transfer phase will make efforts to synthesize internal and external assessments of data utility and provide an efficient means to access this information. Web-based and workshop formats will be used to engage potential users.

Applications for Fire-Severity Data

Central to the missions of both the Western Wildland Environmental Threat Assessment Center and the Eastern Forest Environmental Threat Assessment Center is the early detection, identification, and assessment of multiple environmental threats such as insects, disease, invasive species, fire, loss or degradation of forests, and weather-related risks. The MTBS project will contribute to successfully accomplishing this mission by informing and supporting a variety of fire severity-related analysis applications.

It is beyond the scope of this paper for us to describe, in detail, specific examples of fire-severity data use. The few examples that follow suggest that burn severity or fire severity, or both, have been reported in a variety of research applications including surface runoff and sediment yields (Robichaud and Waldrop 1994), burned area relationships to natural reforestation (Lopez-Garcia and Caselles 1991), forest stand conversion and regeneration establishment (Blackwell and others 1992, 1995), restoration of natural fire regimes with prescribed fire programs (Brown and others 1995, Keifer and Stanzler 1995), and wildlife habitat components (Hutto 1995).

Although explicit analytical uses of the MTBS data are not provided in this paper, a general discussion on scale of application is warranted. Multiscale, integrated analysis to support planning at both strategic and tactical levels has

been presented as an effective way to accomplish management objectives within the context of ecological function (Hann and Bunnell 2001). As described in previous sections, MTBS data provide a practical basis for multiscale analysis. Barbour and others (2005) emphasized the need to look across scales in order to understand potential differences in perception of wildfire risk between planning scales and between management objectives. Using the analysis scales described by Barbour and others (2005), a simple conceptual step-down analysis demonstrates how MTBS data can generally be applied at broad, mid, and fine planning and monitoring scales.

At the broad scale, general spatial and temporal patterns of fire can be juxtaposed against generalized depictions of biophysical setting, current vegetation, and historical vegetation conditions to identify landscapes that are experiencing fire frequency and behavior outside estimated historical ranges of variability. This information may serve the purpose of streamlining resource allocation decisions at the national level and provide risk strata for larger scale analysis. Landscapes with potential for higher risk can be assessed at the midlevel for specific patterns of fire occurrence and magnitude that threaten resource and social values (e.g., sensitive species habitat, hydrologic function, rural communities, and recreational opportunities) identified in regional and forest plans. The spatial resolution of MTBS products align closely with resource data layers commonly found at the regional and forest unit levels of the Forest Service (Brewer and others 2003, Franklin and others 2000, Mellin and others 2004). The scalable nature of dNBR values also allows for severity characterizations specific to analysis needs. Management activity planning and design intended to specifically address issues identified in midscale planning efforts are based principally on fine-scale analyses. Management activities designed to mitigate potential future threats, including severe fire and insect and disease outbreaks, require precise information about landscape condition and disturbance history. At site-specific scales, MTBS data reveal important information about severity patterns within fires, which are necessary to understand current condition and to relate past management activities.

It is our belief that the geographic and temporal extents, along with the consistency of MTBS data products, will provide a rich data legacy from the project. These data will provide the analytical basis for research that would have been logistically impossible without the MTBS project. These data provide the opportunity to stratify by appropriate biophysical environment settings and generate efficient and effective sampling strategies for agents such as insects, pathogens, and invasive plant species. They also provide the basis to substitute space for time to evaluate factors such as climatic effects or site water balance characteristics on timing and duration of water yield. Whereas these data do not address all environmental threat assessment needs, they do provide high-quality, consistent data and a contextual framework for the keystone disturbance agent for many wildland ecosystems.

Conclusions

The MTBS project will develop the data and information necessary to meet the strategic analysis objectives of WFLC and other policymaking and monitoring bodies. In addition to meeting the primary objective of providing the information necessary to support an assessment of recent trends in fire severity, a valuable data legacy will be available to support a broad range of research and operational uses at multiple scales. High-resolution fire-severity data generated at the individual fire scale, yet assembled at broad geographic extents, offers the potential to support analysis ranging in scope from event-specific to continental. Future applications will explore the utility of these data to support threat assessments across a wide range of ecological systems and provide a spatial and temporal framework to better understand the immediate and longer term inter-relationships of wildland disturbance agents and risk factors in postfire settings.

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Information Needs, Acceptability of Risk, Trust, and Reliance: The Case of National Predictive Services Customers

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Abstract

Making complex risk-related decisions involves a degree of uncertainty. How that uncertainty is addressed or presented in reports or data tables can be tailored to meet information users' needs and preferences. Involving the recipients of risk-related information in the design of information to be delivered (including the types of information delivered, format, and approach to risk) follows Fischhoff's (1995) recommendation for involving the recipients of risk information in the crafting of the risk message. Here, we describe a study conducted with people who use risk-related information. We contacted users and potential users of National Predictive Services, an information clearing-house for people who work with prescribed and wildland fire. Specifically, this service supplies information to fire managers and associated personnel that will help them make short-term (30-day) decisions regarding fire personnel and resource deployment. Each potential user is a member of the fire-management community, including fire management officers, meteorologists, information officers or public affairs personnel, and members of various fire-management teams. The information users focused on in this study were employed within Federal agencies. Respondents completed a self-administered survey via a Web-based service. Findings examine the types of information users need, preferred formats for information delivery, and the likelihood of applying information from Predictive Services in decisions made about fire. Of interest to this paper's risk management/risk communication focus are the reported views on acceptability of risk and tolerance for errors, implications of risk, and trust and confidence in the information delivered through Predictive Services. Barriers and facilitators to utilization of Predictive Services are illuminated.

Keywords: Information needs, National Predictive Services, reliance, trust, user-needs assessment.

Introduction

A Needs-Assessment Approach to Evaluation

A framework for program evaluation is found in needs assessment. According to Rossi and others (1999), in a needs assessment, a program is assessed in light of the presenting conditions that make the program necessary. Current and prospective service recipients may be surveyed to explore such pertinent issues as target audience for the services or program, service utilization, services desired, shortcomings of existing services, and barriers to service utilization. Additional items of interest in a user-needs assessment are (1) a detailed examination of the characteristics of the target audience (e.g., gender or time in position of employment); (2) need for specific products and services; (3) program design, including preferred delivery systems (e.g., the Internet, in-person briefings) and delivery styles (e.g., maps and graphs); and (4) program operations (i.e., whether potential Predictive Services clients are actually using the products and services, and if not, why not). A careful examination from the perspective of key informants (users and potential users) facilitates a formative evaluation process. This process includes adjustment of existing products and services to better meet user needs and development of new products identified as necessary but not currently offered. The evaluation can result in a negative appraisal of a program, causing some consternation among program sponsors and enthusiasts. However, findings of a careful needs assessment that result in subsequent adjustment can help increase program value and effectiveness. Such a process might be taken in stride as part and parcel to setting up a program designed to meet a specific need or set of needs, and growing pains in adjusting and further developing a program through its life. Undertaking a user-needs assessment represents an openness and a commitment to service. Evaluation should be a part of every serious risk communication effort (Slovic and others 1990).

Crafting of Risk Messages to Meet Users' Needs—

Access to information is crucial to effective management of risk. Information available prior to a risk-related event can assist in advance planning, including allocating and distributing resources. Information available during a risk-related event could help determine how severe the event might become, adequacy of management resources, and the type and degree of intervention that is needed to protect human, structural, and natural resources. Information following a risk-related event might be used to aid recovery efforts and conduct analysis directed toward future risk-related responses. Wildfire is one example of a risk-related event; fires represent threat and potential harm to natural, structural, and human resources.

Factors That Affect Acceptability and Use of Risk Information—

Not all audience members targeted for risk-related information will have the same information needs or interests; thus, variability in message content is an important consideration in crafting risk-related messages. Multiple factors have been demonstrated to influence perceptions of risk and risk-related decisions, including gender (Finucane and others 2000, Satterfield and others 2004, Siegrist 2000), age (Otani and others 1992), time in decisionmaking role, and degree of experience with risk situation (Payne and others 1992, Reyna 2004), educational level (Vaughan and Nordenstam 1991), expertise in the topic area (including expert vs. layperson views) (Fischhoff and others 1984, Plough and Krinsky 1987, Slovic 2000), and individual worldviews (e.g., culture, attitudes, and values) (Slovic and Peters 1998, Vaughan and Nordenstam 1991, Weber and others 1998). Contextual and situational factors further influence risk perceptions and decisionmaking (see, for example, Kneeshaw and others 2004).

In addition, trust in an information source and confidence in the information received have been repeatedly demonstrated as essential to how information will be perceived, responded to, and accepted (Borrie and others 2002, Cvetkovich and Winter 2003, Siegrist 2000, Siegrist and others 2000) and has direct applications to fire-management issues (Cvetkovich and Winter 2004; Shindler and others

2004; Winter and others 2002, 2004). Trust in the information source tends to foster greater acceptance and belief in the risk-related message.

The Present Study

This analysis presents findings from a study initiated in 2005 through request from the National Predictive Services Group (NPSG), a group chartered under the National Fire and Aviation Executive Board (NFAEB) that provides oversight, leadership, and strategic direction to the Predictive Services program. The NPSG identified a user-needs assessment as one of its program-related goals, and they have played a central role in execution of the assessment. The assessment examines the Predictive Services program, which offers products and services through Web sites, briefings, and emails administered through the National Interagency Fire Center (NIFC) and the Geographic Area Coordination Centers (GACCS). The main purpose of this service is to integrate climate, weather, situation, resources status, and fuels information into products that will enhance the ability of managers to make sound short- and long-term strategic planning and resource allocation decisions and to ensure the safety of firefighting and emergency personnel. Predictive Services is a multiagency effort, with support from the USDA Forest Service, the USDI Bureau of Land Management, National Park Service, Fish and Wildlife Service, Bureau of Indian Affairs, and the National Weather Service. Products are aimed at the fire management community in each of these agencies as well as State and county agencies.

The NPSG requested assistance from the authors to conduct a user-needs assessment; to gain information toward improvement of current products and services, if necessary; and to identify additional products and services that might be offered. Evaluation results are more likely to be utilized if they directly address the information needs of decisionmakers (Patton 1986). Thus, the survey and sample were constructed in close collaboration with the NPSG in order to ensure the assessment met their needs. Respondents included fire management officers, fire weather meteorologists, suppression personnel, dispatchers, fire behavior analysts, fuels specialists, fire researchers, incident

management team members, fire use team members, multi-agency coordinators, aviation personnel, and public affairs personnel. Study findings may interest other agencies and individuals examining dimensions of risk management, including fire and threat information needs.

Method

Respondents—

A sample of email addresses representing users and potential users of Predictive Services products and services was compiled using key contact and snowball approaches. Sources of addresses included the NPSG, a list of incident information officers, the National Wildland Fire Management Directory, contacts at various Federal agencies, and online directories. We compiled a national list of 2,999 Federal contacts. This initial list was composed of Federal sector fire management personnel within the selected agencies (see respondent description below) with a focus on assuring that fire management officers, fire behavior analysts, incident meteorologists, GACC managers, regional coordinators, public affairs and information officers, dispatchers, incident management team members, fire use personnel, and aviation personnel were included. A census listing was not available through any of the agencies, so a compilation from email lists containing fire management types, a training record, and other preestablished lists and directories was used. Because the Predictive Services group wanted as many respondents as possible, we used all members in our list whose email addresses could be obtained. The sample was intended to be as comprehensive as possible given the lack of a census listing.

Beyond those included in the sample, an additional number of individuals responded as volunteers. Two circumstances prompted volunteering. The first occurred when initial contacts forwarded the survey link to others after completing it themselves. In some cases, initial contacts felt they were not the best person to complete the survey and forwarded it to another contact within their agency.

The respondents included 1,078 individuals (including 63 volunteers or 5.8 percent of the sample). The majority (69.1 percent) were male, employed with the USDA Forest Service (53.3 percent), National Oceanic and Atmospheric

Administration and the National Weather Service (14.3 percent), Bureau of Land Management (12.6 percent), the National Park Service (10.0 percent), U.S. Fish and Wildlife Service (4.7 percent), and Bureau of Indian Affairs (3.5 percent). The remainder were employed within a Federal interagency group (0.6 percent) and various other Federal agencies (0.9 percent). Respondents had typically been in their current job for 3 years (median response). We had a final response rate of 36.5 percent with less than 1 percent of the sample refusing to participate (12 individuals). A random sample of nonrespondents was contacted by telephone and asked to complete a brief phone survey covering reasons for nonresponse, use of various GACC Web sites, and familiarity with products and services. The main reasons for nonresponse were lack of familiarity with the program and lack of time during the study period. Nonrespondents were similar to respondents in geographic location and agency of employment.

A comparison of the sample respondents and volunteers revealed that the volunteers were twice as likely to be employed within the Bureau of Indian Affairs (7.9 versus 3.3 percent of each sample) and Bureau of Land Management (22.2 versus 12 percent), and were less likely to be from the USDA Forest Service (31.7 versus 54.7 percent). The average length of employment was significantly different, with volunteers reporting fewer years (2.9 years for volunteers and 6.2 years for the original sample, $t = 3.326$, $p = 0.001$). Gender distribution was similar for the two groups. The volunteers and original sample members are combined for the purposes of this paper because further analyses showed that there were few differences between these two groups.

The Survey—

Topics addressed in the survey included sociodemographics (e.g., employing agency, years in current position, and gender), who the Predictive Services audience should be, preferred information formats, preferred products and services, acceptability of risk and tolerance for errors, implications of risk in making decisions, trust and confidence in the products, reliance on Predictive Services products, reliance on other information, and facilitators and barriers to using

Predictive Services information. The survey included closed-ended (including semantic differentials, checklists, and other formats) and open-ended questions. Some survey items were modeled after recent studies conducted by another Federal agency to allow for comparison, whereas others were developed specifically for this study's purposes. A draft instrument was submitted through peer review and review of the Predictive Services group commissioning the study. The instrument was pretested with a random sample of respondents, and adjustments were made to items that seemed unclear or were described as confusing by pretest participants. The survey was posted on the Web service Question Pro (<http://www.questionpro.com>). A Web-based survey was desirable because of the significant cost reduction achieved by eliminating printing and mailing costs, greater availability of email addresses than mailing addresses for the sample members, and increased familiarity of Web-based instruments among Federal personnel. Failed addresses were typically bounced back within minutes rather than days, allowing for attempted correction and remailing, or, when appropriate, elimination from the sample.

Procedure—

Respondents were emailed an invitation and brief letter describing the study, along with a link to the survey site. Three reminders were sent over the course of the data collection period, with a total of 42 days allowed for response. The first reminder was sent 10 days after the initial mailing, the second was sent 14 days after the first reminder, and the last was sent 10 days after the second mailing, 1 week prior to the close of the survey site. Each of the reminders contained a brief message and the link to the survey site. Reminders were sent to sample members who had not completed the survey as well as those who had not been removed from the sample owing to email failures.

Results

Identified Audiences—

Respondents were asked whom Predictive Services should include as the primary audience for their products. The primary audiences selected by the majority were local and district fire managers (75.8 percent), regional and State

fire managers (75.3 percent), and national fire managers (65.5 percent). Nonfire land managers were listed as a primary audience by about one-third (33.5 percent), and the public was listed by about one-fourth of the respondents (27 percent; note that percents do not sum to 100 because respondents could provide multiple answers on this item).

Information Used and Utility of Information—

Preferred Formats

Respondents were asked to indicate how useful each of 11 styles and formats of presenting information was to them. The average ratings of all items except one fell above 3 (the neutral point on the scale). From greatest to least, the most useful formats were regional or national maps ($\chi = 3.91$, $sd = 1.03$, $n = 879$), satellite maps ($\chi = 3.76$, $sd = 1.15$, $n = 870$), brief executive summaries of data ($\chi = 3.75$, $sd = 1.08$, $n = 858$), brief annotations that accompany data ($\chi = 3.56$, $sd = 1.02$, $n = 850$), radar maps ($\chi = 3.53$, $sd = 1.19$, $n = 857$), data in table form ($\chi = 3.53$, $sd = 1.05$, $n = 863$), bar charts or figures that summarize data ($\chi = 3.37$, $sd = 1.09$, $n = 856$), data in text form ($\chi = 3.33$, $sd = 1.03$, $n = 849$), Web-based ArcIMS maps with user-defined layers and scales ($\chi = 3.31$, $sd = 1.23$, $n = 832$), and data in spreadsheet form ($\chi = 3.21$, $sd = 1.10$, $n = 853$). Least useful to respondents was non-Web-based Geo database files ($\chi = 2.62$, $sd = 1.12$, $n = 793$).

Preferred Products and Services

Thirty-eight products and services were listed in the survey. Some of these products are available elsewhere as well and are provided as a courtesy to the Web site users. For each item, respondents were to indicate if they had not used the product, and, if they had, to rate the usefulness of that product. Several of the products stood out because at least 70 percent of the respondents had used the products and rated them as useful or very useful. They included daily fire weather forecasts, red flag warnings (this term is used by fire weather forecasters to alert users to an ongoing or imminent critical fire weather pattern, <http://www.nwcg.gov/pms/pubs/glossary/index.htm>), Incident Management Situation reports, drought information, and Interagency Situation reports. A number of products were used by a majority (at least 50 percent), although ratings of usefulness varied. A few items offered through Predictive Services had

been used by a minority of respondents and were not rated as very useful by those who had used them. Among the products in this category were regional monsoon updates, upper air soundings, Predictive Services forms, and state-of-the-fuels reports.

Suggestions for Improved Formats and Products

Respondents gave several suggestions for improvement in response to open-ended questions. In terms of format, there were several suggestions for improving Web site performance, including making sure that GACC and Predictive Services employees could direct people to the right location, streamlining information searches by allowing users to bookmark relevant information, having a professional Web designer improve the sites' navigability, and removing information that is no longer accurate. In terms of expanding products and services, there is a desire for more location-specific products and more two-way conversations between Predictive Services and people who work on the local level. This communication would support local decisionmaking and possibly increase the relevance and quality of information provided by Predictive Services. People working in off-season or prescribed burning capacities, or both, suggested more year-round coverage. Additional topics for Web site content were offered including information on smoke management, fuel moisture, safety, real-time information, and current fire behavior.

Acceptability of Risk and Issues of Accuracy—

Respondents were asked to choose the statement that best fit their preference regarding error in predicting risk. The majority (67.3 percent) chose "Statements of danger or risk be issued with a greater margin of error allowing for an early response, knowing that this may lead to unnecessary alarms and response (better safe than sorry)" over "Statements of danger or risk should only be given with certainty, knowing that this may allow a few dangerous events to emerge that were not anticipated (don't cry wolf)" (chosen by 23.9 percent of respondents). In other words, the majority preferred erring on the side of caution when reporting on fire danger and high fire potential.

Open-ended responses pointed to concerns surrounding information accuracy. Among the topics of concern were

the need for clear statements of the limitations of the data and known degrees of accuracy (for example, some would like to see confidence intervals reported along with data). There was also an interest in sources and assumptions used in creating the products offered.

Implications of Risk in Decisionmaking—

The perceived impacts of inaccurate information were examined. To address this concept, two items were used. The first was "Inaccurate Predictive Services information would decrease my ability to predict fire behavior" rated on a scale from 1 to 5 (1 = strongly disagree, 5 = strongly agree). The majority leaned toward slight agreement with this statement ($\chi = 3.36$, $sd = 1.16$, $n = 712$) with 12 percent indicating strong agreement and another 20 percent indicating agreement. Another 18.2 percent neither agreed nor disagreed with this statement.

The second item was "Inaccurate Predictive Services used in my decisionmaking may adversely impact firefighter or public safety" again rated on the 1 to 5 scale. The majority leaned toward slight agreement on this statement as well ($\chi = 3.48$, $sd = 1.18$, $n = 744$), with 20.3 percent indicating agreement and 16.0 percent indicating strong agreement; 17.8 percent neither agreed nor disagreed with this statement.

Written comments pointed to concerns surrounding accuracy in data gathered to make predictions and communication issues. These comments revealed a disconnection between Predictive Services and local field units. Comments indicated that Predictive Services might benefit from a better awareness of local weather and fire problems. Communication-related comments addressed concerns over the need for consistency in content, streamlining of information, and concentration on materials directly relevant to fire-use decisions.

Trust, Confidence, and Reliance—

Trust and confidence in the information provided by Predictive Services were assessed in a general item "How much trust and confidence do you have in the information provided by Predictive Services?" rated on a scale from 1 to 5 (1 = none at all, 5 = a great deal). Very few respondents selected 1 (none at all, 8.8 percent) or 2 (5.3 percent) on this

item. About one-fourth (25.7 percent) indicated some trust and confidence, whereas almost half selected either 4 (35.4 percent) or 5 (12.8 percent), indicating a majority of respondents had trust and confidence in the information provided.

In addition, when asked about three specific trust-related issues that might be barriers to using Predictive Services, very few indicated that trust was an issue. Only 3.5 percent indicated that they did not trust the products and services, 1.4 percent indicated a lack of trust in advice about using the products, and less than one percent indicated a lack of trust in information produced by multiple agencies. These specific items suggest that most had trust in the information provided.

Comments specific to trust and confidence included the desire among respondents to have a working relationship with the people who provide the information, as exemplified by this quote: “The local weather service offices continue to provide one-on-one support for weather products. The level of trust in a forecast product is directly related to the personal conversations I have had with the forecasters.” In spite of some trust, to a great deal of trust expressed by the majority of respondents, the majority do not rely on Predictive Services in decisionmaking. About 10 percent (9.6) relied on Predictive Services a great deal (a rating of 5 on a 1 to 5 scale, 1 = none at all, 5 = a great deal). About one-fifth (21.2 percent) provided a rating of 4, and about one-third indicated little to no reliance on Predictive Services information (12.5 percent gave a rating of 2; 21.5 percent a rating of 1).

When asked how true the statement “I rely on other sources more heavily than the products and services provided by Predictive Services,” the majority indicated that this statement was somewhat to very true (51.1 percent), whereas 16.8 percent indicated the statement was not at all true. The likelihood of taking action based on Predictive Services information received or gathered from a Web site suggested respondents were somewhat likely to take action ($\chi = 2.96$, $sd = 1.23$, $n = 979$, on a 1 to 5 scale, 1 = not at all likely, 5 = very likely).

Facilitators and Barriers to Utilization—

Two facilitators to utilization were queried based on accessibility and utility. The first of these was “I can access

and apply Predictive Services information as part of my job duties.” Almost half (46.3 percent) agreed or strongly agreed with this statement. Approximately another fifth were neutral (18.9 percent), and almost one-third did not supply a response (27.7 percent). The second item related to utilization facilitators was “Predictive Services information helps me perform my job with greater precision,” with which 13.7 percent agreed, whereas almost one-third (31.4 percent) disagreed or strongly disagreed. About one-third (32.5 percent) did not respond to this item.

Barriers to utilization were explored through a general item related to uniqueness of the information “I think there is overlap in the type of information that I can obtain from Predictive Services and other sources.” More than half (56.5 percent) indicated that this statement was somewhat to very true. Respondents noted overlap between the products and services offered by Predictive Services and other sources, particularly the National Weather Service. Some suggested a closer coordination between the two providers in order to reduce or eliminate redundancies.

Specific barriers to utilization not related to trust of the information (already presented above) were examined. The most frequent barrier selected was “I never thought about it,” (indicated by 26.9 percent). Other barriers selected by at least one-tenth of the respondents included “My current management practices don’t require the types of information provided by Predictive Services,” (14.7 percent); “I don’t know how to use these products,” (14.1 percent); and “I need information that is site specific” (13.5 percent). Some respondents also mentioned a lack of resources as a barrier (lack of time mentioned by 9.3 percent; lack of technology by 4.0 percent; and lack of money at 1.4 percent).

Open-ended responses offer additional insights into barriers in using Predictive Services including levels of awareness and access. Some respondents were either unaware of the products and services or indicated a limited knowledge of the array of available information and its potential uses. Respondents made several suggestions that would address this situation, including advertising to targeted markets, annual notices of new and existing products and services, and developing Web-based orientation or training, or both, to familiarize potential users with the

Table 1—Regression results for predicting reliance on Predictive Services information

Independent variable	β	t	sr^2 ¹
Trust and confidence	0.687	29.266	0.460
Years in current position	-.021	-.903	<.001
Gender	-.099	-4.264	<.001
Education	-.031	-1.351	<.001

¹ Squared semipartial correlation is a measure of the unique contribution of the independent variable to the amount of variance explained within that set of independent variables. According to the numbers shown, trust and confidence is the only variable contributing a substantial unique variance beyond the other independent variables (Tabachnik and Fidell 2001)

products. Respondents also suggested presenting information in lay terms, including a glossary of acronyms to further enhance understanding, and creating a Web feature that allows users to earmark their most relevant Web links. Respondents suggested that improved graphics might ease information utilization.

Predicting Reliance and Use of Predictive Services Information—

The ability to predict reliance on Predictive Services information, and the likelihood of taking action were examined through simultaneous multiple regression. Approximately 50 percent of the variance ($R^2_{adj.} = 0.50$, $F_{4, 927} = 234.16$, $p < 0.001$) in “How much do you rely on the information provided by Predictive Services to assist in decisionmaking” was predicted by trust and confidence in the information provided, gender, years in current position, and educational level (Table 1).

Male respondents were significantly more likely to rely on Predictive Services information ($t = 6.36$, $df = 483.68$, $p < 0.001$, males $\chi = 3.00$, females $\chi = 2.42$). Reliance had an inverse relationship with years in position in job ($r = -0.086$, $p = 0.006$, $n = 1,003$); federal employees with longer tenure were less likely to rely on Predictive Services information. Those expressing greater trust and confidence in Predictive Services were far more likely to rely on the information in decisionmaking ($r = 0.704$, $p < 0.001$, $n = 944$). There was not a significant linear relationship between reliance and education.

Approximately 48 percent of the variance ($R^2_{adj.} = 0.481$, $F_{4, 922} = 215.71$, $p < 0.001$) in “How likely are you to

Table 2—Regression results for taking action based on Predictive Services information

Independent variable	β	t	sr^2 ¹
Trust and confidence	0.677	28.303	0.449
Years in current position	-.042	-1.767	<.001
Gender	-.079	-3.329	<.001
Education	-.020	-.841	<.001

¹ Squared semipartial correlation is a measure of the unique contribution of the independent variable to the amount of variance explained within that set of independent variables. According to the numbers shown, trust and confidence is the only variable contributing a substantial unique variance beyond the other independent variables (Tabachnik and Fidell 2001)

take action based on Predictive Services information that you gather or receive from a Web site” was predicted by trust and confidence in the information provided, gender, years in current position, and educational level (Table 2).

Male respondents were significantly more likely to take action based on Predictive Services information ($t = 4.57$, $df = 429.25$, $p < 0.001$, males $\chi = 3.08$, females $\chi = 2.66$). Taking action had an inverse relationship with years in position ($r = -0.120$, $p < 0.001$, $n = 979$). Those expressing greater trust and confidence in Predictive Services were far more likely to take action based on the information ($r = 0.688$, $p < 0.001$, $n = 939$). There was not a significant linear relationship between reliance and education.

Conclusions and Discussion

Current and Desired Services and Format: Where Are the Gaps?

In keeping with Rossi and others (1999) recommendations for a user-needs assessment, we explored the issue of target audience. According to our key informants, users and potential users of Predictive Services, fire managers at the local, district, regional, State, and national levels should be the primary target audience for products and services. Although the public was listed by about one-fourth of respondents as an audience, serving this target audience presents challenges. There is ample evidence that the public differ in their degree of knowledge about risk and, in particular, fire (see, for example, Winter and Cvetkovich 2003). Whereas experts might want technically relevant and appropriate information, the public may want culturally relevant

and value-based information (Fischhoff and others 1984, Plough and Krinsky 1987). Serving the layperson through the products and services would have different implications than serving the fire management community. For example, executive summaries or annotations attached to data presentations may be of greater interest to lay people. Some of our key informants from the fire-management community expressed an interest in this type of summary information.

The overlap between data offered through Predictive Services and other resources (such as National Weather Service) may need to be re-examined by Predictive Services. Overlap was sometimes viewed as a redundancy and, perhaps, a misallocation of resources. The daily fire weather forecasts, red flag warnings, Incident Management Situation reports, drought information, and interagency reports all seemed to be on target as products currently offered that are of high utility to our respondents. To augment these services, the Web sites might be adjusted to streamline information searches and to have more real-time updates (including removal of information that is no longer accurate or timely). Products that seemed of little interest and might be deleted included regional monsoon updates, upper air soundings, Predictive Services forms, and state-of-the-fuels reports. However, before these are dropped entirely, it might be helpful for the NPSG to explore with some key contacts what the intended purpose and barriers to use are. It may be that the information is of interest and would be useful in a different form. Some products are offered regionally rather than nationally, so consideration of availability is essential to interpreting low levels of use among these nationally distributed respondents.

A number of respondents seemed to have little awareness of the products and services offered and expressed a desire for more information on Predictive Services. Communication aimed at the fire management community to gain increased awareness of Predictive Services seems in order. In addition, respondents expressed a desire for a Web-based orientation or training, and a glossary to assist the user. These comments suggest that respondents who are currently aware of Predictive Services see benefit in imbedding more user support into the products to facilitate

utilization of the products as well as to ease comprehension of the products.

Lessons From Acceptability of Risk

Our respondents expressed a clear desire for accurate and timely information. When uncertainty was characteristic of the data, some indicated a desire to have a clear disclosure of the information's limitations and constraints. When respondents were asked to choose between erring on the side of caution when uncertainty might be involved in reporting, the majority leaned toward a cautious approach. However, we did not present detailed narratives or scenarios with tangible situations in our questions (see, for example, Kneeshaw and others 2004), which may have yielded a more complex picture of risk-reporting preferences. Specifically, if one had asked the respondent to contrast uncertainty in risk estimates for fires involving human life versus those burning in uninhabited areas, the results might have been different.

The Role of Trust and Confidence in Reliance

Most respondents expressed trust and confidence in the information provided by Predictive Services. Few respondents selected trust-related barriers as impediments to utilization. A few comments suggested that a personal working relationship with Predictive Services personnel would be essential to building and maintaining trust. In spite of this expressed trust, respondents tended toward not relying on Predictive Services in decisionmaking. A majority indicated they rely more on other sources than Predictive Services and were somewhat likely to take action based on Predictive Services. The regressions predicting reliance and taking action suggest that trust and confidence is the significant predictor in both cases (with gender, years in position, and level of education contributing to the overall equation, but not as individual significant contributors). This finding affirms the importance of trust and confidence in the delivery of Predictive Services products and services. The inverse relationship between years in current position and reliance and taking action support Reyna's (2004) finding that more experienced decisionmakers tend to capture the gist of factors leading up to a decision, rather than relying

heavily on technical details. Expertise leads to a different way of information processing, most likely reflected here. This also falls in line with the work of Siegrist and colleagues (2000) that shows an inverse relationship between the level of knowledge about a topic and the importance of trust. According to this line of research, those who know more about an issue tend to rely less on trust in making determinations about issues than do those with less knowledge.

Lessons for Risk Communications and Information in Threat Management

Our findings offer some lessons for risk communications and information in threat management. The importance of marketing the products and services to potential users was reflected in a desire among many respondents to know more about what Predictive Services offers. Additionally, risk information could be of greater value if Predictive Services provided tools to facilitate comprehension (e.g., appendices, glossaries, executive summaries). Stating constraints or assumptions used to gather, analyze, and report data could facilitate utilization further. This information would allow the user to better understand and make informed decisions about using the information presented. Although Reyna (2004) found a tendency to use the gist of information in arriving at decisions, she also found that decisionmakers prefer to distill information and arrive at this gist or fuzzy information on their own. Therefore, we do not recommend that a distilled approach be the only method of information presentation. The perception that there is an undesirable overlap between resources available from Predictive Services and other sources was an interesting revelation. Whereas overlap is intentional in this case, the driving force for it was user convenience. It may be that establishing Predictive Services as a unique niche for information would be preferred. Then links to other reliable sources for distinctly different information could be presented. This would eliminate overlap, but point the user to where they could find other information of interest.

Opportunities to Enhance Risk Communication

Respondents indicated that building relationships, dialogue with those providing the data, and assistance with

interpretation and underlying assumptions might facilitate their use of Predictive Services. Because trust and confidence influence both users' reliance on products and services and their likelihood of taking action, more of a direct connection to the fire-management community might be desired. This could be facilitated through the Geographic Coordination Centers and might be addressed through a hosted chat link or a hotline that users could call for assistance. Face-to-face briefings might also facilitate familiarity and relationships.

Tools to Move Us Forward

This user-needs assessment highlighted products used by fire personnel and factors that might be facilitators and barriers to Predictive Services usage. It helped clarify informant views on reliance on the products and services and the role of trust in that reliance. Ideas for refinement were offered that might help improve the existing products. These include considering the target audience when developing communication strategies, addressing overlap in available products and services, providing accurate and timely information that discloses assumptions and limitations, and developing Web-based tools that facilitate use. Study results also demonstrated the importance of trust in respondents' decisions to use Predictive Services information. Fostering relationships with users by involving them in the development and maintenance of products and services might increase user trust and usage of Predictive Services.

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Shared Values and Trust: The Experience of Community Residents in a Fire-Prone Ecosystem

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Abstract

The risk and impact of fires have been significant on the San Bernardino National Forest. It is important to understand how residents of areas surrounded by the forest perceive the impact of fires. If fire management agencies understand these perceptions, fire management agencies will be better equipped to communicate with publics about risk-reduction efforts that agencies, community residents, and property owners need to take. Issues of interest include residents' responses to fire risk, beliefs about personal and agency responsibility for addressing risk, personal experiences with fire, and stressors associated with living in a fire-prone area. These issues are examined in light of values perceived as being shared with the Forest Service and other community residents, as well as trust.

A series of studies of natural resource management issues surrounding risk to habitat, nonhuman species, and humans has informed our understanding of the role of perceived similar salient values and trust. Trust continues to be highlighted as an essential element of fire management and communication, and risk management and communication in general. However, the functions of salient values similarity and trust have not been explored in the context of the experience of residing in a community in a fire-prone area.

The authors arranged for residents of fire-prone communities surrounding an urban national forest to participate in focus-group discussions and complete self-administered surveys. It was found that most study participants had multiple fire-related experiences, and that many regarded the risk of fire as part of living in the mountains. Although participants considered the Forest Service and the California Department of Forestry to be primarily responsible for

reduction of fire risk, they also rated personal and community responsibility highly. When participants saw their own values and those of the Forest Service as similar with respect to fire management, they seemed to consider the consistency of agency actions with those values an important basis for making judgments to trust the agency. Public meetings with the Forest Service were supported, although some participants stipulated that the meetings needed to involve dialogue. Other means of communication were also supported. Implications for communication and collaboration, education, and management actions are discussed in light of the role of salient values similarity and trust in a risk environment.

Keywords: Beliefs, fire-prone communities, risk communication, risk management, salient values, San Bernardino National Forest, stresses.

Introduction

Trust and Risk Management/Risk Communication

Trust has been identified as an important component in examinations of public response in risk situations (Siegrist 2000, Siegrist and others 2000). Examinations of trust of publics in fire-management agencies have also been applied to fire-management issues (Cvetkovich and Winter 2004, Liljeblad and Borrie 2006, Schindler and others 2004, Winter and others 2002, 2004). General trends towards trust (Winter 2003) and distrust (Liljeblad and Borrie 2006) have been presented. Trust seems to be target specific and situation specific and involves degree of risk and perceived impacts (Kneeshaw and others 2004, Langer 2002, Winter and others 2004). Trust has been documented as an essential component of effective communication surrounding risk management (Covello and others 1986, Freudenberg and Rursch 1994, Johnson 2004, Slovic 2000). Those who trust the source of a communication about risk are more likely to believe the communicated message and more likely to accept initiatives designed to address the risk, including actions they must take themselves.

Past Studies on Values, Trust, and Natural Resource Management—

A series of studies examining the interactions between salient values similarity and trust has been conducted. Across these studies, salient values similarity has been a significant predictor of public trust in the Forest Service to address a number of natural resource management issues including a proposed program of research (Cvetkovich and others 1995), a recreation fee demonstration program (Winter and others 1999), and acceptance of approaches to management of threatened and endangered species (Cvetkovich and Winter 2003, Winter and Cvetkovich 2000, Winter and Knap 2001). Other significant influences that have been explored in conjunction with this line of inquiry include community of interest and place, ethnicity, gender, concern, and knowledge about the target topic. In one study (Cvetkovich and Winter 2003), participants repeatedly raised the issue of consistency between Forest Service actions and similar salient values. From this we built a pair of items and tested them with publics regarding issues of endangered species management (Winter and Cvetkovich 2008) and fire management (Winter and Cvetkovich 2007). We confirmed that consistency and validity of inconsistency are instrumental in further understanding patterns of trust and distrust among publics. These findings are outlined in greater detail elsewhere (Cvetkovich and Winter 2004). However, the study of attitudes towards fire and fire management (Winter and Cvetkovich 2007) involved random samples of residents residing in four Southwestern States, including those with little direct experience with fire.

Values and Trust in a Fire-Prone Community—

Variation in degree of experience with fire is undeniably an important consideration. It may be that direct experience represents an opportunity to develop greater personal knowledge about fire, which, based on past work, would then be expected to reduce the reliance on trust in making judgments about fire management issues (Siegrist and others 2000). Another explanation may be that trust has been blurred with issues centered on direct experience and reflects confidence rather than assessments of trust (Cvetkovich and Winter 2007, Earle and others 2001, Siegrist and others 2003).

The Present Study

The present study examined trust and salient values similarity among residents in fire-prone communities surrounded by a southern California forest. These residents were assumed to have direct personal experiences with fire, based on the fire regime and recent fire history of this forest.

Method

Participants—

Residents and homeowners ($n = 89$) in fire-prone communities on the San Bernardino National Forest participated in this study. Participants were invited through fire-safe councils, local announcements in newspapers and on radio, a forest district email tree focused on partnerships, and personal phone calls from the investigators. The majority (57.3 percent) of participants were male, white (92.1 percent), 55 years of age or older (68.6 percent), with at least some college education (85.3 percent, with 30.3 percent reporting some graduate study). Participants' total annual household income was between \$50,000 and \$74,999 (13.5 percent), or mostly greater (42.7 percent).

Survey Instrument—

A self-administered survey was created for the purposes of this study and included a number of Likert-type items focused on concern about fire and fire management (concern held by respondents and respondents' judgments of concern held by residents in general), knowledge about fire management (self, residents, and Forest Service), salient values similarity and trust, consistency and validity of inconsistency, personal impact of fire, and perceived effectiveness of fire-risk-reduction efforts. There were also items that examined personal experiences with fire (a series of yes/no items), actions taken (a series of yes/no items), and stress-related experiences related to fire risk (a series of yes/no items, adapted from the Impact of Event Scale-Revised, cited in Weiss and Marmar 1996). To assess perceived responsibility of various agencies, political representatives, scientists, visitors and tourists, and members of the community, participants were provided a list of 10 parties and were asked to assign points to each, where the sum was to be 100. An "other" option was provided so that respondents

Table 1—Perceived responsibility and performance of selected parties

Party	Number	Range of responsibility points	Average responsibility	Grade point average ¹
USDA Forest Service	85	5 - 80	$\chi = 18.68$, SD = 11.89	2.94
California Department of Forestry	79	2 - 50	$\chi = 14.25$, SD = 10.02	2.95
Local fire departments	76	5 - 40	$\chi = 11.51$, SD = 8.32	3.30
Federal legislators and representatives	65	1 - 40	$\chi = 8.80$, SD = 8.60	1.78
State legislators and representatives	61	1 - 25	$\chi = 6.70$, SD = 6.23	1.67
Local community	76	3 - 50	$\chi = 10.79$, SD = 9.06	2.37
Visitors and tourists	60	1 - 30	$\chi = 8.10$, SD = 6.35	1.27
Local business owners	52	1 - 12.5	$\chi = 6.55$, SD = 2.98	1.98
Scientists and researchers	52	1 - 20	$\chi = 7.06$, SD = 3.38	2.40
Me and the people who live with me	76	1 - 80	$\chi = 11.81$, SD = 12.39	3.10

¹ Represents the average of grades assigned by respondents for each party, ranging from 1 = F to 5 = A.

could add parties to the list. Respondents could leave point assignments blank or enter 0 in case of no responsibility for reduction of fire risk. Paired with that responsibility was a followup question, wherein respondents were asked to assign a grade in the range A through F to any party they had assigned points to (Table 1). The grade was to reflect how the party had performed in the past 12 months in reducing the risk of wildland fires in the San Bernardino Mountains.

Focus Group Protocol—

Participants were led through a series of discussion items regarding fire and fire management on the San Bernardino National Forest. These items included objectives for fire management, concerns in fire management, alternatives to accomplish fire management objectives, and shared values and trust in Forest Service fire management.

Procedure—

Each session lasted one and one-half hours and started with a statement of purpose of the study, the voluntary nature of responses, importance of respect of other views in the discussion, and ability to opt out of any items that made the participant uncomfortable. Participants completed the self-administered questionnaire and then were led through the discussion topics. Each discussion was audiotaped; a note taker recorded key comments and concepts to help anchor the transcription of audio records. A total of 10 sessions were conducted over a 3-week period.

Results

Resident Experiences With Fire and Fire Risk—

Participants reported a number of personal experiences with fire during their lifetimes. The vast majority had experienced seeing a wildland fire (91 percent), smoke from a wildland fire (89.9 percent), or road closure owing to a fire (87.6 percent). Additional experiences shared by the majority included evacuation from their homes on account of fire (69.7 percent), went without power, which was shut off to reduce fire risk (65.2 percent), and a prescribed burn near their homes (62.9 percent). Less common were having a family, friend, or close neighbor who suffered loss or damage to personal property (44.9 percent); loss or damage to personal property (15.7 percent); health problems or discomfort (14.6 percent); personal injury (5.6 percent); and a family, friend, or neighbor injured by wildland fire (5.6 percent). Reported health problems were primarily smoke related. Of the 11 potential personal experiences, an average of six were reported. Participants judged the direct, personal impact of fire on the San Bernardino National Forest. A majority (61.8 percent) selected a 6, 7, or 8 on the 8-point impact scale (1 = no impact, 8 = extensive impact), although almost one-tenth (9 percent) selected no impact (a rating of 1).

Stresses Experienced—

A list of 22 possible difficulties was presented, and respondents indicated which, if any, they had experienced in

the past 7 days with respect to wildland-fire risk. Almost one-third of our participants had not experienced any of the items listed (the modal response was 1). Slightly more than one-third (38.2 percent) indicated that “I avoided letting myself get upset when I thought about it or was reminded of it,” and almost one-third (29.2 percent) reported “any reminder brought back feelings about it,” as well as “I felt watchful or on guard” (29.2 percent). About one-fourth (25.8 percent) reported that “other things kept making me think about it,” and “pictures about it popped into my mind” (24.7 percent). About one-fifth (18.0 percent) thought about it when they didn’t mean to. Approximately one-tenth of our respondents reported “I had waves of strong feelings about it” (13.5 percent), “I tried not to think about it” (11.2 percent), “I felt irritable and angry” (9.0 percent), and feeling like they were back in a time when there was no fire (9.0 percent). Reporting of physical symptoms (sweating, trouble breathing, or nausea) was rare (only 3.4 percent). However, more than one-third (41.0 percent) indicated that more than one difficulty was experienced within the past 7 days.

Beliefs—

Beliefs about responsibility and performance in meeting that responsibility are reported in terms of responsibility points assigned to each and overall grade point average (or GPA) representing the average of the letter grades assigned to each responsible party (Table 1).

About Agency Responsibility

The three agencies listed were among the parties assigned the most responsibility. The USDA Forest Service received the highest average responsibility points. This means that respondents felt that the Forest Service had the greatest amount of responsibility for fire management in San Bernardino (Table 1). The agency received a B grade on performance. California Department of Forestry received the second highest average number of responsibility points and was also given a B grade on performance. Local fire departments had somewhat lower responsibility ratings—about equal to the ratings the respondents gave themselves and those who lived with them. Local fire departments were given a somewhat higher average grade for performance (about B+). A few respondents listed city and county plan-

ning agencies under the “other” option. When they did so, they tended to assign or give the agencies high responsibility ratings and below-average performance grades. Planning regulations and codes were also listed as having responsibility, and these were assigned below-average or failing performance grades.

About Responsibility of Federal and State Representatives

Responsibility of Federal and State legislators and representatives was also rated and was scored much lower than that of most other parties. Among the 10 parties listed, Federal legislators and representatives were ranked about sixth in responsibility for fire management. Although they were perceived as having less responsibility than agency-affiliated parties, Federal representatives were given a performance grade of C- (1.78 GPA). State legislators and representatives were seen as having even less responsibility (last among the parties listed), but were also given a grade of C- (1.67 GPA).

About the Responsibility of Other Parties

Other parties listed that did not fit the agency or representative categories included the local community, visitors and tourists, local business owners, and scientists and researchers. The local community was fifth in average responsibility, at the middle of the parties rated. The average performance grade for the local community was a C+ (2.37 GPA). Whereas visitors and tourists ranked only seventh in responsibility, and were given a below-average performance grade of D+ (1.27 GPA). This was the lowest grade assigned to any of the parties and open-ended comments reflected significant concern about the role of recreationists and tourists in fire. Local business owners were assigned the least average responsibility and were given a passing grade for performance (GPA of 1.98, or a C). Scientists and researchers were also assigned relatively little responsibility compared to other parties (eighth out of the 10 parties rated), and were also given a passing grade for performance (GPA of 2.40, or a C+). A few respondents mentioned environmentalists and environmental groups. In such instances, poor grades were given.

About Personal Responsibility

Most respondents (76) assigned at least some responsibility to themselves and the people who live with them.

Respondents and the people who live with them ranked, on average, as the third most responsible group out of the 10 rated. They received a B average (GPA of 3.10) for performance.

A number of actions that could be effective in reducing fire risk were reported. The vast majority of respondents had read about home protection from wildland fires (97.8 percent), implemented defensible space around their property (94.4 percent), and attended a public meeting about wildland fire (93.3 percent). A majority had also reduced flammable vegetation on their property (75.3 percent), worked with a community effort focused on fire protection (75.3 percent), made inquiries of the local fire safety council or volunteers on how to reduce fire risk (73.0 percent), made inquiries of the local fire department on how to reduce fire risk (64.0 percent), or made inquiries of the local forest ranger (56.2 percent). About a third had changed the structure of their home to reduce risk (38.2 percent) or worked on a wildland-fire-suppression effort (38.2 percent). Others had volunteered through various efforts or had worked through a fire-safe council.

An overall judgment of the effectiveness of those actions in reducing the risk of losing one's home during a wildland fire was requested. Effectiveness was rated on a scale from 1 to 8 (1 = not at all effective, 8 = extremely effective), and was rated positively ($\chi = 6.01$, $SD = 1.55$, median=6).

Participants were queried about barriers to effective reduction of fire risk, with one-half of the participants (50.6 percent) selecting "my neighbors have not done their part," Other entities that had not done their part according to respondents included public agencies (29.2 percent selected this barrier) and the Forest Service (22.5 percent selected this barrier). About one-fifth of respondents indicated various barriers to reduction of risk including: "I don't have adequate financial resources" (21.3 percent), "My own physical limitations" (21.3 percent), "I don't want to change the landscape" (21.3 percent), "I don't want to change my roof or other built structures" (20.2 percent), and "I'm not worried about fire risk" (19.1 percent). A few indicated, "I am not sure what will really work" (13.5 percent), or "I don't know who to call/hire to help" (3.4 percent). Other barriers

added by respondents were focused on land use policies, growth and housing, community restrictions on removal of trees and vegetation, a lack of coordination between agencies, and environmentalists.

Trust and Shared Values and Other Evaluations—

Using the salient values similarity model of trust, we examined dimensions of trust and shared values regarding community and the Forest Service.

Community

Participants were asked to rate how concerned San Bernardino National Forest community residents are regarding fire and the risk of fire. Using an eight-point scale (1 = not at all concerned, 8 = very concerned), residents were rated as concerned about fire ($\chi = 6.71$, $SD = 1.38$, median = 7, $n = 87$). This was only slightly below the average level of concern of respondents, who rated their own concern about fire at 7.38 ($SD = 1.00$, $n = 88$). Participants believed that San Bernardino National Forest community residents were somewhat knowledgeable about effective fire management ($\chi = 3.92$, $SD = 1.47$, median = 4, $n = 88$; using an eight-point scale, 1 = not very knowledgeable, 8 = very knowledgeable). However, they saw themselves as more knowledgeable ($\chi = 6.13$, $SD = 1.60$, median = 6, $n = 88$). When asked the extent to which fellow community residents share participants' values about fire management, the average response was above the midpoint on the scale, indicating moderately shared values ($\chi = 5.58$, $SD = 1.55$, median = 6, $n = 81$).

Significant positive relationships were found between rating of one's own concern about fire and rating of concern of community residents ($r = 0.305$, $p < 0.01$, $n = 86$), rating of one's own knowledge about fire and rating of knowledge of community residents ($r = 0.355$, $p = 0.001$, $n = 88$), and concern of community residents and perceived shared values with community residents ($r = 0.424$, $p < 0.001$, $n = 80$). Participants who believed that neighbors had not done their part and that this was a barrier to effective fire management did not rate others' concern, others' knowledge about fire, or perceived shared values differently (t-tests, $p > 0.05$) than did participants who did not believe this.

Forest Service

We were also interested in how knowledgeable participants believed the Forest Service to be regarding effective fire management on the San Bernardino National Forest. Participants' Forest Service knowledge ratings averaged 6.86 (SD = 1.32, median = 7, $n = 88$), indicating they believed the agency to be fairly knowledgeable.

Participants' ratings of the salient values similarity items indicated a perception of shared values (values: $\chi = 6.61$, SD = 1.53, median = 7, $n = 85$; goals: $\chi = 6.37$, SD = 1.75, median = 7, $n = 84$; views: $\chi = 6.31$, SD = 1.56, median = 6, $n = 81$). Less than 6 percent of the participants provided ratings below the midrange on each of these items, indicating dissimilar values. Participants were also asked to what extent they trust the Forest Service in their fire management efforts. Based on an 8-point scale (1 = I completely distrust the Forest Service, 8 = I completely trust the FS), responses leaned toward trust ($\chi = 5.85$, SD = 1.68, median = 6, $n = 86$), with the majority (64 percent) providing a rating of 6, 7, or 8 on the trust item.

We asked participants to indicate how often the Forest Service makes decisions and takes actions consistent with their values, goals, and views. A small portion selected never (1.1 percent) or rarely (5.6 percent), and about one-fourth (25.8 percent) selected sometimes. About one-third (33.7 percent) indicated Forest Service actions were usually consistent with their values, another fourth (24.7 percent) chose almost always, and a few (2.2 percent) said Forest Service actions were always consistent with their values. Participants were then asked to respond to "If or when the Forest Service makes decisions or takes actions inconsistent with my values, goals, and views, the reasons for doing so are valid." A few disagreed with the statement (3.4 percent completely disagreed, and another 15.7 percent disagreed). Almost one-third (31.5 percent) neither agreed nor disagreed. Almost half agreed that inconsistency between values and Forest Service actions was valid when it occurred (39.3 percent agreed, 4.5 percent completely agreed). One participant expressed this balance between trust and the perception that there are valid reasons why the agency might not get things done as follows: "I would trust

one of them with my life. The only problem is red tape and money constraints." Another participant pointed to policy-related constraints: "What I am thinking is that the people in the Forest Service have the rulebook and are playing by the rulebook and the negligence comes with the change in policy. Maybe we need to have a more flexible policy. I trust the Forest Service people, but they are stuck with the policy and they need to figure a way to change policies."

Relationship Between Trust and Salient Values

A significant portion of the variance in trust of the Forest Service was accounted for by similarity ratings for values, goals, and views ($R^{2adj.} = 0.468$, $F_{(3, 76)} = 2.4129$, $p < 0.001$). When consistency and validity of inconsistency were added to the trust prediction model that was based on salient value similarity for values, goals, and views, the resulting model accounted for an increased proportion of the variance in trust ($R^{2adj.} = 0.582$, $F_{(5, 70)} = 21.893$, $p < 0.001$). The most influential predictors in this equation were consistency of action with values ($t = 3.870$, $p < 0.001$) and shared values ($t = 2.546$, $p = 0.013$).

Relationship Between Trust and Other Fire-Related Items

The relationship between trust in the Forest Service and personal actions taken to reduce fire risk and number of perceived barriers to reduction of fire risk was examined. Neither actions nor barriers had a significant relationship with trust. Participants who had experienced more fire-related impacts (such as smoke or knowing someone who lost property) were less likely to trust the Forest Service ($r = -0.293$, $p < 0.01$, $n = 83$). Likewise, participants who reported more difficulties (such as having waves of strong feelings or feeling watchful and on guard) tended to trust the Forest Service less ($r = -0.366$, $p < 0.01$, $n = 80$).

Finally, grades assigned to the Forest Service in fire risk reduction efforts were related to trust ($r = 0.648$, $p < 0.001$, $n = 81$). An ANOVA to examine average trust ratings by grade was completed ($F_{(4, 76)} = 17.850$, $p < 0.001$). Participants who had assigned an F to the Forest Service had a mean trust rating of 2.50, the D group had a mean of 3.17, C's had an average of 5.67, B's had an average of 5.71, and A's had an average rating of 7.00.

Reasons for Reliance on the Forest Service

We asked participants to indicate whether or not a series of items were reasons to rely on the Forest Service's fire management on the San Bernardino. A majority felt that the following were not reasons to rely on the Forest Service: media coverage of Forest Service fire management (60.7 percent said this was not a reason), and Congress holds the Forest Service accountable for its fire management (52.8 percent said this was not a reason). The majority agreed or strongly agreed that the following were reasons they relied on the Forest Service: procedures that ensure the Forest Service uses effective fire management (67.4 percent), personal relationships I have with Forest Service personnel (59.6 percent), the Forest Service's past record of fire management (58.4 percent), and the laws controlling the Forest Service's fire management (52.8 percent). Participants were almost equally divided on "opportunities that I have to voice my views about fire management" (38.2 percent said this was not a reason, 46.1 percent said it was a reason).

Communication and Education

Participants had many relevant views on approaches to communication, collaboration, and education. The most preferred sources of information were public meetings the Forest Service leads so the community can ask questions (88.8 percent) and community meetings (84.3 percent). Other preferred information sources included a Web site (79.8 percent), brochures and pamphlets available on request (77.5 percent), articles in the local paper (77.5 percent), an email tree sent by a Forest Service representative and forwarded by fire-safe council volunteers (75.3 percent), local television/radio spots put on by a local Forest Service ranger (64.0 percent), and information and displays at Forest Service visitor center (60.7 percent). Additional suggestions included emails directly from the Forest Service, signs, a hotline or number residents could call to speak directly with someone knowledgeable, and messages on community bulletin boards. Flyers and newsletters left on residence doors were also brought up as a means of getting the word out. It should be noted that the strong support for community meetings and direct engagement with the Forest Service was expressed by participants who themselves had

come to participate in a meeting. There were many residents who did not attend. From many we heard about scheduling conflicts, burnout from so many meetings, or the need to have a direct, tangible outcome from the meeting before they would commit to participation.

Some residents did not participate because of road closures or weather-related concerns (an unusual series of late-season snowstorms and fog occurred during the study period). However, others told us they felt there was not adequate notice about our meetings. This was in spite of the radio and newspaper announcements, including media Web sites, as well as email notices and telephone calls from the researchers or through fire-safe councils. Identifying the most effective communication networks, including those that are community based, was an important part of our research effort, and we only had partial success. On one forest district, many of our contacts came through an email list derived from various partnership and collaborative efforts. This proved an invaluable resource to us, and the direct contact with a Forest Service employee who was known to residents helped pave the way. In sum, we found that a number of routes and contacts were necessary. These routes varied greatly and, in some ways, reflected the unique nature of the communities we tried to reach.

Discussion and Conclusion

Experiences in These Fire-Prone Communities

The majority of participants reported multiple fire-related experiences, though a minority had suffered personal injury or personal property loss. However, almost half knew others who had suffered loss or damage. Comments about fire risk revealed that many took the risk of fire in stride as part of living in the mountains. The one exception to this surrounded discussions about prescribed fire where participants mentioned the risk of fires getting out of control and the concern surrounding that management technique. A majority indicated that fire had an impact on them directly, but the reporting of stress-related experiences within the past 7 days reflected the time elapsed from the last fire event to the study period. We expect that this timing had something to do with the reports of stress-related events being somewhat low. Another factor may have been the active

role participants have taken in direct actions to reduce fire risk and to educate themselves about fire. This would be an interesting area for further research.

Responsibility and Performance

Participants were most likely to view agencies as having a majority of responsibility for reduction of fire risk, with personal and community responsibility following closely. Agencies, including the Forest Service, personal households, and community were viewed as doing fairly well, although respondents suggested that the Forest Service and neighbors might not always have done their part in reducing fire risk. Although assigned little responsibility overall, tourists and visitors were viewed as doing poorly in reducing fire risk. Comments indicated support for limitations on tourists and visitors, including more limits on access or more limits on forms of use (such as no fires at yellow post sites).

The Interplay Between Values, Trust, Risk, and Response

Whereas perceived salient values and trust were significantly related to each other, consistency between perceived shared values and actions taken by the Forest Service seemed to be more influential in determinations of trust than were the shared values alone. This may have been due to the relatively high average rating of perceived value similarity, paired with low variability. Direct personal experiences with fire and stressful impacts were both negatively associated with trust. Trust was significantly related to perceived effectiveness of the Forest Service in reducing fire risk. Given the role of trust in acceptance of agency actions and communications, we expected to find a relationship between direct actions and trust. However, the number of actions taken had no relationship to trust. This has interesting implications for study of the relationship between trust and public response. Perhaps only those actions directly advocated through the Forest Service might be expected to be influenced by trust and perceived similar salient values. Reliance on procedures and personal relationships seemed to be a factor in deciding to rely on the Forest Service's fire management efforts. The past record of fire management

seemed a bit less important but was still held by a majority to be a reason for reliance.

Implications for Communication and Education

A majority of the participants supported public meetings with the Forest Service, although comments made clear the need to have an open forum where they could ask questions and receive answers from a knowledgeable source. Most of the methods of communication listed are already employed in these communities to some extent, although some residents expressed the feeling that it had been a while since they had met with the Forest Service and that they were starting to feel out of touch with what was going on. Others who did not attend the study sessions expressed a sense of overload on meetings. Clearly, various kinds of contacts need to be used on an ongoing basis, and the use of community organizations and networks, including the fire-safe councils, seems to be an effective vehicle to include. Although media were included in the means of contact, the local newspaper received more support than television or radio spots. A Web site for current and community-based information seemed to receive strong support. One community declined participation because they were waiting for the agency to act on commitments made in prior meetings. This demonstrates the importance of following up with community members after meetings and keeping them informed on an ongoing basis. It would probably be helpful even to report efforts to meet commitments. If barriers were met, those could also be reported, as it seemed participants understood that funding, policies, and other challenges could prevent the Forest Service from taking action.

Gaps and Where We Go From Here

Participants were fairly similar and not representative of the overall populations within these forest communities. Although we made a concentrated effort to recruit seasonal residents, only a few actually participated. Whereas a past study sheds light on differences between seasonal and year-round residents of the San Bernardino mountain communities (Vogt 2008), some participants suggested that seasonal residents and those leasing or renting their properties were less concerned and less similar in values

than were the year-round community members. Additional studies of the differing perceptions of seasonal and year-round residents, including how the Forest Service and other fire management agencies view these two groups, would be of interest. The lack of relationship between personal actions taken and trust levels was somewhat surprising, although the relatively small sample size and little variance in trust may have suppressed any relationship between these two variables. The interest in meetings and information from the Forest Service, and an interest in maintaining an ongoing dialogue, were made clear. The need to report on actions taken, progress made, and barriers experienced by the Forest Service in its fire management efforts was affirmed. These steps would assist the agency in continuing to develop trust and a positive basis for interaction in these communities who sometimes view themselves as very alone in their efforts to reduce risk.

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Pests/Biota Syntheses

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Representing Human-Mediated Pathways in Forest Pest Risk Mapping

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Abstract

Historically, U.S. forests have been invaded by a variety of nonindigenous insects and pathogens. Some of these pests have catastrophically impacted important species over a relatively short timeframe. To curtail future changes of this magnitude, agencies such as the U.S. Department of Agriculture Forest Service have devoted substantial resources to assessing the risks associated with recent or potential forest invaders. These assessments of risk typically include a mapping component; among other things, this presents a useful way to organize early-detection/rapid-response procedures. However, forest pest risk mapping is often limited to readily available and manageable data sets, which results in representations of risk that heavily favor climatic factors or estimates of host species distribution. Detailed examinations of human-mediated pathways of spread are often neglected in forest pest risk analyses owing to a lack of spatial data or uncertainty about a pest's predictive model parameters.

Humans are the most important facilitator of forest pest introduction and spread. With expanding global trade and interstate commerce, the number of potential forest invaders is likely to rise, making the analysis of human-mediated pathways particularly timely. In this synthesis, we present a number of spatial data sources, collected by Federal agencies and private companies for a range of purposes, which can be utilized to represent these human-mediated pathways. Although general in nature, queries can often be used to tailor these data sets to address specific pests. Perhaps, most importantly, the source data can usually be acquired for free or at negligible cost.

Using the sudden oak death pathogen (*Phytophthora ramorum*) and other pests as examples, we illustrate how some of these data sources can be employed for mapping risks associated with human-mediated pathways. First, we demonstrate the use of foreign import cargo statistics—marine, airborne, and transborder—to assess the risk of introduction of new species at United States ports of entry. Second, we examine the utility of inland waterway cargo statistics, freight analysis networks, and other databases on domestic commodity traffic for mapping regional and local spread of forest pests. Third, we explain the diverse applications of business databases, not only to identify clusters of high-risk businesses, but also to rank these businesses using a suite of socioeconomic factors. Finally, we discuss the limited availability of up-to-date land use/landcover data, and present alternative data sources for representing high-risk areas of current urbanization as well as the forest-urban interface.

Whereas many of these data sets are imperfect depictions of human-mediated pathways, integration of several can add significant depth to early-detection/rapid-response projects. To facilitate further applications, we discuss user considerations, future information needs, and potential sources of additional data regarding human-mediated pathways.

Keywords: Commodities, dispersal, forest pests, freight, human-mediated, risk, WUI.

Introduction

In 2000, annual forest losses and control costs in the United States due to nonindigenous forest insects and pathogens were estimated at \$4.3 billion (Pimentel and others 2000), and that figure is likely to rise due to the increasing transport of species beyond their native habitats (Brockerhoff and others 2006b). In ecological terms, these pests alter forest composition, structure, and productivity; in some cases, pests basically remove host tree species from forest ecosystems (Brockerhoff and others 2006b, Levine and D'Antonio 2003). A case in point is the once-dominant American chestnut (*Castanea dentata* (Marsh.) Borkh.),

which was virtually eliminated from eastern forests by chestnut blight (*Cryphonectria parasitica*) within 50 years of the pathogen's introduction from Asia (Liebhold and others 1995). Over a similar timeframe, the hemlock woolly adelgid (*Adelges tsugae* Amnand) has caused extensive mortality throughout the Northeastern United States and has recently spread into the southern Appalachian Mountains, raising fears that it will decimate both eastern hemlock (*Tsuga canadensis* (L.) Carriere) and Carolina hemlock (*T. caroliniana* Engelm.) populations in the region. Notably, the adelgid was considered a mere nuisance pest of ornamental hemlocks for a few decades after its accidental introduction around 1953, until it began to spread to natural forest stands during the mid-1980s (Souto and others 1996). A related pest, the balsam woolly adelgid (*A. piceae* Rutzeburg), was also accidentally introduced in the latter part of the 20th century, and has removed more than 90 percent of Fraser fir (*Abies fraseri* (Pursh) Poir.) from already sparsely distributed spruce-fir communities in the southern Appalachians (Pimentel and others 2000). In a recent example, the emerald ash borer (*Agrilus plannipennis* Fairmaire) now infests an estimated 40 000 km² in the Great Lakes region and Ontario, where it has killed at least 15 million ash (*Fraxinus* spp.) trees and has regularly jumped quarantine boundaries. It is likely that the insect was introduced to the United States 10 or more years before it was first recognized in 2002, during which time populations were able to establish and spread undeterred (US-GAO 2006).

Forest Pest Risk Assessments and Risk Maps

Forest pest risk assessments are intended to provide forest managers with information to prepare for such situations, and in turn, reduce the number of invasive species that become harmful pests (Byers and others 2002). A risk assessment for a particular pest categorizes or quantifies its risk of introduction, establishment, spread if established, and potential economic and environmental impacts (UN-FAO 1995). The spatial representation of such an assessment is a map depicting the varying level of risk of introduction or establishment of a pest throughout a geographic region of interest (Andersen and others 2004b). Risk maps facilitate early-detection/rapid-response procedures by providing a

template for the design of regulatory programs and detection surveys or, if a pest has already been established in one part of the geographic area of interest, through control tactics that prioritize areas for which the risk of pest spread is the highest (Andersen and others 2004a).

Importance of Human-Mediated Pathways

Pest risk maps are typically assembled using spatial data from three principal subject areas: host species distribution, environmental factors affecting pest persistence (e.g., climate), and pathways of pest movement (Bartell and Nair 2004). Risk maps often focus on climatic factors or host species distribution or both (e.g., Meentemeyer and others 2004), perhaps because of a lack of information regarding the relevant pathways for a given pest. Nonetheless, human activities—particularly in the area of commercial trade—have received much attention for enabling the spread of invasive pests. Human-mediated pest transport is largely unintentional (Jenkins 1996): forest pests may travel undetected on a variety of materials including wood products, airline baggage, nursery plant stock, solid wood packing materials, hikers' clothing, and passenger vehicles (Campbell 2001, Liebhold and others 2006, McCullough and others 2006, Tkacz 2002, Work and others 2005).

In 2004, a joint workshop by the Society for Risk Analysis Ecological Risk Assessment Specialty Group and the Ecological Society of America Theoretical Ecology Section focused on the standardization of methods for risk assessment as a component of a National Invasive Species Management Plan. Workshop participants noted a critical need for data to represent pathways of introduction of nonindigenous pests that are related to human trade and transport (Andersen and others 2004b). This information need has been echoed by policymakers focused specifically on forest pests (Chornesky and others 2005). For forest pest risk mapping, the challenge is threefold. First, the amount of spatial data specifically collected for forest pests is limited. Second, it can be a challenge to apply available information on pest biology and other fundamental characteristics to the tasks of spatial data mining and interpretation, especially since that fundamental information may be spread across a wide range of journals, government documents, and

other literature (Ricciardi and others 2000). Finally, many human-mediated pathways operate on broad (i.e., landscape, regional, national, or even continental) spatial scales, impeding their analysis.

Goals of This Synthesis

During the past couple of years, we have collaborated with scientists from the U.S. Department of Agriculture Forest Service Forest Health Technology Enterprise Team (FHTET), the U.S. Department of Agriculture Animal and Plant Health Inspection Service (APHIS), and other organizations to develop national-scale risk maps for a suite of forest pests. In the process, we have gathered an assortment of data sets for analysis and representation of human-mediated pathways. These data are maintained and updated by Federal agencies, academic institutions, nongovernmental organizations, and, in some cases, private companies for a range of purposes. Many of the data sets are statistical and not explicitly spatial but can be used in combination with more readily available spatial data. Queries can often be used to tailor these data sets to depict spatial patterns or temporal trends, or both, that are relevant for specific forest pests. Perhaps most importantly, the source data can usually be acquired for free or at negligible cost, typically through Internet download. It should be noted that because they are generally not collected with forest pest risk assessment in mind, some of the data sets' characteristics (format, classification scheme, and resolution) limit their straightforward application. Based on our knowledge of human-mediated pathways, we have sometimes adopted simplifying assumptions in order to apply the data sets. Nevertheless, it is our belief that the data significantly improve the forest pest risk assessment process, despite their limitations.

Whereas it is infeasible to create an exhaustive list, in this paper we present an overview of data sets from four categories generally pertaining to human-mediated pathways of forest pest movement:

- Foreign cargo statistical data
- Domestic commodity movement data
- Business data
- Land use/landcover data

For each of these categories, we discuss conceptual linkages between the available data and the depiction of forest pest introduction or spread risk. We describe some specific data sets from each category, highlighting characteristics that should be considered by the potential user. We then illustrate their application using a series of examples related to several current forest pest threats. Finally, to facilitate further applications, we highlight some future information needs and note where researchers may look for additional data regarding human-mediated pathways.

Foreign Cargo Statistical Data

International trade comprised 12 percent of the country's freight tonnage in 1998, and that percentage is expected to double by 2020 (USDOT-FHA 2002b). Many nonnative species are transported to the United States in import cargo shipments. A fairly conservative projection suggests that international trade will result in the establishment of 120 nonnative insects and plant pathogens in the United States between 2000 and 2020 (Levine and D'Antonio 2003). For forest insects, the largest introduction risk seems to be associated with solid wood packing materials (Haack 2003, 2006). Some of these materials may sit unattended in distribution facilities near ports of entry for weeks or more (Campbell 2001). Several nonnative insects now detected in parts of the United States have been regularly associated with solid wood packing materials, including the Mediterranean pine engraver (*Orthotomicus erosus* (Wollaston)) and the sirex woodwasp (*Sirex noctilio* Fabricius) (Haack 2006, Hoebeke and others 2005, Lee and others 2005). Joint risk assessments by APHIS and the Forest Service suggest that the importation of unmanufactured wood, especially in raw log form, is a high-risk pathway of introduction for a diverse range of insects and diseases (Tkacz 2002). The commercial trade of nursery stock and plant materials is another likely pathway of introduction; evidence suggests that the sudden oak death pathogen (*Phytophthora ramorum*) was likely introduced via the commercial plant trade (Ivors and others 2006).

The Port Interception Network (PIN) database, maintained by APHIS since 1984, provides information on plant pests intercepted at U.S. ports of entry (Work and others

2005). A number of analyses have made use of the PIN database, along with other data sources such as the APHIS Agricultural Quarantine Inspection Monitoring (AQIM) protocol, to analyze trends in the places of origin and commodities most likely to carry nonnative pests, primarily insects (Brockerhoff and others 2006a; Haack 2003, 2006, McCullough and others 2006; Work and others 2005). Although informative, such analyses are constrained by the fact that only 2 percent of cargo is inspected (NRC 2002), and, thus, many pest introductions may go undetected. As an alternative, publicly available statistical data on international trade offer a means to rank the Nation's ports of entry according to forest pest risk. In particular, filtering such data for countries or commodity types of interest or both may reveal ports where the likelihood of intercepting a specific pest is greatest. In addition, certain data sets provide information on eventual inland destinations, facilitating simple but nationwide risk analyses.

A number of government agencies keep databases on some segment of foreign imports to the United States. The associated data are used to generate national reports on current import patterns, as well as changes in those patterns through time. The databases are similar in format; typically, each database record contains fields listing a particular port of entry, a country or region of origin, an import tonnage, and (sometimes) the commodity type. Records can usually be queried and summarized through time to determine the amounts of high-risk cargo arriving from high-risk locations of origin at individual U.S. ports of entry. Readily available coordinate information for these ports can be used to develop maps depicting relative risk of introduction for forest pest(s) of interest.

Foreign Marine Cargo Statistics

In 2001, more than 78 percent of the total U.S. import tonnage was transported on cargo ships (USDOT-BTS 2003). The U.S. Army Corps of Engineers, Navigation Data Center, Waterborne Commerce Statistics Center issues annual summary reports on inbound and outbound foreign marine cargo. The agency also provides public Internet access to statistical data used to generate the reports (USACE-NDC 2006a). Annually compiled data tables are available for

1997 through 2004. Each annual table, available in ASCII or DBASE format, has greater than 100,000 records listing the total weight tonnage of cargo shipments for a specific foreign port (and country) of origin, U.S. destination port, and commodity type. Commodities are classified into one of 41 categories based on the Navigation Data Center's Lock Performance Monitoring System. The U.S. destination ports listed in the tables may be mapped using geographic information systems (GIS) layers provided by the Navigation Data Center or with ports data included in the National Transportation Atlas Databases (USDOT-BTS 2005).

Foreign Air Cargo Statistics

The U.S. Department of Transportation, Bureau of Transportation Statistics publishes monthly commercial air carrier survey information as part of its T-100 databank (USDOT-BTS 2006c). A portion of the T-100 databank focuses specifically on inbound and outbound international flights, listing foreign airport and country, passenger counts, and total pounds of enplaned mail and freight. Currently, ASCII data tables may be downloaded for 1990 through 2005. Unfortunately, freight commodity categories are unspecified in the T-100 databank.

As a possible alternative, the U.S. Department of Transportation, Federal Highway Administration, Office of Freight Management and Operations publishes the Freight Analysis Framework (FAF) database in Microsoft Access format. The FAF compiles data from several sources, including international and domestic waterborne commerce data from the U.S. Army Corps of Engineers as well as the T-100 air cargo data. The most recent version, the FAF² database, compiles data from 2002 on freight movement by all transportation modes. Notably, although FAF² is intended to supercede the previous FAF¹ database (constructed using 1998 data), a supplementary data table in FAF¹ focuses specifically on international air freight shipments and contains projections of likely freight totals for 2010 and 2020 (USDOT-FHA 2002a). In this supplementary data table, commodities are classified by a two-digit code based on the Standard Transportation Commodity Code system (resulting in 40 categories). The geographic precision of this data table is limited, with shipment destinations

reported as States (rather than individual airports) and shipment origins designated only by trade region: Asia, Europe, Latin America, Canada, Mexico, and the rest of the world.

Trans-Border Cargo Statistics

The Bureau of Transportation Statistics publishes land border crossing data collected by the U.S. Department of Homeland Security, Customs and Border Detection Division (USDOT-BTS 2006a). The data detail the total number of loaded and empty truck and rail containers passing through specific United States-Canada and United States-Mexico land border and international ferry crossings. ASCII data tables are available for 1995 through 2004, but as with the T-100 air carrier databank, commodity categories are unspecified. The aforementioned FAF² database is perhaps a more useful source, presenting import and export land border crossing data from 2002 in a dedicated table of greater than 200,000 records (USDOT-FHA 2006a). Each record lists a unique combination of origin, destination, border crossing location (sometimes reported as a specific city, but often just a State), the cargo transport mode, the two-digit Standard Classification of Transported Goods commodity category (43 categories), and the total tonnage of that commodity. Origin and destination are reported as Canada, Mexico, or 1 of 114 U.S. metropolitan areas or rest-of-State statistical regions (USDOT-FHA 2006a).

Commodity Coding Systems

A consideration when using the above-described data sets is that any commodity information provided may be of limited specificity. The two FAF databases and the Navigation Data Center's marine cargo statistics all use two-digit commodity codes. A category that might be of specific interest for a forest pest, such as nursery stock, is likely subsumed by a much broader category (e.g., noncereal-grain agricultural products). This may mean that to use the data requires some assumptions, or instead, the use of other data to estimate the breakdown of the broader category. In addition, all of these databases use different commodity coding systems. For example, the Lock Performance Monitoring System, used with the foreign marine cargo data, has a category labeled 41 – forest products, lumber, logs, woodchips. In contrast,

the Standard Transportation Commodity Codes, used with the FAF¹ database, include category 24 – lumber and wood products, and the Standard Classification of Transported Goods, used with FAF², has two corresponding categories: 25 – logs and other wood in the rough and 26 – wood products. There appears to be movement towards use of the Standard Classification of Transported Goods, which would be beneficial for more sophisticated, multimodal analyses. Documentation for the FAF² database includes cross-walk tables for the three commodity coding systems mentioned here (USDOT-FHA 2006b).

Domestic Commodity Movement Data

Human activity enhances the spread of many invasive pests by enabling long-distance dispersal beyond their natural abilities. For instance, insect eggs, larvae, and adults may be accidentally carried on cars or other vehicles, as has been the case with the range expansion of the gypsy moth (*Lymantria dispar* (L.)) in the Eastern United States (Marshall 1981). Perhaps more significantly, domestic shipments of commodities may carry undetected insects and pathogens from ports of entry to previously uninfested locations with favorable conditions. For example, research suggests that the normal spread of the emerald ash borer through local flights has been greatly exacerbated in the Great Lakes region by human transport of infested saplings or firewood (Muirhead and others 2006).

As with the international cargo data, a number of government agencies have developed data sets that track the flow of commercial freight via various transportation modes across the country. These databases may be categorized as either networked or non-networked in structure, depending on whether their focus is on the flow of commodity types or on an arc-node pattern for modeling a particular mode of transport (e.g., freight trucks). The user must consider attribute resolution (i.e., the level to which commodities are specified) and what this allows in terms of filtering and querying the data for application in forest pest risk maps.

Non-Networked Commodity Data

The U.S. Census Bureau and the Bureau of Transportation Statistics jointly issue Commodity Flow Survey reports and

an accompanying database as part of a national Economic Census. The most recent iteration of this joint effort was completed using 2002 data (USCB and USDOT-BTS 2005). The Commodity Flow Survey database is a key source of geographically summarized data on the country's domestic commodity movement. The database is delivered with a customized interface that allows data to be filtered for commodity types, transportation modes, distance traveled, and origin and destination regions of interest. Moreover, previous versions (using 1993 and 1997 data) may be downloaded from the Bureau of Transportation Statistics' TranStats data warehouse (USDOT-BTS 2006b), permitting analysis of change through time.

The Commodity Flow Survey 2002 database comprises summary tables for several levels of geographic detail: national, regional, State, and metropolitan area. Significantly, the level of specificity for commodity information reported in a table is tied to the level of geographic detail. Far more specific commodity codes are available for broad-scale (e.g., region-to-region) flows than fine scale (e.g., metro-to-metro) flows. This can be an important issue to consider for pest risk mapping. For instance, *P. ramorum* has been spread to nurseries throughout the United States by infected nursery stock. In the Commodity Flow Survey database, the commodity category "nursery crops and live plants" is only available at a regional scale; finer scale tables use two-digit commodity codes, meaning nursery stock is lumped into an "other agricultural products" category. In such cases, it may be necessary to use ancillary data or combine tables from different spatial detail levels to adjust tonnages accordingly. Moreover, although the list of industries covered by the survey has expanded each time it has been completed, certain industries are generally excluded, including transportation, construction, and some retail industries (USCB and USDOT-BTS 2005). Farms and fisheries are also excluded, meaning that the survey's agricultural commodities data only illustrate shipment from market area to market area, rather than directly capturing areas of commodity production (USCB and USDOT-BTS 2005).

The FAF is a practical alternative to the Commodity Flow Survey database. The FAF² database includes a single

large table on domestic commodity flow (at the metropolitan area level) that incorporates Commodity Flow Survey data as well as other data sources (e.g., U.S. carload waybill for rail traffic, inland waterborne commerce statistics). The FAF² database incorporates these additional sources using a modeling approach; approximately 40 percent of the total commodity tonnage depicted in the FAF² domestic commodity table is outside the scope of the Commodity Flow Survey and is thus supplied by the modeling effort (USDOT-FHA 2006c). This adjustment is intended to create consistent nationwide coverage. Nonetheless, it is important to recognize that many of the reported tonnages are model-based estimates, and the degree of error in these estimates is not reported. Commodities are categorized in one of 43 categories (from the Standard Classification of Transported Goods), which is comparable to the level of detail for metro-to-metro flows in the Commodity Flow Survey.

Spatially Explicit Transportation Networks

Spatially explicit networks are regularly used for transportation planning and summary analysis. For example, the Federal Highway Administration's FAF¹ database includes a Highway Truck Volume and Capacity component based on 1998 data, with a similar component for the FAF² database due in late 2006 (USDOT-FHA 2002a). The Highway Truck Volume and Capacity component consists of a database table of freight truck traffic flows along major U.S. roadways, which can be linked to corresponding features in a vector GIS road layer. Daily estimates of the number of truck trips along more than 96,000 road segments are calculated programmatically from domestic commodity flow tonnage data (Tang 2006). This permits ranking of road routes most likely to carry trucked freight and the identification of major arteries of interstate freight movement. The network does not include commodity categories, nor have nodes in the network been linked to corresponding populated places or other meaningful locations, although this can generally be addressed using ancillary data sets.

The National Transportation Atlas Databases (NTAD) are a collection of GIS data sets compiled and edited by the Bureau of Transportation Statistics. The most recent version of the NTAD collection was issued in 2005 (USDOT-BTS

2005) and contains two arc-node networked data sets: the National Highway Planning Network and the National Rail Network. The National Highway Planning Network contains greater than 176,000 arcs and 139,000 nodes, whereas the National Rail Network contains greater than 167,000 arcs and 130,000 nodes. Although neither data set contains explicit information on commodity flow along arcs in the network, the nodes are linked to populated places and other meaningful locations (e.g., airports, railyards). In short, these networked data sets depict routes at a fine scale but may need to be linked to commodity information from some other data source to make them useful for assessing broad-scale patterns of forest pest risk.

The U.S. Army Corps of Engineers Navigation Data Center publishes annual Waterborne Commerce Statistics of the U.S. data sets (USACE-NDC 2006b). The annual data sets, spanning the years 1993 to 2004, are broken into four reporting regions: Atlantic, Pacific, Great Lakes, and Mississippi Valley/gulf coast. This is probably the most complete of the networked data sets, providing information on the tonnages of commodities moving along all major inland rivers. Each record in an annual data set has codes for commodity type, direction of traffic (e.g., inbound receiving or outbound shipping), and the specific waterway, which can then be linked to corresponding GIS data. The 146-category commodity coding system is the most elaborate of any of the domestic or international commodity data sets (e.g., five categories for wood products, including fuel wood, wood chips, wood in the rough, lumber, and forest products not elsewhere classified). Nevertheless, it is important to consider that the vast majority of domestic commodity movement is by truck (with rail a distant second), so the networked waterway data may have limited utility for analyzing potential forest pest pathways.

Business Data

Business data sets relevant to the topic of human-mediated pathways fall into two general categories: those that describe general geographic patterns of business activity and those that can be used to pinpoint and analyze individual businesses. The former can be used to describe regional trends in business activities that may increase or decrease

Table 1—Example of the North American Industrial Classification System

NAICS 2002 code and brief description	
11	Agriculture, forestry, fishing, and hunting
111	Crop production
1114	Greenhouse, nursery, and floriculture production
11141	Food crops grown under cover
111411	Mushroom production
111419	Other food crops grown under cover
11142	Nursery and floriculture production
111421	Nursery and tree production
111422	Floriculture production

An example from the 2002 NAICS coding system, showing the hierarchy of sub-classes related to nursery production within the primary business sector Agriculture, Forestry, Fishing and Hunting (NAICS Code 11).

forest pest risk. For example, it may be possible to identify geographic areas in the United States with high levels of retail nursery sales, which may be relevant to the risk of introducing a forest pathogen. Unfortunately, such data sets cannot identify specific businesses that may represent key nodes in pest dispersal pathways. It may be useful to identify those key nodes as well as elucidate meaningful spatial patterns in terms of certain of their characteristics (e.g., business location size or sales volume). For example, a large number of retail nurseries in the Eastern United States received *P. ramorum*-infected plants from west coast nurseries during the last few years. Because natural forests around infected nurseries face an elevated risk of potential infection, as do the residential landscapes served by those nurseries, the mapping of specific nursery locations may help determine the best placement of detection survey plots. This is possible using commercially available business location databases.

Industry/Business Classification Systems

There are two primary industry/business classification systems. The North American Industrial Classification System (NAICS) has become the accepted system for the United States, Canada, and Mexico, replacing the older Standard Industrial Classification (SIC) codes (USCB 2006c). Most of the business data sources discussed in subsequent sections use NAICS codes (or both), and cross-walk tables between SIC and NAICS codes are available (USCB 2006c). The NAICS is a hierarchical classification system built upon

20 primary business sectors (USCB 2006c). The system is regularly updated on the same time step as the U.S. Economic Census (described below). The 1997 and 2002 versions of the NAICS have been released, and the 2007 version is in development. A 1,400-page manual describing the classes in the 2002 system is available from the NAICS Web site. Table 1 shows a sample of the hierarchy related to nursery production. Notably, the finest level of detail separates nursery and tree production from floriculture production. Whereas this is more specific than the commodity coding systems described previously, it is still broad in some regards (e.g., not separating nurseries that specialize in trees from those that specialize in shrubs).

General Business Data

The U.S. Census Bureau is a good source of general business patterns data. In particular, the bureau releases an economic census every 5 years, with the most recent version compiled using 2002 data (USCB 2006a). The data are summarized for geographic units as fine-scale as ZIP code areas. Types of information that are available include per-area dollar values of sales, receipts, or shipments for a particular industry class, as well as per-area counts of the number of businesses within a particular industry class. By linking the economic census data to GIS data layers, it is possible to map region-to-region patterns in business activities, and—using information from a previous economic census—to highlight short-term growth or decline in business sectors of interest. Economic census data sets may be downloaded via the Census Bureau's American Fact Finder file transfer protocol interface (USCB 2006b) or are available on DVD-ROMs purchasable by subscription (USCB 2006a).

With respect to the agricultural business sector, the U.S. Department of Agriculture, National Agricultural Statistics Service (NASS) issues summary reports on production of a wide variety of crops, including nursery plants (see the NASS Web site for access to a wide range of reports, <http://www.nass.usda.gov/index.asp>). Data tables in these summary reports are typically available as ASCII files. Similar to the Commodity Flow Survey, the attribute resolution for a given data table depends on the degree of

geographic detail. For example, regarding *P. ramorum*, one might be interested in nursery stock production coming out of California, Oregon, and Washington. A national-scale NASS report on nursery crops lists the total number of producers and sales quantities for 18 States (including the above three States), broken down into specific nursery stock type (e.g., broadleaf evergreens, a category that includes the most significant hosts of *P. ramorum*) (USDA-NASS 2004b). However, the report only details operations with greater than \$10,000 in annual sales. In contrast, regularly issued State-level Census of Agriculture reports list the number of all farms that normally have greater than \$1,000 in sales, as well as the total greenhouse and open-air acreages per county for broader crop categories such as all nursery stock (e.g., USDA-NASS 2004a). Notably, if a county has just a few operations in a given crop category, greenhouse and open acreages are withheld to protect farmowner privacy.

Business Location Databases

ReferenceUSA is a subscription-based online business database (InfoUSA 2006). Users have access to more than 13 million detailed records on business locations in the United States and Canada. The data were compiled from phone books, business registries, and phone call verifications of individual businesses. Records include fields describing a business's primary industry classification, as well as other industry classes with which it might be associated. Each record has been assigned geographic coordinates. Attribute fields that might be used to sort records include size (square footage), estimated sales volume range, and number of employees. The database may be searched and filtered by location (city, State, ZIP code, and others) or industrial classification codes. Selected records may be downloaded as ASCII files and compiled in statistical software or mapped in GIS software.

Other companies (e.g., Dun & Bradstreet) have developed similar packages, but none are free. Many libraries and government agencies have subscriptions to ReferenceUSA, so at least limited access is available to many users. A related issue that might be of concern to a user is the limitation on the number of records that may

be downloaded at one time (the size limitation depends on the paid subscription price). A user must also remember the sometimes-broad scope of industry classifications, and that an individual business may define itself by a number of industry classifications.

Land Use/Landcover Data Sets

The last data category has a loose definition because it involves a wide assortment of data formats and sources. It is beyond the scope of this article to describe all land use/landcover information that may depict some aspect of human activity relevant to pest spread. Instead, we will highlight what we see as two main uses of these data. The first purpose is to map areas that have appropriate conditions to sustain forest pests and that, ultimately, may permit them to spread into naturally forested landscapes. These data sets depict areas where a pest is likely to invade a region and have access to at least a few host individuals. A chief example is spatial data depicting the extent and nature of the forest-urban interface in a given region. The second purpose is to find areas of land use/landcover change owing to human development. The fragmentation or disturbance of natural forests owing to urbanization leads to more chances for forest pest introduction or establishment or both (Chornesky and others 2005). For managing pest risk, it would be preferable to pinpoint areas currently undergoing such changes or facing such changes in the near future. This is a significant challenge, as we discuss below, owing to the timeframe at which most land use/landcover data sets are currently collected and published.

Forest-Developed Interface Data

Landcover maps derived from satellite imagery—such as the U.S. National Land Cover Data (NLCD) (Vogelmann and others 2001)—may be analyzed using moving-window functions (e.g., Riitters and others 2002) to highlight edges between forest and developed landcover types. This may be a straightforward way of identifying interface areas with a high level of pest risk, but it may be even easier to take existing data products and apply them in a forest pest context. For instance, the SILVIS Lab at the University of Wisconsin has created GIS coverages of wildland-urban

interface (WUI) for the conterminous United States. The coverages, composed of U.S. Census blocks, distinguish between WUI (where a block contains less than 50 percent wildland vegetation but is within 2.4 km of an area at least 5 km² in size with greater than 75 percent wildland vegetation) and intermix (where wildland vegetation occupies greater than 50 percent of a block). SILVIS Lab researchers combined housing density information, landcover percentages derived from 1992 NLCD, and forest proximity measurements to classify blocks into 1 of 13 classes of WUI or intermix. For each block, they calculated the degree of interface/intermix based on both 1990 and 2000 housing density estimates, allowing spatial analyses of changes through time (UW-SILVIS 2006). Although created with a primary focus on fire risk, the WUI data are easily translatable to forest pest risk (Radeloff and others 2005). High levels of intermix may be of particular interest; such intermingling between wildland and residential land may significantly reduce the functional distance between a forest pest and its host species.

Contemporary Land Use/Landcover Change

The wall-to-wall land use/landcover classification provided by the NLCD is sufficient for many analyses. However, remote sensing-derived landcover data sets present some difficulties. In particular, the most recently available complete NLCD coverage dates from 1992, although the 2001 NLCD version is approaching completion. This is because the processing of satellite imagery into classified maps, and subsequent accuracy assessment, is a costly, labor-intensive process. Unfortunately, this means that there will typically be a long time step between subsequent wall-to-wall land use/landcover data sets. This fails to address short-term or recent land use/landcover changes, both of which may be of great interest when assessing forest pest pathways.

As an alternative, it may be possible to use vector GIS data sets that are more regularly updated to depict more recent land use/landcover changes. For instance, many GIS users have access to fine-scale (1:100,000-scale or greater detail) roads data through the Environmental Systems Research Institute (ESRI) Streetmap extension. These roads data are supplemented versions of U.S. Census Tiger

data (ESRI 2005), with new versions of the data issued almost annually. By creating a nationwide grid of cells and intersecting a roads data set with this grid, it is possible to derive an estimate of road density (i.e., the total road length per grid cell). This process can be easily repeated for data sets of any given year, enabling analysis of road density changes. Road density change can serve as an indirect surrogate for areas of general development (i.e., new roads lead to nearby residential and/or commercial construction). This may be an efficient method to highlight areas of increased disturbance owing to construction and areas of potential interaction between pests and hosts.

Application Examples

In the following sections, we detail ways that data for representing human-mediated pathways of forest pest spread have been or may be used for analytical purposes. All of these examples describe ongoing projects, and we would point out that validation or other assessment procedures have not been completed. Although that is an ultimate goal, for now we wish to simply illustrate some of the capabilities of the described data sets and, thus, encourage users to integrate them in additional forest pest risk mapping efforts.

Sirex Woodwasp and Foreign Marine Cargo Statistics

The sirex woodwasp primarily attacks *Pinus* spp. but, occasionally, will attack other conifers in the Pinaceae family. It is a native of Europe, Asia, and northern Africa, where it is rarely a pest. However, it has been accidentally introduced to many parts of the world, resulting in outbreaks in pine plantation forests in Australia, New Zealand, South Africa, and parts of South America (Hoebeke and others 2005). *Sirex noctilio* was intercepted at U.S. ports of entry more than 100 times between 1985 and 2000 (Hoebeke and others 2005). In early 2005, the Forest Service's FHTET organized a team to map the risk of *S. noctilio* introduction and establishment in the United States. An important component of the *S. noctilio* introduction risk map was a ranking of U.S. port locations based on their relative likelihood of receiving cargo carrying the pest. This ranking was constructed using the U.S. Army Corps

of Engineers foreign marine cargo statistics database. The database was queried to select records (from all years available at the time, 1997 to 2003) associated with countries with a known or accepted *S. noctilio* presence as well as with commodities that might harbor stowaways of this pest. The commodity categories were selected by consulting APHIS PIN data to identify commodities typically linked to the pest; commonly, these were goods that are associated with wood packaging materials. The resulting map of 151 marine ports (Figure 1) suggests that numerous ports in the Eastern United States face a substantial risk of accidental *S. noctilio* introduction. In particular, several high-risk points of entry in the Southeastern United States are close to large amounts of potential host forest. Notably, a single specimen of *S. noctilio* was found in a trap a short distance from the port of Fulton, New York, about the time this project was undertaken. Subsequent delimiting surveys during summer 2005 detected the pest more than 30 mi inland from the port, and 2006 surveys have detected the pest as far south as Pennsylvania. Surveys for *S. noctilio* in other parts of the United States are ongoing, with the survey design driven by the risk maps developed for this project.

Wood Product Flows From Canada and Mexico Into the United States

Cargo statistics data can also be used to explore risks associated with general pathways rather than specific pests. As an example, in 1998 the Forest Service issued a pest risk assessment regarding the importation of unprocessed *Pinus* and *Abies* logs from Mexico (Tkacz and others 1998). The assessment noted substantial risks of forest pest introduction through wood flows from Mexico. This issue was addressed in recent Federal regulations, which now require treatment of raw wood products from Mexico (USDA-APHIS 2004a). Currently, wood may travel from Canada under a general permit, but with recent restrictions imposed on pine shoot beetle host material (USDA-APHIS 2004b). We used FAF² trans-border cargo statistics (from 2002) to map the regional flow of wood products into the United States via land transport from Mexico and Canada (Figure 2). Notably, although "wood products" and "logs and other wood in the rough" are separate commodity categories in the FAF² database,

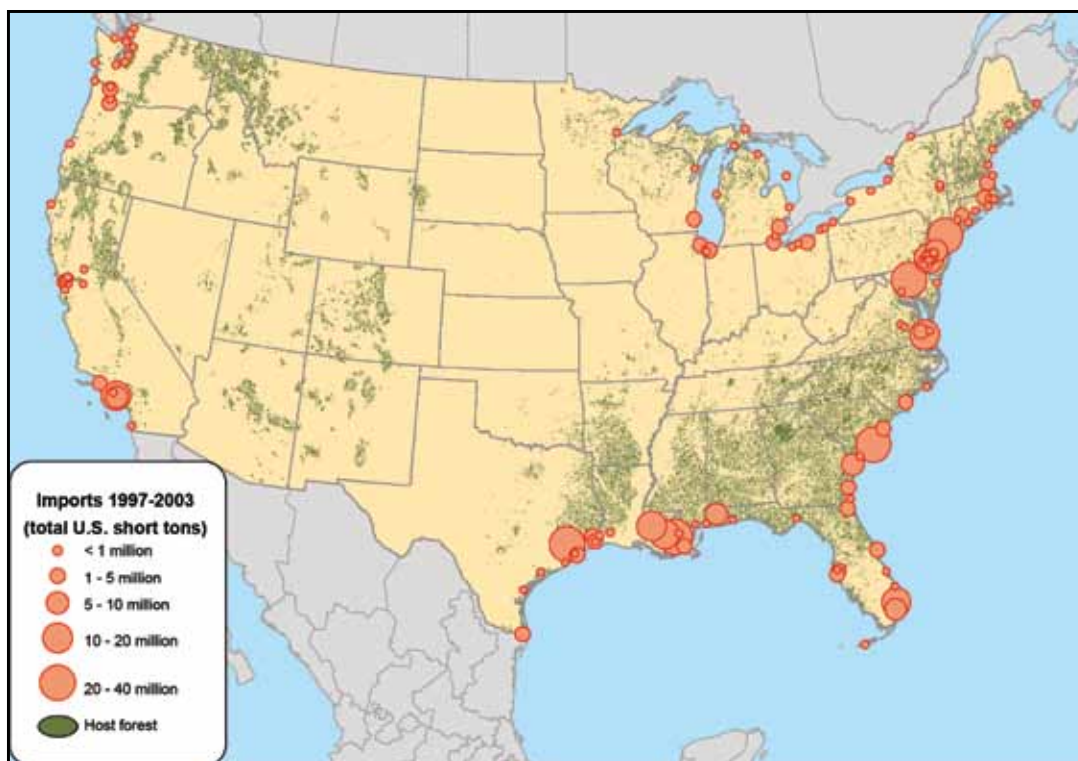


Figure 1—Potential introduction risk of *Sirex noctilio* at U.S. marine ports of entry. Map of U.S. marine ports handling commodities from countries where *S. noctilio* is known or believed to be established. Port locations are scaled in size according to the total tonnage (U.S. short tons) of high-risk commodities received from 1997 to 2003. The background is the distribution of susceptible forest (primarily *Pinus* spp.) in the conterminous United States, generated through spatial interpolation of Forest Inventory and Analysis phase 2 plot data.

they are not distinguished in the border crossing data table. In this simple analysis, Mexico accounts for a very small percentage of total trans-border wood product shipments; however, most States, particularly Texas and California, do receive some volume of Mexican wood products. Most of the Canadian wood products are destined for States just across the border, but some southern metropolitan areas such as Atlanta, Los Angeles, and Houston also receive a large quantity of wood products from Canada.

Truck Traffic Data and Risk of *S. noctilio* Spread Beyond Points of Entry

The example in “*Sirex* Woodwasp and Foreign Marine Cargo Statistics” described a process for ranking U.S. marine ports of entry according to their relative risk of importing cargo infested by *S. noctilio*. This example builds on that by using spatially explicit transportation network data to predict likely locations of *S. noctilio* spread beyond

those initial ports of entry (Figure 3). We employed the FAF¹ Highway Truck Volume and Capacity data to classify U.S. roadway line segments into five classes based on daily truck traffic (roughly 8,000 of the > 96,000 line segments in the network reported no measurable freight traffic). We then defined a set of 96 cities (i.e., urban area polygons defined by the U.S. Census) as ports by tracing the highest-traffic route leaving each marine port of entry point and identifying the first census-designated city along that route. If there were two equally high-traffic routes leaving the port, we selected the first cities encountered along both routes. Naturally, some marine points of entry were located within major urban areas, but a number were geographically distinct, making the tracing of high-traffic routes particularly important. To create a set of potential markets, we intersected the Highway Truck Volume and Capacity data with a coverage depicting census-delineated populated areas. If a polygon in this coverage intersected a

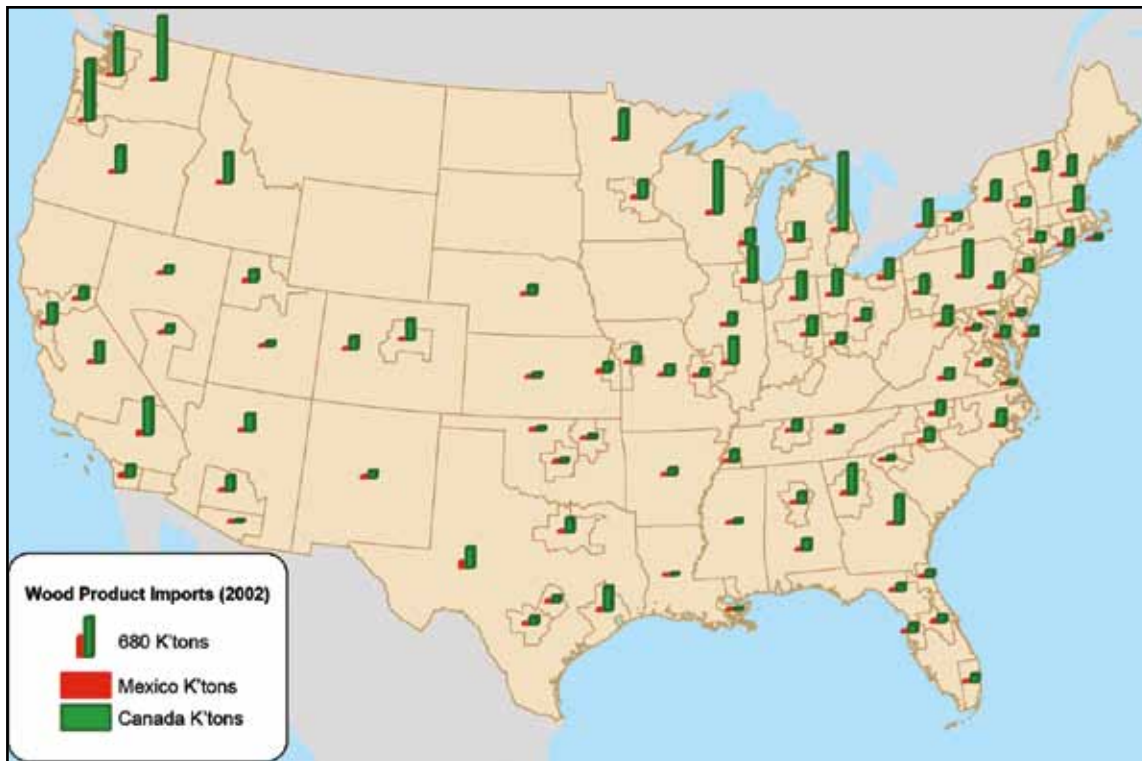


Figure 2—Trans-border wood product flows to United States from Mexico and Canada. Map of 114 metropolitan areas or rest-of-State statistical regions in the conterminous United States. The bar graphs show 2002 wood product imports from Mexico and Canada to each region, reported in thousands of tons.

line segment with a measurable level of truck traffic, then it was incorporated in the final markets layer, which included 2,921 urbanized areas. The port and market layers generated by these analyses, as well as the marine ports data described in “Sirex Woodwasp and Foreign Marine Cargo Statistics,” became the primary pieces of the national-scale risk map for *S. noctilio* introduction. As was noted earlier, this map is being used to guide *S. noctilio* detection surveys throughout the country.

Network Model of Nursery Stock Flows

First recognized in the mid-1990s, *P. ramorum* has infected live and red oaks in coastal forests in California and Oregon, with mortality rates of approximately 40 percent (Garbelotto and others 2001, Rizzo and Garbelotto 2003). More critically, the pathogen affects dozens of shrub host species. Many of these shrubs persist after infection and yield large quantities of aerially dispersed spores (Davidson and others 2002, Tooley and others 2004). A number of these shrubs (e.g., rhododendrons, azaleas, camellias) are

popular landscaping plants that are often sold as nursery stock (Garbelotto and others 2001, Tooley and others 2004). In the past few years, wholesale nurseries in California, Oregon, and Washington unknowingly shipped hundreds of infected plants to retail outlets in 39 States (Stokstad 2004). Although surveys have not detected *P. ramorum* in natural forests outside the west coast, there is some concern that this pathway may lead to spread of the pathogen from infected nursery stock planted in home landscapes to susceptible forest, of which there is a substantial acreage in the Eastern United States. As already noted, nursery stock is not well represented in the Commodity Flow Survey, so we have adopted an integrated modeling approach to examine the issue of potential *P. ramorum* dispersal via nursery stock. By combining two networked data sets (the FAF¹ Highway Truck Volume and Capacity data and the National Highway Planning Network), we created a nationwide, arc-node GIS infrastructure for dynamic modeling of nursery stock flows (see Figure 4 for regional example). We labeled

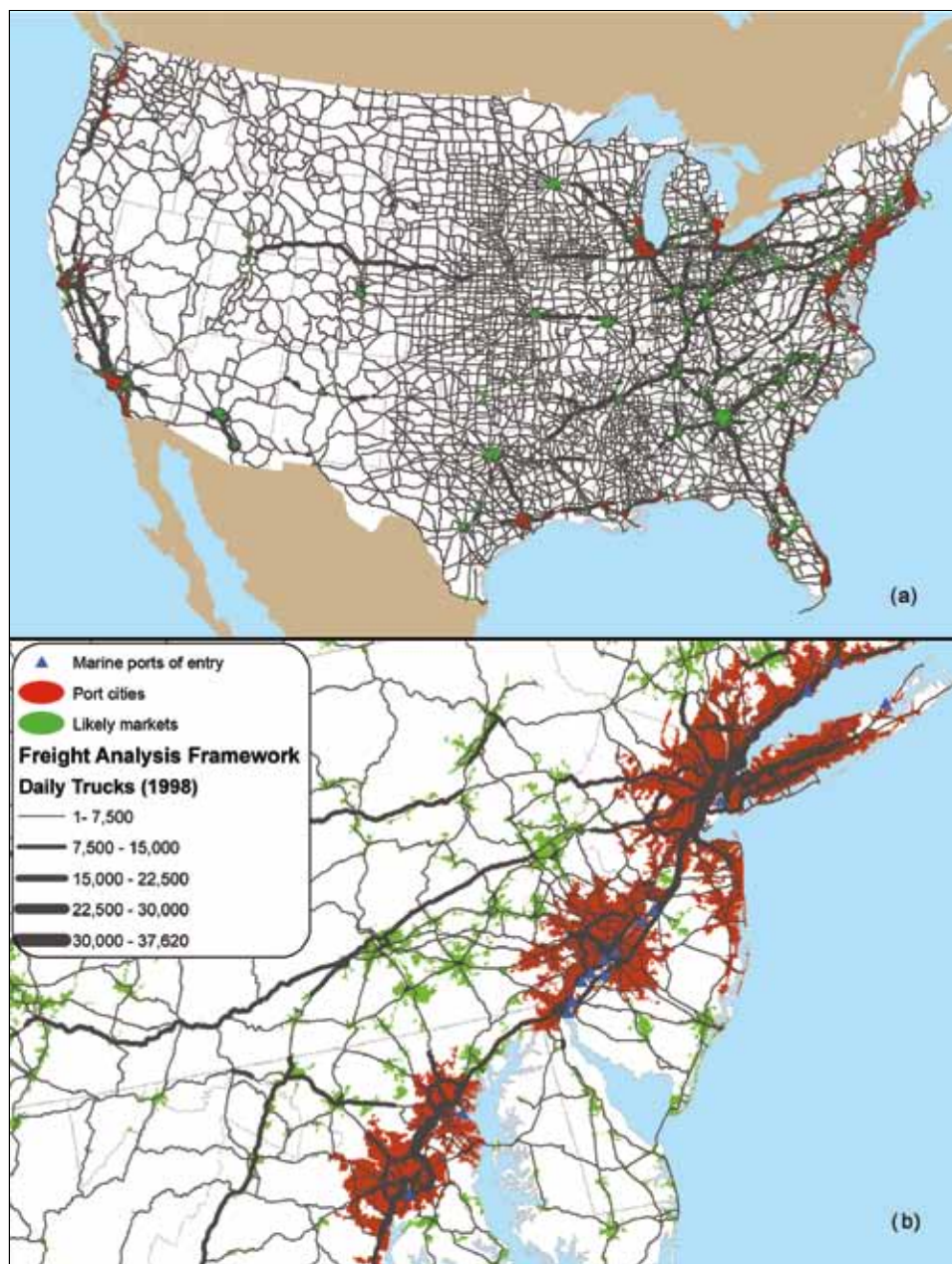


Figure 3—Port and market cities at risk for spread of *Sirex noctilio*. Figure 3a is a map of U.S. urban areas that have a high risk of *S. noctilio* spread. These areas are either port cities or likely markets for imports associated with the pests. Figure 3b is a closeup view of the Washington, DC-New York City corridor, showing roadway routes leaving marine ports of entry, ranked by their volume of truck traffic.

nodes as entry, destination, or transit depending on their roles in the model: nodes in proximity to U.S. wholesale or production nurseries are designated as entry nodes, destination nodes correspond to populated places in the United States, and transit nodes are unweighted link points in the network. We have adopted a Bayesian approach to model

probabilities of transmission of *P. ramorum* from entry to destination nodes within the network. For destination nodes, probability of infection is based on the number of plants received from entry nodes as well as the probability that some of the received plants are infected with *P. ramorum*. The number of plants shipped from any entry node is based

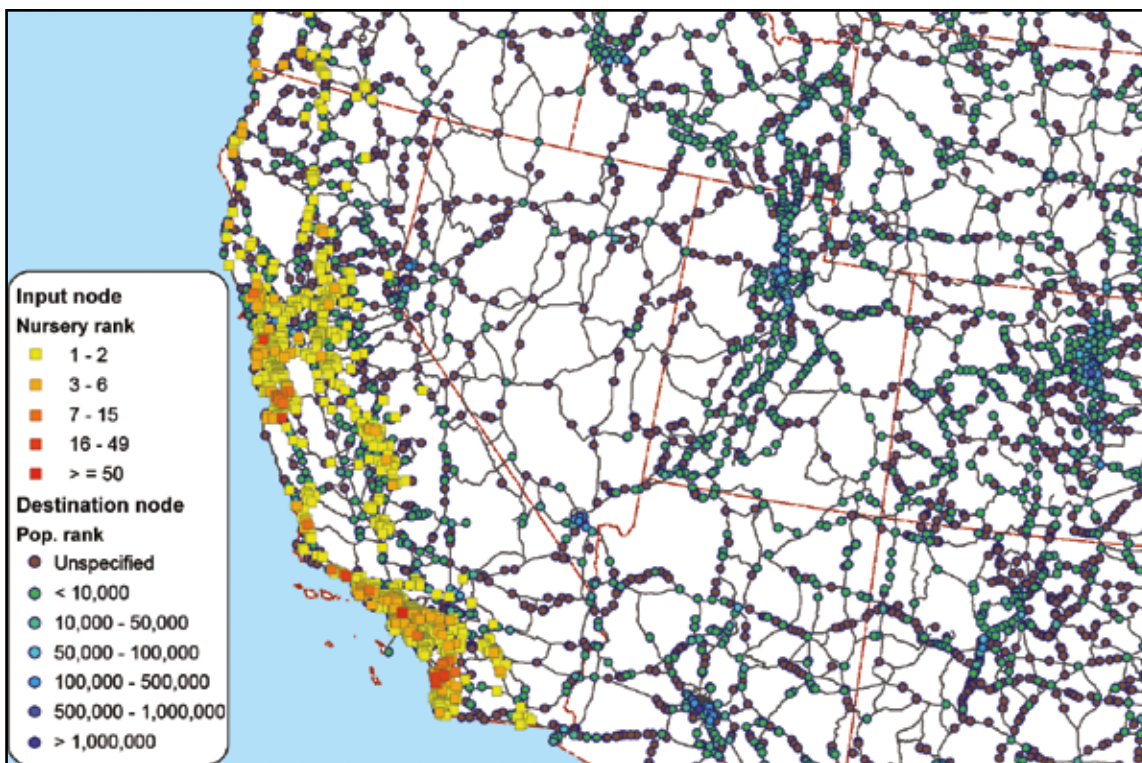


Figure 4—Linear network modeling of *Phytophthora ramorum* spread via nursery stock flows. Regional closeup (California and the Southwestern United States) of a national arc-node network model for predicting potential long-distance dispersal of the *P. ramorum* pathogen on infected nursery stock. The model links wholesale and production nurseries on the west coast to cities and towns throughout the region and the rest of the country. Input nodes are ranked and color-coded by the number of wholesale or production nurseries or both in close proximity. Destination nodes are ranked and color-coded by U.S. Census population.

on the number of nurseries in proximity and the likely volume of outgoing nursery stock, calculated from the U.S. Commodity Flow Survey, the FAF² database, and NASS reports on county-level nursery production (described in more detail in “General Business Data”). The probability that some plants received at a given destination node are infected with *P. ramorum* is calculated from infection rates observed in trace forward data (i.e., inspections of plants shipped from known infected production nurseries to retail outlets), and trace back data (i.e., determination of the source of plants found infected at retail outlets) collected by APHIS. Probabilities are also modified by factors affecting how much nursery stock is sold and planted in a given area (e.g., U.S. Census population). We believe the network modeling framework, when completed, will be general enough that it should be applicable for a variety of commodity flow analyses relevant to forest pest risk.

P. ramorum and Production Nurseries

Production nurseries present a higher risk of infecting nearby forests than retail nursery outlets due to several factors: large quantities of stock grown onsite; artificially high moisture levels conducive to the pathogen; and disproportionately large amounts of interface with natural forests. We queried the Reference USA database for all businesses with nursery and tree production as their primary industry classification. This query yielded more than 6,500 business locations nationwide. We ranked the selected businesses into four classes based on their total square footage (from the Reference USA database) and mapped them in relation to overstory host species distribution, represented in the example by a surface depicting the forest percentage of red oak, live oak, or both calculated from Forest Inventory and Analysis data. Figure 5 shows a subset of the nationwide coverage that includes the States of North Carolina and

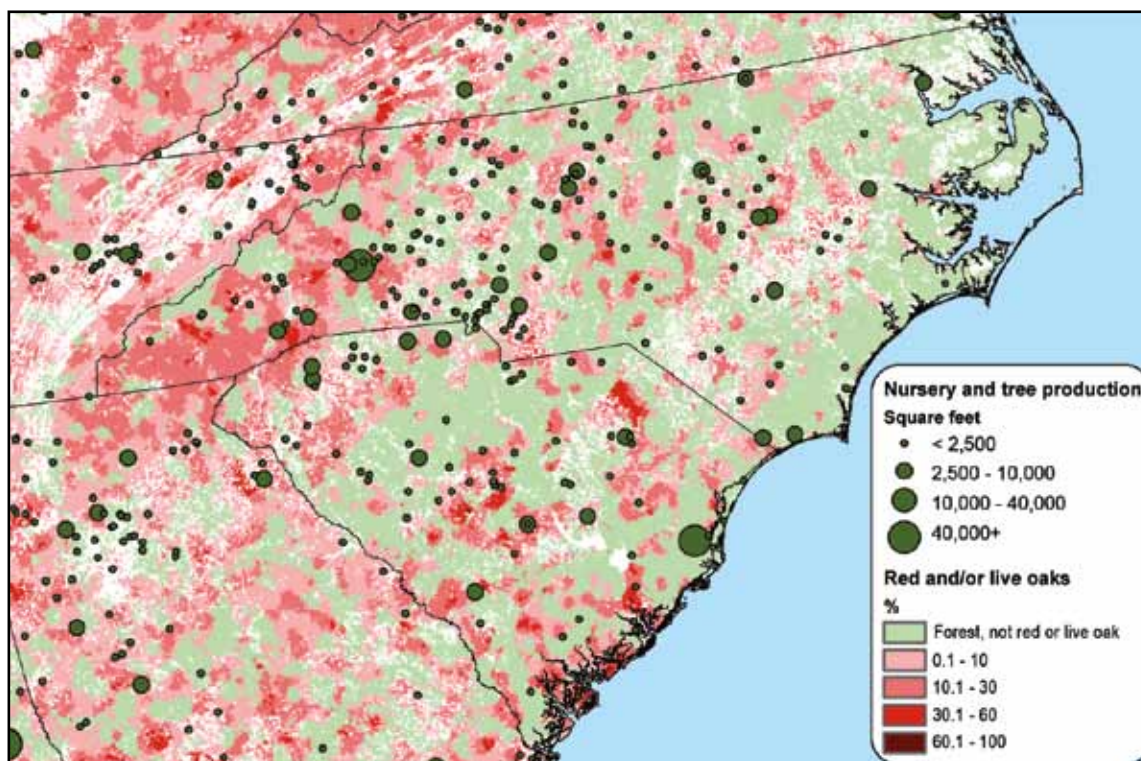


Figure 5—Geographic relationship between production nurseries and *Phytophthora ramorum* host distribution in North Carolina and South Carolina. Map of production nurseries in North Carolina and South Carolina ranked by size class (in square footage). Backdrop map shows overstory red oak or live oak distribution or both.

South Carolina. Larger production nurseries seem to be associated with areas of moderate oak distribution. Whereas this may just be a geographic anomaly, it indicates the kinds of analyses that are possible using specific business location data from ReferenceUSA or similar resource. The production nurseries data generated from ReferenceUSA will be incorporated in a refined version of a preliminary *P. ramorum* risk map developed by the U.S. Department of Agriculture Forest Service (USDA-FS 2004).

Wildland-Urban Interface Data and Areas at Risk for *P. ramorum* Introduction

We used WUI coverages developed by the SILVIS Lab to examine spatial patterns of potential residential-to-forest landscape spread risk for *P. ramorum*. We focused our analysis on an 11-State region encompassing the areas in the Eastern United States that were identified as high risk for *P. ramorum* introduction or establishment or both in the Forest Service's preliminary risk map (USDA-FS 2004).

We combined the individual State coverages into a single map and converted this map to a grid format (1-km² cells). We then reclassified the 13 original WUI coverage classes into 5 classes: little or no intermix, low intermix, moderate intermix, high intermix, and very high intermix. (Areas labeled as “interface” in the original WUI coverages were classified as somewhat lower risk than intermix areas—see “Forest-Developed Interface Data”). We examined changes in the intermix pattern between 1990 and 2000. Our reclassified maps (Figure 6) depict an expansion of low intermix into new areas at the rural fringes of metropolitan areas, as well as a subtle, but significant, increase in intermix in many suburban areas, particularly along the interstate highway corridor from Atlanta to northern Virginia. Whereas all areas with intermix face some *P. ramorum* risk, areas with greater than a one-class increase between 1990 and 2000 may be of most immediate concern; such areas appear to be widely scattered across the entire 11-State region. As with the production nurseries data, some form of the WUI

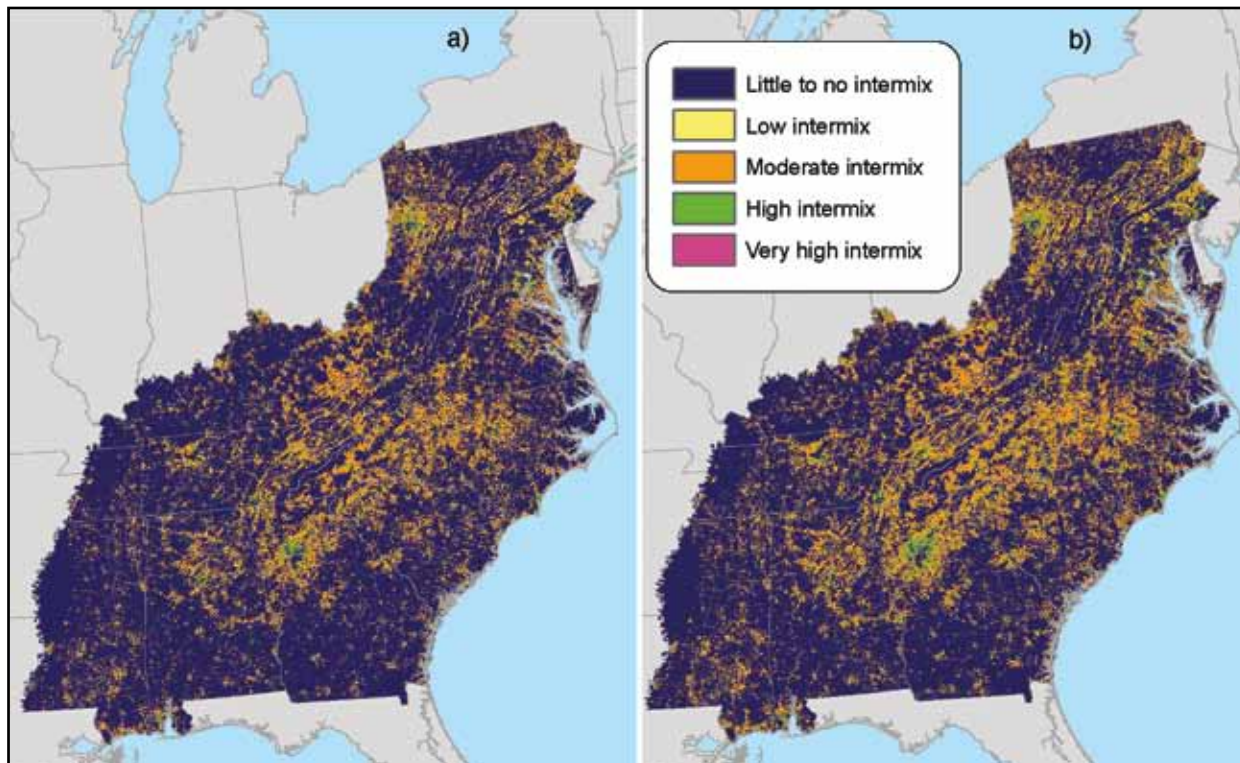


Figure 6—Wildland-urban intermix in an 11-State region, 1990 and 2000. Raster maps (1 km^2 resolution) for 1990 (Figure 6a) and 2000 (Figure 6b), depicting the distribution of five wildland-urban intermix levels (from little to very high intermix) across 11 States in the Southeastern and mid-Atlantic regions.

data will be utilized for the refined version of the Forest Service's national *P. ramorum* risk map (USDA-FS 2004).

Road Density Change and Increased *P. ramorum* Risk

A major concern for the refined *P. ramorum* risk map is identification of areas that may receive large quantities of nursery plant stock that are potentially infected by the pathogen. As noted previously, areas of new residential or commercial development or both are strong candidates as potential introduction sites for *P. ramorum*. As an indirect measure of areas recently or currently undergoing such development, we calculated road densities (total length of all road segments within a cell) for a nationwide grid of 4-km^2 cells using ESRI roads data for 1995, 2003, and 2004. We then calculated road density changes by subtracting the 1995 and 2003 road densities from the 2004 road density for each cell and mapped these changes in relationship to U.S. urban areas. Figure 7 shows a subset of the national

maps, focusing on the Research Triangle region of North Carolina. For this subset, the areas with the greatest road density change between 1995 and 2004 do correspond, at least anecdotally, with areas that underwent substantial urbanization during this time period. The 2003 through 2004 difference image also seems to accurately depict areas of recent construction, although the image may highlight major roadwork and not necessarily current residential or commercial expansion, both of which are likely to occur in these areas in the future.

Final Considerations

One of our goals with this synthesis was to provide examples that would stimulate discussion and the interest of other researchers in looking for ways to represent human-mediated pathways in forest pest risk mapping efforts. Although there is widespread awareness that humans contribute to the spread of nonnative pests, there has not been much attention paid to this issue in a spatial analysis

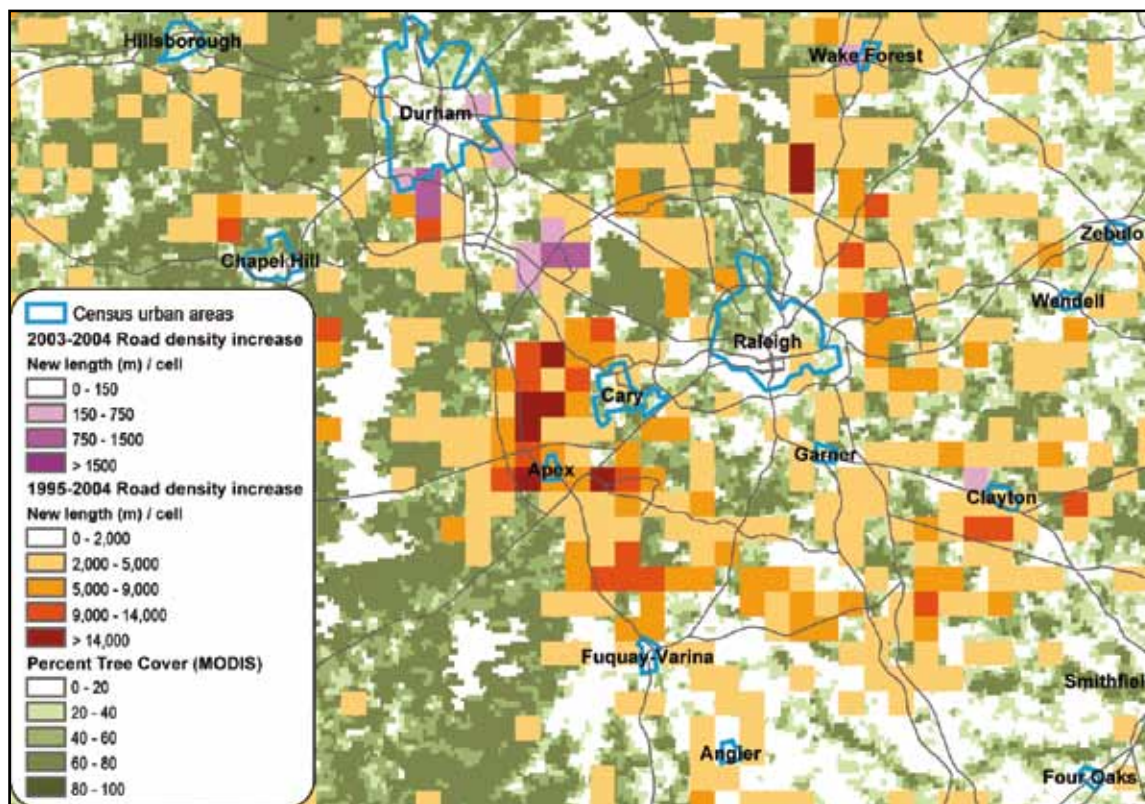


Figure 7—Use of road density change data to identify areas at risk for *Phytophthora ramorum* introduction. Changes in road density (total length of all road segments) for the Research Triangle region of North Carolina between 1995 and 2004 and between 2003 and 2004. Minimum resolution of individual cells is 4 km². Cells are color-coded according to the degree of change (cells with minimal change are omitted). This information is overlaid on a percent-age tree cover map developed from Moderate Resolution Imaging Spectroradiometer (MODIS) satellite imagery.

context. We hope this will change, but we would also like to note some considerations for researchers who decide to perform similar investigations.

Utility of Above-Described Data Sets

There are basically three overriding issues with respect to these data sets: attribute resolution, spatial scale, and imperfect representation of the pathway-related variable of interest. Attribute resolution refers to the level at which a variable of interest is aggregated. As noted, commodity classifications in the foreign and domestic cargo data are usually quite broad, so it is difficult to track many specific commodities of interest (e.g., nursery plants). Rather, users must do one of two things. One way is to work with the tonnage values of the broader commodity category and assume that the spatial flow pattern of this broader category mirrors the pattern of the specific commodity of interest,

so the relative ranking of different locations remains consistent. The alternative is to use ancillary data or some other means to estimate what percentage of the tonnage of the broader category belongs to the specific commodity of interest and how this percentage varies spatially across the area of analysis. The industry classification in the business location databases such as ReferenceUSA is, at times, similarly broad (e.g., we can identify nurseries, but not exactly which plants they grow or sell). This is a caveat that business location databases may be good for examining spatial patterns and trends, but should be used sparingly, if at all, for analyzing individual locations.

Related to the issue of attribute resolution is spatial scale, or the size of the geographic units at which the data are summarized. We have noted how the Commodity Flow Survey summarizes domestic commodity flow data at multiple scales, but the finest scale is at the level of

metropolitan area. The Freight Analysis Framework databases are similarly scaled at the metropolitan area level. This is not necessarily problematic, although it should be considered when choosing a working scale for a risk map that incorporates multiple pathway data layers as well as climate and host distribution information. In cases where the resolution of attribute information is tied to the output spatial scale (e.g., the Commodity Flow Survey database), the user should evaluate whether more specific attribute information is worth the tradeoff in geographic specificity. Naturally, this may depend on the pest of interest.

Finally, many of the data sets are imperfect, surrogate representations of a particular risk of interest. This is especially true of the land use/landcover data. As an example, change in road density may generally suggest where there are areas of increased residential or commercial development, and this, in turn, may suggest where nursery plants infected with a pathogen or infested by an insect are likely to be planted. These may be reasonable assumptions for portraying the variability of relative risk across a landscape. However, it is important to consider the potential uncertainty that comes with using indirect measures.

Availability and Accessibility

Many of the data sets described here are free or can be purchased for a small fee. Most may be downloaded directly through the Internet, although some must be ordered on disk (e.g., the Commodity Flow Survey and the National Transportation Atlas Databases, both of which will be shipped to the requesting individual at the agency's expense). Some of the statistical data sets are already in database format (e.g., the FAF databases, which are in Microsoft Access), although most are comma-delimited data that may be assembled into a database by the user. In terms of data accessibility, there are two notable exceptions. First, the business location databases typically require a subscription (see "Business Location Databases"). Also, the roads data used to assess contemporary land use/landcover change are only available to ESRI ArcGIS software users with a license for the optional StreetMap extension.

The timespan of the available data varies widely owing to both when the data are collected, and how often their

parent agencies analyze them and report the results. The current pattern suggests that many of the commodity flow data sets will be updated roughly every 5 years, although this likely depends on how much analysis time is required. For data sets that have previous versions, there have often been changes in how the data were processed or summarized between versions. We would advise potential users to carefully note these differences if combining multiple time steps of one of these data sets. More generally, it is important to consider differences in temporal scale, as well as spatial scale, if working with multiple data sources.

If initiating an analysis similar to what has been described in this synthesis, a few Internet sites offer a good place to start when searching for additional data sets:

- U.S. Army Corps of Engineers, Navigation Data Center: <http://www.iwr.usace.army.mil/ndc/index.htm>
- U.S. DOT Bureau of Transportation Statistics, TranStats Data Warehouse: <http://www.trans-tats.bts.gov/>
- University of Wisconsin-Madison, SILVIS Laboratory: <http://www.silvis.forest.wisc.edu/>
- U.S. Census Bureau, Business and Industry page: <http://www.census.gov/econ/www/index.html>
- U.S. Census Bureau, Geography page: <http://www.census.gov/geo/www/index.html>

Areas of Future Research

We will continue to search for additional data sets to represent human-mediated pathways in forest pest risk maps, but such a large data mining task would clearly benefit from the contributions of additional researchers and analysts. Some areas of significant data need include urban forest spatial distribution and condition, as well as recreational or informal pathways (e.g., movement of firewood) by which humans may spread forest pests. In addition, some of the data sets we have identified in our pest-focused research may be useful for examining other forest threats. For instance, we have only discussed insect and pathogen forest pests, but many of the same pathways are likely applicable

for invasive plants. There are strong possibilities for researching the cross-applicability of these data sets.

A major challenge for forest pest risk mapping is the integration of multiple data sources with different attribute, spatial, and temporal scales in a single model for a given pest. This includes the combination of disparate data on host species, environmental factors, and pathways of spread. Basically, forest pest risk mapping has many of the same problems as other analyses that involve ranking or classification with large, complex data sets. First, it can be difficult to select the most significant variables out of a large suite of model inputs. Second, even if the most important variables can be selected, it is also difficult to define appropriate threshold values for ranking risk. It is our experience that these thresholds are often determined ad hoc or through expert opinion. Although this can be an adequate approach, additional research on quantitative techniques for building robust, parsimonious risk models would be fruitful (e.g., Downing and others, this volume). Moreover, there may be opportunities to combine quantitative risk assessment with quantitative analysis of different management options, revealing how they might affect the distribution of risk (Woodbury and Weinstein, this volume).

Spatial and thematic accuracy, the propagation of error, and the characterization of uncertainty in spatial data sets—as well as how these characteristics may influence the forest pest risk mapping process—are often downplayed in the need to quickly derive a useful risk map product for forest planners. The issues have really never been considered for the human-mediated pathways data described in this synthesis, which have only recently been adapted to a risk mapping context. Research opportunities abound in terms of developing quantitative techniques to address some of these issues in the forest pest risk-assessment process.

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Decision Making Under Risk in Invasive Species Management: Risk Management Theory and Applications

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Abstract

Invasive species management is closely entwined with the assessment and management of risk that arises from the inherently random nature of the invasion process. The theory and application of risk management for invasive species with an economic perspective is reviewed in this synthesis. Invasive species management can be delineated into three general categories: exclusion, detection, and control. Key ideas and approaches in current literature and potential applications of existing theory are presented in this synthesis. Economic literature tends to emphasize either individual management strategies, such as preventing invasive species from entering an ecosystem or controlling extant populations. There is also a growing focus on the optimal allocation between multiple activities for the same species such as between prevention and control. The key biological and economic relationships included vary across frameworks and objectives.

The synthesis is organized into sections, which cover the salient aspects of invasive species management by separating the major veins of economic literature on decisionmaking. Section 1: invasive species management is discussed and a brief overview of risk management and economic theory is provided. Section 2: an overview of the key factors causing invasions is presented. Section 3: exclusion activities to prevent introductions is the focus. Section 4: control activities after the species has been successfully introduced are emphasized. Section 5: the tradeoffs between multiple management strategies are addressed. Section 6: a discussion concludes the synthesis.

Keywords: Decisionmaking, economics, invasive species, non-native, risk management.

Introduction: the Role of Risk Management in Invasive Species Management

An overview of decisionmaking under risk in invasive species management, with an emphasis on the economic literature, is provided in this synthesis, and the following topics are discussed: an overview of invasive species management; the key factors causing invasions; the exclusion activities aimed at preventing introductions; postintroduction control activities; and tradeoffs between multiple management strategies. Scientists have studied the impacts of invasive species on nonnative ecosystems for many years. Recently, the public has become more aware of the problems associated with invasive species, due largely to greater levels of media coverage and public information campaigns regarding the issue. Government agencies deal explicitly with risk in their ongoing invasive species management programs; thus, risk management plays a crucial role in these programs.

The synthesis is organized into sections, which cover the salient aspects of invasive species management by separating the major veins of economic literature on decisionmaking. In section 1, invasive species management is discussed, and a brief overview of risk management and economic theory is provided. In section 2, an overview of the key factors causing invasions is offered. The focus is on exclusion activities to prevent introductions in section 3. Control activities after the species has been successfully introduced are presented in section 4. The trade-offs between multiple management strategies are addressed in section 5. The synthesis concludes with a discussion in section 6.

Owing to the breadth of this risk management, and discrepancies among discipline-specific terms, the sections below clarify the usage of terminology in this synthesis.

Defining Risk

The word risk has three common definitions: (1) an [adverse] event (as in “non-native invasive species are a

risk to ecosystems”); (2) the probability that the event will occur (as in “the goal of management is to reduce the risk of invasive species introductions to new ecosystems”); and (3) the probability that an event will occur weighted by the consequences of the event (as in “the possibility that invasive species will harm oak forests is a substantial risk”). In economics, the term is most often used to describe situations in which the results of a decision follow some sort of probability distribution. The probability distribution may be objectively determined, either through a priori reasoning (such as the probability that a fair die will show the number 6) or through repeated experiments. Probabilities may also be subjectively determined without clear experimental evidence. These are known as beliefs. The term ambiguity applies when probabilities are not known with certainty. When probabilities are involved, it is possible to define (and maximize) objective functions that are weighted by these probabilities. We adopt the economics usage of the term risk, referring to a situation in which management decisions affect outcomes and their probabilities.

There may be cases when it is not possible to assign probabilities to outcomes. One example is the case of global climate change as discussed by Woodward and Bishop (1997). Although it is possible to assemble a panel of experts to glean their beliefs about possible dangers, and it is possible to delineate the range of possible options, Woodward and Bishop argue that it is not reasonable to assign probabilities to these options based on the number of experts sharing a particular belief. Woodward and Bishop call this pure uncertainty, following the distinction as defined by Frank Knight in 1921. Others term this ignorance (Arrow 1972 and Hurwicz). In these cases, it is not possible to define a function that is weighted by probabilities.

However, the term uncertainty has a number of different definitions, including simply: uncertainty arises whenever a decision can lead to more than one possible consequence (Hammond 1998). This definition includes situations such as lotteries where probabilities are well-established. In the risk assessment literature, uncertainty arises due to the lack of precise knowledge about parameters, models, or scenarios. It can also come from differences among modelers (Linkov and Burmistrov 2003).

We adopt the usage in the risk analysis literature, where uncertainty refers to this lack of precision, and use the term pure uncertainty to refer to the case where it is not possible to assign probabilities.

Risk Assessment and Risk Management

Risk analysis consists of risk assessment and risk management activities. Risk assessment is defined as “the systematic, scientific characterization of potential adverse effects of human or ecological exposures to hazardous agents or activities” (The Presidential Congressional Commission on Risk Assessment and Risk Management 1997). Risk assessment informs risk management decisions by supplying this characterization of potential outcomes and probabilities. In invasive species management, risk assessment informs two specific areas: the risk surrounding introductions of new species, including vectors, species, and potential damages; and the risk associated with existing invasives, including the potential spread and damages caused by established species (Andersen and others 2004). For example, the date of the introduction of a pest, like the Siberian moth (*Dendrolimus superans sibiricus* Tschetverikov) may not be known. However, numerous factors can be used to construct probabilities to characterize the chances of the Siberian moth being introduced at any particular date. These factors include the number of pathways that it has, the level of interaction between the United States and its native territory, and the introduction success of similar species. The frameworks for assessing invasive species risk vary substantially (Woodbury and Weinstein, this volume).

Risk management, as the Presidential Commission (1997) states, is “the process of identifying, evaluating, selecting, and implementing actions to reduce risk to human health and to ecosystems.” Risk management relies heavily on underlying risk assessments to establish the potential for adverse events occurring as a consequence of a particular action. In addition, risk management must account for resource, social, ethical, political, and legal constraints. In this synthesis, with our focus on the economic literature, we pay particular attention to how resource constraints guide risk management decisions.

A (Brief) Primer in Economic Theory

Some common economic concepts and terms used throughout the review are briefly covered in this section. Welfare economics focuses on the implications of alternative resource allocation methods on social welfare, both in market and nonmarket settings. One criterion used to judge whether an outcome is efficient is Pareto optimality; a Pareto optimal allocation is one in which no one can be made better off without making someone else worse off. A competitive equilibrium will be Pareto optimal unless there are market failures. One example of a market failure is an externality, where a consumer or producer generates costs that they do not bear themselves. Invasive species represent an externality in that accidental introductions of invasives impose a cost to society and can occur as a consequence of consumption or production activities. Therefore, one role of management is to align individual incentives with social goals—to internalize external costs—so that a Pareto-optimal allocation will result. One possible tool that can be used to align incentives is a tax on the externality, calculated as the difference between private cost and social cost at the optimum. This is known as a Pigovian tax. Another tool is a system of tradable permits in which an agency sets the total allowable level of the externality, but the allocation of that level emerges out of a permit market. Both of these have been suggested as ways to handle the invasive species that are introduced as a byproduct of economic activity.

When situations are risky, it may not be possible to guarantee a particular outcome, but it may be possible to choose among alternative probability distributions by choosing the level of conditioning variables. Economists have developed theories of rational choice that are appropriate in risky situations, including the Expected Utility theory of von Neumann and Morgenstern (1944) for objective probabilities and the Subjective Expected Utility Theory of Savage (1954) for subjective probabilities. For example, it may be possible to reduce the probability of species introductions by altering trade levels. Expected utility theory is one framework that can guide these choices. The objective function consists of the expected level of utility, where utility represents an individual's satisfaction derived from their preferences for consuming or experiencing goods

and services. Utility functions, or numerical representations of individual preferences, can account for costs and benefits that accrue under unknown future events. The vonNeumann-Morgenstern utility function offers one characterization of an individual's preferences over all potential outcomes. Expected utility theory essentially states that such a characterization exists if an individual's preferences conform to certain axioms. Such representations allow for identification of optimal behavior that maximizes expected utility. They also provide a basis to compare different risk preferences, or attitudes toward risky situations. For example, an individual may be risk-averse (i.e., he or she prefers a situation with little or no risk to a more risky situation even if the expected outcome in the risky situation is higher).

When probabilities cannot be assigned to possible outcomes—situations of Knightian pure uncertainty—it is possible to use other criteria, such as maxi-min (choose the course of action that leads to the best case when the worst state of nature occurs) or mini max-regret (choose the course of action that leads to the lowest possible regret, Loomes and Sugden 1982). Ciriacy-Wantrup (1964) promoted the Safe Minimum Standard criterion, which would suggest that irreversible losses should be avoided. This is echoed in the Precautionary Principle of Perrings (1991). These criteria can be justified as rational in situations where possible future outcomes are knowable, but the probabilities of those outcomes are not.

The theme of optimization over time is prominent in the area of natural resource economics, and the balance of the productivity of natural assets with that of other assets typically characterizes an optimal solution. In the context of renewable resources, the optimal harvest rate is one that equates the returns from the stock of the resource to the returns one could achieve in an alternative investment. Extinction can be an optimal outcome, particularly if there are no nonmarket benefits associated with the resource. This theme is echoed in the literature on invasive species, where both corner solutions (eradication) and interior solutions (control at some positive population level), are possible. The probability of random catastrophic events has been shown to increase the appropriate discount rate (risk

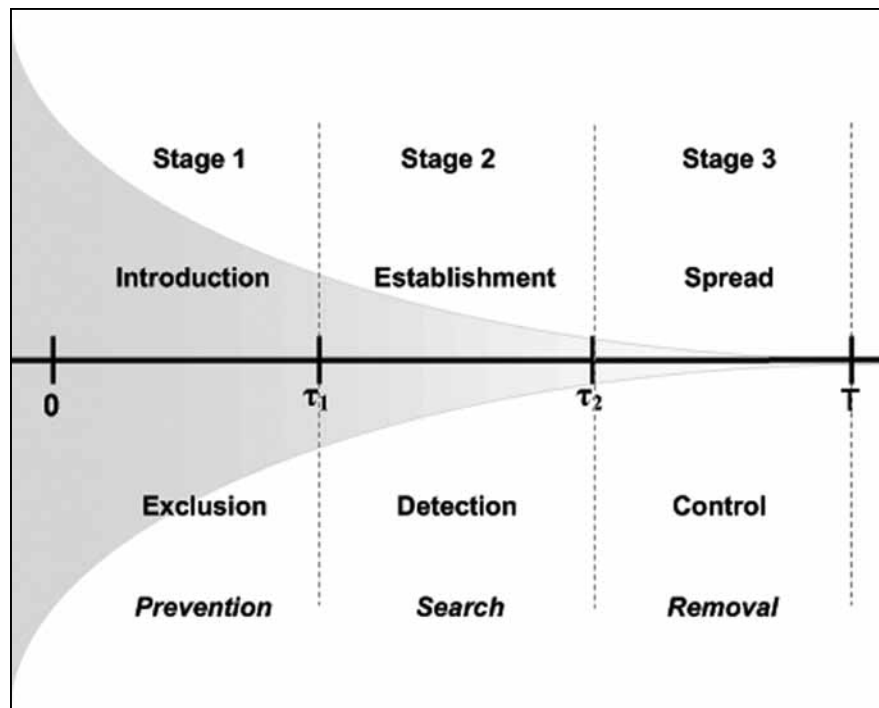


Figure 1—Management activities and corresponding invasion process stages. This diagram depicts the major management activities—exclusion (prevention), detection (search), or control (removal), which correspond to the stages of the invasion process—introduction, establishment, and spread—as used in this synthesis. Whereas many of these activities occur concurrently, the focus in the synthesis is largely on the individual management activities (sections 3 and 4) with a lesser emphasis on the relationships between multiple activities (section 5). To illustrate the simplest example, the finite-horizon timeline represents a situation where an agency is managing a single species and activities are undertaken in isolation. Thus, the manager first engages in preventing introductions and, if that fails, switches to detection (at time period τ_1) and then potentially to controlling the species if it establishes and spreads (at time period τ_2)

adjusted discounting) and accelerate harvest (Reed 1984). It is also possible to factor in the stochasticity associated with population growth (Pindyck 1984), and this can either introduce caution or intensify harvest pressure. An alternative approach is to specify some probability of extinction and then set management levers so that this probability is not exceeded; for example, one possible criterion is that the probability of extinction should never be higher than 1 percent (Haight 1995, Montgomery and others 1994). Analogous criteria can be used in invasive species management.

Invasive Species Management

Invasive species management spans a variety of activities that can be grouped into three areas: exclusion, detection,

and control (Figure 1). The management activities concentrate on different parts of the invasion process, which comprises three main stages: introduction, establishment, and spread. Even though agencies engage in additional activities, these categories capture the majority of decisions facing managers. Thus, the synthesis is arranged according to the general management categories. Although the theory behind the risk management models is discussed, the emphasis is on the potential outcomes.

Various species may be in different stages of the invasion process so that management agencies engage in exclusion, detection, and control activities simultaneously. Additionally, the dynamic nature of the invasion process implies that it is optimal to make management decisions in a forward-looking manner by accounting for future stages

in current actions. Existing literature often analyzes such aspects of the relationships between the management activities. Many papers in the economic literature, for example, consider the optimal allocation between exclusion and management activities for the same species.

Factors Fueling the Invasion Process

Key factors are focused on in this section that are thought to contribute to the risk of invasions, including factors that can be controlled for the purposes of risk management. One commonly held view is the disturbance hypothesis (Dalmazzone 2000), which asserts human activities and their accompanying disturbances to the environment primarily cause both species' introductions and their subsequent invasions of new ecosystems. In addition, human and commodity movement provide the major vectors for species to enter new ecosystems. Risk management requires the knowledge of which vectors pose the greatest chance for new introductions of invasives, and this translates to a reliance on comprehensive risk assessment (e.g., cargo data in "Factors Fueling the Invasion Process;" Koch and Smith, this volume). Current risk assessments typically do not use an economic framework and economic data to understand the relationship between trade and invasion risk. Costello and others (2007) provided one of the few examples of such a framework when they parameterize a model based on invasive species introductions data to find that the threat of new invasions depends on the past trade level with a region and the past exposure to invasive species. Using invasion data from the San Francisco Bay over a period of 138 years, they identify trade partners from the Atlantic/Mediterranean and West Pacific regions as posing the greatest risk of introductions to the San Francisco Bay Area. Explicitly incorporating economic aspects can potentially alter the results of a risk assessment and, in turn, change suggested approaches to risk management. In this case, risk management can potentially reduce risks through targeted trade restrictions and economic mechanisms. However, such restrictions can produce unintended negative consequences if inappropriately targeted or implemented.

Economic activities can produce other externalities such as land disturbance and the loss of biodiversity.

Scientists have long argued that biodiversity loss increases an ecosystem's susceptibility to invasion. This idea stems from a theory postulated by Elton (1958) that diverse habitats can better fend off invasions. One major reason for biodiversity loss to affect invasions arises from the interactions between native and non-native species. Many experts believe that interactions between native and exotic species are generally detrimental, often stemming from competition over limited space or food (e.g., Shigesada and Kawasaki 1997; Tilman 1982, 2004). However, studies have shown that the relationship between native and non-native species is quite complex, and some species may actually benefit due to mutualism (Bruno and others 2003). Hence, a simple classification based on the number of species in a geographic area can not necessarily predict the susceptibility to new invasions; instead, the effects of the interaction often depend on the spatial scale (Fridley and others 2004, Meiners and others 2004).

Apart from the externalities from economic activities, each species' unique biological traits largely affect their impacts in a new ecosystem. Thus, risk assessment emphasizes biological traits of potential invasive species as indicators of potential risk (Downing and others, this volume; Iverson and others, this volume; Pontius and others, this volume). However, given the numerous possible factors and characteristics that contribute to a species' invasiveness, the selection of appropriate characteristics poses much difficulty. For example, Rejmanek (1999) posited that all salient biological characteristics provide some information on the potential risk posed by a species. Besides the species specific biological traits, landscape characteristics of the potential habitat also provide important indicators (Page-Dumroese and others, this volume; Royo and Carson, this volume; Shore and others, this volume).

Other researchers, such as Williamson (1996), believe that very few characteristics provided suitable predictors for invasions. Williamson (1996) argued that most invasion patterns elude generalization over wide taxonomic ranges due to the specificity of success factors. Still, propagule pressure, habitat suitability, and prior invasion success serve as rough indicators of invasion success. Of these, propagule pressure, or the number of organisms in an area,

is the variable that can be most directly altered through risk management. As the propagule pressure increases, the chances of survival increase, whereas the effects of predators, stochasticity, and the Allee effect (“Eradication as an Optimal Strategy”) diminish. Unsurprisingly, a positive correlation exists between propagule pressure and disturbed land. Disturbed land tends to have greater economic activity, which translates to increased exposure to vectors for exotic species.

In addition to establishing general predictive factors, Williamson (1996) formulated the Tens’ Rule, which postulates that of the exotic species introduced to an area, 10 percent will become established and 10 percent of those will spread (approximately 1 percent of introduced species). Although this is a very general rule-of-thumb, assigning probabilities with educated guesses for the underlying probability distributions can produce an approximation of the actual invasion process (e.g., Lockwood and others 2001). Using such approximations, a decisionmaker can perform a more structured analysis, which potentially widens the possible management options available (Perrings 2005, “Understanding Risk Mitigation Versus Adaptation”). However, whereas general rules, such as the Tens’ Rule, serve as useful benchmarks to provide a sense of the magnitude of potential invasions at the aggregate level, they do not predict outcomes at a micro-level scale. The pine shoot beetle (*Tomicus piniperda*) is an example of how aggregate level predictions can fail at the regional level. Pine-shoot beetle sightings were met with strict quarantines due to its classification as a high-risk pest causing potentially high economic damages. The pine-shoot beetle actually produced relatively low damages, but the quarantine measures resulted in significant losses to the pine industry in the affected areas (Haack and Poland 2001). As the authors state, “... it is difficult for agencies like APHIS to change course once they have enacted a federal quarantine given that the concerns of the uninfested states have been heightened by the initial establishment of the quarantine.” Thus, whereas risk assessment can inform risk management, it could potentially exclude key impacts such as in the economic consequences of quarantines on industry or the irreversibility of certain actions.

Risk management frameworks incorporate these relationships into the decisionmaking problem facing the agency manager along with the spatial, temporal and stochastic dimensions. However, there is no clear rule dictating which relationships must be included or how they should be included. Inclusion of specific relationships depends largely on the agency manager’s objective, and the choice of relationships will impact the model’s outcome.

Exclusion Strategies and Risk Management

Exclusion strategies occupy much of the limelight in risk management for invasive species because the majority of species introductions have been attributed to human activities (“Factors Fueling the Invasion Process”). Reducing the risk of species introduction involves managing the potential pathways that species use to enter a new ecosystem. Market-based mechanisms, such as tariffs or permits, can regulate trade behavior and produce socially optimal outcomes (“Policy and Market-Based Mechanisms to Manage the Risk of Introductions”). The optimal strategy necessary to prevent a species introduction may be substantially affected by whether or not you know the probability distribution of the invasion process.

Policy and Market-Based Mechanisms to Manage the Risk of Introductions

The varying policies and market-based mechanisms aimed at reducing introductions produce substantially different outcomes and can lead to unintended effects such as economic losses as illustrated by the papers in this section. Increased trade volume creates greater opportunities for species to engage in ecosystem hopping, leading some to argue for tighter trade restrictions to reduce this risk (Jenkins 1999). However, the sheer volume of trade coupled with strong political resistance and social welfare losses requires a targeted approach. The first step is identifying the high risks within trade (“Factors Fueling the Invasion Process”). The next step involves evaluating mechanisms, which reduce this risk. Market-based mechanisms can ensure socially optimal outcomes, but their implementation is often confounded by political issues and the inability to

properly capture the full implications of these mechanisms. Optimal tariff policy is discussed in a majority of these papers, but the role of politics should not be overlooked. Several policies, such as phytosanitary measures (<https://www.ippc.int/IPPC/En/default.jsp>), aim to reduce the potential damages of pests that arrive through trade. However, it can be argued that extant tariffs and trade policies often stem more from political motives than the conscious desire to mitigate environmental damages produced by trade. As Margolis and others (2005) illustrated, delineating protectionist tariffs from those designed to mitigate invasion risk requires knowledge regarding the social costs inflicted by invasive species and the value the government places on societal welfare. Without this information, it is hard to gauge the consequences of tariffs. Achieving socially optimal outcomes involves two steps: (1) identifying the mechanisms that induce behavior to produce the socially optimal outcome, as demonstrated in the papers in the following sections, and (2) implementing these mechanisms. The first step pertains to this synthesis; however, the obstacles that arise during implementation must also be kept in mind while evaluating these mechanisms.

Trade Policy—

Models that utilize taxes to manage invasive species introductions are discussed in this section. Although the economic focus on invasive species is relatively new, frameworks from the environmental economics literature can inform decisionmaking models. For example, Costello and McAusland 2003, Knowler and Barbier 2005, and McAusland and Costello 2004 draw from the pollution literature by treating invasives as a negative externality comparable to pollution. Knowler and Barbier (2005) evaluated the extent to which market-based mechanisms, such as taxes, can produce a socially optimal level of exotic plant imports. Private industry and agriculture rely heavily on exotic species for a range of purposes (McNeely 1999). To address the unintended consequences of intentional introductions, Knowler and Barbier assessed the effects of Pigovian taxes for the private nursery industry. Pigovian taxes impose a penalty on the loss associated with an agent's actions; in the pollution literature, Pigovian taxes are levied against firms based on the amount that they pollute ("A [Brief] Primer

in Economic Theory"). Setting optimal levels of Pigovian taxes requires perfect information on firm practices, and, more importantly, the impacts of those actions. Assessing the contribution of individual firms on overall species' introductions is difficult. However, from a social welfare perspective, optimal Pigovian taxes provide a better alternative than total prohibition of exotic imports. As shown in the pollution literature, market-based mechanisms such as Pigovian taxes can internalize the externalities of private actions to result in a socially optimal outcome.

To assess the impacts of a Pigovian tax, Knowler and Barbier developed a model to analyze the decision facing a policymaker regulating a private industry. Specifically, they studied the commercial nursery industry, where importing, breeding and selling behaviors often occur without consideration of the potential loss to society from unintended spread following the sale of the exotic species. The social benefits of plant imports are represented by the discounted aggregate profits of the private nursery industry. The expected losses depend on the quantity of land invaded by the species once the invasion occurs. The overall net benefits are the total profits until the time when the invasion occurs minus the discounted losses following the invasion. A hazard function characterizes the probability that the species will arrive by a particular date. The hazard function in their model incorporates the salient factors driving invasions, such as the invasiveness of the species and the number of firms in the industry. Solving the dynamic optimization problem yields a time path of the optimal number of firms in the industry. The application of this model requires empirical analysis. However, the uncertainty in several relationships, such as potential damages, potential invasiveness, and the time of the invasions, necessitate assumptions based on educated guesses and a priori beliefs. To facilitate the model's implementation, Knowler and Barbier made several simplifying assumptions in their analysis of the saltcedar (*Tamarisk* spp.), an ornamental shrub, which became invasive. The hazard function is estimated from a survey of decisionmakers on their perceived risk of invasiveness. Based on USDA data for the horticultural industry, the industry's profit function is fitted at the state

level using a second-order polynomial equation. The analysis of four model specifications for different treatments of the relationship between the number of firms and the hazard function, coupled with varying specifications of the hazard level (low and high hazard) and four levels of profitability illustrate two key points: (1) the optimal number of firms is always lower than the optimal long-run equilibrium without invasion risk and (2) the optimal number of firms and level of Pigovian taxes is sensitive to the hazard level. Under some conditions, the optimal number of firms is zero (i.e., it is optimal to prohibit all imports of the exotic species).

Conventional wisdom states that tariffs, or any protectionism, will reduce invasion risk as a consequence of a reduction in trade; Costello and McAusland (2003) demonstrated that this may not be the case. Protectionist policies can achieve a reduction in overall invasive species introductions. However, the failure to account for the role of agricultural damages skews the interpretation of the true efficacy of protectionist policies, which may actually increase invasion risk. This result is an example of the aforementioned disturbance hypothesis (“Factors Fueling the Invasion Process”), and, in this case, human disturbance is often believed to increase the chance of invasion. Higher tariffs on agriculture will reduce agricultural imports, so domestic producers will increase production. This, in turn, increases land disturbance as more land is converted to agricultural production. This increase in land disturbances facilitates invasions by extant invasives as well as new ones, thus, any reductions in invasion risk from the tariffs are offset by increased risk from land disturbance. A complex theoretical framework presented in their paper incorporates the potential damages from different trade levels and commodities, the supply and demand elasticities for various commodities, and the level of protectionism on these commodities. Though their model is theoretical and difficult to parameterize, the analysis illustrates that setting policy based on conventional beliefs may lead to suboptimal results. As Costello and McAusland stated, “...the rate of introduction causing crop damages provides minimal (if not outright misleading) information about the rate of ecologically damaging invasions. This has important implications for the use of existing estimates of invasion-related damage;

while existing estimates are staggering, they omit invasion-related costs to biodiversity and other non-monetized assets.”

Whereas Costello and McAusland’s model does not incorporate averting behavior by farmers or the role of invasive species management activities, several salient observations are provided in their paper such as the usefulness of current introduction estimates and the impact of using incorrect empirical models. Also illustrated is the value of expanding the breadth of the analysis to better inform decisionmaking by including both direct and indirect consequences of incentive mechanisms, in addition to the underlying stochastic relationships. The crucial component is acknowledging and incorporating the economic aspects, such as price distortions and demand and supply responses to import changes. An often overlooked aspect of invasive species policies is addressed—the potential behavioral responses by the import-competing industries who respond to the supply changes resulting from the tariffs impact on the importing firms. Tariffs may produce positive effects for some firms while concurrently altering production choices by other firms, which can lead to other changes such as an increase in domestic agriculture intensity. The need for analysis to encompass the full extent of changes resulting from a policy is illustrated in this paper.

In a subsequent paper, McAusland and Costello (2004) analyze the combination of tariffs and monitoring practices to achieve the socially optimal level of prevention activities. The assumption that all components are known is shown in their model. Whereas tariffs and cargo inspections reduce the introductions of invasive species, the omission of explicit stochastic elements excludes this model from frameworks that can be implemented directly for risk management. The results of the analysis, though, are worth mentioning as they can provide insight for risk management practices:

1. It is always optimal to have a positive tariff although it may be optimal not to have inspections in some situations (i.e., if the level of infection of the partner is so high that it is optimal to not inspect but instead to set a high tariff).

2. Higher infection rates necessitate higher tariffs but not necessarily greater inspections.
3. Extending the time horizon results in greater inspections but not necessarily higher tariffs.

They draw an analogy between this situation and the one involving pollution emissions, which requires monitoring to determine the pollution levels to levy the correct taxes. Here, the monitoring entails inspections that sort the uninfected goods from the infected ones. The findings from the inspections determine the amount of monitoring needed in the future and the level of taxes that should be set. At sufficiently high levels of infection, the optimal strategy is to discontinue monitoring and rely solely on taxes to repay the costs of invasives.

Permit Trading Models—

Horan and Lupi (2005) explore a tradable permit program as an alternative to current regulation of ballast water to reduce the number of invasive species entering the Great Lakes. Permits allow commercial vessels to release ballast water, which carries species. However, releases are unobservable; thus, they must be estimated as a function of vessel characteristics and management practices. The authors find that although permit trading produces the most efficient outcomes, appropriately targeted technology regulations can lead to similar results. Emission permits are considered more efficient than regulation because they provide a performance-based mechanism, which is why permits are preferred for pollution control. However, unlike pollution emissions, the existence of invasive species in ballast water is unknown beforehand, and, even after the species have been released, the species introductions are not observed due to a lack of appropriate monitoring technology. Also, a potentially lengthy lag between introductions and spread (Crooks and Soulé 1999) further obfuscates the ability to pinpoint the individual polluter. This means that a specific vessel cannot be connected to a specific species introduction, because the outcome of a vessel's actions is not directly observable. To overcome this lack of information, tradable permits can act as a proxy for the potential risk posed by each vessel via a performance measure that incorporates vessel-specific characteristics and firm actions aimed at reducing introduction risk.

Horan and Lupi's permit trading model (2005) relates to a previous one introduced in Horan and others (2002) regarding prevention strategies in risky and uncertain scenarios ("Pure Uncertainty Versus Risk in Assessing Prevention Strategies," below). In this recent model, each firm introduces a range of species, measured by their biomass, through their vessels. The potential biosecurity actions that a firm can employ to reduce the risk of transporting species are considered inputs. There are two stochastic relationships: (1) the size of the biomass, which depends upon the biosecurity inputs and the firm's characteristics, and (2) the post-introduction probability of establishment and spread, which depends on the given control strategy, the biomass, and the biosecurity inputs. Combining these two separate stochastic relationships characterizes the probability of a firm introducing an invasive species. The authors characterize the total probability of an individual species introduction as the sum of the separate probabilities of introduction over all firms. This total invasion probability drives the expected damages resulting from a species invasion. The policymaker's objective is to minimize the social costs, which are represented by the costs of biosecurity inputs for all firms plus the expected damages resulting from successful invasions. Focusing on ballast water released by vessels in the Great Lakes, Horan and Lupi illustrate the model using data and estimates for probability of introduction and successful invasion, firm characteristics, biosecurity inputs, and costs.

Optimally, the marginal cost of an action (choice of biosecurity input) for each firm equals the expected marginal benefits of that action, or the decrease in expected damages. Permit trading requires much information including vessel- and firm-specific characteristics and actions, all potential invaders and expected damages, and probabilities for introductions and successful invasions as they relate to new habitats and firm behavior. Creating permits based on the specific characteristics of each vessel and firm and for each specific species would be the first-best option. However, with this scheme, the multiple permits for each species and each vessel would result in a cumbersome system. A second-best permit trading scheme, related to the first-best option, produces a desirable outcome but with only

one permit for all species, instead of several different ones targeting different species. Whereas less efficient than the first-best approach, a single permit reduces the information requirements because the policymaker does not require detailed information for each specific firm and vessel. This approach finds that the first-best scheme provides the least costly option followed by the second-best trading scheme when simulating three risk scenarios for different mechanisms: (1) the first-best trading scheme with varying permits, (2) the second-best scheme with one permit, and (3) various technology regulations. This result holds for relatively moderate permitted levels of invasive species introductions; the difference between the regulation mechanisms fades with stricter permitted levels. Interestingly, their outcomes suggest that technology regulations can inexpensively mitigate risks if suitable technology is chosen and appropriately regulated. Although direct implementation requires several assumptions, this model offers a useful tool to analyze potential regulatory policies while accounting for the major stochastic relationships. Also, it takes the Knowler and Barbier (2005) analysis ("Trade Policy") one step further by demonstrating the potential loss incurred by simplifying assumptions to address information gaps.

By characterizing the unknown elements of any situation, managers can employ explicit frameworks to evaluate the potential outcomes of various policy mechanisms as illustrated in these papers. The authors were forced to make several simplifying assumptions to deal with a lack of data or to address the stochastic elements, but they still provide valuable analysis. They also illustrate the importance of evaluating policies in an explicit economic framework to capture the full extent of the repercussions such as the social welfare loss from posing industry-wide taxes or implementing tariffs without accounting for the changes in industrial behavior.

Pure Uncertainty Versus Risk in Assessing Prevention Strategies

Whereas it is assumed in most papers that decisionmakers have some information that can help characterize risk in invasive species management, there may be cases (Knightian pure uncertainty) where it is misleading to

assign probabilities, and information is lost when the true lack of knowledge is overlooked. Horan and others (2002) tackle this issue using an aggregate model to capture preinvasion decisions by firms whose actions can introduce invasive species. Horan and others argue that invasions cannot be analyzed using standard economic theory, which assigns probabilities to all situations regardless of the level of uncertainty. Traditional risk-management models function similarly by characterizing all risk, irrespective of the level of uncertainty, with probability distributions. The authors argue that standard expected utility theory (or traditional risk-management) does not apply to low probability events, especially when the events are catastrophic, as they could be in the case of invasive species. Non-native species invasions can be considered catastrophic since irreversibility of invasions poses potentially very high costs and irrevocable damage to native ecosystems. To illustrate the effects of incorporating differing levels of uncertainty into the decisionmaking framework, the authors identify optimal prevention strategies by firms under the traditional risk-management framework (with assumed probabilities) versus an ignorance model (full uncertainty, which is not characterized by probabilities), which appropriately reflects the circumstances before the invasion occurs.

In the traditional risk management model, the probability of introducing a species depends on the firm's characteristics and the control strategies chosen by each individual firm. A species' successful invasion depends on the biomass of the introduced species, the characteristics of the environment, and the firm's characteristics. From the perspective of a policymaker regulating firms in an industry, the concept presented in the paper by Horan and others (2002) creates a framework where the stochastic elements are the species introductions and the success of the invasion. The framework is fairly theoretical and the information necessary to implement this model directly may not be available. General aggregate-level decisions are focused on in their paper. The probability of an invasion follows a Bernoulli distribution that depends on the actions of all firms in the industry. As the number of firms increases, the probability of an invasion approaches one. The present value of damages facing society depends on expected damages, expected costs, and

the possible set of all species that may be introduced. The risk management problem is static meaning the state-of-the-world remains the same for the single planning period. The firm minimizes expected damages caused by the species plus the control, or abatement, costs that lessen the probability of a species introduction. The major distinction between the traditional framework and the ignorance model is the potential set of invading species; in the traditional framework, all species that can invade are known, whereas under the ignorance model, the set of potential species contains a subset of species that is completely unknown. This approach gives rise to the idea that events are associated with different levels of potential surprise.

According to the traditional risk-management framework (i.e., the expected value approach), the policymaker has two potential optimal strategies: (1) all firms should be unregulated or (2) all firms should undertake expensive measures so that the probability of an invasion is driven down to zero. Also, with a large number of firms, abatement is not optimal because the chances of invasion are high regardless of the control strategies pursued by individual firms. In both frameworks, the optimal strategy is to set marginal costs equal to expected marginal benefits, or the negative of damages. Under ignorance, though, firms will evaluate the marginal impacts of the events and subsequent potential outcomes quite differently. The difference in valuations of the marginal costs and damages leads to different outcomes for the two approaches. In the expected value scenario, low abatement is an optimum strategy for all firms, whereas that is not the case for full uncertainty because the firms' values are significantly different. Subsequently, policies using the uncertainty framework establish uniform performance limits for all firms as opposed to the risk management framework where limits vary for each firm. Straightforward application of this framework is unlikely; however, the theoretical model, which illustrates the importance of considering alternative decision frameworks when elements of the model are unknown is the greater contribution of this paper. Due to the importance of uncertainty in the invasion process, continued reliance on the traditional approach for characterizing risk could lead to a severely restricted view of the true situation facing us.

This does not mean that the expected value approach is not valuable, but it is crucial to be aware of other characterizations and unspoken caveats of these models.

Control Strategies

After the species successfully establishes, the decision-maker may employ several control strategies: eradicate the population, slow the spread of the population through spatial control strategies, or take no action. As in the other stages of the invasion process, a species' spreading success relates to its biological characteristics and the interaction with its surrounding habitat and species. Unlike previous stages, there may be more available information on the species' characteristics at this stage, which can inform decisionmaking. From an ecological perspective, eradication may yield the most desirable outcome. However, it may be costly to achieve under conditions such as larger spatial scales or substantial population sizes. Consequently, eradication attempts often fail to reach their objectives. Section 4.1 focuses on the spatial aspects of control. "Eradication as an Optimal Strategy" highlights the efficacy of eradication as a control strategy.

Spatial Control Strategies

Invasive species management is inherently about the management of land, or space. Ecological literature provides the theoretical framework to capture the spatial aspects (e.g., Shigesada and Kawasaki 1997); however, the majority of the economic literature fails to explicitly incorporate the spatial aspect. Discussion of the spatial effects on management is only found in the literature pertaining to control strategies following successful establishment. This literature is quite limited and does not include any stochastic aspects.

Barrier zones reduce the spread of species either through eradication of small populations or quarantining a population. Using a dynamic framework, Sharov and Liebhold (1998a) assess the management of barrier zones for gypsy moths in the United States. To assess the efficacy of barrier zones, the authors construct a model of pest dispersal, which factors in the monetary damages and benefits of control. Model application requires information about the specific population: the length of the population front,

the shape of the population, the spread rate, the cost of the barrier zone, the damages caused by the species, and the discount rate. The conceptual framework (based on the spatial, economic, and biological components) is evaluated for three different spatial population spread scenarios: a strip with a constant width, a rectangular area, and a circular area. Parameterizing this framework with gypsy moth data and information from the Slow-the-Spread program (<http://www.gmsts.org/operations/>), the authors demonstrate the benefits produced by containment and eradication strategies over disparate geographic areas.

The authors indicate that eradication is viable for a species with a limited range, whereas slowing the species spread can be optimal in several scenarios. Using gypsy moth data as a case study, the model analysis shows eradication is optimal for certain small or isolated populations or both, whereas slowing the spread is better for larger, more established populations. Slowing the spread, as a control strategy, can yield substantial reductions in population spread (Sharov and Liebhold 1998b). The merger of economic and ecological relationships into a spatial model is demonstrated for one of the first times in this paper. Hof (1998) constructed a spatial model to illustrate how the effectiveness of barrier zones is reduced by the dynamics of the managed population. As the population grows, it can extend the size of the barrier zone or splinter, thus reducing the viability of barrier zones as an optimal management tool. However, an important caveat is pointed out in these two papers that, as with most papers reviewed in this synthesis, implementing such a framework has certain limitations. The choice of functional forms, model structure, and the data greatly influence the outcomes. Altering assumptions on these functional forms or other relationships included in the model can lead to varying outcomes. However, Sharov and Liebhold's model provides a spatial framework with explicit economic aspects that can be expanded to incorporate several scenarios and could potentially be extended to analyze decisions before the species begins spreading.

Building upon the framework set forth by Sharov and Liebhold (1998a), Cacho and others (2004) analyzed the critical points that govern the optimal control strategy:

eradication, containment, or doing nothing. Their model focuses on plants and includes several parameters such as maximum rate of spread, seed longevity, and costs of control. The authors represented the unknown length of seed longevity in differing environments by using a range of values in the biological parameters. They determined the switching points at which eradication and control are no longer optimal strategies by employing Scotch broom (*Cytisus scoparius*, L.) data and estimates. Based on this analysis, the salient characteristics are seed longevity and the spread rate. As the spread rate increases, the two switching points move closer together indicating that management should emphasize eradication.

Useful frameworks for incorporating the spatial dimensions into risk management strategies are proposed in these papers. Sharov and Liebhold (1998a) provided a caution in their paper, which is applicable to all models: "Control of natural resources may depend considerably on social factors; thus the model...cannot automatically generate decisions." Further work to understand and incorporate societal and other factors will increase the viability of these frameworks. Overall, very little literature explicitly analyzes the spatial aspects, and future work should definitely focus on the spatial dimension as it is one of the most crucial components in the invasive species management problem. Perhaps researchers can learn from areas with substantial existing spatial research such as wildfire prevention or land conservation.

Eradication as an Optimal Strategy

Eradication as a control strategy yields the most desirable outcome—total elimination of the invasive species from the habitat—but this strategy often fails due to numerous obstacles that impede complete removal, leading many to question the circumstances when eradication is feasible and optimal. Myers and others (2000) cited several successful eradication cases noting that success relies upon certain key conditions. Simberloff (2001) argues that eradication in itself is not impossible, but is idiosyncratic and contingent upon several criteria:

1. Sufficient resources to successfully complete the project.

2. Clear and identifiable authority to oversee the project.
3. Fairly good information regarding the biological characteristics of the species; i.e., the same basic criteria needed for successfully implementing any activity involved in invasive species management.

He mentions resource constraints but without explicitly employing economic frameworks to assess the management options in the control stage. Several authors have addressed this gap by identifying the economic conditions under which eradication is optimal (e.g., Eiswerth and van Kooten 2002, Olson and Roy 2002, Regan and others 2006, Taylor and Hastings 2004).

Olson and Roy (2002) focused on the costs of control and damages of species currently under management (i.e., they captured the decision of a manager who must choose future strategies for an existing population). The policymaker minimizes the expected discounted control costs plus the damages caused by the remaining population conditional upon the species' growth function. The growth function incorporates environmental disturbances as a random process. As these disturbances increase, so do the chances of the population growing. Using this framework, they develop a rough guide of conditions favoring eradication. For small populations with marginal damages greater than marginal control costs, eradication is always optimal. When marginal damages are less than marginal costs, eradication is still optimal if the growth rate is sufficiently high. Irrespective of population size, eradication is optimal if the damages significantly outweigh the control costs in the worst possible scenario of environmental disturbances. Whereas this stylized framework is fairly general and cannot be directly implemented, it provides an approximate rule-of-thumb to ascertain the optimality of eradication as a management strategy. The one drawback is the information requirements; the marginal costs relative to the marginal damages must be known fairly well to determine the optimal management strategy.

Eradication not only depends on the relative costs and damages of controlling the population, but also upon the

tenuous relationships between the population and its habitat. Environmental and demographic stochasticity and the Allee effect can drive low-density populations towards extinction (Liebhold and Bascompte 2003). The Allee effect works similarly to the critical depensation point or a threshold under which a population cannot survive. The Allee effect has been observed for low-density populations, but could apply to other populations as well. This effect contributes to an extinction threshold; if a species' population is low enough, extinction will automatically occur. All species exhibit this effect, except asexual organisms like some plants. In general, management methods should be aimed at increasing the probability of extinction. Extinction is highly likely if an adequate number of the population is removed, although achieving 100-percent eradication is difficult. Taylor and Hastings (2004) utilized *Spartina alterniflora* (a non-native grass in Washington that exhibits a weak Allee effect) to test this theory while accounting for economic aspects. Their analysis of the *Spartina alterniflora* data indicates that, in the absence of an Allee effect, the optimal strategy involves the removal of isolated, high-growth, low-density species. The model analysis establishes a relationship between budget and optimal strategy: lower budgets necessitated the removal of low-density plants, and the optimal strategy with larger budgets is to focus on eradicating high-density areas. For this particular species, the Allee effect does not lead to cheaper eradication. Hence, the Allee effect plays a role in determining eradication strategies, but it must be considered on a species-specific basis and in conjunction with budget constraints.

Regan and others (2006) constructed a theory to analyze the optimal time needed to survey an area before declaring that an eradication attempt has been successful. Evaluating the efficacy of eradication strategies depends on the reliability of survey strategies, which, in turn, depends on the amount of time and resources devoted to detection. These authors postulated that managers facing budget constraints may prematurely cease surveying, which could result in a new eruption of the pest if the species was not fully eradicated. The authors develop a simple rule of thumb for the optimal number of consecutive zero surveys by

minimizing the sum of survey costs and expected damages. They compare this rule of thumb with the results of a fully optimal forward-looking solution derived using stochastic dynamic programming. The key difference between the two approaches is that stochastic dynamic programming incorporates all the possibilities that can occur in the future, including the possibility that the plant will re-emerge, and a new attempt at eradication will have to be undertaken and then finds the best decision. The authors parameterize these two seemingly different approaches—the rule-of-thumb and the stochastic dynamic problem—using bitterweed (*Helenium amarum*) data. The authors state that this rule-of-thumb can reduce variability in decision strategies while increasing evaluating the sensitivity of their decisions to various parameters in the eradication programs.

Eiswerth and van Kooten (2002) argued that the categorization of risk in subjective terms necessitates the use of fuzzy membership functions, which differ from the traditional expected value approaches (similar to Horan and others in “Pure Uncertainty Versus Risk in Assessing Prevention Strategies”). Subjective risk assessments can produce widely varying outcomes depending on the scientists or experts administering the assessment (e.g., Woodward and Bishop 1997). A stochastic dynamic model maximizing the agricultural producers’ discounted present value of net returns is presented in the paper. The objective function consists of the agricultural production, which depends on the size of the invasion and the choice of control technology. The objective function is conditional upon the species growth function, which includes a stochastic term. As part of this research, the authors surveyed land managers to gauge their management choices under risk. The authors parameterize this model using results of this survey and extant data for the yellow starthistle (*Centaurea solstitialis*). The analysis illustrates that land managers tend to aggressively control a species even when the economic criteria do not warrant such a stringent control regime. The optimal control strategy involves managing the spread of yellow starthistle instead of full eradication, even though this weed has significantly impacted agriculture.

Allocating Resources Among Multiple Strategies

Management activities in one stage have direct consequences on other stages although specific stages of the invasion process are the focus of several papers. For example, scarce resources necessitate allocation between several activities. Decisionmakers determine these allocations concurrently, thus the framework should incorporate the relationships between these stages. Economic literature often focuses on the introduction and postestablishment stages of the invasion process to identify the optimal strategies between exclusion and control activities. The allocation between control and other activities, such as postintroduction detection, is the focus of some papers. The interaction between mitigation and adaptation activities is discussed in “Understanding Risk Mitigation Versus Adaptation.” Optimal resource allocation strategies amongst differing activities are addressed in the other sections.

Understanding Risk Mitigation Versus Adaptation

Risk analysis often treats mitigation and adaptation separately, but invasive species risk analysis needs to account for both of these actions for effective management practices. Shogren (2000) discussed the distinction between mitigation—actions where people actively reduce the probability of a bad state, and adaptation—actions which reduce the magnitude of a bad state if it happens (as with insurance). He proposed the need to account for both of these actions simultaneously owing to the fact that an action to reduce risk affects the consequences if the species does invade. His model is based on endogenous risk theory to analyze risk-benefit tradeoffs for explosive invaders, and it depicts the problem facing a representative policymaker allocating scarce resources. These ideas stem from economic theory on decisionmaking under risk and uncertainty as addressed in “Defining Risk” and “A (Brief) Primer in Economic Theory” (de Finetti 1974, Drèze 1987, Savage 1954, Von Neumann and Morgenstern 1944).

Perrings (2005) built upon Shogren’s framework and extended it to examine decisionmaking practices aimed at allocating resources between these two strategies. He classified management strategies addressing risk into the

same categories: mitigation and adaptation. Mitigation refers to actions that alter the chances of an event occurring. In invasive species literature, mitigation activities reduce the likelihood of invasions. Adaptation refers to actions that alter the value of the outcome. These activities would reduce the impact cost of invasions without changing the probability of the invasions themselves. Decisions regarding mitigation and adaptation activities often occur simultaneously. The chosen strategy relates to where the species is in the invasion process (i.e., whether the species has just been introduced or whether it has already established). The manager must also assess whether the situation is observable or controllable. Perrings pointed out that there are two schools of thought regarding the predictability of invasions (“Factors Fueling the Invasion Process”). The first school, including Williamson (1996), argues that invasions can rarely be predicted beyond a few indicators such as propagule pressure and the past invasion history of the species. Others, such as Rejmánek (1999), believe that the invasiveness of a species and the susceptibility of the land can be predicted by analyzing a wider range of salient characteristics.

Using probability transition matrices that follow a Markov Chain, Perrings evaluated four possible outcomes once a species has been introduced:

1. It may not establish.
2. It may establish irrespective of management activities.
3. It may establish, and its population will depend on the state of nature (including management activities).
4. It may establish and have an unstable population in the long run.

The probabilities in the transition matrices represent the overall resilience of the land against invasion. If these probabilities are known, a model of the system’s dynamics and the value function (both dependent on the probability transition matrix) can guide the optimal choice of strategies. In addition to the probabilities, the model requires knowledge of the expected net benefits and costs of different control regimes, and a feedback matrix that links control choices to the probability transition matrix. If this information is known, the outcomes of control measures (e.g., those

that only reduce population size, can be assessed for their long-run effectiveness).

Mitigation is an appropriate option if the expected outcomes of management activities can be assigned some probabilities. In situations where probabilities for the connections between actions and outcomes are unknown, mitigation cannot occur, and managers are left with adaptation as the only possible strategy. Perrings’ main objective was to draw attention to the need to quantify unknown aspects as he stated at the end of his paper, “In an environment in which decisionmaking is increasingly dominated by non-probabilistic ‘scenarios’ which drive decisionmakers to focus on adaptation, it is important to remind ourselves that this may be both inefficient and inequitable.” This argument arises from the idea that any structured analysis based on some quantitative information is better than the alternative because conventional wisdom does not necessarily lead to optimal strategies, such as the case of tariffs to reduce invasion risk (“Policy and Market-Based Mechanisms to Manage the Risk of Introductions”).

Maximizing Welfare Through Invasive Species Management Activities

Unlike the previous papers in this synthesis, the focus in this section is on the tradeoffs between management strategies and their social benefits and costs. Welfare functions allow the analyst to capture the overall benefits and losses of a management decision. The use of welfare functions is employed in several papers in their objective functions to assess optimal resource allocation strategies (e.g., Finnoff and Tschirhart 2005, Finnoff and others 2005, Leung and others 2002).

Leung and others (2002) showed that prevention is more cost effective than control. Stochastic dynamic programming captures the situation facing a policymaker allocating resources between prevention and control activities on an aggregate level. Welfare consists of the profit function minus the costs of invasive species management activities. The invasive species grows according to a logistic function plus some stochastic term representing uncertainty in species growth patterns in the new environment. The planner’s maximization problem optimizes welfare over

a probability transition matrix that reflects the probability of moving across States (i.e., different invasion outcomes) given various allocations between exclusion and control strategies.

Implementing the Leung and others (2002) model requires the following: data on a species' growth function, the costs of controlling that particular invasive species, the efficacy of control, the total inputs and costs for the industry, the total outputs and prices for the industry, the monetary loss caused by the invasive species, and the probability of invasion. Data on zebra mussels and powerplants, coupled with estimates of certain biological characteristics and the probability of invasion, are used to simulate three possible scenarios for lakes: uninverted over a 25-year time horizon, invaded over 25 years and uninverted for 5 years. The simulations determine the optimal allocation for prevention strategies given two control options (do nothing or do something) for 10 years. The optimal expenditures for prevention activities yield the greatest welfare. However, the difference in cumulative welfare resulting from optimal expenditures, suboptimal expenditures, and taking no action is relatively small. Engaging in optimal prevention activities is ideal over the longer time horizon (25 years), whereas the optimal strategy with the shorter time horizon (5 years) is to not take any action. As in several other papers, Leung and others (2002) employ data from a highly invasive species with high growth rates and high damages (in this case, the zebra mussel). Using such a species illustrates the worst case scenario. Applying this model to less insidious invasive species may produce different outcomes. The advantage of this model is the explicit linkage between private industry and management activities. Whereas actual data may not exist for all components of the model, estimates can be used to analyze the potential scenarios facing the policymaker for diverse industries and invasive species.

Leung and others (2005) follow up their previous work with an attempt to bridge the gap between theory and application by proffering a framework to identify general rules-of-thumb for resource allocation over various invasive species management activities. Extending the concepts in their earlier paper, the authors establish the relationships underlying optimal choice of exclusion and control strategies. The

policymaker endeavors to maximize cumulative social welfare conditional on several factors: (1) the welfare in an invaded state, (2) the welfare in an uninvaded state, and (3) the probability of invasion dependent on the prevention strategy, invasion parameters, and the efficacy of prevention. Based on the model analysis, optimal control expenditure increases with the system's value and decreases with uncontrollable damages (amongst other rules). The optimal prevention expenditure is closely tied to the preventability of invasions. Several rules characterize the optimal expenditure including one stating expenditures decrease as the probability of unpreventable invasions increase. The authors provide a detailed list of data required to implement the model as well as a thorough comparative statics analysis of the interaction between the various parameters and variables. This model's strength lies in its application using available data. However, the simplified framework comes at a cost—several strong assumptions (e.g., the specific functional forms, the relationships included or excluded in the framework, the availability of data necessary to implement the framework, etc.) underlie the model. The loss of specificity translates to a gain in the ease of implementation and a decrease in the time needed to reach general management rules.

Building upon the underlying tradeoff between prevention and control, Finnoff and others (2007) evaluated the effect of manager's risk preferences on the optimal investment in management activities. Risk preferences dictate the valuation and incorporation of risk into decisionmaking frameworks. The authors postulated that, based on their endogenous risk model (an extension of Shogren 2000, "Understanding Risk Mitigation Versus Adaptation"), risk-averse models tend to over-invest in control while under-investing in prevention. As a manager's risk aversion increases, so does the propensity to implement control activities. This behavior results in increased invasions as indicated in their paper. This paper was based upon an earlier one (Finnoff and others 2005) where a similar endogenous risk framework analyzed the role of feedback between decisionmakers (i.e., the firms or the manager) and biological and economic aspects associated with invasions.

Here, feedback refers to the ability of the decisionmakers to update beliefs based on changes in the situation. If decisionmakers neglect to respond to these changes, the results could range from minor efficiency loss to severe biological and economic consequences as a result of invasions.

Determining Optimal Allocations Based on Inter-Species Relationships

Like humans, plants can be thought of as welfare maximizing organisms whose survival success depends on certain biological traits, which can predict outcomes from interaction with other plants, humans, and their environment (Finnoff and Tschirhart 2005). Contrary to previous papers on species management, the focus in this paper is on the species (i.e., the plant) as an optimizing agent, which aims to maximize its biomass conditional on specified parameters and the presence of competitors in the habitat. Finnoff and Tschirhart explain the uniqueness of this model compared to previous ones: "In the plant community model herein, the theory starts prior to population updates by first assuming the individual plant behaves as if it is choosing its optimum biomass. Optimization is done given the plant's parameters and the presence of other competing plants in its own and other species." Using this model, the authors evaluate individual plant behavior and species interactions as they result from plant-specific traits, environmental factors such as temperature, and human interaction. Each scenario analysis offers a rough guideline for plant behavior given certain conditions.

The authors classify invasive species as redundant or successful. Redundant species fail to invade successfully but remain in the habitat as biological insurance until environmental conditions become favorable for them. Successful species effectively invade the new habitat from the start. These two categories are mutually exclusive, but species can switch groups over time as the environmental circumstances change. The plant's efficient energy usage dictates its growth function, which updates the model. An individual plant's optimization problem—maximizing net energy—includes the leaf size, the flow of solar radiation, the biomass, and the energy expended for the plant's

functioning. Additionally, as a member of a plant community, the population size and growth vis-à-vis the available land capacity combine to enter as a space constraint that also influences the individual plant's optimization. The model of plant behavior is then incorporated to a policymaker's welfare maximization problem because there is feedback between human decisions such as agricultural management and species success. Through this framework, the authors capture the interconnection between ecological and economic relationships in a situation with multiple species. The policymaker chooses prevention and control efforts to manage an invasive species. The probability of invasion depends solely on prevention efforts. Accounting for human effects on species population, the plant's growth function has altered to now include population reductions through harvest and control measures.

Based on the relationships and factors in just the plant relationships, the analysis determines that the optimum plant biomass in the steady state depends largely on plant-specific traits, namely those related to respiration activity. Expanding this result to multiple species provides criteria to predict species success in steady-state scenarios. Factors beyond the plant-specific parameters, such as temperature, also drive the optimization behavior. After augmenting the aforementioned plant relationships by temperature, the authors analyze the optimization behavior to find that any number of species can co-exist regardless of the resource constraints in this model. This outcome deviates from previous papers in that the number of resources dictates the maximum number of coexisting species populations. The authors construct a conceptual framework encompassing the major economic and ecological factors that impact plant success. Although the authors do not apply empirical data to the model, this can be done using data for current species and estimates for potential species. The majority of the model is deterministic except for the probability of invasions, thus the information necessary to implement the model should be available. By explicitly incorporating complex species interactions, a creative, albeit unorthodox, approach for evaluating the ecological consequences of human actions is proffered in this paper.

Focusing on the Cost Versus Damage Tradeoff to Identify Optimal Strategies

Optimal strategies for multiple activities can be found by focusing on the tradeoffs between the management costs and the species' damages deterred by engaging in the particular management activity. The optimal resource allocation between prevention and control activities with a stochastic initial population size depends on the marginal damage function of the invading species (Olson and Roy 2005). Whereas this model cannot be directly implemented due to the theoretical nature of the framework, their analysis produces general rule-of-thumb principles for optimal resource allocation between prevention and control activities. The framework represents a situation where an invasive species has been controlled, and the decisionmaker must allocate resources for potential management strategies for the same species. As an example, the gypsy moth (*Lymantria dispar*) presents such circumstances; it has been controlled in certain areas and requires continuous management. The management options can differ from exclusionary activities for preventing new introductions of the gypsy moth, to control strategies for managing remaining gypsy moth populations.

The policymaker chooses the level of prevention and control activities. The costs of control and prevention are assumed to be known, but the damages from the resulting invasion are driven by a stochastic relationship representing the risk of an unknown invasion. The policymaker selects the prevention and control strategies simultaneously prior to the invasion, which reflects the decisionmaking process in risk management. However, the established population size is known indicating that the invasion had already occurred and these management decisions focus on potential invasions going forward from either the same species or other species.

The role of risk on optimal resource allocations is highlighted in this paper. An increase in risk is represented by an increase in the variability associated with the chance of an invasion. The optimal choice between prevention and control following such an increase in risk depends mainly on the shape of the marginal damage function. Thus, the

species' damage function must be known to apply this framework. Data on past damages from the species can be used to estimate the damage function, which can then determine the optimal management strategy for the species in an uninvaded area or a reoccurrence of the species in the same area.

Whereas the focus is on the introduction and spread stages in many papers, there are few where the detection stage between the introduction of the species and the subsequent establishment and spread are explicitly addressed. The unclear relationship between species that are intercepted or discovered during the introduction stage and the established species being found in ecosystems is due to the fact that successful introductions do not often translate to successful invasions (Williamson 1996). Even those species that successfully establish often begin to spread after long lag periods (Crooks and Soulé 1999). Lags occur for many reasons such as natural lags in population dynamics or changes in the environment and the genetic composition of extant species. Additionally, past experiences with species do not provide good indicators of their future invasiveness due to an ever-changing environment and the response to and by other species. Also, species introduced many years ago may now have populations that are sizeable enough to detect (Costello and Solow 2003). These factors contribute to the uncertainty surrounding the establishment stage of the invasion process.

If populations are detected early in the invasion process, either before they fully establish or as they are establishing, control strategies can commence sooner and, possibly, at a lower cost. Some species, such as the black-striped mussel (*Mytilopsis* sp.) in Australia, have been eradicated due to detection activities, which included constant surveying followed by quick mobilization upon detection (Myers and others 2000). Mehta and others (2007) captured the stochastic and dynamic aspects of this tradeoff between detection and control activities. The model focuses on a decisionmaker minimizing costs and expected damages for a single invasive species by choosing a constant optimal search level at the detection stage. The time of detection is stochastic and depends on the effort devoted to search activities and how easy it is to detect the species.

Based on simulations representing four types of species, the model analysis indicates that it is often optimal to devote significant resources to detection efforts for species causing high damages, even if the species is difficult to detect. The optimal strategy for species that do not have a high-damage potential involves undertaking no action if the population is sufficiently small, if the detection is quite difficult, or if post-detection control activities are costly. Even if a species causes a high level of damage, it may not be optimal to invest in detection when the post-detection control strategy is relatively costly (i.e., the control costs are near or greater than the damages produced by the species). The simulations show that the biological parameters are more influential than the economic parameters. This may be an artifact of the specific model, but it is a point worthy of further exploration. It is demonstrated in the paper that the optimal detection strategy relies greatly on the detectability of the species, similar to findings from Cacho and others (2006) who applied search theory to a spatial model aimed at analyzing detection and control strategies. Whereas the Cacho and others model does not include any economic aspects, it does incorporate the risk underlying these activities and the role of detection on subsequent eradication strategies to illustrate the importance of detectability in the optimal detection strategies for weeds.

Some characterizations of the tradeoff between the costs of managing invasive species and the damages inflicted by them are provided in these papers. The variety of potential methods of addressing resource allocation amongst several activities is also touched upon. General guidelines for resource allocation are established as well. However, direct application of these models is fairly difficult. Specific models, or examples, of these strategies in practice would be quite instructive and useful for pragmatic application.

Discussion

An overview of some of the existing frameworks for evaluating risk management from an economic perspective is provided in this synthesis, as the field of invasive species management literature continues to evolve and expand. New

collaborations and new knowledge have spawned, and will continue to create, a wide range of methodologies aimed at identifying optimal strategies and mechanisms for diverse management cases and objectives. The individual sections illustrate that several creative and insightful decisionmaking frameworks have already been explored. Nonetheless, there are numerous potential research areas that need to be investigated.

Space and invasive species are closely intertwined. Models, which explicitly incorporate the spatial and economic aspects are crucial to the invasive species management problem, yet very few currently exist. Also, current economic models focus on only three major management activities. Other management activities, such as restoration and public outreach, offer high returns for invasive species management and ought to be considered in the risk management framework as they occupy a place in the decisionmaking framework for agency managers. The set of activities included in risk management framework should be expanded, as well as the number of activities included in resource allocation frameworks. Realistically, management activities are undertaken concurrently and the theoretical frameworks should reflect this.

Only a few models incorporate multiple species, so this should be expanded to understand the interaction between species as well as optimal resource allocations across species. Approaches that transcend the traditional risk management, or expected values and framework are employed in some papers; they highlight crucial issues involving the levels of risk facing managers. Increasing an awareness of different methodologies for incorporating stochastic elements will help agency managers and expand the number of tools available for characterizing management risk. Overall, these models tend to be general. Whereas that is important for establishing overall frameworks and guidelines, future work should focus on specific species to emphasize the link between theory and application. Also, the focus tends to be solely on insidious species in some papers, whereas agencies face a wide gamut of invasive species. These frameworks should be applied to a variety of different types of species, and the first step towards this has been taken through the range of simulations used in these papers.

The interdisciplinary body of literature in this field is constantly growing. As such, certain key papers have been focused on in this synthesis while acknowledging that other recent or related papers may have been omitted. The purpose of the synthesis is to provide a basic overview of the existing state of invasive species risk management literature from an economic perspective. Hopefully, this review will encourage readers to continue to push the boundaries of this research by engaging across the disciplines to discover novel and exciting approaches for decisionmaking tools for invasive species.

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The Formation of Dense Understory Layers in Forests Worldwide: Consequences and Implications for Forest Dynamics, Biodiversity, and Succession

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Abstract

Alterations to natural herbivore and disturbance regimes often allow a select suite of forest understory plant species to dramatically spread and form persistent, mono-dominant thickets. Following their expansion, this newly established understory canopy can alter tree seedling recruitment rates and exert considerable control over the rate and direction of secondary forest succession. No matter where these native plant invasions occur, they are characterized by one or more of the following: (1) the understory layer typically has greater vegetation cover and lower diversity than was common in forest understories in the past; (2) this layer can delay stand renewal and alter species composition by inhibiting tree regeneration; and (3) once this layer is formed, it can resist displacement by other species and remain intact for decades. In this paper, we evaluate the processes that trigger the expansion of several plant species native to forests and review their ecological characteristics to provide general guidelines in assessing native invasion risk in forest stands.

We argue that major anthropogenic changes to disturbance and browsing regimes bring about the monopolization of the forest understory by native plants. In all cases reviewed, aggressive understory plant expansion followed alterations in overstory disturbance regimes. Although these disruptions included predictable and manageable impacts such as tree harvesting, other less predictable overstory disturbance agents including catastrophic fires, insect outbreaks, and pathogens were involved. Assessing and managing risk from these alternative threats is challenging as their occurrence is often erratic, hard to control, and not limited

by land ownership and administrative boundaries. In many cases, the risk to forest understories was particularly acute if the effects of multiple stressors occurred in a stand, either in tandem or within a short period of time. Specifically, the synergy between overstory disturbance and uncharacteristic fire regimes or increased herbivore strongly controls species richness and leads to depauperate understories dominated by one or a few species.

We suggest that aggressive expansion by native understory plant species can be explained by considering their ecological requirements in addition to their environmental context. Some plant species are particularly invasive by virtue of having life-history attributes that match one or more of the opportunities afforded by multiple disturbances. Increased overstory disturbance selects for shade-intolerant species with rapid rates of vegetative spread over slower growing, shade-tolerant herbs and shrubs. Altered fire regimes select for only those species that can survive the fire or resprout thereafter. Finally, overbrowsing selects for only those species that are well defended or tolerant to browsing. Ultimately, these processes create novel conditions that favor only a small subset of species that possess some combination of the following life-history characteristics: rapid vegetative growth, relatively shade intolerant, fire tolerant, and herbivore tolerant. The result is a low diversity but dense understory that can persist for long periods of time even if the canopy closes.

The framework advanced by this review could aid land managers in implementing informed management policies and practices that both limit the spread of these plants and target control and remediation treatments directed at the precise mechanism of interference. We suggest vigilant monitoring of stand conditions to ensure that alterations to the overstory and understory disturbance regimes do not operate concurrently, particularly when control over these factors falls under the purview of different management agencies (e.g., wild game vs. forestry management agencies). Finally, we caution that decisions regarding partial or complete overstory removals should consider a site's

understory conditions including inadequate advance regeneration, presence of clonal understory plants, fire history, and high herbivore impact.

Keywords: Competition, interference, invasive, recalcitrant understory layer, regeneration.

Introduction

Major anthropogenic changes in the frequency and severity of natural disturbance regimes can radically alter understory species composition and threaten the long-term sustainability and biodiversity of plant ecosystems (Alpert and others 2000, Roberts 2004, Rooney and others 2004). These changed disturbance regimes often trigger rapid expansion of native plant species that previously occupied a relatively minor portion of the understory flora (de la Cretaz and Kelty 1999, Mallik 2003, Vandermast and Van Lear 2002). Following their release, these herbs, shrubs, trees, and vines aggressively colonize and overtake disturbed patches forming persistent, nearly monospecific, and seemingly impenetrable thickets. This layer, identified in the literature as competing vegetation, interfering plants, low canopy, non-crop vegetation, native invasives, recalcitrant understory layer, or weeds, creates conditions below its canopy that reduce tree seedling establishment and survival, inhibit seedling growth into the sapling-size class, and alter species composition (Bashant and others 2005, Horsley 1993a, Messier and others 1989, Nilsen and others 2001, Tappeiner and others 1991). The impacts of this interfering layer alter the rate, direction, and composition of tree regeneration so profoundly that forest recovery following disturbance may contrast sharply with the predicted patterns of vegetation development for a particular forest type. Thus, these dominant understory layers often are the crucial factor determining success or failure of tree regeneration following harvest, thus threatening sustainable forest management (Ehrenfeld 1980, Gill and Marks 1991, Huenneke 1983).

In this paper, we first review the processes that cause the formation of recalcitrant understory layers. Second, we describe how these layers alter the rate and direction of forest succession. Third, we review published work to identify how these layers control tree recruitment, growth, and survivorship and, thus, patterns of tree regeneration and

succession. Fourth, we identify the most prominent causal mechanisms for the formation of these layers and outline the consequences of their formation on successional dynamics and forest regeneration. Finally, we discuss how recalcitrant understory layers may reduce floristic diversity, we argue for their incorporation into forest successional models, and we explore management options for mitigation of their impacts.

On the Development of Recalcitrant Understory Layers Worldwide

Recent changes in disturbance and browsing regimes have strongly impacted species composition in forest understories worldwide (Coomes and others 2003, de la Cretaz and Kelty 1999, Mallik 2003, Vandermast and Van Lear 2002). Typically, these changes have led to large increases in the density and cover of a small number of native understory plant species (e.g., Mallik 2003). In many cases, these species expand to form persistent, monodominant layers that, in some cases, are nearly impenetrable (Figure 1, Tables 1 and 2). We term these dense strata recalcitrant understory layers. No matter where they occur worldwide, they are characterized by sharing one or more of the following attributes: (1) the understory layer is often more dense with greater vegetation cover and lower diversity than was common in forest understories in the past; (2) this layer can alter successional trajectories and slow the rate of succession by creating conditions in the understory near ground level that are inimical to seeds and seedlings of many tree species (e.g., very low light at the soil surface); or (3) once this layer is formed, it can resist displacement by other species and remain intact for decades, even beneath closed-canopy forests. These layers and species have been termed low canopies (Schnitzer and others 2000) and native invasives (de la Cretaz and Kelty 1999), respectively. We prefer recalcitrant understory layer because it emphasizes that the effect of this layer occurs in the understory and is resistant to displacement. Additionally, the term native invasive suggests these species, similar to exotic invasives (e.g., exotic Japanese barberry, Amur honeysuckle; reviewed by Richburg and others 2001), are invading novel habitat when in fact the species that formed these layers were present throughout the

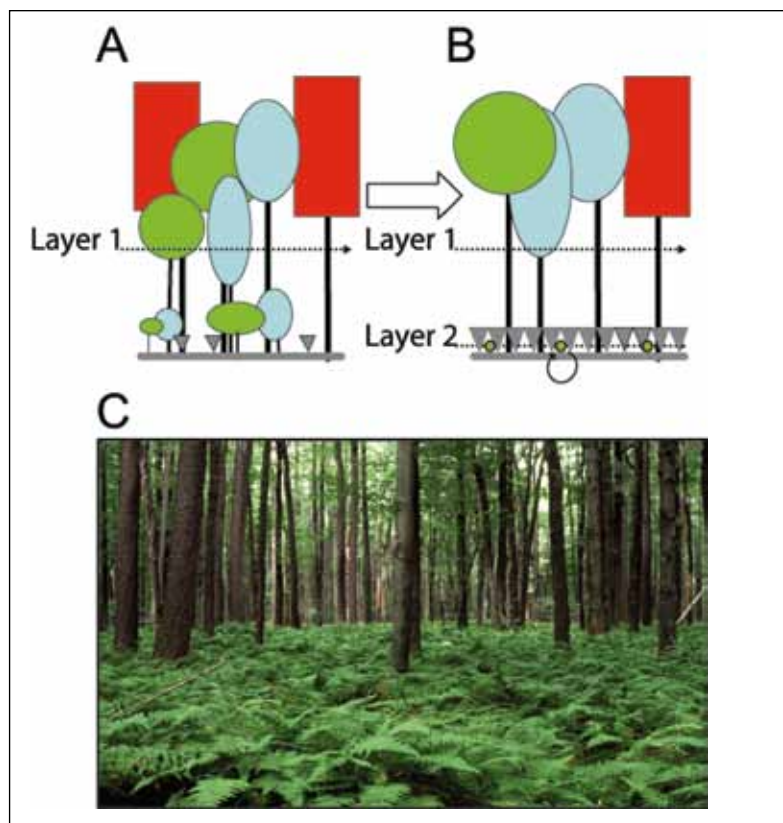


Figure 1—Diagrammatic representation of the conversion from (A) forests containing a diverse and structured advanced regeneration layer with sparse understory plant abundance (shown as grey inverted triangles), to (B) forests where a native understory species expands and monopolizes the understory. The dense herbaceous or shrubby cover represents a new vegetation layer that exerts direct and indirect interference effects and prevents seedling (shown as small green disks) recruitment into the sapling class. (C) Photographic example with hay-scented fern in northwestern Pennsylvania forests (Photograph by Alejandro Royo, USDA Forest Service)

habitat at varying degrees of abundance. Overall, we argue that models and theories of forest succession must now consider that many forests have a strong understory filter that determines which tree species are present to take advantage of a newly formed gap. In many cases, these recalcitrant understory layers are dramatically altering forestwide species diversity and patterns of succession.

Processes Causing the Formation of Recalcitrant Understory Layers

In this section, we discuss how natural processes, including such stressors as (1) overstory disturbance, (2) elevated herbivore regimes, and (3) altered fire regimes may be treated either as threats or benefits to forest communities.

Overstory disturbances reinitiate stand development (Oliver and Larson 1996), and characteristic fire and herbivore regimes often promote species coexistence (reviewed in Bond and Keeley 2005, Huntly 1991). What constitutes a threat or a risk often is intrinsically linked (and, thus, often critiqued) to a subjective value of what constitutes a loss in biological or ecological diversity, function, or service (see Power and Adams 1997 for a vigorous debate). To be workable, we narrowly categorize an uncharacteristic disturbance regime as a threat or risk if its occurrence results in a persistent negative impact on the ability of the disturbed stand to regenerate its predisturbance tree species composition. Utilizing that definition, it is clear that alterations to disturbance regimes can constitute a threat to forest

Table 1—Interfering species examples^a

Species	Family	Areas affected	Rapid vegetative growth	Herbivory	Fire	Overstory disturbance	Mechanism	References
<i>Dennstaedtia punctilobula</i> (Michx.) T. Moore	Dennstaedtiaceae	NE US	✓	✓ (T)	-	✓	1,2,3,5 [4]	Cody and others 1977; Anderson and Egler 1988; Drew 1988; McWilliams and others 1995; de la Cretaz and Kely 1999, 2002; George and Bazazz 1999 a,b; Horsley and Marquis 1983; Horsley 1977, 1993 a,b; Horsley and others 2003; Hill 1996; Hill and Silander 2001; Brach and others 1993 and Gleissman 1978; Gleissman and Muller 1972, 1978; Dolling 1996, 1999; Dolling and others 1994; Ferguson and Adams 1994; Den Ouden 2000
<i>Pteridium aquilinum</i> (L.) Kuhn	Dennstaedtiaceae	Worldwide	✓	-	✓	✓ (T)	1,2,3,5 [4]	Hill and Silander 2001
<i>Thelypteris noveboracensis</i> (L.) Nieuwl.	Thelypteridaceae	NE US	✓	✓ (T)	-	✓	[1]	Coomes and others 2003, Cunningham 1979
<i>Blechnum</i> spp.	Blechnaceae	New Zealand	✓	✓ (T)	-	✓	-	Jane and Pracy 1974, Rogers and Leathwick 1997
<i>Cyathea</i> spp.	Cyatheaceae	New Zealand	✓	✓ (T)	-	✓	-	Maheswaran and Gunatilleke 1988, Russell and others 1998
<i>Dicranopteris linearis</i>	Gleicheniaceae	Sri Lanka, Hawaii	✓	-	✓	✓	-	Guariguata 1990, Walker 1994, Walker and Boneta 1995, Russell and others 1998, May 2000
<i>Dicranopteris pectinata</i> (Burm.) Underw.	Gleicheniaceae	Puerto Rico	✓	-	✓	✓	[1]	Guariguata 1990, Walker and Boneta 1995
<i>Gleichenia bifida</i> (Willd.) Ching	Gleicheniaceae	Puerto Rico	✓	-	✓	✓	-	Kochummen and Ng 1977
<i>Gleichenia linearis</i> (Burm. f.) C.B. Clarke	Gleicheniaceae	Malasia	✓	-	✓	✓	-	Cater and Chapin 2000, Hogg and Liefers 1991, Liefers and others 993
<i>Calamagrostis canadense</i> (Michx.) P. Beauv.	Poaceae	N. America	✓	-	✓ (T)	✓	1,5 [4]	Jane and Pracy 1974, Rogers and Leathwick 1997
<i>Cortaderia</i> spp.	Poaceae	Boreal Forests	✓	✓	-	✓	-	Berquist and others 1999
<i>Deschampsia flexuosa</i> (L.) Trin. ex	Poaceae	New Zealand, European Boreal Forests	✓	✓ (T)	-	✓	-	

Species	Family	Rapid Areas affected	vegetative growth	Herbivory	Fire	Overstory disturbance	Mechanism	References
<i>Astrogynne martiana</i> (H. Wendl. ex Drude)	Areaceae	Costa Rica	-	-	-	-	1,2,[6]	Denslow and others 1991
<i>Geonoma cuneata</i> H. Wendl. ex spruce	Areaceae	Costa Rica	-	-	-	-	1,2,[6]	Denslow and others 1991
<i>Oenocarpus mapora</i> H. Karst.	Areaceae	Costa Rica	✓	-	-	-	6,[1]	Farris-Lopez and others 2004
<i>Asplundia uncinata</i> Harling	Cyclanthaceae	Costa Rica	-	-	-	-	1,2,[6]	Denslow and others 1991
<i>Aechmea magdalenae</i> (André) André ex Baker	Bromeliaceae	Panama	✓	-	-	✓	[1]	Brokaw 1983, Pfitsch and Smith 1988
<i>Sinarundinaria gangiana</i>	Poaceae	China	✓	-	-	-	-	Taylor and Zisheng 1988
<i>Chusquea</i> spp.	Poaceae	W. South America	✓	-	-	✓	[1,4]	Lusk 2001, Veblen 1982, Gonzalez and others 2002
<i>Guadua sarcocarpa</i> Londono & Peterson, sp. nov.	Poaceae	S. America	✓	-	✓	✓	[1,6]	Griscom and Ashton 2003
<i>Fargesia denudata</i> Sasa spp.	Poaceae	China	✓	-	-	-	-	Taylor and others 1995
<i>Yushania microphylla</i> Munro	Poaceae	Japan Bhutan	✓	-	-	✓	[1,4] [1]	Abe and others 2002, Iida 2004, Wada 1993; Gratzer and others 1999
<i>Calluna vulgaris</i>	Ericaceae	W. Europe	✓	-	✓ (T)	✓	[3]	Mallik 1995
<i>Kalmia angustifolia</i> (L.)	Ericaceae	E. Canada & United States	✓	-	✓ (T)	✓	[2,3]	Hall and others 1973, Meades 1983, Mallik 1995, Inderjit and Mallik 1996, Mallik and Inderjit 2001, Bradley and others 1997, Yamasaki and others 1998
<i>Kalmia latifolia</i> (L.)	Ericaceae	E. United States	✓	-	✓ (T)	✓	[1]	Woods and Shanks 1959, Monk and others 1985, Lipscomb and Nilsen 1990, Mallik 1995

Species	Family	Areas affected	Rapid vegetative growth	Herbivory	Fire	Overstory disturbance	Mechanism	References
<i>Rhododendron maximum</i>	Ericaceae	E. United States	✓	-	✓	✓(T)	1,2,3,4,5]	Monk and others 1985; Lipscomb and Nilsen 1990; Clinton and others 1994, Clinton and Vose 1996, Walker and others 1999; Nilsen and others 1999, 2001; Beckage and others 2000; Lei and others 2002; Beckage and Clark 2003, Christy 1986; Hille Ris Lambers and Clark 2003; Vandermast and Van Lear 2002 Beckage and Clark 2003
<i>aylussaccia baccata</i> (Wang.) K. Koch	Ericaceae	E. United States	✓	-	-	✓	1,2	
<i>Gaultheria shallon</i> Pursh	Ericaceae	W. Canada Pacific Northwest	✓	-	✓(T)	✓	3	Price and others 1986; Messier and Kimmins 1990, 1991; Messier 1992, 1993; Chang and Preston 2000, Chang and others 1996a,b; Bunnell 1990
<i>Vaccinium myrtillus</i> (L.)	Ericaceae	European boreal forests	✓	-	✓(T)	✓	1,2,3	Maubon and others 1995, Jäderlund and others 1997, Moola and Mallik 1998, Frak and Ponge 2002
<i>Empetrum hermaphroditum</i> (Lange ex Hagerup) Böcher	Ericaceae	European boreal forests	✓	-	-	✓	1,2,3	Michelsen and others 1995, Mallik 2003, Nilsson 1994, Wardle and Nilsson 1997, Zackrisson and others 1997, Nilsson and others 2000
<i>Pseudowintera colorata</i>	Winteraceae	New Zealand	-	✓(T)	-	✓	-	Godley and Smith 1981, Allen and others 1984, Coomes and others 2003, Husheer and others 2003
<i>Rubus</i> spp.	Rosaceae	Pacific NW & Europe	✓	-	-	✓	[1,4,5]	Franklin and Pechanec 1967, Tappeiner and others 1991, Schreiner and others 2000
<i>Rhus glabra</i>	Anacardiaceae	E. United States	✓	-	-	✓	1,2	Putz and Canham 1992
Lianas (various genera)	-	Tropical Forests & E. United States	✓	-	✓	✓	1,2,[6]	Smith 1984, Dillenberg and others 1993, Perez-Saliciup and Barker 2000, Schnitzer and others 2000, Schnitzer and Bongers 2002, Gerwing 2001, Grauel and Putz 2004
<i>Cornus</i> spp.	Cornaceae	NE United States	✓	-	-	✓	1,2	Ehrenfeld 1980, Huenneke 1983, Putz and Canham 1992
<i>Corylus cornuta</i> Marsh.	Betulaceae	Lake States United States	✓	-	-	✓	1,2	Sabhasri and Ferrell 1960; Batzer and Popp 1985, Tappeiner 1971, Tappeiner and John 1973, Tappeiner and others 1991, 2001; Kurmis and Sucoff 1989
<i>Acer spicatum</i> Lam.	Aceraceae	SE Canada, NE United States	✓	-	-	✓	[1]	Aubin and others 2000

^a Occurrence of dense, monodominant understory species. Information in this table summarizes whether the species possess rapid vegetative growth; if their increase in abundance is linked to alterations in the herbivory, fire, or overstory disturbance regimes; and whether they are fire or browse tolerant (Ts). "Mechanism" indicates the specific interference mechanisms exerted by a species (1 = aboveground competition, 2 = belowground competition, 3 = allelopathy, 4 = seed/seedling predation, 5 = litter, and 6 = mechanical damage). Mechanisms were tested using manipulative field experiments, unless they are in brackets, in which case they are speculative.

regeneration if they result in a degraded understory plant community composition monopolized by a select species that interferes with tree regeneration.

We found that major anthropogenic changes to disturbance and browsing regimes underlie the development of most recalcitrant understory layers (see Hobbs and Huenneke 1992 for their similar conclusion regarding exotic invasives). Indeed, overbrowsing, altered fire regimes, and increased overstory disturbance were implicated in 18, 34, and 82 percent, respectively, of the cases in Table 1. More importantly, our review suggests that the formation of a dense understory canopy layer arises approximately 53 percent of the time in the cases when overstory disturbances and altered understory fire and browsing regimes occur in tandem (Table 1). Additionally, these understory layers are depauperate because repeated canopy disturbances combined with other processes (i.e., fire and browsing) strongly favor a small subset of species.

Increased Overstory Disturbance

Direct and indirect human-induced disturbances (including logging, fires, insect outbreaks, and pathogens) have increased the extent and particularly the frequency of overstory disturbance over the past century (Carson and others 2004, Seymour and others 2002, Sharitz and others 1992, Youngblood and Titus 1996). These disturbances typically increase resource availability (e.g., light) in the understory both in the short and long-term. There is little doubt that these disturbances increase the establishment and growth of seedlings and saplings of canopy trees at least in the short term (Canham 1989, Canham and others 1994, Denslow 1987, Finzi and Canham 2000, Hartshorn 1978, Runkle 1982). However, these extensive and repeated overstory disturbances may be most beneficial to a few understory species that possess high rates of growth and vegetative expansion when exposed to high light (Ehrenfeld 1980, Huenneke 1983, Schnitzer and others 2000) (Table 1). These species are typically shade intolerant, yet highly plastic, so that they can persist at low-light levels following canopy closure by utilizing sunflecks or clonal integration (e.g., Brach and others 1993, Lipscomb and Nilsen 1990, Messier 1992, Moola and Mallik 1998).

There are numerous examples worldwide whereby canopy disturbances lead to the formation of recalcitrant understory layers (Table 1). Tappeiner and others (1991) found that logging increased the formation of salmonberry (*Rubus spectabilis*) tangles by nearly 300 percent over uncut stands. Throughout the Tropics, large-scale disturbances can create bamboo and fern thickets that persist for decades (Griscom and Ashton 2003, Guariguata 1990, Russell and others 1998, Walker 1994). In temperate and boreal forests, both native and exotic insect outbreaks open up vast areas of forest canopies (e.g., Gypsy moth (*Lymantria dispar* L.) and Spruce budworm (*Choristoneura fumiferana* Clemens.) often leading to an increase in the density and dominance of a few shrub species (Aubin and others 2000, Batzer and Popp 1985, Ehrenfeld 1980, Ghent and others 1957, Hix and others 1991, Muzika and Twery 1995). Fungal pathogens have opened up canopies in central New York (Dutch elm disease, *Ophiostoma ulmi*) causing the formation of widespread and dense patches of *Alnus*, *Cornus*, and *Viburnum* spp. (Huenneke 1983). Both Huenneke (1983) and Ehrenfeld (1980) argued that these dense shrub layers would delay canopy formation and alter its composition. Likewise, Chestnut blight (*Cryphonectria parasitica*) apparently led to the aggressive expansion of *Rhododendron maximum* in the Southern Appalachians (Vandermast and Van Lear 2002). In general, any process (whether anthropogenic or not) that increases light availability in the understory has the potential to lead to the formation of recalcitrant understory layers. Nonetheless, it appears that several processes must be altered in combination before these recalcitrant layers can form.

The Interaction of Elevated Herbivore and Canopy Disturbance

This section describes how extended periods of elevated browsing by either introduced or native mammalian herbivores (e.g., white-tailed deer in the Eastern United States; reviewed by Côte and others 2004, McShea and others 1997, Russell and others 2001) often coincide with large-scale canopy disturbances leading to the development of dense interfering layers. Frelich (2002) characterized chronic overbrowsing as a low-intensity disturbance, which,

Table 2—Estimates of spatial coverage by understory native plant invasions in forested areas at both local and regional scales

Species	Location	Type	Estimated forest (ha)	Proportion of forested area affected	References
<i>Dennstaedtia punctilobula</i>	Pennsylvania	Temperate hardwoods	2.1 million	0.33	McWilliams and others 1995
<i>Dennstaedtia punctilobula</i>	Allegheny National Forest, Pennsylvania	Temperate hardwoods	241,000 - 303,000	Approx. 0.47 - 0.59	Royo, unpublished data
<i>Rhododendron maximum</i>	Southern Appalachian Region, U.S.	Temperate hardwoods	2.5 million	-	Monk and others 1985
Lianas (various genera)	Barro Colorado, Rep. of Panama	Tropical forests	-	Approx. 0.22 of all gaps	Schnitzer and others 2000
<i>Pteridium aquilinum</i>	Fontainebleau Forest, France	Old-growth Beech forest	-	0.02 - 0.17	Koop and Hilgen 1987
<i>Pteridium aquilinum</i>	The Netherlands	Temperate hardwood and conifer	288,500	Approx. 0.7 of entire country	den Ouden 2000
<i>Kalmia angustifolia</i>	Newfoundland, Canada	Black spruce	-	0.55	English and Hackett 1994
<i>Gaultheria shallon</i>	British Columbia	Cedar/hemlock forest	100,000	-	Weetman and others 1990
<i>Guadua sarcocarpa</i>	Amazonia	Tropical forests	18 million	-	Nelson 1994
<i>Calamagrostis canadensis</i>	Western Canadian Provinces	Boreal forests of all cutover forest	-	0.25 - 0.5	Lieffers, pers comm.

Note: dash (-) in "Estimated area affected" and "Proportion of forested area affected" means that no information was available for this entry.

^a Coverage data convey either the total forested land area (in hectares) or the proportion of forested area dominated by a particular species within a region.

over time, can lead to depauperate understories composed almost entirely of highly browse tolerant or unpalatable species (Banta and others 2005, Frelich and Lorimer 1985, Horsley and others 2003, Rooney and Dress 1997, Waller and Alverson 1997). If these browse-tolerant or unpalatable species happen to be clonal shrubs or herbs, then any canopy disturbance that suddenly elevates understory light levels can cause the rapid expansion of these species. One of the best examples of the interplay between long periods of overbrowsing and canopy disturbance is hay-scented fern (*Dennstaedtia punctilobula*). This species historically occupied <3 percent of the understory in Pennsylvania (Lutz 1930) but currently forms a recalcitrant understory layer covering more than a third of the forested area in that State (Table 2) and is abundant throughout much of the Northeastern United States (de la Cretaz and Kelty 1999). Essentially, years of overbrowsing created a depauperate forest understory and suppressed woody establishment in the advance regeneration layer. When light levels increased, continued overbrowsing prevented successful seedling establishment and growth while the unpalatable hay-scented fern rapidly spread into this sparsely occupied habitat-forming dense monospecific stands (Figure 1). Other examples include Sweden, where clearcutting and overbrowsing convert forests to unpalatable grass-dominated communities (e.g., *Deschampsia flexuosa*; Bergquist and others 1999) and in New Zealand where arboreal herbivory by marsupials opens up the canopy, and, in combination with deer overbrowsing, leads to stands of unpalatable plant species (Allen and others 1984, Coomes and others 2003, Jane and Pracy 1974, Rogers and Leathwick 1997, Wardle and others 2001). In parts of New Zealand, forest area cover by shrubs, ferns, and grasses has increased from < 1 percent to nearly 30 percent in just 30 years (Batcheler 1983).

The Interaction of Altered Fire Regimes and Canopy Disturbance

This section discusses how human alterations to the frequency or severity of fire in various ecosystems (Attiwill 1994, Mallik 2003, May 2000) are often linked to the increase in interfering species. Frequent understory

fires thin the understory by reducing seedling and sapling density, thereby increasing light availability, and favoring species that can survive the fire or resprout thereafter (Abrams 1992, Collins and Carson 2003, Donlan and Parker 2004). When canopy disturbances and surface fires occur in tandem or within a relatively short period, the increase in light can contribute to the development of a recalcitrant understory layer (Mallik 2003, Payette and Delwaide 2003). For example, in boreal forests, Payette and Delwaide (2003) found that a “synergy” existed between fires and overstory disturbance, which created shrub-dominated heathlands. These heathlands became dominated by shrub species, mainly *Calluna*, *Kalmia*, and *Vaccinium* spp., which can rapidly resprout and spread clonally following severe fires (Mallik 1995, Meades 1983). Similarly, in tropical forests, various shade-intolerant ferns (*Dicranopteris*, *Gleichenia*, or *Pteridium* spp.) or bamboo (*Guadua*) that also spread clonally can rapidly colonize and monopolize areas following catastrophic fires (Dolling 1999, Finegan 1996, Gliessman 1978, May 2000, Nelson 1994).

Alternatively, canopy disturbances that coincide with a decrease in fire frequency can lead to the development of recalcitrant understory layers. Mallik (2003) hypothesized that long-term fire suppression in logged or defoliated stands led to forest “conversion” to *Kalmia*, *Calluna*, and *Gaultheria* heathlands. In temperate forest systems, fire suppression and canopy disturbances contribute to the spread of rhododendron and mountain laurel (*Kalmia latifolia*). These species now form recalcitrant understory layers that cover an estimated 2.5 million hectares in the Southeastern United States alone (Table 2; Monk and others 1985, Vandermast and Van Lear 2002). Furthermore, studies from the Coweeta Basin in North Carolina confirm the expansion continues with a doubling of rhododendron cover in only 17 years (Nilsen and others 1999).

The separate and combined effects of disturbances and browsing act as strong filters on species richness creating depauperate understories dominated by one or a few species. The degree of control or release of specific species will depend on the degree to which disturbance and browsing regimes are altered as well as the life-history characteristics

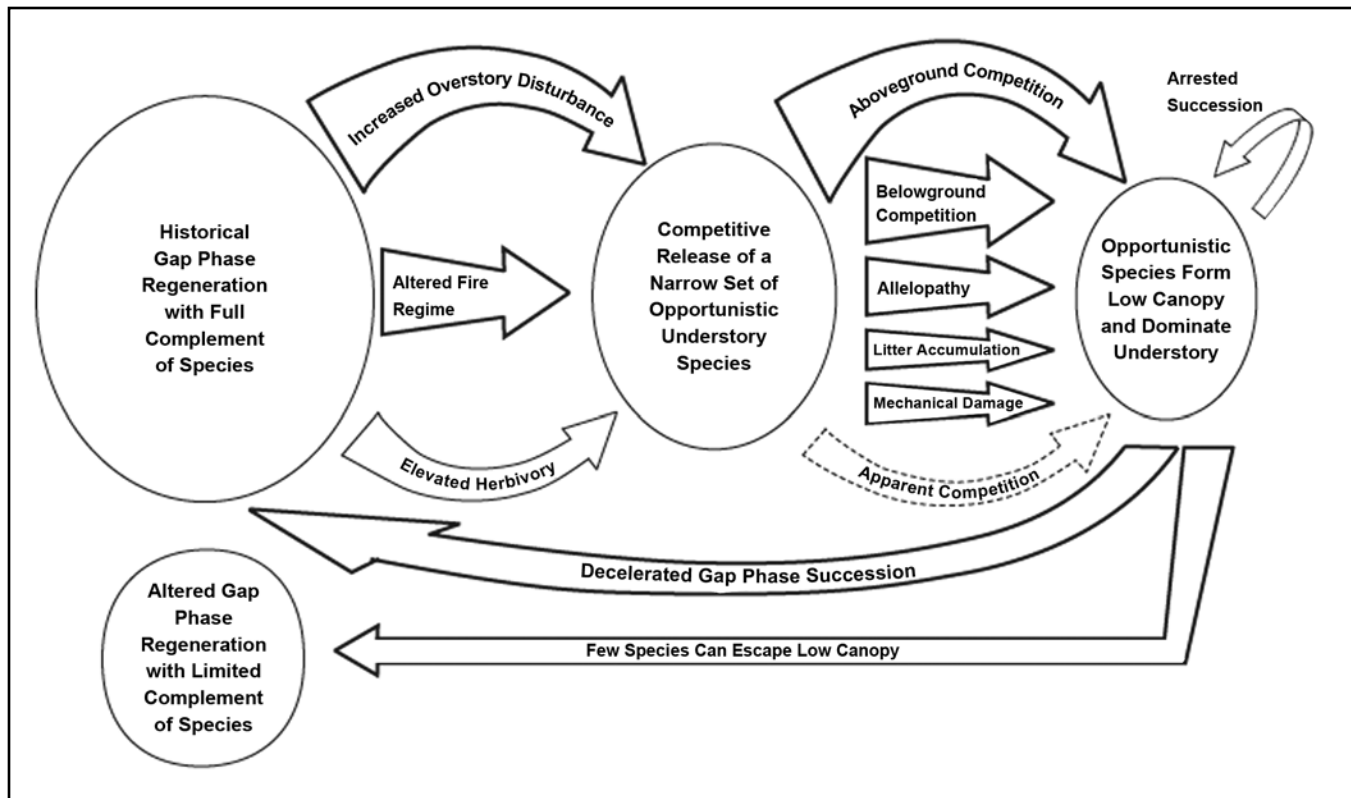


Figure 2—Conceptual model illustrating factors precipitating change from historical gap-phase regeneration into low-canopy dominance. The model also reveals various interference mechanisms and illustrates the ensuing successional pathways. The size and boldness of the arrows denote the relative importance of each transition as revealed by our review.

of the understory plant species (Roberts 2004). Overbrowsing selects for only those species that are well defended or tolerant to browsing (Banta and others 2005, Horsley and others 2003). Frequent fires select for only those species that can survive the fire or resprout thereafter (Gliessman 1978, Mallik 2003, Payette and Delwaide 2003). Finally, increased overstory disturbance selects for shade-intolerant species with rapid rates of vegetative spread vs. slower growing shade tolerant herbs and shrubs (Ehrenfeld 1980, Moola and Mallik 1998, Schnitzer and others 2000). Ultimately, these processes create novel conditions that favor only a small subset of species that possess some combination of the following life-history characteristics: rapid vegetative growth, relatively shade intolerant, and herbivore tolerant (Table 1; see also Roberts 2004). The result is a low diversity but dense understory that can persist for long periods of time even if the canopy closes.

Recalcitrant Understory Layers Arrest, Delay, and Alter Forest Succession

This section describes different ways that a recalcitrant understory layer can influence forest regeneration and stand development following a disturbance event. In the following sections we briefly review the literature to evaluate the evidence for three different successional pathways (Figure 2). These pathways include (1) indefinite suppression of subsequent tree regeneration (arrested succession), (2) a protracted period of stand establishment (delayed succession), and (3) a differential reduction of tree seedling recruitment that constricts species composition in the ensuing forest stand (altered succession).

Arrested Succession

In a small number of documented cases, recalcitrant understory layers appear to exclude tree regeneration for extended periods of time. This pathway is described by a

variety of terms including self-perpetuating climax community (Horsley and Marquis 1983), alternate stable state (Schmitz and Sinclair 1997, Stromayer and Warren 1997), polyclimax (Tansley 1935), or arrested succession (Niering and Goodwin 1974). Although the long-term stability of these systems is difficult to confirm (Connell and Sousa 1983; Peterson 1984; Sutherland 1974, 1990), there are compelling examples where shrubs and ferns have persisted for decades in stands formerly dominated by trees (Den Ouden 2000, Horsley 1985, Koop and Hilgen 1987, Latham 2003, Mallik 2003, Niering and Egler 1955, Petraitis and Latham 1999, Raich and Christensen 1989). It is unclear whether these layers are self-sustaining (e.g., via strong interference; Stromayer and Warren 1997) or if continued browsing or frequent fire is required to perpetuate them and retard the reestablishment of trees (Hill 1996, Mallik 2003).

Delayed Succession

A recalcitrant understory layer can slow the growth rate of tree species, thereby slowing the rate of successional change without altering the eventual tree species composition. For example, in boreal forests, the grass *Calamagrostis canadensis* suppresses the regeneration of dominant tree species, including white spruce (*Picea glauca*). This delays stand development by 20 to 30 years until saplings eventually emerge through the *C. canadensis* canopy, and the stands revert to forest (reviewed by Lieffers and others 1993). Delayed successions also occur in other boreal forests where a dense ericaceous shrub layer suppresses the-growth and emergence of tree species including western redcedar (*Thuja plicata*), Sitka spruce (*Picea sitchensis*), and Norway spruce (*Picea abies*) (Mallik 1995, Maubon and others 1995, Messier and Kimmins 1991, Messier and others 1989).

Additionally, a recalcitrant understory layer may reduce tree species survivorship sufficiently to delay succession. For example, in tropical forests, gaps promote the expansion of resident understory lianas (Schnitzer and others 2000). These understory lianas can become so dense after gap creation that they inhibit the subsequent growth and survival of both pioneers and shade-tolerant trees, thus stalling succession for decades (Schnitzer and others 2000). This dynamic of delayed gap-phase regeneration occurs

in tropical and temperate forests where lianas, fern, and bamboo thickets effectively clog gaps (Abe and others 2002, Guariguata 1990, Kochummen and Ng 1977, Schnitzer and others 2000, Taylor and Zisheng 1988, Walker 1994). In time, trees emerge from this layer and reach the canopy, apparently with little impact on species composition or the ensuing successional trajectories (Abe and others 2002).

Altered Forest Succession

A recalcitrant understory layer may differentially reduce establishment among co-occurring tree species, thereby controlling the composition of the advance regeneration layer (George and Bazzaz 1999a, 1999b). Dense understories create conditions near the soil surface that are inimical to tree germination and early growth and survivorship. For example, understory layers that generate a thick litter layer may inhibit germination of small-seeded species (Farris-Lopez and others 2004, George and Bazzaz 1999a), whereas those that strongly preempt light can preclude the establishment of many shade-intolerant and intermediately tolerant species (de la Cretaz and Kelty 2002, Gonzalez and others 2002, Horsley 1993a). These dense layers may substantially suppress tree recruitment by a combination of at least six different types of interference mechanisms (Table 1). Consequently, only a few tree species may possess the necessary traits to persist under, and eventually emerge through, this understory layer to constitute the advance regeneration layer (Connell 1990, Runkle 1990). If so, then the species composition of the advance regeneration layer and subsequent successional dynamics will contrast sharply in forests with a recalcitrant understory layer vs. one without.

Mechanisms of Interference Over Tree Establishment, Survival, and Growth

This section describes different ways that a dense understory canopy can suppress regeneration. Because most studies fail to distinguish among these mechanisms, Muller (1969) proposed the term interference to describe the suppression of one species or layer on another species. In the following sections, we briefly review the literature to evaluate the evidence for six different mechanisms of interference between the understory layer and co-occurring

tree species. These mechanisms include (1) resource competition, (2) allelopathy, (3) physically impeding seedling germination and growth, (4) through modifications of inter-specific interactions (Figure 2). We suggest that the most efficient and cost-effective remediation of the deleterious effects of these recalcitrant understory layers will require a greater understanding of how these layers alter patterns of forest regeneration and succession.

Resource Competition

In forested systems, perhaps the most prominent interference mechanism exerted by a recalcitrant understory layer would be direct competition for above- and belowground resources. In closed-canopy forests, dense understories exacerbate the degree of light attenuation caused by the midstory and canopy (Beckage and others 2000, de la Cretaz and Kely 2002, Messier and others 1998, Nilsen and others 2001). Photosynthetically active radiation (PAR) levels can drop well below 5 percent of full sun beneath these layers (Aubin and others 2000, Clinton and Vose 1996, George and Bazzaz 1999a, Hill 1996, Horsley 1993a, Kelly and Canham 1992, Lei and others 2002, Lusk 2001, Nakashizuka 1987, Wada 1993, Walker 1994). Additionally, these dense, low canopies can reduce light quality (e.g., red: far-red wavelengths), thereby preventing germination, altering internode elongation, and inhibiting flowering (Horsley 1993a, Mancinelli 1994, Messier and others 1989). Furthermore, dense, low canopies decrease the availability of sunflecks particularly for seedlings (Denslow and others 1991, Lei and others 2002, Nilsen and others 2001). Finally, if canopy gaps do form, they may not operate as gaps at all if seedlings remain trapped beneath a dense understory layer (Beckage and others 2000, Lusk 2001, Webb and Scanga 2001). Under this scenario, regeneration may be limited to only a few individuals of those few species that are highly shade tolerant.

Dense understories may also exacerbate belowground competition (Dillenburg and others 1993, Messier 1993, Putz and Canham 1992). Some studies infer resource limitation by detecting increased growth or survival of target plants following fertilization or measuring lower nutrient and water concentrations in soil beneath dense understory

cover vs. more open areas (e.g., Inderjit and Mallik 1996, Messier 1993, Nilsen and others 2001, Yamasaki and others 1998). Similarly, vine-covered saplings often have lower foliar nitrogen levels, reduced preleaf water potential, and decreased diameter growth when compared to vine-free saplings (Dillenburg and others 1993, Perez-Salicrup and Barker 2000). The above studies are suggestive of resource limitation though they typically do not distinguish between competition for water vs. soil nutrients. Because nutrient and water availability covary, decoupling these two factors is difficult (Casper and Jackson 1997, Nambiar and Sands 1993). Additionally, few experiments use factorial manipulations to disentangle a dense understory layer's aboveground vs. belowground effects and their interactions (McPhee and Aarssen 2001).

Horsley (1993a) experimentally tested the influence of aboveground vs. belowground competition. He tied back hay-scented fern fronds while leaving their roots and rhizomes intact, thereby reducing light competition and isolated seedlings within PVC tubes, thereby reducing root competition. He found that light attenuation, not belowground competition, was the mechanism of interference (Horsley 1977, 1993a, 1993b). Putz and Canham (1992) conducted similar aboveground and belowground manipulations. They found that a dense shrubby understory layer reduced tree regeneration primarily because of belowground competition (see also Christy 1986), although this varied with soil fertility. Belowground competition was more important in infertile sites, whereas aboveground competition was more important in fertile sites. Clearly well-replicated factorial experiments are required to ascertain the relative importance of aboveground vs. belowground competition, although other processes may confound the results of these experiments (e.g., allelopathy; see section on allelopathy).

Allelopathy

This section discusses the potential effects of the phenomenon called allelopathy: i.e., the inhibition of growth or survivorship of one plant species by chemicals produced by another species. Direct field evidence for allelopathy remains equivocal and elusive. In forests that have dense

understories dominated by ericaceous shrubs, the phenolics and other phytochemical compounds produced by these shrubs can disrupt nitrogen mineralization and inhibit ectomycorrhizal fungi; this significantly reduces conifer growth and survivorship (Walker and others 1999; reviewed by Mallik 1995, 2003 and Wardle and others 1998). In these systems, Nilsson (1994) used factorial manipulations of aboveground and belowground competition and allelopathy to identify how the boreal shrub *Empetrum hermaphroditum* suppressed tree regeneration. She found that both belowground competition and allelopathy were important, but that belowground competition played the primary role. Similarly, Jäderlund and others (1997) found that *Vaccinium myrtillus* interfered with Norway Spruce (*Picea abies*) primarily through belowground competition. In forests where ferns form dense understories, bioassays and greenhouse studies have suggested the potential for strong allelopathic effects on tree regeneration (Gliessman and Muller 1972, 1978; Horsley 1977); however, further field experimentation failed to find strong allelopathic effects (Den Ouden 2000, Dolling 1996, Horsley 1993b, Nilsen and others 1999). Despite these results, too few studies have tried to experimentally disentangle resource competition from allelopathy via field experiments. Future research must move beyond merely documenting the existence of phytotoxic exudates in greenhouse and laboratory studies (Fuerst and Putnam 1983, Inderjit and Callaway 2003, Weidenhamer 1996, Williamson 1990).

Seed Predation

This section discusses how a dense understory layer can increase the activity of small mammals, thereby increasing the rate and impact of seed and seedling predation (Den Ouden 2000, George and Bazzaz 1999a, Gliessman 1978, Schreiner and others 2000, Wada 1993). This can create a situation where it appears that low seedling densities are caused by resource competition (e.g., light attenuation) when, in fact, they are caused by seed and seedling predation (Connell 1990; Holt 1977, 1984). Connell (1990) defined this as a type of apparent competition (*sensu* Holt 1977, 1984). Experiments that use canopy removals confound the direct competitive release caused by the removal of the

understory layer with the indirect benefits of removing this layer, particularly the decrease in seed and seed predation by small mammals (Reader 1993). Even though small mammals are abundant, forage preferentially beneath dense vegetative cover, and consume copious quantities of seeds, few experiments have attempted to evaluate the role of seed or seedling predators vs. resource competition. Nonetheless, long-term studies in other plant systems have documented that selective seed and seedling predation can lead to rapid changes in plant community composition (e.g., Brown and Heske 1990, Gill and Marks 1991, Howe and Brown 2001, Ostfeld and Canham 1993).

Litter Accumulation

A thick litter layer typically reduces plant species diversity and density through a wide variety of direct and indirect mechanisms (see Facelli and Pickett 1991). For example, George and Bazzaz (1999a) found that a thick fern litter layer directly limited the establishment of small-seeded tree species (see also Beckage and others 2000, Farris-Lopez and others 2004, Lei and others 2002, Veblen 1982). Alternatively, in boreal forests, the insulative properties of a dense grass litter layer results in decreased soil nitrogen mineralization, water uptake, and seedling photosynthetic rates, thus indirectly diminishing conifer growth and survival (Cater and Chapin 2000, Hogg and Lieffers 1991, Lieffers and others 1993). Aside from these examples, there are few experimental tests that unravel the many facets of litter interference or evaluate its importance relative to other mechanisms (e.g., resource competition). However, in forests characterized by a recalcitrant understory litter layer, it is clear that this alternative remains a viable and potentially important mechanism.

Mechanical Interference

A dense understory layer can reduce tree seedling regeneration via non-competitive, physical interference. Clark and Clark (1991) demonstrated that the passive shedding of branches and leaves of subcanopy palms smothered seedlings present in the understory. Similarly, collapsing *Guadua* bamboo culms can reduce tree seedling growth and survival (Griscom and Ashton 2003). Additionally,

the physical weight of a large liana load may suppress tree seedling and sapling growth (Gerwing 2001, Putz 1991, Schnitzer and others 2004). If tree species respond differentially to these physical stresses, then this mechanism alone can potentially alter understory tree species composition and modify future successional trajectories (e.g., Gillman and others 2003, Guariguata 1998).

The Relationship Between Mechanisms of Interference and Phenology

This section discusses how the intensity and duration of any particular interference mechanism can vary temporally as a result of the species' life history, whether evergreen, deciduous, or monocarpic. In fact, this trait may provide clues to understand both the strength and type of interference. For example, evergreen species may pose a greater impediment to tree regeneration as their effects are exerted throughout the year on all tree seedling life-history transitions (Givnish 2002). In contrast, herbaceous perennials that senesce in the fall or deciduous shrubby species only exert competitive effects during the growing season (e.g., de la Cretaz and Kelty 2002, Nilsen and others 2001). This delayed expansion of the recalcitrant understory layer provides a brief window of opportunity for evergreen tree species, species with early germination (e.g., *Acer rubrum* L.), or species with early leaf expansion (e.g., *Betula lenta* L.) to overcome the understory stratum's deleterious effects on early establishment. This temporal advantage can provide sufficient photosynthetic and growth opportunity to enable trees to survive and eventually grow through a fern layer (de la Cretaz and Kelty 2002). Additionally, if the intensity of seed and seedling predation decreases with senescence of the low canopy, then the impact of pervasive seed predation may decrease in the fall. This timing of senescence may generate increased predation on early seed dispersers (e.g., *Quercus* spp.) relative to later dispersers (e.g., *A. saccharum* Marsh., *Fagus grandifolia* Ehrh.).

Causes and Consequences of a Recalcitrant Understory Layer

In this section, we discuss our contention that the expansion and monopolization of the understory by a narrow set

of plant species is often an inadvertent outcome of policies and management decisions that deviate from natural forest overstory disturbance, fire, and herbivory regimes. We propose a general conceptual model through which alterations in the dynamics of the overstory, understory, or both generate increases in a select few understory plant species (Figure 1). These alterations involve changes in the frequency and scale of overstory disturbance, increased or decreased fire frequency, or increased herbivory that release a restricted set of understory species from prior competitive constraints. Once released, these species increase dramatically in abundance and cover over large portions of the forested landscape (Table 1). Following its establishment, this recalcitrant understory layer interferes with tree regeneration through a variety of direct and indirect mechanisms including above- and belowground competition, allelopathy, microhabitat-mediated seed/seedling predation, litter, and mechanical damage. Consequently, this recalcitrant layer itself inhibits tree regeneration and strongly influences which tree species establish and survive beneath its canopy (e.g., Cater and Chapin 2000, Clinton and others 1994, Dolling 1996, Veblen 1982). The strength and selectivity of this filter can retard succession, alter the tree species participating in the successional sere, or potentially arrest succession.

We found only 25 percent of the published studies reviewed the reported results of manipulative field experiments designed to identify the existence of one or more particular interference mechanism(s) (Table 1). Above- and belowground competition and allelopathy were the predominant mechanisms tested (37, 32, and 13 percent, respectively; Table 1). Apart from competition and allelopathy, various interference mechanisms were speculated on in many papers, but few, if any, were tested experimentally. Given the paucity of information for most systems, we lack the information needed to clearly establish by which mechanism a recalcitrant understory layer inhibits tree regeneration (see Levine and others 2003 for a similar conclusion on exotic invasives).

We argue that a move towards a more mechanistic understanding of the "interference" phenomenon could begin by considering the most limiting resource(s) within a given system. For example, on a coarse scale, forested

ecosystems differ in the identity of the most limiting resource(s) (e.g., light, soil nutrients, and water), and these differences could provide insight into the most plausible interference mechanism. Boreal and cool-temperate forests are typically nutrient poor (primarily N) and less light limited relative to their temperate and tropical counterparts (Attiwill and Adams 1993, Kimmins 1996, Krause and others 1978, Reich and others 1997) (reviewed by Coomes and Grubb 2000 and Ricard and others 2003). We found that dense low canopies in these forest types suppress regeneration directly via belowground competition and indirectly via allelopathic interactions that mediate resource availability and uptake (Table 1; Christy 1986, Jäderlund and others 1997, Nilsson 1994). In contrast, temperate deciduous and tropical rain forests tend to be more light limited (Finzi and Canham 2000, Pacala and others 1994, Ricard and others 2003). In these systems, we found that other mechanisms including aboveground competition and seed predation were generally more important than belowground competition (Table 1; Den Ouden 2000, Denslow and others 1991, Horsley 1993a). Ideally, the best tests would link a series of carefully controlled laboratory or greenhouse studies with field experimentation in order to identify which mechanisms merit further investigation. Furthermore, we strongly argue that manipulative field experiments remain among the best tools to test the relative importance of each factor independently as well as any interactions.

Floristic Diversity and Forest Succession

The increasingly common development of recalcitrant understory layers worldwide plays a strong, yet vastly under-appreciated role in determining future successional patterns, forest composition, and diversity because of their tendency to selectively suppress tree regeneration. Indeed, studies examining the regeneration success of a variety of tree species demonstrate that a majority of tree species suffer decreased seedling densities and limited height growth underneath recalcitrant understory canopies (e.g., de la Cre-taz and Kelty 2002; George and Bazzaz 1999a, 1999b; Hille Ris Lambers and Clark 2003; Horsley and Marquis 1983). The presence of this additional filter on floristic diversity in forest understories together with increased herbivory and

altered fire regimes strongly restricts the number of species that can successfully regenerate. The potential consequences of these ecological filters (*sensu* George and Bazzaz 1999a, 1999b) on species composition remains poorly understood. Nevertheless, we suggest that floristic diversity in such areas is so severely constricted that succession may move steadily toward monodominance or complete regeneration failure. These extreme cases include the fern-and grass-covered orchard stands in Pennsylvania where 50- to 80-year-old failed clearcuts remain devoid of tree regeneration (Horsley 1985) or bracken-covered tropical regions of Central America that have persisted for centuries following forest removal (Den Ouden 2000).

Forest Dynamics Models

Computer-based forest successional models (e.g., JABOWA-FORET [Shugart and West 1977, Smith and Urban 1988] and SORTIE [Pacala and others 1994]) remain the best tool to explore long-term successional outcomes; however, forest dynamics models typically fail to include a dense understory layer's impact on early seedling survival and growth. For example, in the original SORTIE calibrations, the growth and mortality parameters derived from saplings (15 to 750 cm in height) are applied to small seedlings as well (Kobe and others 1995, Pacala and others 1994). Additionally, the authors acknowledge their recruitment parameter estimate is potentially unreliable as the survival of individuals < 5 years old is highly variable, and mortality is often intense (Pacala and others 1994). Indeed, researchers have documented that density dependent (e.g., Packer and Clay 2000) and density independent mortality can dramatically alter initial seedling distribution patterns, particularly under a dense understory layer (Hille Ris Lambers and Clark 2003, Schnurr and others 2004). By constraining the model and its parameters to the 5-or more-year-old age class, SORTIE assumes away part of the early dynamics that may occur low to the ground underneath a recalcitrant understory layer and help shape the composition sapling class.

As originally calibrated (Pacala and others 1994), SORTIE did not include the effects of a recalcitrant understory layer into its resource (light) submodel. More recent developments note that SORTIE can underestimate light

attenuation (Beaudet and others 2002) and the long-term development of shade-intolerant tree species following major disturbance (Tremblay and others 2005). It is suggested in both papers that this may be due to the lack of an understory layer component in the model, and it is stressed that this goal is an ongoing research focus (see also Aubin and others 2000 and Beaudet and others 2004). We know of only one effort that has integrated a recalcitrant understory layer into SORTIE. Hill (1996) incorporated hay-scented fern abundance as a function of light as well as hay-scented fern's impact on light availability as a function of frond density. With the increased light limitation imposed by fern cover, successional projections indicated faster reductions in shade-intolerant species abundance and an accelerated shift towards dominance by shade-tolerant species (Hill 1996). Nevertheless, none of the simulations containing a dense fern layer reflected the pattern of complete regeneration failure documented in the field (Hill 1996). We concur with Hill that the inconsistencies between model projections and observable field patterns likely result from overestimates in seedling growth and underestimates in seedling mortality inherent in SORTIE. We argue these inconsistencies are due to (1) ignoring the early (fewer than 5 years) seedling dynamics, and (2) failure to incorporate additional interference mechanisms causing seedling mortality (e.g., seed and seedling predation) beyond light competition.

Forest Management

Understanding the autoecological characteristics of interfering plant species may allow land managers to preemptively limit the aggressive spread of these species as well as provide alternative options for their control. We found that alterations in forest canopy disturbance, fire, and herbivory regimes may lead to the establishment of recalcitrant understory layers, particularly when alterations to the overstory and understory disturbance regimes occur in tandem (e.g., Payette and Delwaide 2003). We suggest managers monitor overstory and understory conditions to ensure that modifications to either of these strata do not operate concurrently in an effort to mitigate invasion risk. Furthermore, if control over overstory and understory factors falls under the purview of different agencies (e.g., wild game vs.

forestry management agencies), then communication and coordination between them is essential in order to minimize the chance of concurrent or overlapping disturbance events. We caution that decisions regarding partial or complete overstory removals should consider the site's understory conditions including inadequate advance regeneration, presence of clonal understory plants, and high herbivore impact (e.g., Marquis and others 1990). We further suggest the implementation of management practices that more closely resemble natural disturbance levels.

Knowledge of a species life-history traits and interference mechanisms may also provide managers with alternative treatments to promote tree regeneration when conventional treatments like herbicides are not desired or permitted (Berkowitz and others 1995). For example, mowing or cutting of ferns, grasses, and shrubby interfering vegetation may successfully ameliorate their aboveground competitive effects and enhance regeneration (Biring and others 2003, Davies 1985, Marrs and others 1998). Alternatively, if belowground competition is the major interference mechanism, fertilizer application may mitigate the competitive effects of interfering plants and promote tree regeneration (Haywood and others 2003, Prescott and others 1993). Additional remediation techniques tailored to other interference mechanisms could include direct seeding of propagules coated with small mammal repellent (Campbell 1981, Nolte and Barnett 2000), soil scarification or controlled burning to reduce litter interference (Nyland 2002), and activated carbon as a treatment to mitigate allelopathic interference (Jäderlund and others 1997). A basic understanding of possible successional outcomes following the establishment of a low canopy may further aid land managers. In areas where the low canopy simply stalls succession, successful regeneration will ultimately occur without any silvicultural techniques. Finally, where the recalcitrant understory layer filters tree species composition or arrests succession, managers could manipulate the rate and direction of regeneration by underplanting tree species relatively unaffected by the interfering layer (e.g., shade-tolerant species) in order to attain a desirable and diverse mixture of regeneration species (Löf 2000).

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Methods to Assess Landscape-Scale Risk of Bark Beetle Infestation to Support Forest Management Decisions

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Abstract

The objective of our paper is to provide practitioners with suggestions on how to select appropriate methods for risk assessment of bark beetle infestations at the landscape scale in order to support their particular management decisions and to motivate researchers to refine novel risk assessment methods. Methods developed to assist and inform management decisions for risk assessment of bark beetle infestations at the landscape scale have been diverse, ranging from simple empirical correlations to complex systems models. These approaches have examined different bark beetle species, forest types and systems, and management questions, and they differ in spatial and temporal precision, the types of processes included, and the form of output. Bark beetle risk assessment methods, however, share a common theme: they aim to quantify expected levels of attack and loss due to beetles. By focusing on this commonality, we present a gradient in which methods can be classified and ranked, ranging from more structural, pattern-oriented methods to more functional, process-oriented methods. Our objective is to describe a framework for comparing methods in terms of how risk is represented and in terms of the complexity of application. To illustrate how diverse methods can be cast within a common frame of reference, we describe and provide brief examples of four types of methods that we have used in British Columbia, Canada, to examine landscape-scale risk of mountain pine beetle attack in lodgepole pine

forests. We then provide some guidance on how to select an appropriate method for a given system and set of questions. The most appropriate method is the simplest one that can address the questions, minimize uncertainty, and inform the decision process in the required timeframe. It is important that researchers and practitioners can view bark beetle risk-assessment methods as a toolkit and select appropriate tools for a given task, as no single method is best for all situations.

Introduction

A variety of risk assessment tools have been designed to help managers quantify expected losses from bark beetles, losses which can be quite severe. We present a framework to help managers choose the tools that are most appropriate to their needs.

Several bark beetle species, mostly in the family Curculionidae, subfamily Scolytinae, have the potential for dramatic population increases under favorable forest and climate conditions, which can result in landscape-scale mortality to the host tree species (e.g., Wood and Unger 1996). For example, the mountain pine beetle (MPB, *Dendroctonus ponderosae* Hopk.) has killed much of the mature lodgepole pine (*Pinus contorta* Dougl. ex. Loud.) over an area of approximately 9 million hectares in British Columbia in recent years (Westfall 2005), and the spruce beetle (*D. rufipennis*) has infested several hundred thousand hectares of spruce forest in southwestern Yukon, Canada (R. Garbutt, pers. comm.). These events have widespread implications for current and future forest management, ranging from effects on timber supply and operations, wildfire-urban interface, wildlife habitat, and aesthetics.

Landscape-scale risk assessment of bark beetle infestation aims to quantify the spatial and temporal likelihood of attack extent and severity. Methods to assess risk can range from structural risk (i.e., strictly assessing patterns) to functional risk (i.e., assessing interactions and feedbacks between pattern and process). Susceptibility and risk rating systems (Bentz and others 1993) generally classify stands

without a temporal dimension in a relatively simply spatial context and are useful for a quick overview of landscape state and general patterns. Landscape connectivity methods (O'Brien and others 2006) can join stands of higher susceptibility into a network that can help give an integrated landscape perspective but are still temporally static. Connectivity assessments have been useful in areas with limited current attack to provide an assessment of the spatial pattern of hosts and likely pathways of attack. Empirical projection models explicitly model outbreak dynamics based on historical temporal patterns and have been useful to assess very broad-scale dynamics and potential interactions with management (Eng and others 2005). Population models capture system dynamics and feedback between host patterns and beetle demographics in detail (Dunning and others 1995) and can help to gain insight into likely trends of outbreak development and to explore management alternatives in relatively fine detail.

These methods, and others in the literature, share a common theme: they aim to quantify expected levels of attack and loss due to beetles. Although the way in which risk is quantified may differ among methods, risk can always be cast in terms of a probability distribution that represents likelihood of losses or impacts. This provides a common frame of reference with which methods can be compared on a gradient from simpler, structure-based methods to more complex, process-based methods, allowing methods to be assessed in terms of tradeoffs between data required, difficulty of application, and precision of results. To demonstrate this framework, we examine a diverse suite of tools useful to assess risk of bark beetle attack at broad spatial scales. For each, we provide an overview of the method as applied to bark beetle infestation risk, with a focus on data requirements and outputs. We highlight the pros and cons and outline the types of management questions that can be addressed and present an example of management application.

Management Implications of Bark Beetle Infestations

There are three main management strategies for major bark beetles in forestry: prevention, direct control, and salvage

(Shore and others 2006b). Preventive management is used when beetles are at, or below, endemic levels, and managers have the opportunity to be proactive in making trees, stands, and landscapes less susceptible to large infestations (Whitehead and others 2006). Direct control is used in the situation when an infestation is underway and management efforts are reactive and primarily directed at killing beetles in order to reduce population size and spread (Carroll and others 2006). Salvage occurs either during or following those outbreaks that are too large for effective control (Eng and others 2005).

Bark beetle management requires decision-support tools to provide information on which to base decisions, including identification of infested trees and susceptible stands. Many decisions are involved in resource allocation including budgets, risk to surrounding stands, access to the infested trees, other resource constraints, allowable harvest levels, and treatment efficacy to name a few. Landscape-scale risk assessment can provide information to support some of these decisions.

What Is Risk Assessment in This Context?

Risk can be defined as the likelihood of an undesirable outcome combined with the magnitude of impact. Events with low likelihood of occurrence (e.g., meteorite impact) are not classified as high risk, nor are events with minor negative impacts (e.g., attack by secondary bark beetles on weaker, smaller trees in a stand). We define landscape-scale risk of bark beetle infestation as the probability of a given magnitude of loss of standing timber due to attack and concurrent management response. That is, conceptually, it is a probability distribution that quantifies the potential of attack at broad spatial scales (Figure 1). The specific shape and expected values of this distribution will depend on the landscape under study (e.g., the configuration and composition of hosts and beetles), and the method used to assess risk (e.g., the quantity reflected by the method, such as proportion of stand volume at risk or proportion of stands that may be attacked within a given timeframe). In general, this distribution cannot be fully mapped owing to uncertainties in future events (e.g., weather) and data (e.g., infestation locations and severity). In addition, management

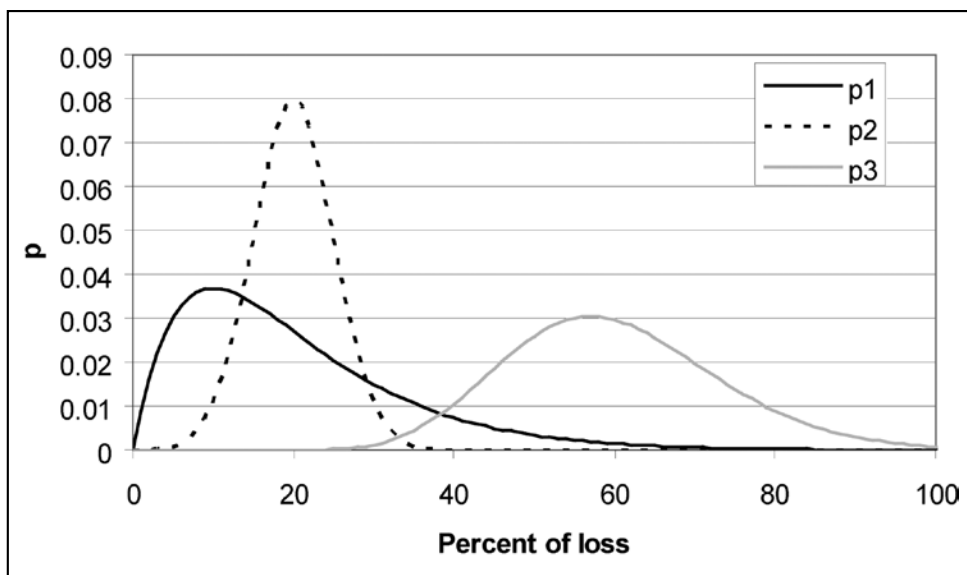


Figure 1—Conceptual diagram of landscape-scale risk of bark beetle infestation. Landscape-scale risk of bark beetle infestation can be conceptualized as a probability distribution of loss of standing timber owing to attack and management response (where probability of loss within a given range is the area under the probability density function curve, p , within the specified range). Loss may be quantified as area attacked, volume killed, or volume lost owing to decay. Variance about the mean may be due to the source data (e.g., if the distribution was formed from frequency information) or uncertainty in future processes (e.g., weather). Three example risk distributions are shown. The first two ($p1$ and $p2$) have lower overall risk relative to the third ($p3$). Based on a 67 percent confidence interval, $p1$ has an expected loss of between 3 and 24 percent (mean of 20 percent), $p2$ has an expected loss of between 16 and 24 percent (mean of 20 percent), and $p3$ has an expected loss of between 45 and 70 percent (mean of 59 percent).

decisions influence risk, so each management option results in a different risk probability distribution. Often, however, risk focuses mostly on the mean, or expected, value of the distribution (Fall and others 2004). Hence, a high-risk scenario is one with a high probability of large levels of loss (e.g., example distribution $p3$ in Figure 1). A medium-risk scenario may be due to a high probability of medium levels of loss (e.g., example distribution $p2$ in Figure 1), or a medium probability of high levels of loss (e.g., example distribution $p1$ in Figure 1) (Shore and Safranyik 1992).

Why Landscape-Scale Risk Assessment Is Needed

Forest management decisions in the context of potential or existing bark beetle infestations require practical information in a timely manner (MacLauchlan and Books 1994, Safranyik and others 1974). Forest managers want to know the most likely outcomes of a range of alternative choices, both to help select an option and to communicate rationale.

When a landscape has no current infestation, managers want to know the likelihood of an infestation starting (e.g., if one is imminent or a more remote possibility), and which stands are most likely to be affected first. In landscapes with a current infestation, managers want to know the impact of different types and levels of management effort (e.g., harvest levels, fell and burn treatment levels, global positioning system [GPS] surveys), on the timing, location, and magnitude of loss (Shore and others 2006b). In essence, the choices involve appropriate resource allocation and how to deal with uncertainty in order to assess tradeoffs and costs/benefits. Landscape-scale risk assessment provides key information to support this decision process.

A key role of researchers providing decision support is to help decisionmakers frame their specific questions in terms that can be addressed using risk assessment methods. Getting at the fundamental question helps to identify the best methods to apply that can balance information desires with the limitations of data availability, system knowledge,

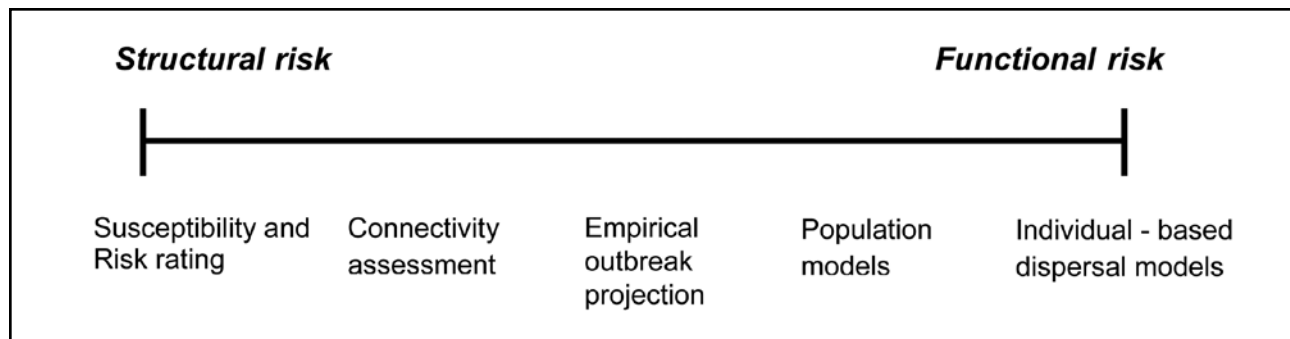


Figure 2—Gradient of landscape-scale methods for assessing risk of bark beetle attack. Landscape-scale methods for assessing risk of bark beetles can be placed on a gradient from structural to functional risk. Structural risk methods focus on patterns, such as stand susceptibility or connectivity networks. Functional risk methods focus on process, such as demographics and movement.

and timeframe. Researchers also need to recast risk assessment results back into a language and format that can be communicated to decisionmakers in a comprehensible and useful manner (Gustafson and others 2005).

Risk-Assessment Methods

This section provides information and example applications for categorizing risk methods, susceptibility/risk rating systems, graph-based connectivity assessments, empirical outbreak projections, population modeling, and other risk-assessment methods. The examples provided in this section illustrate how we categorize risk methods. These examples are based on a range of tools we have used to assess potential impacts of MPB at scales from stands to the entire province of British Columbia. Although the specific details of the methods differ substantially, they are essentially just different approaches to assessing risk. They differ fundamentally in terms of the degree to which ecological and management processes are taken into account and can be viewed along a gradient (Figure 2). **Structural** approaches to risk focus mostly on landscape patterns and correlations between past outbreak behaviour and stand structure, whereas **functional** approaches focus on underlying processes and interactions in the system (cf. distinction between structural and functional habitat connectivity, Taylor and others 1993).

Susceptibility/Risk-Rating Systems

Susceptibility and risk rating systems classify each stand or location in a landscape according to local characteristics

(e.g., forest age, distance to nearest attack). We define susceptibility in terms of conditions inherent to a stand (i.e., how suitable is the stand for the beetle species) and risk rating as a function of susceptibility and beetle pressure (Shore and Safranyik 1992, Shore and others 2006a). Stand susceptibility rating systems provide the forest manager with a tool that identifies the likelihood of damage to a stand should a beetle infestation occur in it. When implemented on a map, these tools form the starting point for setting management priorities by identifying stands and landscapes that are more susceptible than others. When combined with maps of beetle locations a manager can look at the risk of loss. Highly susceptible stands in closer proximity to large numbers of beetles can be given management priority.

Data Requirements—

To apply an existing rating system requires basic digital spatial forest cover data on attributes such as stand age, basal area, percentage of host and other tree species. Data on the location of each stand (elevation, latitude, longitude) and infested tree locations are also required. To develop a rating system, however, requires substantial fieldwork to identify correlations between beetle biology and stand characteristics (e.g., Perkins and Roberts 2003).

Output—

The main output is a spatial map of relative or absolute susceptibility rating, defined as the likely proportion of susceptible volume that would be killed **if** beetles arrived in the stand. Structural risk rating systems output a spatial map of

the estimated loss and likelihood of attack based on proximity to existing attack. These maps can be cast as frequency distributions, creating a form consistent with Figure 1.

Pros and Cons—

This method has the distinct advantage of simplicity of application and minimal data requirements. The mountain pine beetle susceptibility and risk rating system (Shore and Safranyik 1992; Shore and others 2000, 2006a) remains one of the most widely used MPB landscape risk assessment tools. On the other hand, these approaches are temporally static. Likely pathways cannot be identified with this method, and there is a limited ability to incorporate management. Susceptibility and risk rating have a limited capacity to use spatial information of known outbreak locations. In general, distances are simply Euclidean (i.e., do not account for direction or intervening land types), and the magnitude of pressure from nearby outbreaks is difficult to incorporate.

Management Application—

Susceptibility rating systems are best used to help prioritize harvest in landscapes without current or imminent attack (preventative management). Risk-rating systems are designed for landscapes with some attack but are best used in the early stages of an outbreak and are of limited use during epidemics (Shore and others 2006a).

Example Application—

We contrasted two landscapes of approximately equal size (Figure 3): Nadina Forest District in west-central British Columbia is approximately 3.0 million ha in size, and Dawson Creek timber supply area (along with a portion of Tree Farm License 48) is approximately 2.6 million ha. The general pattern of susceptibility differs substantially in these two landscapes (Figure 4). Nadina has larger areas of high susceptibility and more continuous cover, whereas Dawson Creek has overall lower susceptibility, but with a substantial amount of moderate susceptibility in the southeast. In terms of the present outbreak, Nadina has experienced very high levels of impact and mortality, whereas Dawson Creek, being east of the Rocky Mountains, is more isolated from the predominate outbreak populations

in British Columbia, and population buildup is more recent (Eng and others 2005). We can quantify these differences as a frequency distribution of susceptibility (Figure 5). The percentage of forested area with very low susceptibility is higher in Dawson Creek than Nadina, whereas Nadina has more area with higher susceptibility rating for most cases above 25.

Graph-Based Connectivity Assessment

Examining the network of inter-connections between susceptible host patches can provide a broad perspective of landscape patterns. A variety of methods are available for assessing habitat connectivity (Gustafson 1998, Schumaker 1996, With and others 1997), which is defined as the degree to which a landscape facilitates or impedes movement of organisms among habitat patches (Taylor and others 1993, Tischendorf and Fahrig 2000). Graph-based methods are emerging as an effective approach that supports multiscale analysis and that provides a good balance between field-intensive studies that aim to directly measure functional connectivity, but are limited to small areas and less vagile species (Tischendorf and Fahrig 2000) and pattern analysis methods that focus on structural connectivity (Calabrese and Fagan 2004, Urban and Keitt 2001). We have developed an extension to graph theory (Harary 1972) that we call **spatial graphs**, which captures features relevant to geospatial ecological analysis (Fall and others, in press; O'Brien and others 2006). Unlike conservation situations, management of bark beetles generally aims to reduce connectivity of host habitat. Hence, a key objective of connectivity analysis is to help identify where opportunities exist in a landscape to increase the level of fragmentation. This can be done by analyzing the spatial scales at which patches of susceptible hosts are well-connected, in particular, connected to areas with existing attack.

Data Requirements—

Digital spatial forest cover data and information on beetle movement and current infestation locations are required. Spatial graph connectivity assessment requires two spatial inputs: a patch layer (e.g., areas of high susceptibility) and a **cost surface** (e.g., relative difficulty or speed of movement or spread through different cover types). These require a

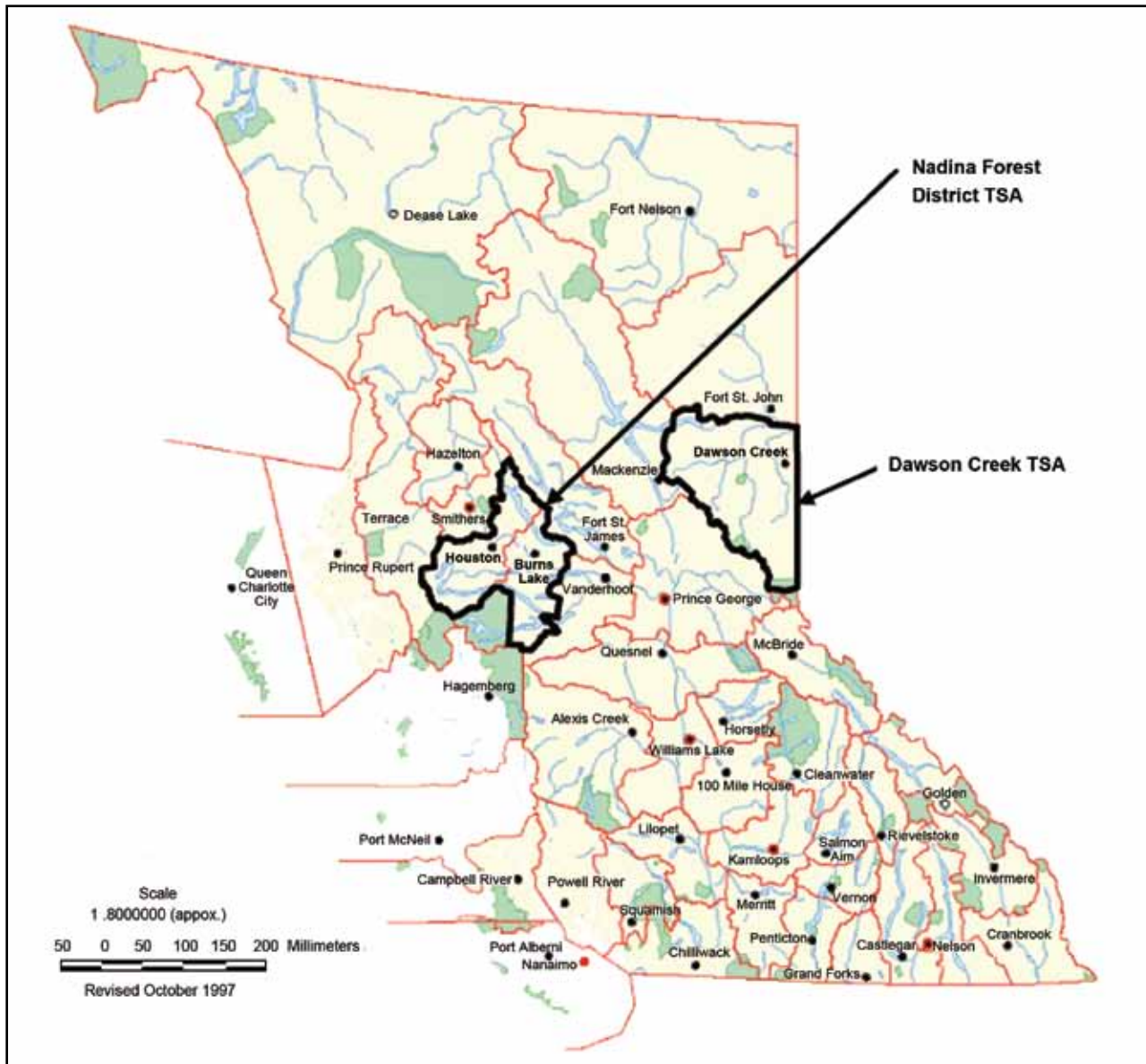


Figure 3—British Columbia, Canada, showing the Nadina Forest District (Lakes and Morice timber supply areas [TSA]) and Dawson Creek TSA.

susceptibility rating to define habitat patches and information on the permeability of different cover types in the matrix between patches (O'Brien and others 2006). Current infestation locations can be used to analyze the resulting connected network of patches in terms of effective distances from susceptible hosts to current attack.

Output—

The main output of graph-based connectivity analysis is a spatial graph that shows the location of connections

between host patches and the cost of those links, derived from the patch map and cost surface. Analysis of this graph can identify scales (effective distances) at which large increases in host connectivity occur—these can be examined spatially using the graph to draw connections at those scales. Given infestation locations, the graph can be reoriented to identify scales at which hosts become connected to current attack. The results can then be cast as a distribution of high susceptibility stands according

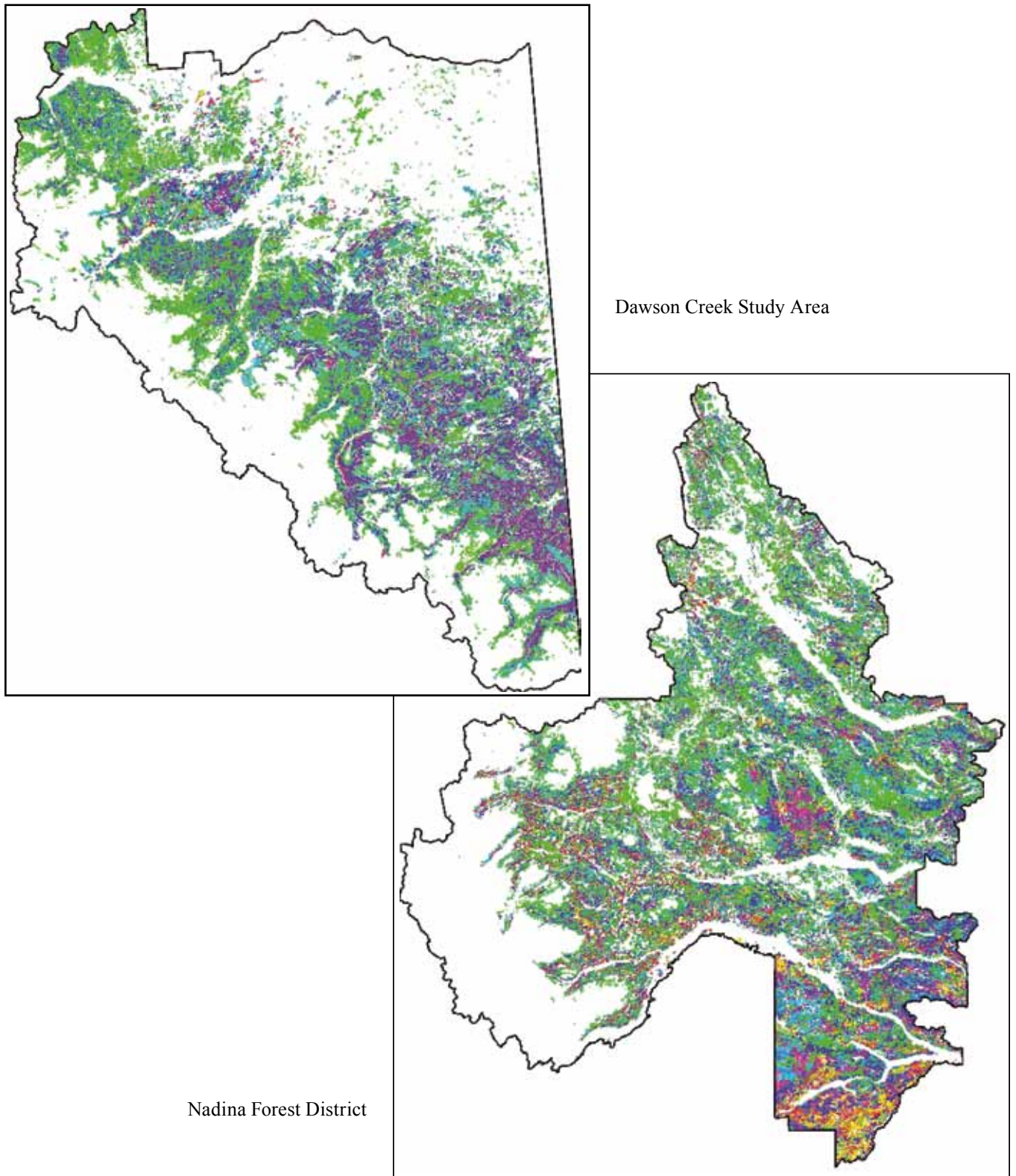


Figure 4—Susceptibility of stand loss owing to mountain pine beetle attack in the Nadina and Dawson Creek study areas (based on Shore and Safranyik 1992). The Nadina forest district (right) has high overall susceptibility and large contiguous areas of moderate to high susceptibility. Susceptibility in the Dawson Creek study area (left) is more moderate and fragmented.

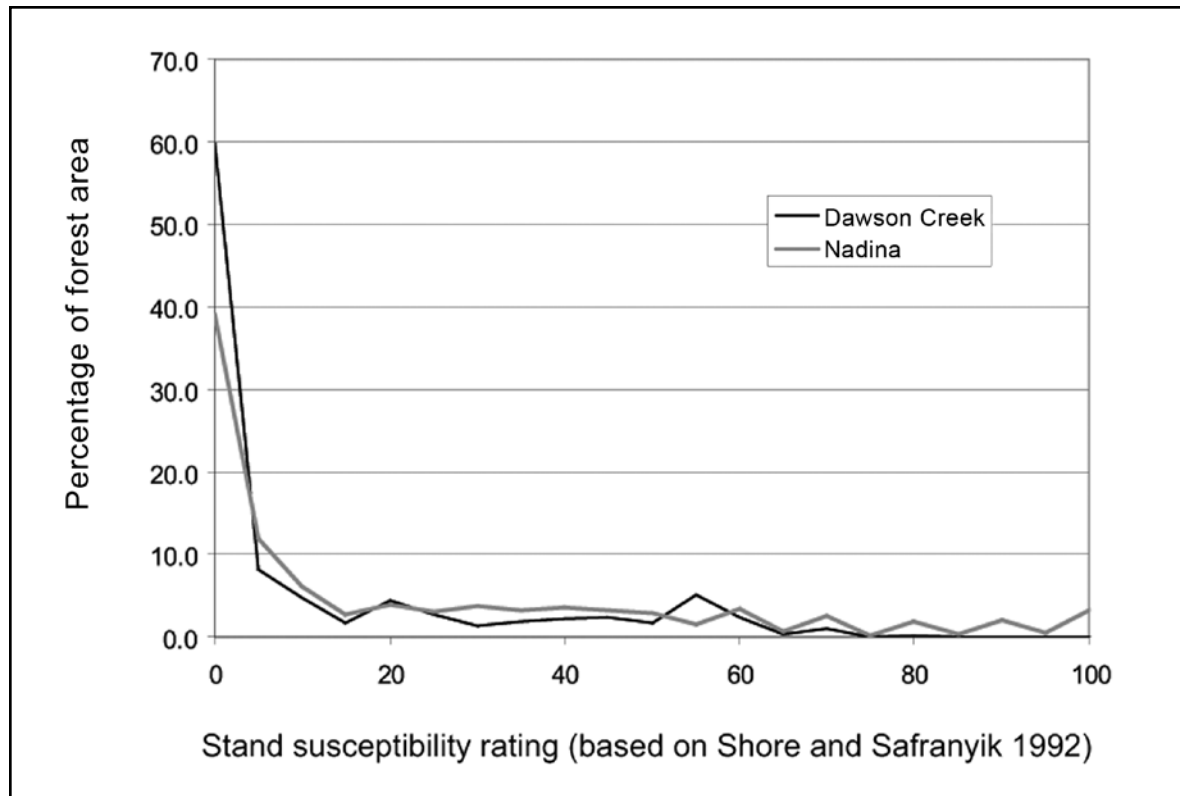


Figure 5—Frequency distribution of stand susceptibility rating for mountain pine beetle in Nadina and Dawson Creek study areas as percentage of forested area. Susceptibility was defined on a range from 0 to 100 as an index of potential basal area mortality according to Shore and Safranyik (1992) and Shore and others (2000). The Nadina distribution has a mean of 23.3 and a standard deviation of 28.8. The Dawson Creek distribution has a mean susceptibility of 12.9 and a standard deviation of 20.0.

to distance from current attack. One key benefit of using spatial graphs is the ability to translate from a graph back to the landscape, which is critical for effective communication and decision support.

Pros and Cons—

These methods required a low to moderate effort to apply, and data requirements are modest. The required information on movement cost/impedance can be challenging to parameterize with statistical confidence (O'Brien and others 2006), but cost surfaces derived using simpler methods, such as expert opinion, can be used for some less precise analysis. Although still static in nature, likely pathways can be identified as well-connected corridors, especially between areas with current attack and areas with clusters of high susceptibility. In addition, large areas can still be

processed efficiently. However, it provides a static perspective on a dynamic problem and has a limited ability to incorporate management.

Management Application—

Graph-based connectivity assessment is best used in situations with low or no attack. In a sense, this approach can be viewed as increasing the spatial dimension of susceptibility and risk-rating systems. The results can help to focus management effort on stands to reduce connectivity within management limits. The utility is based on the premise that the most important stands to treat in a landscape are not necessarily just the most susceptible, but the susceptible stands that are most connected. This is especially important in landscapes for which the area of susceptible stands greatly exceeds management capacity.

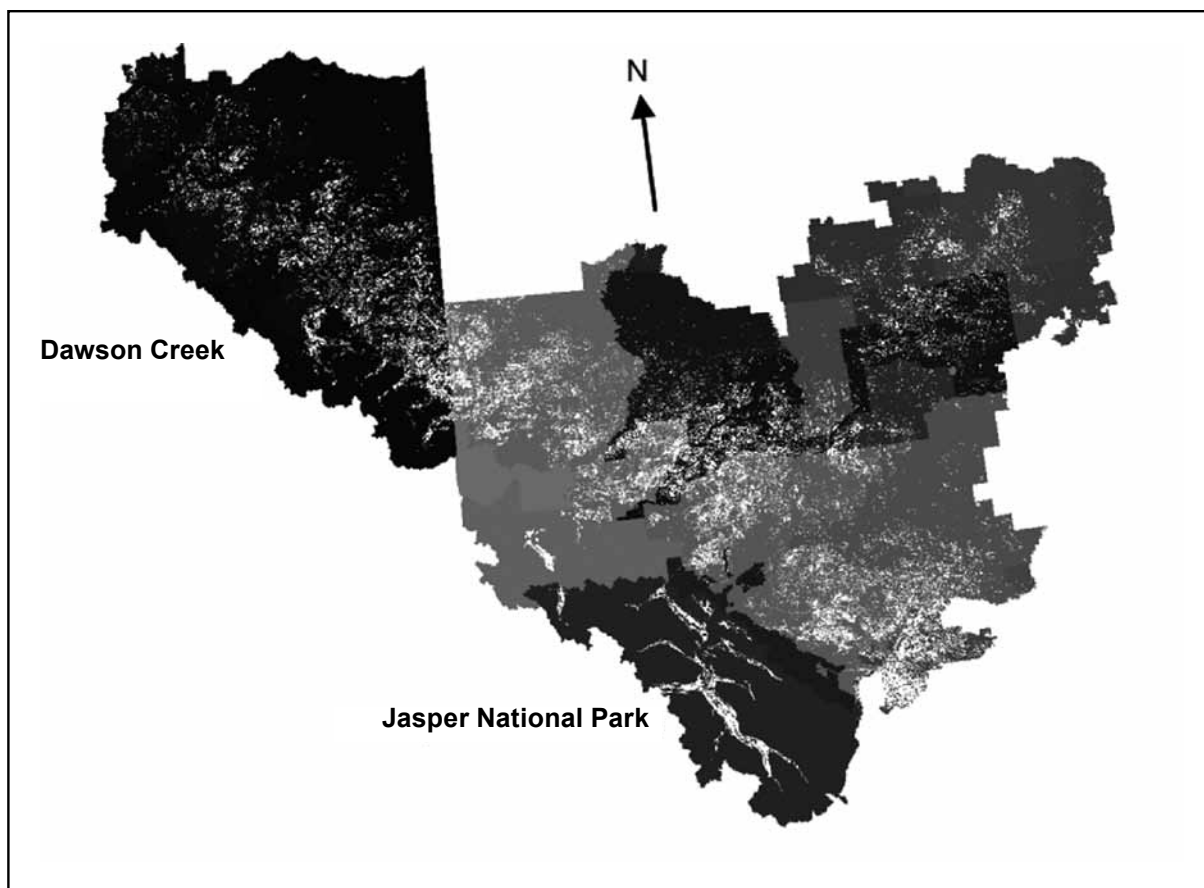


Figure 6—Patches with high susceptibility to mountain pine beetle attack and loss used for connectivity analysis. High susceptibility patches (those with greater than or equal to 65 on the Shore and Safranyik [1992] susceptibility rating system) for the Dawson Creek and west-central Alberta study area for which connectivity analysis was performed (white areas in the Figure). The area east of Dawson Creek and north of the Alberta portion of the study area is farmland. The southwest boundary is the Rocky Mountain divide. The different shades of grey in the underlying map denote different management units (national parks, provincial parks, timber management areas called forest management units in Alberta and timber supply areas or tree farm licenses in British Columbia).

Example Application—

We used spatial graphs to assess the potential of the current mountain pine beetle outbreak in British Columbia to spread into the boreal forest of northern Alberta. The study area consists of about 11.2 million ha, with Dawson Creek (Figure 3) on the western side and extending east to Slave Lake in central Alberta, north of Edmonton. Susceptibility was defined on a range from 0 to 100 according to Shore and Safranyik (1992). High susceptibility patches were defined as contiguous areas of susceptibility greater than, or equal to, 65 (Figure 6). The cost surface was produced using the reverse of susceptibility (i.e., increasing cost with decreasing susceptibility), based on the assumption

that beetles will spread more effectively through higher susceptibility stands, and that this increases linearly with susceptibility. The base graph extracted joined host patches into a network (Figure 7). This graph was reoriented, so that costs (which can be interpreted as effective distance), through the graph correlated with distance to the nearest infested patch.

This resulting graph was analyzed by thresholding: at each threshold from 0 to 600 km in units of 100 m (effective distance in cost units), all connections longer than the threshold were discarded, and only patches connected to current attack in Dawson Creek and western Alberta (mapped by heli GPS, M. Duthie-Holt, pers. comm.) were

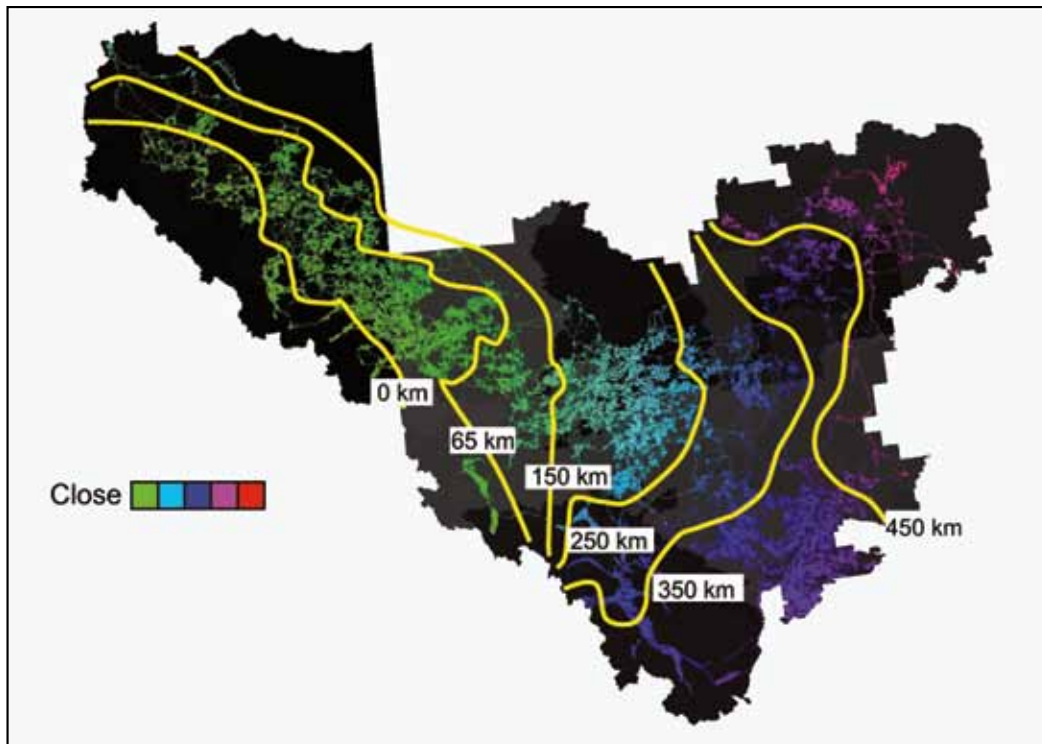


Figure 7—Isolines (of the results of graph thresholding. Mapping the results of graph thresholding in Dawson Creek/west Alberta study area by drawing isolines lines of equal effective distance from current attack) at the critical scales identified in Figure 8. Effective distance is the cost through the graph from infested patches to other patches.

retained (Figure 8), (Fall and others, in press). Hence, at a threshold of 0, only patches containing current attack were retained. As thresholds increase, the area of host joined to current attack increases until all host area is included. The pattern of these increases provides insight into pattern across spatial scale. Steep areas indicate scales with rapid increases in connectivity to current attack. These critical scales can then be cast back onto the original map of susceptible patches using isolines (Figure 7). Areas corresponding to more gradual increases in connectivity likely represent areas with higher likelihood that management can reduce connectivity, and, hence, risk in a timely manner. In this example, the areas corresponding to thresholds 65 to 150 km and 250 to 350 km appear to provide the best opportunity to reduce the risk of spread across this landscape.

Empirical Outbreak Projection

The development of methods for modeling and analyzing spatially and temporally autocorrelated data such as

the historical spread of a bark beetle outbreak across a heterogeneous landscape is a current and active area of research (e.g., Augustin and others 2007, Wikle 2003, Zhu and others 2005). However, appropriate statistical modeling techniques are not yet sufficiently well developed or disseminated for timely and practical application in many situations. Semi-Markov models are a standard method of projecting vegetation dynamics (Acevedo and others 1995, Baker 1989). Where data are abundant, the statistical challenge of modeling transition probabilities may be avoided by categorizing observations and using observed transition probabilities directly. Until robust statistical methods are available, this direct approach is reasonably well suited for practical and timely decision support. The approach can be extended to include multiple predictive factors in a probabilistic state-transition table. A time series of infestation progression is collected and categorized into infestation intensity classes. That time series is combined with spatial data about the physical environment. All of the data are

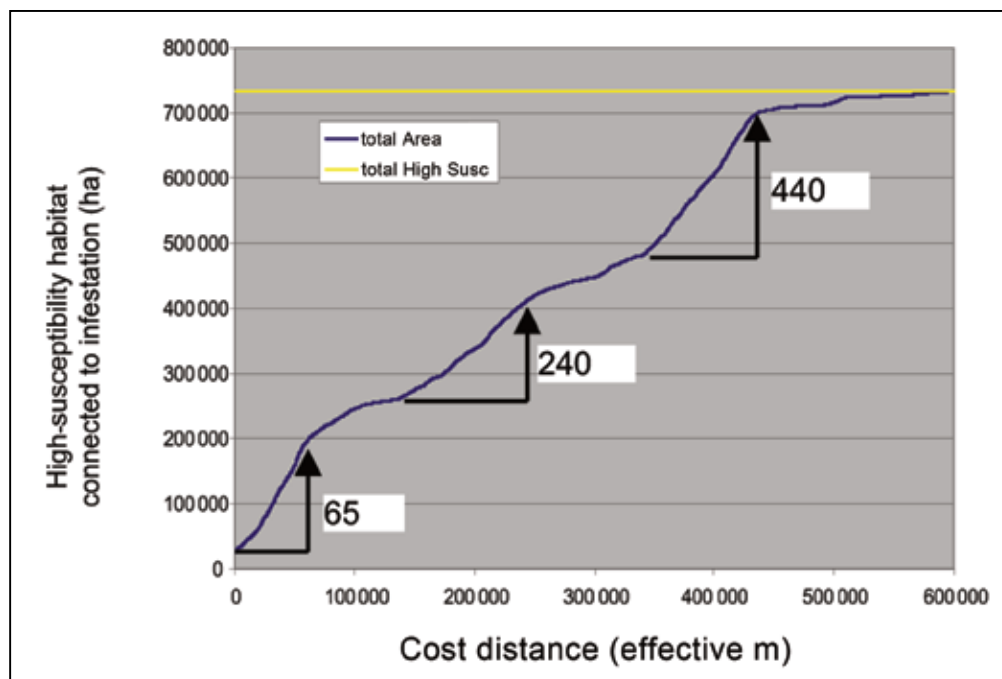


Figure 8—Results of graph thresholding in Dawson Creek/west Alberta study area. The total area of high susceptibility to mountain pine beetle attack and loss (greater than, or equal to 65, according to the Shore and Safranyik [1992] rating system) habitat is just over 700 000 ha (yellow line). The amount of high-susceptibility habitat connected to current attack in Dawson Creek through the graph increases as cost thresholds increase (blue line). At a threshold of 0, only host patches containing current attack are connected. As the threshold increases, more patches are joined at distances less than or equal to the threshold. At an effective distance of just under 600 km, all habitat is connected in a single cluster. Arrows indicate scales over which there are rapid increases in connectivity indicated by changes in slope.

cast in a grid cell (raster) environment. Two kinds of factors determine the state of a cell:

1. Factors based on the state of the infestation itself such as the history of the infestation in a particular cell, and some measure of the influence of infestations in other parts of the landscape. Simple neighborhood rules (e.g., number of infested cells within some distance) are a common method of characterizing spatial effects, but we have found a more biologically informed model of dispersal pressure to be more useful.
2. Factors based on the nature of the physical environment such as forest age, species composition, and elevation.

Transition probabilities, from one state to another, are calculated directly from the observed transitions. Care must

be taken not to over-specify the model (ensure adequate sample sizes for probability calculations). The model may be refined somewhat by introducing hierarchy into the transition table. For example, the factors that best predict the probability of infestations starting in cells with no previous infestation history may be different than the factors that predict how infestations proceed once they have arisen. The state transition table is used to project an infestation through time.

Data Requirements—

Spatially explicit information about infestation history is required, categorized into states that represent the level of damage caused. In addition, information about the physical environment relevant to the progression of the infestation, such as forest cover mapping, must be available in digital form.

Output—

The principal output is a spatially explicit projection of the effect of an infestation on the forest resource. The spatial resolution of the output will be the same as the resolution of the input layers, whereas the precision (e.g., the level of the effect) will depend on the resolution of the historical outbreak information. Output is in the form of the state of the infestation (in terms of severity classes) in a given year and grid cell, based on the states represented by the input infestation maps. Numerous other outputs can be derived from the projection of the state; for example, spatially explicit projections of the volume of timber that is killed and tabular summaries of the area affected. As such models are stochastic, each scenario can be used to generate an expected distribution of attack, such as the form illustrated by Figure 1. However, mean values are generally used to communicate spatial and temporal dynamics and compare alternative scenarios.

Pros and Cons—

This approach is relatively data intensive, and a significant amount of analysis is required to develop the state transition tables. The approach will not provide new insights into the behavior of an infestation. As a strictly empirical approach, a key assumption is that the future behavior of the outbreak will resemble the behavior in the past. This approach, however, is one of the simplest ways to provide a spatially and temporally explicit dynamic projection. Although the data requirements are reasonably onerous, there is only a limited requirement for understanding the processes that govern the progression of the outbreak. A key advantage of the approach is that the spatially and temporally explicit projection can be integrated with other management or planning models to explore interactions between the effect of the infestation and the management response.

Management Application—

This approach could be applied over an area of any size. It is only applicable in situations where there is enough historical data on all outbreak phases to develop a useful state transition table. Its primary use is to project potential infestation trajectories, which can be used to clarify

management options and consequences and guide strategic policy development.

Example Application—

British Columbia is currently in the midst of the biggest MPB outbreak in recorded history. We used an empirical outbreak projection to forecast the possible impact of the outbreak over the entire province for the next 20 years (Eng and others 2005, Figure 3). We obtained 7 years of infestation history collected through the Provincial Aerial Overview of Forest Health (Ebata 2004, Figure 9). Forest cover information and a host of other data regarding the physical environment and management regime were collected primarily from the Province of British Columbia's Land and Resource Data Warehouse (<http://lrdw.ca>). An outbreak projection model along with a forest management response model was implemented using the SELES (Spatially Explicit Landscape Event Simulator) spatio-temporal modeling tool (Fall and Fall 2001). SELES combines a spatial database for a landscape with a high-level, declarative modeling language used to specify key processes and a discrete-event simulation engine that interprets and executes such models.

Based on the most recent infestation mapping, we estimate that approximately 25 percent of the merchantable pine volume in the province was observed to be dead (red or grey crowns) during the summer of 2005 (Figure 10). Because trees killed during the summer cannot be detected through aerial surveys (their crowns are still green), we rely on the projection model to estimate that an additional 10 percent of the pine volume was killed during that summer. The difference between the two projections is due to the dramatic increase in infestation in 2005. We show both to illustrate how empirical projection models are driven by observation, but focus on the one driven with a more complete data set (i.e., including 2005 data). This difference, however, does not alter the primary conclusions. We project that by 2010 over 60 percent of the merchantable pine volume in the province will be observed as dead, and that about 80 percent will be killed (Figures 10 and 11) by 2013 when the infestation will have largely run its course. Further maps of the input data and the projections can be found at <http://www.for.gov.bc.ca/hre/bcmpb>.

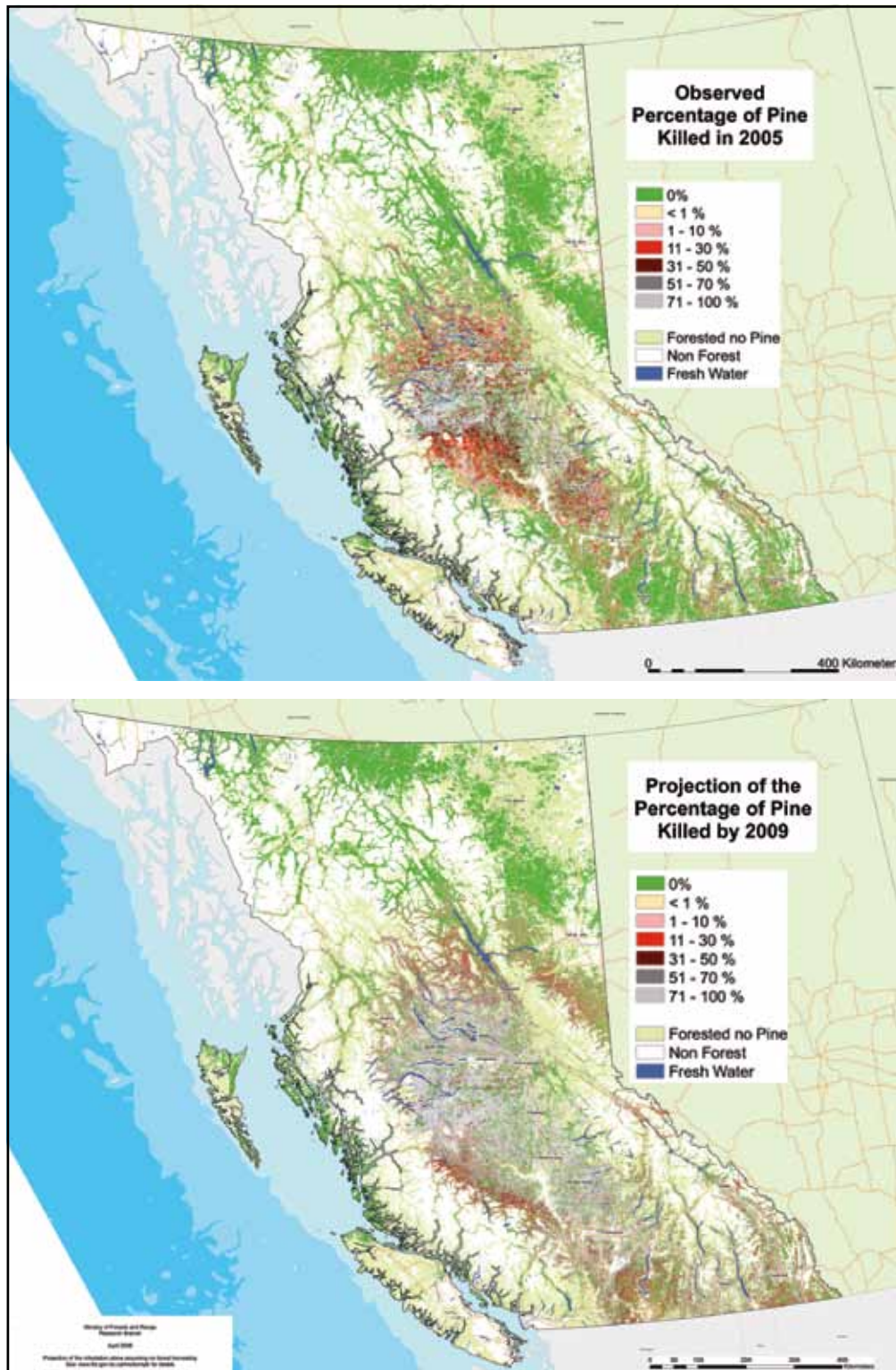


Figure 9—Mountain pine beetle attack from 2005 observation and projected at 2009. Comparison of patterns of mountain pine beetle attack from 2005 aerial overview surveys (top) and projected at 2009 using an empirical model (bottom) (from Eng and others, unpublished, found at <http://www.for.gov.bc.ca/hre/bcmapb>).

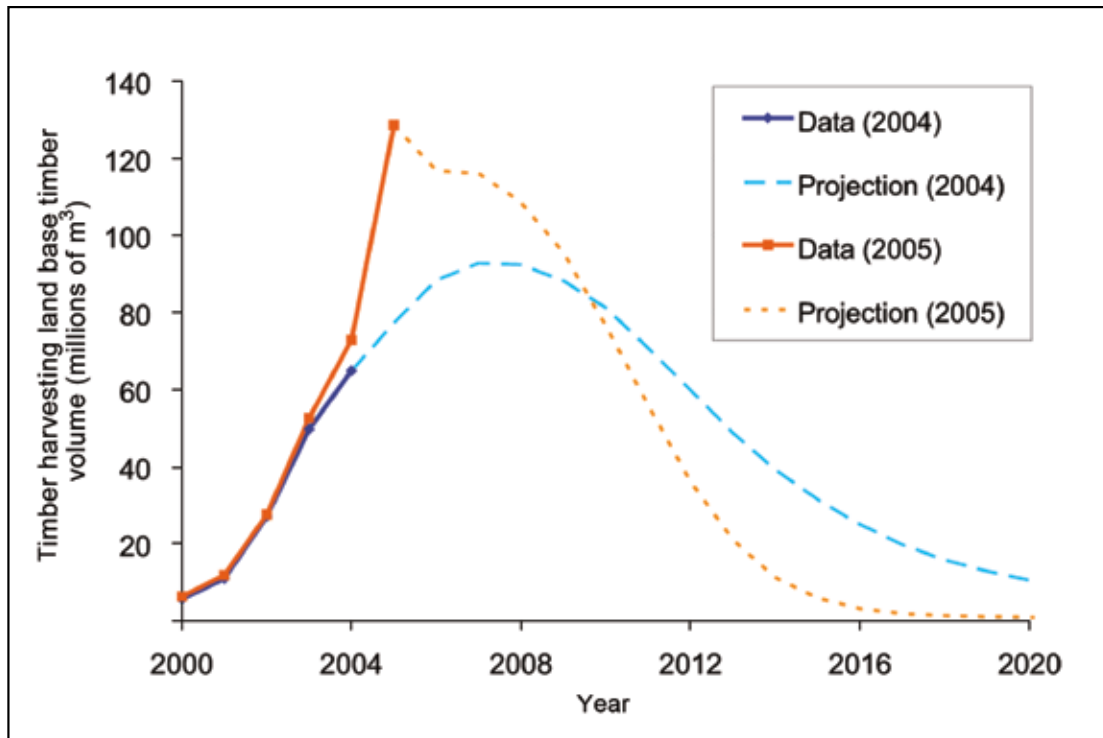


Figure 10—Observed and projected annual kill based on British Columbia mountain pine beetle model runs in the timber harvesting land base (THLB) over the entire province of British Columbia. Based on the 2004 and 2005 Provincial Aerial Overviews (Eng and others, 2005, Eng and others unpublished found at <http://www.for.gov.bc.ca/hre/bcmapb/BCMPB.v3.BeatleProjection.Update.pdf>).

The results of the projection have been used for a variety of purposes. Notably, they have been widely cited in the press and have been extensively communicated to natural resource managers in an effort to increase awareness about the severity of the problem. The results of the interactions between the outbreak projection model and our forest management model have been used to help direct funding for control efforts, examine the impacts of the forest management response on the transportation system, and to investigate the possibility of developing a bioenergy plant in the most severely affected area. The results of the infestation projection itself have been incorporated into detailed modeling for timber supply analyses.

Population Modeling

Population models capture outbreak dynamics by explicitly modeling demographic changes with processes of mortality, birth, dispersal, etc. (Caswell 1989). We designed a landscape-scale MPB population model to assess impacts at scale of ~1 000 000 ha to explore likely trajectory and

broad spatial patterns of an outbreak, to evaluate a range of management options, and to estimate likelihood of different outcomes (Fall and others 2004). The general concept is to project an infestation forward using a landscape model that combines a spatially explicit MPB population model (Dunning and others 1995) with a spatial management model for timber supply, strategic forest management, and fell and burn treatments, so that interactions between management and beetles can be assessed.

The MPB population model scales results from a more detailed stand-level MPB population model, MPBSIM, which projects expected development of a beetle outbreak in a stand of up to several hectares (Riel and others 2004, Safranyik and others 1999). Our approach is to conceptually run MPBSIM in each cell of the landscape. Because it is not feasible or desirable to do this directly, we first run MPBSIM under a wide range of conditions to produce a table linking conditions to consequences. Conditions include stand attributes (e.g., age, percentage of pine),

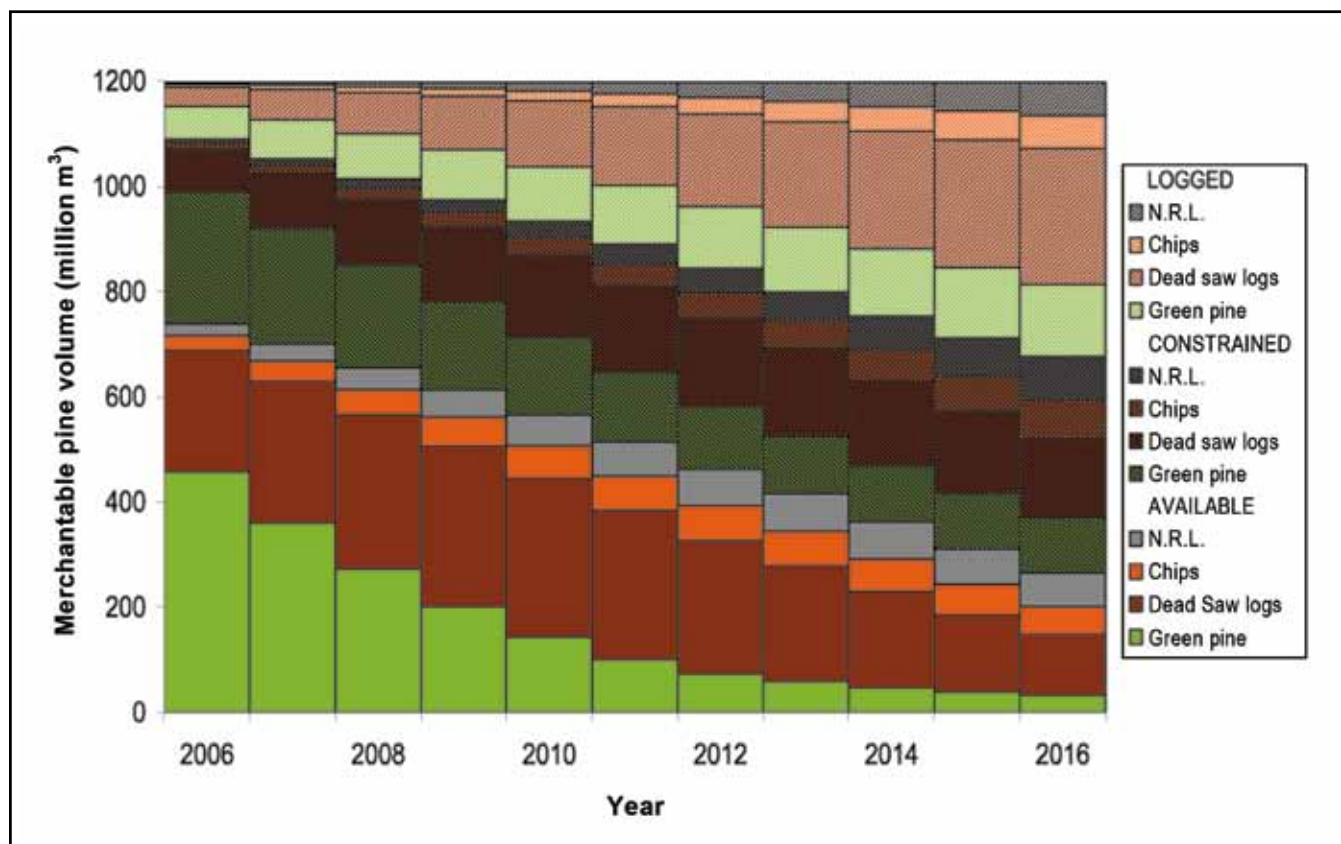


Figure 11—Projection of pine volume from British Columbia mountain pine beetle (MPB) model. This graph indicates the projected proportion of merchantable pine volume on the timber harvest land base in the 20 management units in British Columbia with the highest level of susceptible pine (from Eng and others, unpublished, found at <http://www.for.gov.bc.ca/hre/bcmapb/BCMPB.v3.ReferenceScenario.Update.pdf>). Left diagonal hatching indicates volume logged during the scenario. Right hatching indicates volume not available for harvest owing to policy constraints (e.g., visual areas) or economic limits (road access, minimum volume). No hatching indicates volume available for harvest (standing live and dead merchantable wood). NRL is nonrecovered loss (volume lost owing to decay). Chips are for products with lower requirements for wood quality (e.g., oriented strand board), whereas saw logs are for products with higher requirements for wood quality (e.g., structural lumber). All pine killed by MPB are covered by the NRL, chips, and dead saw logs category. Pine volume not killed by MPB is in the green pine category.

outbreak status (e.g., number of attacking beetles), etc. (Riel and others 2004). Consequences refer to the effect of 1 year of attack under those conditions (e.g., number of dispersers and number of trees killed). The landscape-level model uses this table to project MPB dynamics in each 1-ha cell containing beetles. The stand table includes stochastic variation in number of emerging beetles, and we control this to capture synchronous annual variation and above-average weather conditions.

Dispersal between cells provides the spatial context for an outbreak, leading to an increased beetle population in cells within a current outbreak, or starting an outbreak in a currently uninfested cell, expanding an existing spot, or starting a new spot. The flight period, including beetle local

and long-distance dispersal and pheromone production and diffusion, is modeled as a spatial process. Long-distance dispersal is largely governed by wind speed and direction used to select distance locations for MPB spread, whereas local dispersal is influenced by wind, susceptibility, pheromones, and distance. During attack, beetles kill pine trees, resulting in standing dead volume that may be salvaged by the logging sub-model.

Data Requirements—

Data requirements include detailed digital spatial forest cover data, infestation locations/intensity, beetle population estimates, and demographic parameters. In addition, availability of a stand-level population model to support scaling,

or the information required to develop process sub-models is required. This poses substantial effort and a long-term program to obtain the ecological information for parameterization (Fall and others 2004).

Output—

Nonspatial indicators summarize information across space as time-series output that includes:

1. The MPB outbreak indicators such as volume killed, number of trees killed, and area attacked
2. Growing stock inventory: cubic meters of live forest.
3. Management indicators such as annual volume and area harvested, volume of nonrecovered loss, volume salvaged, and amount of available salvageable wood.

Because the approach is stochastic, multiple replicates of each scenario are run. We designed several spatial indicators that summarize information across time and replicate:

1. The number of runs in which each 1-ha cell was attacked at least once, which can be roughly thought of as the probability that a cell will be attacked at some point in the 10-year horizon.
2. The cumulative volume killed, which shows areas likely to have the highest timber impacts.
3. The cumulative percentage of pine killed, which shows areas likely to have the higher ecological impacts.

Pros and Cons—

This approach requires substantial effort to develop and has fairly high data requirements, in particular the need for a reasonable understanding of beetle biology and interactions with hosts at relatively fine scales. The main advantage is that a population model takes a process-oriented approach to dynamic projections. This provides a closer match with the ecological process and greater ability to assess interactions with management. These methods can be used to identify likely trends over time and can integrate with management models. The process-based perspective enables emergent (bottom-up) properties not possible in a

more strictly empirical (top-down) approach (Korzukhin and others 1996). That is, a population can respond to future landscape conditions that haven't been encountered in the historical record. As with the empirical projection method, each scenario could potentially produce risk information in the form illustrated by Figure 1, but generally mean values are used to facilitate comparison of scenarios and to communicate spatial or temporal dimensions or both.

Management Application—

This method is applicable at landscape scales where cell resolution can be fairly fine (1 ha) and is designed for situations with an existing outbreak. Its strength is the ability to explore dynamic interactions between management alternatives and beetle populations, an approach that we have applied in a number of landscapes across BC (Fall and others 2004, in press). Information on the relative effects of beetle management strategies on area infested and volume killed can be used to assess impacts directly or to serve as input for further analysis of economic, social or ecological cost/benefits.

Example Application—

The MPB attack was first confirmed in the Dawson Creek area in 2002 (Figure 3). The main outbreak in British Columbia was expanding rapidly, and it appeared that there was some long-distance dispersal through the Rocky Mountains. Spots recently detected in Dawson Creek most likely originated from the main outbreak, and were transported over long distance via wind and through mountain passes (A. Carroll, pers. comm.). Growth rates, as indicated by green: red attack ratios have been relatively low (M. Duthie-Holt, pers. comm.), but nonetheless showed potential for population growth. This suggested that recent weather was sufficiently warm to support an outbreak, whereas historical climate likely precluded outbreaks (Carroll and others 2004). There has been substantial effort in Dawson Creek focused on dealing with detection and treatment of spots, with cooperation among licensees, the Forest Service, and parks. A landscape-scale projection of outbreak potential was deemed to be useful to inform this process and to help clarify some tradeoffs between options available (Fall and others, in press).

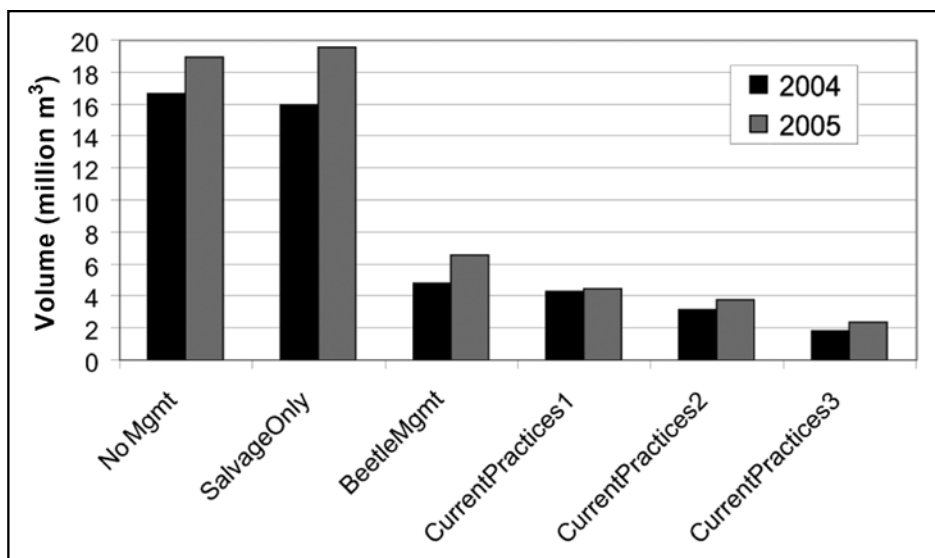


Figure 12—Projected cumulative volume killed in Dawson Creek mountain pine beetle population model over 10 years for the base beetle management scenarios and no harvesting. Initialized using 2004 or 2005 survey information (averages of 10 replicates). These scenarios include estimated changes in climatic suitability (Carroll and others 2004) and immigration from the main outbreak to the west. NoMgmt is no harvesting and no fell and burn. SalvageOnly only allows salvage of dead merchantable timber (i.e., focus is on recovery of dead wood, not population reduction). BeetleMgmt is general beetle management, aimed at reducing populations where possible. CurrentPractices approximates better current management in Dawson Creek, which aims at population reduction as well as reducing risk of spread eastward into Alberta. Three variations were assessed, differing primarily in budget limits on treatment levels.

The main purpose of this study was to apply a population-based model methodology to evaluate the effectiveness of bark beetle management activities in reducing losses to the MPB and to analyze the potential spread, likely trajectory, and impacts of the beetle across the study area. To achieve this goal, we started with the current conditions and projected likely outcomes under various management scenarios representing alternative beetle management regimes derived from workshops held in Dawson Creek with government and industry. We projected system dynamics for 10 years, with 10 replicates per scenario. A wide range of experimental scenarios was also assessed for calibration and sensitivity analysis.

Our results generally showed that this area still has the potential for beetle management to have an impact on population levels, in particular current local practices (Figure 12). In addition to general information to help with strategic planning, this modeling approach can provide spatial outputs to visualize and quantify patterns of the

expected outbreak trajectory under different management and weather conditions (Figure 13). This information is important for communicating risk potential in a landscape and can help with tactical planning of areas that may need management focus.

Other Risk-Assessment Methods

It may be possible to interpret other methods for assessing landscape-scale risk of bark beetle outbreaks in the framework presented (Figure 2), such as spatial temporal statistical methods (Augustin and others 2007, Wikle 2003, Zhu and others 2005) and field experiments. An alternative approach is employed in the Westwide Pine Beetle Model in which contagion forms the basis for a spatially explicit spread model (Beukema and others 1997). Another process-based approach to spatio-temporal modeling of MPB dynamics has been taken in the MPBpde model (Powell and others 2000), in which the MPB-pine interaction is represented by a system of partial differential equations that can

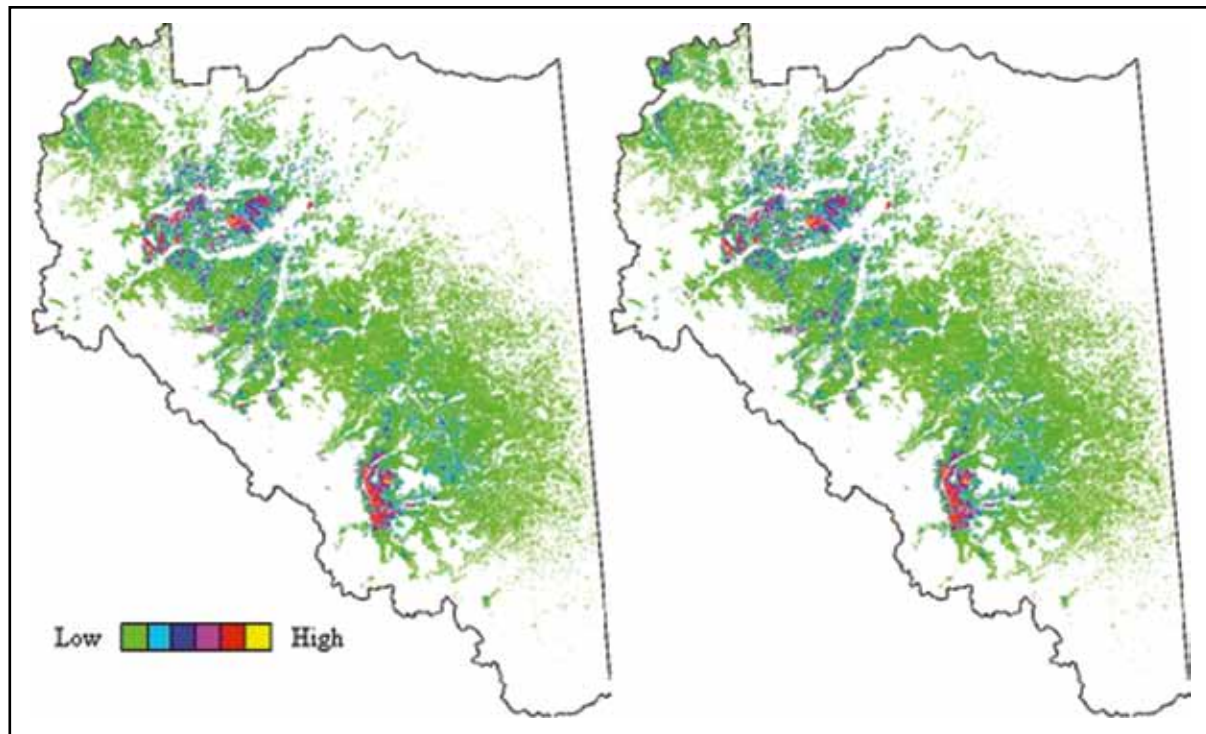


Figure 13—Spatial presentations of landscape risk for the no management scenario in the Dawson Creek study area. Left: projected likelihood of attack (proportion of 10-year runs in which each cell is attacked in projection). Right: projected intensity of attack represented as the percentage of pine killed (average cumulative percentage killed). These include estimated changes in climatic suitability (Carroll and others 2004) and immigration from the main outbreak to the west.

be explored numerically (e.g., Logan and others 1998) and analytically (e.g., Powell and others 2000). Partial differential equation methods can be applied over broad scales, but are challenging to combine with other landscape processes that are more discrete in time and space (e.g., timber harvesting). Hughes (2007) modeled beetles individually at a similar spatial and temporal scale. These models allow detailed exploration of how beetles interact functionally with a landscape. However, individual-based approaches are generally prohibitive at the landscape scale because such models are computationally demanding and because we lack the detailed land cover and beetle data required to parameterize them.

Discussion

In this section, we provide information for use in selecting an appropriate method of risk assessment, provide a discussion on model verification and validation, and discuss future research needs in this area.

Selecting an Appropriate Method

Some key aspects of a given problem can help guide the most appropriate choice of risk assessment method. In general, the best method is the simplest one (Occam's razor) that addresses the desired management questions in the required timeframe, using data available for the study area. The **possible** methods for a given problem are in the intersection of these issues. That is, these issues can help filter infeasible approaches. Generally, management needs emphasize more detailed and precise (more functional) risk assessment methods, whereas timeframe and data/knowledge availability emphasize simpler and coarser (more structural) methods. There is always a tension between the goals of maximizing information and minimizing uncertainty.

The following are some key considerations to help narrow the range of potential methods:

- Identifying management questions: Often, decisionmakers want to “know what will

happen” in the future. However, it is essential to clarify in precise terms the nature of the decision problem. What decisions need to be made? What level of information would be sufficient (as opposed to desirable)? It is important to maintain a transparent and collaborative relationship (e.g., Fall and others 2001) to ensure that results are useful. It is also important to communicate the uncertainties associated with different options and different questions to increase confidence in the chosen method.

- **Decision timeframe:** The timing of decisions plays a key role. For ongoing or long-term decisions, there may be time to collect new field data and develop more complex models. Short timeframes will require the use of currently available information and methods that can be supported by available data. Articulating the timeframe required for different options can help foster a shared understanding of the constraints imposed upon the choices available.
- **Defining area of interest:** Often, the spatial scale of a decision will determine the study area (e.g., harvest levels and strategies are often specified at the scale of a timber supply area). Sometimes the ecological scale of a process influences this decision. For example, a very broad-scale output may indicate that a larger study area is needed to ensure that adequate context is captured. It is critical to avoid scale mismatch problems (Cumming and others 2006).
- **Data availability:** Data availability can impose a significant constraint on which risk-assessment method can be employed. Lack of detailed knowledge about beetle demographics and movement may prohibit a population- or individual-based approach. Lack of readily available spatial information on historical outbreak patterns may prohibit an empirical approach. If the available data are not adequate to support a given method, this should lead to serious con-

sideration of its applicability. The apparent precision provided by applying a more detailed method and using inadequate data may be a false benefit compared to the more accurate but less precise results that would be achieved using a coarser method. Additionally, the data requirements to develop a new method or model are often different (and in many cases more onerous) than the data required to adapt an existing method developed elsewhere.

The most important decision is selection of a good team with a broad range of skills. At the outset, the primary focus should be on the questions or issues to address, and then the team should work backwards towards the tools. That is, clarify the issues and constraints of data and knowledge and timing raised in the preceding subsections, while developing and formalizing conceptual models. Flexibility and often an iterative approach are required to shift course as information (or lack thereof) becomes apparent. The simplest method that meets the needs of the study should be chosen. If temporal dynamics or outputs are not required, a static approach can be used. If spatial interactions or outputs are not needed, a non-spatial approach is appropriate. We presented the example methods in order of increasing complexity. Other methods can be fit into this framework at different points. A suite of methods that may be applicable should be identified, filtering out methods that are not adequate to meet the requirements. The remaining methods should then be contrasted to pick the simplest one because higher complexity and data requirements imply higher uncertainty as well as longer timeframes for application.

Model Verification and Validation

We define **verification** as an assurance that the model is implemented as specified, and **validation** as an assurance of the appropriateness of the model for its intended use (Rykiel 1996). That is, validation relates to the level of certainty one can place in model outputs; (i.e., the degree to which model results differ from expectations). Verification is an essential step and must be considered in model selection and application.

Validation is often defined as the degree to which model output matches an independent data set (Rykiel 1996). More structural approaches to risk assessment facilitate validation more easily than more process-oriented approaches (e.g., Cameron and others 1990, Dodds and others 2004). Static methods such as susceptibility rating can be statistically tested in areas with field data on past and current attack (Dymond and Wulder 2006). Empirical and connectivity approaches are driven by observation, and predictions can be compared with actual outcomes as an outbreak proceeds (and such data can then be used to improve the model parameters). Empirical, or data, validation for a spatio-temporal model is only possible in cases with short time lags in system response or for which suitable replicates exist (e.g., for chronosequence-type comparisons). The exact conditions encountered within large landscape systems cannot be found outside the system (Levin 1992). In addition, observational data isn't available for assessing hypothetical management alternatives. In relatively process-oriented approaches, it may be more appropriate to rely on conceptual and logical validation (Rykiel 1996), where we view the model as a hypothesis and model output as a consequence of the hypothesis. That is, the purpose of such models are to make a clear link between the initial conditions, parameter values, and process behavior, and the consequences of those assumptions, which are projected via simulation (which in this sense is akin to theorem proving), and not to predict the real state of the future forest (i.e., projection not prediction). Logical validation inherently relies on the adequacy of the input information regarding initial conditions, model processes, and appropriate parameter values. Refinement of these can occur over time as ecological knowledge is refined.

Future Research Needs

New methods will be developed, and existing methods will be improved in the area of risk assessment. We suggest that using the proposed framework for comparing tools will assist tool selection for a given situation and improve understanding of the differences between tools in terms of precision, uncertainty, and resources required. In addition

to ensuring that the set of tools forms a cohesive toolkit, it will also be important to improve the application of tools. That is, evaluating the applicability of a tool in a given situation needs to be easy, and usage of the method should be as straightforward and transparent as possible.

Although the examples we present focus on MPB in lodgepole pine forests, the concepts underlying the risk-assessment methods and the classification gradient apply to other bark beetle species and forest systems. Susceptibility rating systems have been developed for MPB in ponderosa pine, *Pinus ponderosa* Dougl. ex Laws. (Chojnacky and others 2000, Negron and Popp 2004) and whitebark pine, *P. albicaulis* Engelm. (Perkins and Roberts 2003). Dodds and others (2004) and Negron (1998) examined risk rating for Douglas-fir beetle (***Dendroctonus pseudotsugae*** Hopk.). Susceptibility rating systems have been developed for spruce beetle (*D. rufipennis* Kby.) in Alaska (Reynolds and Holsten 1996). Connectivity analysis for risk assessment is not very common at present. We have ongoing work to explore risk of spruce beetle (across a large area of southwestern Yukon, Canada, using connectivity methods. In addition to susceptibility rating, statistical methods to examine spatial and temporal autocorrelation of environmental factors (Gumpertz and others 2000) and simulation-based approaches (Mawby and Gold 1984) have been applied to the southern pine beetle (*D. frontalis* Zimm.).

Applying methods in new systems presents a number of challenges and high levels of uncertainty. The MPBs have been expanding the northeastern limit of their range and are approaching boreal jack pine (*P. banksiana* Lamb.) forests in Alberta, Canada (H. Ono, pers. comm.). These changes increase uncertainty owing both to the dynamic character of the changes and because little information is known on MPB—host interactions in these forests. Nonetheless, managers of these systems are faced with challenging decisions, and risk-rating systems can provide some insights. In conjunction with climatic suitability work (Taylor and others 2006), we have ongoing work to adapt and apply susceptibility and connectivity methods in the boreal forest of British Columbia and Alberta, Canada.

Conclusion

We presented a common framework within which methods to assess landscape-scale risk of bark beetle infestations can be classified. This framework has two elements. First, conceptualizing landscape-scale risk as a probability distribution of potential loss provides a common basis to compare methods and allows varying degrees of precision, uncertainty, and stochasticity to be included. Second, viewing methods along a gradient from structural (pattern-oriented) to functional (process-oriented) approaches to risk assessment helps to clarify tradeoffs between precision, uncertainty, data requirements and timeframes for application. A key message is that no single tool or method can address all needs. Viewing methods along a gradient of complexity helps provide a system to classify methods, which, in turn, facilitates comparison and selection for a given set of questions.

Acknowledgments

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Review of Methods for Developing Regional Probabilistic Risk Assessments, Part 2: Modeling Invasive Plant, Insect, and Pathogen Species

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Abstract

We reviewed probabilistic regional risk assessment methodologies to identify the methods that are currently in use and are capable of estimating threats to ecosystems from fire and fuels, invasive species, and their interactions with stressors. In a companion chapter, we highlight methods useful for evaluating risks from fire. In this chapter, we highlight methods useful for evaluating risks from invasive species.

The issue of invasive species is large and complex because there are thousands of potential invasive species and constant movement of new and established plants, plant material, pests, and pathogens. Adequate data are not always available to support rigorous quantitative modeling of the different stages of invasion. However, even a semiquantitative rule-based approach can help to identify locations that contain host species susceptible to specific pathogens or insect pests, and where propagules are more likely to enter based on the current locations of the invasive species, ports of entry, and methods of spread. Predicting long-distance movement is much more difficult, as such events are rare, often poorly understood, and are often influenced by human behavior. Even so, published methods to make probabilistic predictions of pest establishment could be expanded to provide quantitative estimates of spread beyond an initial port of entry. Many invasive species are transported along roads, and so road networks provide some information about the likelihood of introduction into a new region.

Models based on fundamental biological and physical processes, such as population demographics and movement

of organisms, can be more robust than purely statistical approaches. Process-based models may better support extrapolation beyond the range of available or historical data because they use predictor variables that represent physical and biological processes. However, even simple correlative approaches may be useful to quantify the overlap in spatial distribution of stressors and ecological receptors as a screening-level analysis. Furthermore, if predictors are chosen carefully, they may represent important processes. For example, data on nonindigenous species may be quite useful for predicting the occurrence of much rarer invasive species because the correlation is based on the key processes of human-influenced transport, establishment, reproduction, and dispersal of propagules. Ecological niche-modeling approaches are useful because they can use data from museum collections in other countries to make estimates of potential new range areas in the United States. Other spatial data such as road networks may also be useful to predict the number of nonindigenous species or presence of a particular species. Such relationships may also support extrapolation to future conditions if there will be more roads or a higher traffic volume.

As for any regional stressor, the use of multiple models and a weight-of-evidence approach would help to increase confidence in predictions of ecological risks from invasive species. Two approaches to predicting the risk of Asian longhorned beetle (*Anoplophora glabripennis* Motschulsky) throughout U.S. forests make quite different predictions because they focus on different stages in the process of establishment and spread, thus combining such approaches should result in more robust predictions. Invasive species management should be addressed at multiple spatial scales, including reducing importation of new species at border crossings and ports, national and regional mapping of locations of invasive species, methods to reduce long-distance transport, and methods to reduce local movement.

Keywords: Ecological risk assessment, invasive species, probabilistic risk assessment, regional risk assessment, risk analysis.

Introduction

This review provides an overview of issues in probabilistic risk modeling at the regional scale and suggestions for productive directions for future risk assessments and research. Invasive nonindigenous species are a serious and increasing threat to many ecosystems throughout the United States (NRC 2002, Pimentel 2005). For example, invasive species are implicated as threats for more than half of all endangered species in the United States (Wilcove and others 1998). Invasive species are also altering fire regimes, hydrology, nutrient cycling, and productivity of ecosystems in the Western United States, particularly rangelands and riparian areas (Dukes and Mooney 2004). Plant species such as yellow star-thistle (*Centaurea solstitialis* L.), other *Centaurea* species, and cheatgrass (*Bromus tectorum* L.) have overtaken large areas of native ecosystems in the Western United States (LeJeune and Seastedt 2001). Leafy spurge (*Euphorbia esula* L.), knapweeds (*Centaurea* sp.), tamarisk (also known as salt cedar, *Tamarix ramosissima* Ledeb.), nonnative thistles, purple loosestrife (*Lythrum salicaria* L.), and cheatgrass are some of the most severe problems on national forest lands. For example, the number of counties in Washington, Oregon, Montana, and Wyoming where yellow star-thistle has been found has been increasing exponentially during the last 100 years (D'Antonio and others 2004). Furthermore, the number of new exotic species has increased roughly linearly over this time period, reaching a total of nearly 800 by 1997 (D'Antonio and others 2004). Annual costs of selected nonindigenous species in the United States have been estimated at \$120 million (Pimentel and others 2005). However, this estimate does not account for all effects of invasive species on rangelands and forests (Dukes and Mooney 2004), and it is clear that such costs are substantial. Despite the difficulty in quantifying economic damage, there is substantial evidence suggesting that invasive species have many deleterious effects in ecosystems in the Western United States, and that improved management of invasive species in wildlands is crucial (D'Antonio and others 2004). For example, tamarisk alone has been estimated to cost \$133 to \$285 million per year (in 1998 U.S. dollars) for lost ecosystem services including

irrigation water, municipal water, hydropower, and flood control (Zavaleta 2000).

Various aspects of invasive species biology and ecology, as well as policy and management issues (NRC 2002), are addressed in many published reviews. We will review briefly some key issues, but the focus in this piece is on modeling methods suitable for spatially explicit probabilistic risk assessments for invasive species. This chapter, and a companion chapter addressing fire (Weinstein and Woodbury, this volume), present results of a project sponsored by the U.S. Department of Agriculture (USDA) Forest Service, Western Wildland Environmental Threat Assessment Center during its development; but these results should not be construed to represent the views of the center nor its personnel. The overall goal of our project was to identify promising methods for analyzing ecological risks to forest, rangeland, and wildland ecosystems from multiple stressors. The results of such risk analyses are intended to provide information useful for strategic planning and management of wildlands including national forests. The specific goal of this chapter is to identify modeling approaches suitable for making spatially explicit, probabilistic estimates of ecological risks from invasive plant, insect, and pathogen species throughout large regions such as the Western United States. Such modeling approaches ideally should be capable of:

1. Calculating risk of a detrimental environmental effect.
2. Using spatially heterogeneous environmental data to drive calculation of risk at different points throughout a region. Spatial scales of interest include landscape, sub-State region, State, region of the United States, or the entire conterminous United States.
3. Relying primarily on available regional (in United States, state or multi-State) or national data.
4. Being useful for many species, not just a single invasive species.
5. Modeling effects of interaction among multiple stressors.
6. Modeling effects of changes in environmental conditions in the future.

We review selected modeling approaches relevant to the goals listed above, and more detailed analyses of specific aspects of invasive species assessment and management are provided by other chapters within this broader work.

Stages of Invasion and Risk Assessment Frameworks

This section provides an overview of the stages of the invasion process, key factors that affect these stages, and different frameworks that can be used to assess risks due to invasive species. The process by which a nonindigenous species becomes an invasive species can be divided into the following five stages:

1. Uptake/entry into transport system
2. Survival and transport to the United States via land, air, or water, with or without vectors
3. Initial establishment—survival and reproduction
4. Local dispersion
5. Widespread dispersion

Three classes of key factors influence the likelihood that a potential invader will pass through each stage: (A) propagule pressure, (B) physicochemical requirements of the potential invader, and (C) community interactions (Colautti and MacIsaac 2004). However, even successful modeling of all stages of the invasion process still does not address the likelihood or degree of damage caused by the invasive species. For this purpose, an ecological risk assessment approach is required.

The topic of invasive species has begun to be addressed by practitioners of ecological risk assessment (Andersen and others 2004a, 2004b; Stohlgren and Schnase 2006). Andersen and others (2004a, 2004b) reviewed the regulatory framework for invasive species in the United States and some of the issues in extending the approach to ecological risk assessment (originally developed for contaminants) in order to address biological stressors such as invasive species. They also provide information about a series of articles of the journal “Risk Analysis” that report the results of a joint workshop between the Society for Risk Analysis Ecological Risk Assessment Specialty Group and the Ecological Society of America Theoretical Ecology Section.

In addition, they identify research needs for this field.

Of relevance to this chapter, they suggest that “Spatially explicit, multiscale decision-support systems will contribute to better decisionmaking through enhanced credibility, an explicit and direct relationship with managing for sustainability, and explicit illustration of trade-offs and the cost of inaction.” Presented in one of the articles in this series is a model of establishment risks for Asian long-horned beetle (*Anoplophora glabripennis* Motschulsky) introduction via solid wood packing materials (Bartell and Nair 2004). This approach estimates both the probability of establishment at the port of entry and the probability of spread based on environmental factors, host availability, and traits of the invasive species. Uncertainty in key parameters is investigated by means of Monte Carlo analysis. Additionally, there is investigation of the efficacy of different management techniques. Integration of quantitative risk analysis and quantitative analysis of management options within a single analytical framework is much too rare and should be applied more widely. Another article in this series describes how the conceptual model in the relative risk model can be applied to predict the effects of invasive species (Landis 2003). This approach is promising in that it is capable of addressing multiple stressors simultaneously at the regional scale by means of a ranking procedure. Although complete risk assessments are not reported in this article, it illustrates how invasive species risk can be analyzed at the regional scale in the context of multiple stressors and multiple endpoints. A case study of this approach has been implemented for a European green crab (*Carcinus maenas* L.) for a region of Washington State (Colnar and Landis 2007).

Transport to the United States and Within U.S. Regions

Most exotic plant species have been introduced to the United States intentionally, whereas most insects and pathogens have entered the United States unintentionally (Mack and Erneberg 2002). Global travel and trade have increased the amount of plant material, wood, and wood products moving into U.S. ports, increasing the likelihood of introduction of invasive plants, insects, and pathogens. By 2020, it has been predicted that more than 100 new

insect species and 5 new plant pathogens will become established (Levine and D'Antonio 2003). A particularly high-risk pathway for forest insects and pathogens is importation of raw logs (Tkacz 2002). As an example for the Pacific Northwest, surveys of ports, port areas, mills and businesses known to have received or handled imported wood or wood products from 1996 to 1998 found seven species of wood-boring beetles from Asia, Europe, and the Eastern United States (Mudge and others 2001). For the United States as a whole, inspections of all types of products in four cargo pathways at ports and border crossings found the highest rate of insect introductions in refrigerated maritime cargo, with 1 new insect species found in every 54 inspections (Work and others 2005). It was estimated in this study that fewer than half of such new species are detected, and 42 insect species may have become established from 1997 to 2001. These species do not necessarily pose a high risk of widespread infestation or damage, but they do indicate that exotic species are entering the United States at an alarming rate. Many of the issues of invasive species transport and establishment from other countries to the United States also apply to establishment of new populations owing to long-distance transport of invasive species among regions in the United States. Gypsy moth (*Lymantria dispar* L.) is an example species known to cause severe infestation and damage in Eastern U.S. forests (Liebhold and Tobin 2006). Gypsy moth has been long established in the Eastern United States but has been prevented from establishing, to date, in the Pacific Northwest owing to surveillance and eradication efforts (Hayes and Ragenovich 2001).

To manage invasions and reduce risks, it is vastly more cost-effective to prevent establishment, or eradicate an invasive species as soon after entry as possible (Simberloff 2003, Stocker 2004). However, most invasive species are difficult to locate and may not appear to present any significant risk to ecosystems until they have become well established, often many decades after introduction. Thus, most management and control efforts focus on severe known problems rather than preventing future severe problems. Also unfortunately, it is difficult to predict which nonindigenous species will become invasive, and which invasive

species will become severe problems (Smith and others 1999). A number of initiatives have been undertaken in the United States to address various aspects of invasive species monitoring, risk assessment, and management owing to the severity of problems caused by invasive species.

Existing National Invasive Management Programs

A number of international, national, and regional efforts are underway to attempt to reduce the risks posed by invasive species. Some of these efforts for the United States are discussed briefly below, with a focus on programs related to forest and rangeland ecosystems. It is beyond the scope of this review to discuss all international programs that may provide valuable information for invasive species in the United States. However, some sources of global information are mentioned in the subsequent section on invasive species databases.

The National Invasive Species Council (NISC) consists of eight Federal departments and was formed in 1999 by Executive Order 13112. The NISC 2001 National Management Plan called for development of a risk analysis system for nonnative species by 2003. The NISC is intended to provide a gateway to information, programs, organizations, and services about invasive species. Their Web site (<http://www.invasivespecies.gov>) provides information about the impacts of invasive species and the Federal government's response, as well as select species profiles and links to agencies and organizations dealing with invasive species issues.

The USDA Animal and Plant Health Inspection Service (APHIS) protects not only agricultural but also forest, rangeland, and wetland ecosystems. APHIS works closely with the USDA Forest Service and the U.S. Department of the Interior's Bureau of Land Management, National Park Service, and Fish and Wildlife Service. APHIS conducts risk assessments with a dual mission to promote international trade and prevent invasive species that may cause serious harm from entering the United States. Some APHIS activities focus on protecting and managing endangered species as well as migratory bird populations. APHIS maintains the Port Information Database, and there is great potential to strengthen and make broader use of this

database for understanding the pathways taken by invasive species entering the United States (NRC 2002).

The USDA Forest Service, working in conjunction with Federal, State, tribal, and private partners, has developed the Early Warning System (EWS) to detect and respond to environmental threats to forest lands in the United States. The EWS comprises many existing programs, along with new initiatives such as the Western Wildland Environmental Threat Assessment Center and the Eastern Forest Environmental Threat Assessment Center. The EWS addresses potential catastrophic threats such as insects, diseases, invasive species, fire, weather-related risks, and other episodic events. The system is intended to:

1. Improve understanding of the crucial elements involved in early detection and response to environmental threats.
2. Help identify and remedy weaknesses in the current system of early detection and response.
3. Aid for strategic planning and resource allocation.

There are many groups both within and outside the Forest Service that participate in the process of detecting and responding to threats to forests. Further information about some component groups that conduct regional risk analyses is presented in other chapters in this volume. Further information about the EWS is available at the following Web site: http://www.fs.fed.us/foresthealth/programs/early_warning_system.shtml.

The National Aeronautic and Space Administration (NASA) and the U.S. Geological Service (USGS) are developing a National Invasive Species Forecasting System (ISFS) for the management and control of invasive species on Department of Interior and adjacent lands. The system provides a framework for using USGS's early-detection and monitoring protocols and predictive models to process remote sensing data from the Moderate Resolution Imaging Spectroradiometer (MODIS), the Enhanced Thematic Mapper, and the Advanced Spaceborne Thermal Emission and Reflection Radiometer as well as commercial remote sensing data. The goal is to create on-demand, regional-scale assessments of invasive species patterns and vulnerable habitats. Additional information can be found at the

following Web site: <http://bp.gsfc.nasa.gov/>. This approach has recently been used to predict the relative suitability of all areas in the conterminous United States for tamarisk, an invasive woody shrub (Morissette and others 2006). This analysis is reviewed below under the heading of USGS and NASA Invasive Species MODIS-Regression methodology.

Within the USDA Forest Service, the establishment of the two Threat Assessment Centers is a key part of the strategy for improving the management of invasive species. These efforts build upon ongoing programs and projects such as the Forest Inventory and Analysis Program (including Forest Health Monitoring) and Forest Health Protection. Further information about the strategies of these agencies for invasive species management is provided at the following Web site: http://www.off-road.com/land/invasive_species_strategy.html. Recommendations for control of invasives in rangelands are provided at the following Web site: <http://www.fs.fed.us/rangelands/ecology/invasives.shtml>

The USDA Forest Service's Forest Health Technology Enterprise Team (FHTET) is using an expert opinion approach to model risks of invasive pests and tree pathogens at the national scale for national strategic planning purposes. Potential tree mortality risk is modeled based on expert opinion, forest inventory data, and other GIS (geographic information system) data (Marsden and others 2005), also see the following URL: <http://www.fs.fed.us/foresthealth/technology/products.shtml>. Further discussion of this approach is presented below under the heading of "FHTET national risk map."

Availability of Spatial Data

Many kinds of regional data may be useful for developing regional probabilistic risk assessments, including land cover and land use data, transportation networks (e.g., roads and trails), hydrography, climate, digital elevation models, etc. Many such databases are available in GIS format from the National Atlas, which also includes data on selected invasive species (<http://www.nationalatlas.gov/atlasftp.html>). Data on land use is available from the National Land Cover Characterization database that is being compiled across

States as a cooperative mapping effort of the Multi-Resolution Land Characteristics Consortium. Landcover databases are being developed by bioregion based on remotely sensed imagery acquired from 1999 to 2003 and are complete or nearly complete for most portions of the United States, including the West Coast and much of the Southeast (http://www.mrlc.gov/mrlc2k_nlcd.asp). It is beyond the scope of this review to discuss all of these types of data, or even all types of databases specifically on invasive species, but a brief overview of invasive species survey data is presented below.

At the global scale, the Global Invasive Species Information Network is developing an online registry of data sets related to nonnative species (Simpson 2004), and ongoing efforts are being made to develop linkages among national and multicountry invasive species databases (Simpson and others 2006). The Global Invasive Species Programme (Mooney 1999) provides an online list of invasive species databases, including those covering the conterminous United States, Alaska, and Hawaii (<http://www.gisp.org/links/index.asp>). In the United States, a survey was undertaken recently to identify data sets of nonnative species at county, State, region, national, and global scales (Crall and others 2006). Based on a literature survey, Internet search, and responses from surveys sent to 1,500 experts, a total of 319 data sets were identified, and metadata were collected for most data sets (79 percent). Of the total, 57 percent are available online (see the following Web site for further information: <http://www.niiss.org>). Categories of data sets for which metadata are available consist of the following: 77 percent cover vegetation, 38 percent cover vertebrates, 77 percent cover invertebrates, 14 percent cover pathogens, and 9 percent cover fungi. Note that these percentages sum to greater than 100 percent because some data sets cover multiple taxa or categories. The scale of data sets for which metadata are available are as follows: 33 percent are at the county scale, 20 percent at the State scale, 17 percent at the multi-State regional scale, 15 percent are national, and 14 percent are global. Although this number of data sets is encouraging, the authors note that only 55 percent of the data sets have a quality assurance and quality control

procedure, suggesting that the accuracy of many data sets may be questionable or undetermined.

Other sources of data useful for regional assessments of invasive species are databases developed by the Forest Inventory and Analysis (FIA) Program of the USDA Forest Service (<http://fia.fs.fed.us/>). The FIA Program collects data for all land meeting a specific definition of forest land in three phases. Historically, Phase 1 has been based on aerial photography, but now satellite remote sensing imagery is being used. Phase 1 points are used to identify forested and nonforested locations. Phase 2 includes ground measurements such as tree species, height, diameter, disturbance, and stand age on more than 100,000 stratified sampling plots across the country. Historically, the focus was on timber resources that are available for potential harvest, but during recent decades there has been increased emphasis on a broader suite of forest characteristics including forest health and invasive species. In particular, Phase 3 sampling is done on a subset of plots to determine the species, abundance, and spatial arrangement of all trees, shrubs, herbs, grasses, ferns, and fern allies (horsetails and club mosses). This Phase 3 sampling was begun as a separate program called Forest Health Monitoring but is now administered through the FIA Program. As an example, a pilot study collecting Phase three data on plots throughout Oregon found at least 1 nonnative species on 70 percent of all forested plots, and 20 percent of plant cover was nonnative in one of 10 forested plots (http://earthscape.org/r1/ES16479/pnrs_science%20update.pdf; note: membership is required to access this Web site, but free trial membership is available). In addition to data specifically on invasive species, the Phase 2 FIA data are a valuable source of vegetation data because they have been collected in statistically designed surveys for decades. Information on forest type, stand age, and disturbance history are available and can be used in conjunction with data on invasive species to predict vulnerability of forest stands to invasion. Such an approach is underway in the Southern United States (Ridley and others 2006). Phase 2 FIA data are also being used in conjunction with other data to develop regional and national vegetation databases in other research programs including LANDFIRE. See the

topic “Conclusions Concerning the Use of Fire Modeling Systems” in Weinstein and Woodbury (this volume).

Review of Selected Methodologies

In this section, we review selected modeling approaches relevant to the goals listed above in the “Introduction” section. The focus is on invasive species of concern for the Western United States, particularly forest and rangeland ecosystems. Examples were selected to cover a range of analytical techniques with an emphasis on the State or regional scale. In addition, we selected examples of two different methods applied to an invasive pathogen that is the causal agent of sudden oak death disease (*Phytophthora ramorum* Werres, de Cock & In’t Veld) and two methods applied to an invasive insect: the Asian long-horned beetle.

Climatic and Ecological Niche Models

The most common and readily applied approaches to predicting the risk of invasive species occupying sites across a large region rely on biogeographical distribution models. These models are based on information about the biophysical factors that limit where a species can survive. Such models are known as bioclimatic envelope models, biogeographical distribution models, and (ecological) niche models. Such models are generally correlative and may be either statistically based or rule based. As applied to invasive species, such approaches typically attempt to map which parts of a region are suitable for the invading species, and suitability is typically based on habitat requirements. For pests and pathogens, the simplest approach is to map the presence or absence of suitable hosts. Such maps are typically developed from available regional data sets, which often provide relevant but not necessarily ideal data for a particular invasive species. Such maps may be useful for strategic planning at the regional scale, but may be of limited use for managing specific areas presuming that the managers of those areas already know where different species occur.

Niche models typically identify habitats for invasive species based on records of their presence at known locations. Such records can be obtained from museum collections such as herbaria, but currently, only 5 to 10 percent

of such data are available in electronic form worldwide (Graham and others 2004). To define the niche or bioclimatic envelope, biophysical data for each such location are often extracted from regional databases, usually in a GIS. The most important distinction among such approaches is whether they use absence data in addition to presence data. In other words, whether locations where the invasive species does not occur (absence) are used to define biophysical conditions that are outside of the niche. Either approach is problematic for invasive species because, typically, they have not yet occupied all possible sites. Thus, sites where the species doesn’t occur may not necessarily provide information about the species niche or requirements; instead, those may be sites that the invasive species haven’t yet reached. Presence and absence data can be obtained from the native region of the invasive species, but the species may have a different niche in the part of the world it is invading, as compared with its native region. However, use of data from the native region may be the only reasonable choice for species that have not become widely established in the United States. Even so, there may be substantial uncertainty in such predictions until a species becomes widely established. For example, an analysis of purple loosestrife (a common invasive species in Eastern United States wetland areas) determined that a reliable prediction of the current nonnative distribution in North America was only possible 150 years after initial establishment (Welk 2004).

Many variations of the niche approach are used to predict the niche of the invasive species including:

1. Simple ranges for factors based on mean climatic variables such as the widely used BIOCLIM and DOMAIN models.
2. Fuzzy rather than crisp calculations of the niche (Robertson and others 2004).
3. The use of spatial statistical techniques and newer computational approaches, such as genetic algorithms and support vector machines.

We have evaluated a few examples of such approaches below, with a focus on the Western United States. For each of these examples, we discuss how they meet the criteria listed above.

GARP Niche Modeling Approach

In this family of approaches implemented in a software tool, the potential range of invasive species is predicted based on point data from the species native home range and spatial data including mean annual temperature, rainfall and elevation (Anderson and others 2003, Costa and others 2002, Godown and Peterson 2000, Peterson 2001, Peterson and Cohoon 1999, Peterson and Kluza 2003, Peterson and others 2003b, Stockwell and Peterson 2002, Underwood and others 2004; also see <http://www.lifemapper.org/desktopgarp/>). This approach shares many features with other approaches to predict ecological niches based on bioclimatic data, including climate envelope modeling and other methods for niche modeling. All of these approaches assume that bioclimatic predictor variables (for example, mean annual temperature and precipitation) control the native distribution of an invasive species, and these factors will also control the potential distribution in the United States. This technique differs from others because it uses a machine learning method (also known as artificial intelligence) named Genetic Algorithm for Rule-Set Prediction (GARP). Based on only 15 to 20 records of locations of a species from its native home range (species input data), the method can predict the potential distribution (home range, or niche) of a species. This approach has been used by its developers to model the niche of both invasive species and noninvasive species. The user needs to provide species input data of known points where the species has been found in its native region. These data should be well distributed throughout the species native range and need to be georeferenced. The user also needs to provide environmental data covering the entire area for which predictions are desired, including mean annual temperature and precipitation (modeled surfaces). Potentially, many other input data could be used such as remote sensing images, but they might need to be available for both native region and the analysis region.

The software used is desktopGARP, which can be downloaded from the following Web site: <http://nhm.ku.edu/desktopgarp/>. The user selects a type of inferential tool, such as logistic regression, or bioclimatic rules. The input data are then divided into training data and validation data. The software generates pseudodata via resampling,

and then iteratively tries a large number of rule sets, continuing either until there is no further improvement in the predictions, or 1,000 iterations. The output from the model is a map of species niche as presence/absence, with some confidence values. Modeling may be done for either counties or for grid cells (pixels). The primary prediction is whether a county or a pixel is contained in the species potential (fundamental) niche. A measure of likelihood is generated by using multiple models, and assigning higher likelihoods to counties or pixels predicted to be included in the niche by multiple models (Peterson and others 2004).

In the following citations, only one predicted value is made per county, although the approach could be extended for finer grain analyses if input data are available at finer scales. The methodology (Peterson 2003) and its use to predict the distribution of four alien plant species in North America for a single point in time (the fundamental niche) are described in the references reviewed herein. Invasive plant species analyzed to date include Hydrilla (*Hydrilla* L.C. Rich.), Russian olive (*Elaeagnus angustifolia* L.), sericea lespedeza (*Lespedeza cuneata* (Dum.-Cours.) G. Don), and garlic mustard (*Alliaria petiolata* (Bieb.) Cavara & Grande) (Peterson and others 2003a). To predict the spread of Asian long-horned beetle, the GARP approach has been combined with a spatial model of spread originally developed for forest fire (Peterson and others 2004). The GARP approach has several strengths for the regional risk analysis of invasive species, which are as follows:

1. It has been applied to a number of taxa of invasive and noninvasive species in the United States and elsewhere.
2. A freely available software tool has been developed that implements this approach.
3. Data requirements for this approach are modest.

Most weaknesses of the GARP approach are shared by all niche modeling approaches, which include:

1. Not all of the stages of the invasion process are modeled.
2. Only presence or absence of a species is predicted, not effects of invasive species.
3. Results may be biased, depending on the

source of data and the use of pseudo-absence data (Graham and others 2004).

Other approaches such as support vector machines and generalized additive model (GAM) approaches may be less biased and provide more optimal statistical solutions (Elith and others 2006, Stockwell 2005), but see also Anderson and others (2003) for improving on model selection methods.

FHTET National Risk Mapping Approach

This approach is also a family of related approaches to predict tree mortality risk owing to an invasive insect or pathogen based on expert opinion, forest inventory data, and other GIS data (Marsden and others 2005), and also consult FHTET products Web site: <http://www.fs.fed.us/foresthealth/technology/products.shtml>). Specifically, predictions are made of the potential basal area loss of susceptible tree species owing to an invasive insect or pathogen. The location of suitable host species is interpolated using inverse-distance weighting based on forest inventory data. A multi-criteria risk ranking model is developed based on expert opinion about the factors that influence pest or pathogen establishment, spread, and tree mortality. An iterative process is used to develop risk maps, so the experts and analysts can alter the weighting of difference factors to adjust the maps to match expert opinion. This approach has been used to predict the potential effect of oak wilt in the North Central States and of wood wasp (*Sirex noctilio* Fabricius) throughout the conterminous United States: (http://www.fs.fed.us/foresthealth/technology/invasives_sirexnoctilio_riskmaps.shtml).

The following are the key required input data and their sources:

1. Principal U.S. ports - Army Corps of Engineers, Waterborne Commerce, Foreign Cargo Statistics.
2. Markets for wood products - Federal Highway Administration, Freight Management and Operations, Freight Analysis Framework, Highway Truck Volume and Environmental Systems Research Institute's (ESRI) polygon data.
3. Distribution centers - National Transportation Atlas Database.
4. Species occurrence and basal area of individual tree species – USDA Forest Service, Forest Inventory and Analysis (FIA), National and New York State Christmas Tree Association Web sites.
5. Road density and distance to road – U.S. Geological Survey, Heterogeneous Distribution Indicator.

Use of this approach requires one or more experts on the pest or pathogen, expertise in the use of FIA data, and expertise in GIS software. The spatial scope is the conterminous United States for a single time period. Required software includes ArcView 3.x, Spatial Analyst ModelBuilder (ESRI, Inc.), and IDRISI 32 (a raster GIS software package). Model output includes maps of predicted occurrence based on (1) hosts known to be susceptible and (2) hosts suspected to be susceptible.

For regional and national risk analysis, the approach of mapping factors that influence a stressor and then combining these factors with weightings derived from expert opinion are intuitively appealing and fairly common. This flexible, iterative expert opinion-based approach can be used for virtually any pest or pathogen, and a risk map can be generated fairly quickly because the system is already in place. Other strengths of this approach include the use of national FIA data and the quantification of potential damage in terms of tree mortality. However, the flexible expert opinion-based approach is also a weakness because it is so open-ended, subjective, and difficult to validate. To date, it does not appear that an attempt has been made to determine which environmental factors were actually associated with pest presence, or to quantify uncertainties in GIS layers or predictions. In contrast, a statistical inference approach that made quantitative predictions of pest occurrence would be more useful because it could be better tested against validation data.

Meentemeyer Sudden Oak Death Approach

Meentemeyer and others (2004) used a rule-based function to predict spread of sudden oak death pathogen distributions

in grid cells (30 by 30 m) throughout California. A prediction was made of the likelihood of presence of the disease based on rules derived from expert opinion and published data on plant species susceptibility, pathogen reproduction, and host climate. This method is focused on evaluating a single risk, the probability of oaks on a given site being infected by *P. ramorum*. More specifically, the method begins with mapping five predictor variables in a GIS and then using a set of rules to determine the risk of infection based on these predictor variables. The predictor variables are host species index, precipitation, maximum temperature, minimum temperature, and relative humidity. Host species index is weighted three times as strongly as precipitation and maximum temperature, which in turn are weighted twice as strongly as relative humidity and minimum temperature. Each variable is classified on a relative index, with host scored on a scale from 0 to 10, precipitation, maximum temperature, and humidity scored from 0 to 5, and minimum temperature scored from 0 to 1. The model was tested against 323 field observations in California. The model generally predicted higher risk for sites where *P. ramorum* is currently present and lower risk for sites where it is currently absent. However, it appears that approximately 20 percent of low-risk sites were infected.

Input data for the model include host susceptibility, pathogen reproduction, and host climate suitability. Like many modeling approaches, this approach requires expertise in GIS and database analysis. The model output is in the form of a map with estimated risk of occurrence of the pathogen at a single time period – movement of the pathogen is not modeled. The spatial scope includes all of California, and the map unit is landscape cell (30 by 30 m). The approach uses the CALVEG database (USDA Forest Service RSL 2003) for vegetation alliance and presence of *P. ramorum* and the Parameter-elevation Regressions on Independent Slopes Model (PRISM) for elevation-based regression extrapolations from base weather stations for climate data, which are available for the conterminous United States (<http://www.wcc.nrcs.usda.gov/climate/prism.html>).

The method meets the criterion of calculating the risk of detrimental environmental effect by mapping the probability of pathogen occurrence in each forest grid-cell

and could be extended to predict the presence of pathogens in smaller regions or pixels. But the focus is assessment of effects over a region, specifically bioregions, rather than at all points within a region. The method meets the criterion of using spatially heterogeneous environmental data to drive calculation of risk at different points throughout the Western United States. Potentially, it could be used to evaluate the risk from a number of stressors, but relationships between habitat conditions and probability of stressor occurrence would have to be developed. Potentially, the method could be extended to consider effects of interaction among multiple stressors, but interaction terms would need to be identified and parameterized in a regression model. The approach does not currently consider the effect of changes in environmental conditions over time.

Unfortunately, no attempt was made to determine which environmental factors were actually associated with disease presence. A statistical inference approach that made quantitative predictions of pathogen occurrence would be more useful because it could be better tested against validation data when they become available. The finding that 21 percent of sites predicted to be low risk, yet were found to be infected, suggests that the model has limited predictive power. This limited power is likely due to data limitations as well as lack of precision in rules and weights applied to them. The investigators do state that they plan to use FIA data to improve the predictions. This study was evaluated because it addressed an important risk factor in Western and potentially Eastern U.S. forests, but use of a method that makes more quantitative predictions would be useful in the future.

Nowak Host Range Approach

This approach predicts potential home range of an (invasive) insect or pathogen of trees by modeling the location of suitable host species based on forest inventory data (Nowak and others 2001, <http://www.fs.fed.us/ne/syracuse/Data/Nation/InsectPoten.htm>). A model of urban forests (UFORE) is used to predict urban forest composition based on data from a limited number of cities in the United States. Predictions are also made of the amount of tree cover that could be lost owing to tree death and the costs of replacing killed trees.

A simple model of spread (moving outward at a constant rate from one location) was used to predict the length of time required for invasion to occur in each major city. This approach has been used to predict the potential effect of Asian long-horned beetle throughout all urban areas in the United States (Nowak and others 2001), and preliminary predictions have been made for nonurban areas (<http://www.fs.fed.us/ne/syracuse/Data/Nation/InsectPoten.htm>). Preliminary predictions have also been made for the emerald ash borer (*Agrilus planipennis* Fairmaire) (<http://www.fs.fed.us/ne/syracuse/Data/Nation/InsectPoten.htm>). The main type of required input is appropriate forest inventory data. Model output is in the form of maps of predicted occurrence based on (1) hosts known to be susceptible and (2) hosts suspected to be susceptible. The model has been used at the scale of the conterminous United States for a single point in time.

One strength of this approach is the use of FIA data in conjunction with a model that has been used for many years. Another strength of this approach is the quantification of damage in terms of economic losses of urban trees. For urban trees, such economic losses are quite high, though for wildlands they will be much lower for an individual tree and much harder to estimate for a forested region. A limitation for regional risk assessment and management is that the focus of the model is urban areas. Another limitation, typical of most niche modeling efforts, is that not all steps in the process of invasive dispersion and reproduction are modeled, and that predictions are primarily of the potential host range of the pathogen, not of effects of the pathogen other than economic losses owing to the death of urban trees.

USGS and NASA Invasive Species MODIS-Regression

In this approach, a logistic regression is developed to predict the suitability of each 1-km pixel as habitat for tamarisk throughout the conterminous United States (Morissette and others 2006). Various ground surveys of tamarisk occurrence were integrated into a single database as presence or absence of tamarisk. Land cover, normalized difference vegetation index (NDVI), and enhanced vegetation index (EVI) were derived from MODIS data products. A discrete Fourier transform was used to model a constant

amplitude yearly sine wave to each pixel, and the mean, amplitude, and phase of both NDVI and EVI were used as potential predictor variables along with a fitted parameter for each land cover class in a logistic regression model to predict the likelihood of habitat suitable for tamarisk. The ground data were split into a training set to fit the model (67 percent of data) and a validation set (33 percent of data). The best model included land cover, and seasonal variability in NDVI and EVI. The proportion of correctly predicted observations using a threshold of 0.5 was 0.90. The main model inputs are MODIS data and surveys of tamarisk presence. Because it is a regression procedure, many other input data could be used, such as human population density, trail networks, air temperature, etc. The main model output is a relative ranking of the likelihood of suitable habitat for an invasive species.

This general approach would be useful for regional assessments because it uses remotely sensed data that cover the entire conterminous United States. However, for each invasive species, a large database of ground survey data is required. If FIA or other systematic survey data could be used for this purpose, that would make the approach useful for many more invasive species. A limitation of this approach is that it uses statistical correlation to make predictions, thus it cannot readily predict the effect of future environmental conditions such as changes owing to development, changes in hydrology, or changes in regional or global climate. Other examples of logistic regression to analyze invasive species include multiple species in South Africa (Higgins 1999) and Russian knapweed (*Acroptilon repens* (L.) DC.) in Colorado (Goslee and others 2003).

Dark Invasive Species Spatial Autoregressive Approach

This approach uses spatial statistical analysis to predict the distribution of invasive and noninvasive alien plants throughout all bioregions in California (Dark 2004). Spatial autoregressive (SAR) models were used to assess the relationship between alien plant species distribution and native plant species richness, road density, population density, elevation, area of sample unit, and precipitation. Three predictors were found to be statistically significant for both

invasive and noninvasive plants: elevation, road density, and native plant species richness. The best model (with all predictors) explained about 80 percent of the variance in the number of alien species in each bioregion. Additionally, there was significant spatial correlation for both invasive and noninvasive alien plants. Both invasive and noninvasive alien plants are found in regions with low elevation, high road density, and high native-plant species richness. Spatial data input requirements include a digital elevation model, precipitation (a modeled surface), road networks, native species richness, and occurrence of alien species. Because it is a regression procedure, many other input data could be used, such as population density, trail networks, air temperature, traffic volume, etc. The model has been applied to all of California for a single time, with bioregions as the map unit. Model outputs include maps of the number of invasive and noninvasive alien species by bioregion. The method could be extended to predict the presence of invasive species in smaller regions or pixels.

This general approach would be useful for regional probabilistic risk assessments because it uses widely available data in conjunction with a flexible spatial statistical approach. Additionally, it predicts the total number of nonindigenous (alien) species within a region. This technique could be feasibly extended to predict the probability of occurrence of invasive species based on the occurrence of noninvasive alien species. This would be very useful because noninvasive species were found to be roughly tenfold more common than invasive species for the bioregions. This would be a useful first step for regional risk assessment for large regions such as the Western United States in order to identify areas with higher overall risk for invasive species. The approach could be improved by using more detailed data on vegetation types rather than bioregions. A limitation of this approach is that it uses statistical correlation to make predictions, thus it cannot readily predict the effect of future environmental conditions, such as changes owing to development, changes in hydrology, or changes in regional or global climate. However, it might be feasible to develop statistically based extrapolations from existing data. For example, if the number of nonindigenous species in a region can be predicted based on some measure

of the transportation network, or other environmental factor, one could extrapolate to future conditions with more roads or a higher traffic volume. A future scenario of new road development or greater traffic or both on existing transportation networks could be developed based on planned State and Federal transportation projects. This scenario could be used to predict the subsequent increase in occurrence of nonindigenous species and invasive species.

Guo Support Vector Machine Approach

This method uses a type of machine learning algorithm called support vector machine (SVM) in a niche modeling approach to predict risk of occurrence of sudden oak death throughout California (Guo and others 2005). A useful comparison is made of presence-only (one class SVM) versus presence with pseudo-absence data (2-class SVM). Based on their results, the use of pseudo-absence data does not appear to be a good choice for modeling invasive species—they inherently lead to bias because they conflate environmentally determined absence with absence on account of infestation not having occurred yet in a particular location. Input data include 14 environmental variables including mean annual temperature, mean annual precipitation, distance to roads, distance to patches of hosts, and presence of susceptible species. The use of this approach currently requires an analyst with not only GIS skills, but also substantial programming skill. Also, assistance may be needed from algorithm developers to modify code. Model output is a map of the potential location of the invasive species. The spatial scope includes all of California, and the map unit is a 1-km grid cell for a single time. Two regional databases are used as input data: California GAP and climate surfaces from the DAYMET model (<http://www.daymet.org/>). The software used is LIBSVM, which is a library of generic support vector machine functions developed by Chang and Lin 2001, as cited by Guo and others 2005. In this approach, risk is calculated only as potential presence of the disease. There are some probabilistic components, but many sources of uncertainty are not quantified.

This approach would be useful for regional probabilistic risk assessments because it is a generic machine learning technique applied to niche modeling. Thus, it could be used

for invasive plants, insects, diseases, and possibly other stressors. One-class SVMs appear particularly attractive because they are statistically based and unbiased and theoretically optimum, unlike some other machine learning methods and don't require a lot of model tuning. A weakness of the approach, at least for many potential users, is dependence on a library of computer code functions rather than a more mature and user-friendly software package, and assistance may be required from the library developers to apply the functions in an analysis. This approach also does not account for time, nor does it incorporate spatial processes such as dispersion. It may be difficult to specify weights for each variable. Like all niche models, it is dependent on data quality, and there will likely be issues of spatial support and spatial scaling.

Discussion

The issue of invasive species is large and complex because there are thousands of potential invasive species and constant movement of plants, plant material, pests and pathogens, in addition to established invasive species. It seems clear that the most cost-effective approach is to control invasive species very early in the process of transport from the native range and entry to the United States. This issue has received national recognition as an important threat and should be addressed at the national scale (NRC 2002). Increased international trade is exacerbating the problem, and despite this increase, the budget for APHIS, the first line of defense, has been decreasing in recent decades as a function of the volume of imported material (D'Antonio and others 2004).

Despite the lack of complete data sets and complete information about the biology and ecology of invasive species, it is feasible to develop risk analyses of invasive species at the regional scale that should provide information useful for land managers. Even a semiquantitative rule-based approach can help to identify locations that contain susceptible host species for specific pathogens or insect pests and where propagules are more likely to enter, based on the current locations of the invasive species and methods of spread (for example, Meentemeyer and others 2004, Nowak and others 2001). As discussed above, the use

of regional forest inventory data and detailed vegetation mapping based on these and other data provide an important starting point for regional risk assessments of invasive species.

A broad range of niche modeling approaches are useful because they can use data from museum collections in other countries to make estimates of potential new range areas in the United States. Such data provide information about the fundamental niche of the organism, although this information must be evaluated critically by scientists skilled in taxonomy and biogeography and applied with care (Graham and others 2004). The GARP approach would be useful for regional assessments because a software package is available specifically to apply this method to niche modeling. However, other approaches such as support vector machines and GAM approaches may be less biased and provide more optimal solutions (Elith 2006, Stockwell 2005).

As compared to predicting the fundamental ecological niche of a species, predicting the rate of long distance movement is much more difficult because such events are rare, may not be well understood, and may be affected by human behavior. The approach demonstrated recently by Bartell and Nair (2004) to examine pest establishment and spread could be expanded and adapted to provide quantitative estimates of spread beyond an initial port of entry. There is a large body of work in the spatial ecology literature addressing various aspects of the spread of populations and, more generally, the role of space in structuring populations and metapopulations (Tilman and Kareiva 1997). In recent years, there has been an increase in the number of publications using empirical data in conjunction with modeling approaches to predict the spread of invasive plant species. This process is complex because of the rare, but crucial events of long-distance transport, including movement from the native range to the United States. Whereas simple diffusion models may be useful in some instances, the issue of long distance transport by human vectors needs to be addressed (Hastings and others 2005). Some of the methods discussed above included estimates of spread. One such analysis to assess the risk posed by Asian long-horned beetle combined the GARP niche modeling approach with a simple model of spread from likely ports of entry (Peterson

and others 2004). This approach makes predictions quite different from those based on analysis of species host range, as discussed above (see “Nowak Host Range”).

Models based on fundamental biological and physical processes, such as population demographics and movement of organisms, generally are preferable to correlative statistical approaches. This does not mean that correlative approaches are not valuable for probabilistic regional risk assessments. They may be useful first steps for regional analysis (for example, to quantify the overlap in spatial distribution of stressors and ecological receptors throughout the Western United States). Correlative models such as that of Dark (2004) may be extended with some confidence beyond the range of available data because they use predictor variables that represent physical and biological processes. For example, the distribution of nonindigenous and invasive species was found to be similar, because both must pass through the same environmental filters or stages. The approach of using data on locations of all nonindigenous species to predict the occurrence of much rarer problem-invasive species may be quite useful because the correlation is based on the key processes of human-influenced transport, establishment, reproduction, and dispersal of propagules. In such cases, statistically based extrapolations from existing data should be quite credible and useful. In addition to extrapolating from all nonindigenous species to only invasive species, future environmental scenarios might be developed to predict future risks. For example, one could extrapolate to future conditions with more roads or a higher traffic volume, if the number of nonindigenous species in a region can be predicted based on some measure of the transportation network (Larson 2003, McKinney 2002) or other environmental factor. A future scenario of new road development or greater traffic or both on existing transportation networks could be developed based on planned State and Federal transportation projects. This scenario could be used to predict the subsequent increase in occurrence of nonindigenous species and invasive species.

Risk assessment for invasive species will be most useful if it helps provide information about the degree of potential harm, or damage. For certain invasive plant species, especially serious and common weeds of crop and

rangelands, damage can be quantified in economic terms. However, it can be difficult to quantify the ecological effects of many invasive species, especially for effects on wildlands. For example, it has been assumed that purple loosestrife is a serious threat to wetlands in the Northeastern United States, and considerable effort has been made to eradicate it. However, an analysis of ecological effects found little evidence for damage to wetlands (Hager and McCoy 1998), although one recent publication did find some evidence that it can reduce native plant diversity (Schooler and others 2006). The lack of evidence of severe ecological effects in wildland ecosystems does not mean that such effects don't exist. Rather, such a lack of evidence may indicate a lack of research on wildland ecosystem effects and the difficulty in quantifying such effects in wildland ecosystems as compared to highly managed ecosystems such as agricultural row crops. This difficulty in assessing economic damage of invasive species has been recognized as a key challenge for research (Andersen and others 2004a). Despite the challenge, such efforts may be useful, as they may provide evidence that even large expenditures required for removal of invasive species may provide a valuable economic return. For example, it has been estimated that the costs of eradication of tamarisk throughout the Western United States would be fully recouped within 17 years with continued ongoing benefits beyond that time (Zavaleta 2000).

As for any regional stressor, the use of multiple models and a weight of evidence approach would help to increase confidence in predictions of ecological risks from invasive species. As discussed above, two approaches to predicting the risk of Asian long-horned beetle throughout U.S. forests make quite different predictions because they focus on different stages in the process of establishment and spread. All models have some level of uncertainty both in the data used to drive the model and in the calculations made within the model. A focus on uncertainty as an important type of information is crucial for meaningful assessments of invasive species risk. There is strong evidence of the potential for invasion and damage to occur for certain species such as those already on lists of noxious weeds. The strongest predictor for a species is if that species is already

an invasive species causing substantial damage in another part of the world. For these species, there is generally quite a bit of information about aspects of their life history that are important for predicting risk, such as host range, reproductive potential, and phenotypic plasticity. However, for other species there is little or no information. For example, the causal agent of sudden oak death in California was only discovered because of unusual mortality and morbidity in California live oaks. Investigation revealed a new species; thus, there was virtually no information about the ecology of the species such as host range, climatic requirements, and reproductive potential. Until such information began to be gathered, it was not possible to make any meaningful prediction of invasiveness or ecological risk.

Finally, risk assessments will not be useful unless they provide guidance for management. Land managers could benefit in particular from regional risk assessments that provide information about potential future risks. Invasive species management should be addressed at multiple spatial scales such as:

1. Reducing importation of new species at border crossings and ports.
2. Conducting national and regional mapping of locations of invasive species.
3. Developing procedures to reduce long-distance transport if possible.
4. Developing local procedures to reduce movement of invasive species.

Because many invasive species become established along roadways and trails, it may be easier to locate and eradicate them before they spread. However, costs of eradication can be very high, and the most cost-effective approaches will be at the national and regional scale, rather than the scale of a single national forest. Quantitative approaches to estimate the costs and benefits of management options are needed. The feasibility of estimating such costs has been demonstrated (Bartell and Nair 2004, Zavaleta 2000), but much more work is required. Developing such estimates by bringing together risk assessors and land managers should be considered in developing regional risk assessments that will help focus on key issues for management.

In summary, we offer the following suggestions to be considered when selecting modeling approaches for probabilistic risk assessment for invasive species at the regional scale:

1. Define management options and formulate the risk problem definition at the same time so that predictions will be useful for making management decisions.
2. Ecosystems are spatially explicit, so use spatially explicit data, such as vegetation type, topography, stream networks, and elevation.
3. Use both socioeconomic and ecological information.
4. Do not assume that the initial conditions of a landscape can all be captured by a few regionalized variables because of the large role that site history often plays in shaping future dynamics.
5. Whenever possible, make quantitative predictions of risks rather than using ranks (such as low, medium, and high). Ranked values can lead to erroneous interpretations because it may not be clear what is meant by a high risk and also because of uncertainty about what happens at the boundaries of the rank categories.
6. Quantify important spatial and nonspatial sources of data uncertainty and address these uncertainties in the analysis.
7. Quantify important sources of uncertainty in model equations, including aggregation and scaling issues, and address these uncertainties in the analysis.
8. Whenever feasible, use multiple models based on different approaches and data.

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Pests/Biota

Case Studies

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Developing and Validating a Method for Monitoring and Tracking Changes in Southern Pine Beetle Hazard at the Landscape Level

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Abstract

The objective of this research project is to develop and validate a method for using satellite images and digital geospatial data to map the distribution of southern pine beetle (SPB) habitats across the pinelands of east Texas. Our approach builds on a work that used photo interpretation and discriminant analysis to identify and evaluate environmental conditions suitable for SPB infestation. Because current implementations of Billings and Bryant's method by the Texas Forest Service (TFS) use manual photo interpretation, they are relatively costly, labor intensive, and require sampling. Satellite imagery and geographic information system (GIS) technology present potential means to reduce operational costs and improve accuracy. Here we report the principal results of our work in a pilot area of east Texas, specifically: (1) development and integration of satellite and digital inputs into the Billings and Bryant model, (2) accuracy assessment of model inputs, (3) validation of the model adaptation through comparison of satellite-derived SPB hazard maps to operational maps produced by TFS, and (4) revalidation of the model through comparison of satellite-derived SPB hazard maps to known locations of SPB infestations. Collectively, the results point to the considerable potential of satellite imagery and automated analysis techniques to produce timely, accurate, and cost-effective maps of SPB hazard at the landscape level.

Keywords: *Dendroctonus frontalis*, GIS, hazard rating, remote sensing, risk assessment, satellite data, Texas.

Introduction and Background

The southern pine beetle (SPB), *Dendroctonus frontalis* (Coleoptera: Curculionidae: Scolytidae), is one of the most destructive insect pests of pine forests in the Southern United States, Mexico, and Central America (Thatcher and others 1980). The beetle's range extends from New Jersey to Texas and from New Mexico and Arizona to Nicaragua. Because populations build rapidly during periodic outbreaks and large numbers of trees are killed, this insect generates more concern among managers of southern pine forests than any other insect pest. In the Southern United States, average annual losses may exceed 100 million board feet of sawtimber and 20 million cubic feet of growing stock (Price and others 1998).

Southern pine beetle outbreaks have increased in frequency, severity, and distribution during the past 30 years. Preventive silvicultural practices offer the most promising and long-lasting means of reversing this trend (Belanger and others 1993, Nebeker and others 1985). If pine stands are weakened by drought, flooding, lightning strikes, careless logging, or overcrowding, they become more susceptible to attack by the beetle (Blanche and others 1983, Hicks and others 1980). Mature trees in pure, dense stands have long been considered most susceptible to SPB attack, but, in recent years, unthinned pine plantations have increasingly supported SPB infestations (Cameron and Billings 1988). Trees less than 5 years of age or less than 4 inches in diameter are seldom attacked. Dense pine stands also are more likely to suffer extensive losses from the expansion of established SPB infestations in the absence of direct control (Hedden and Billings 1979).

The most practical approach to minimizing timber losses and avoiding costly short-term suppression projects is to maintain forests in a vigorous, healthy condition (Belanger 1980, Hedden 1978). To manage SPB populations

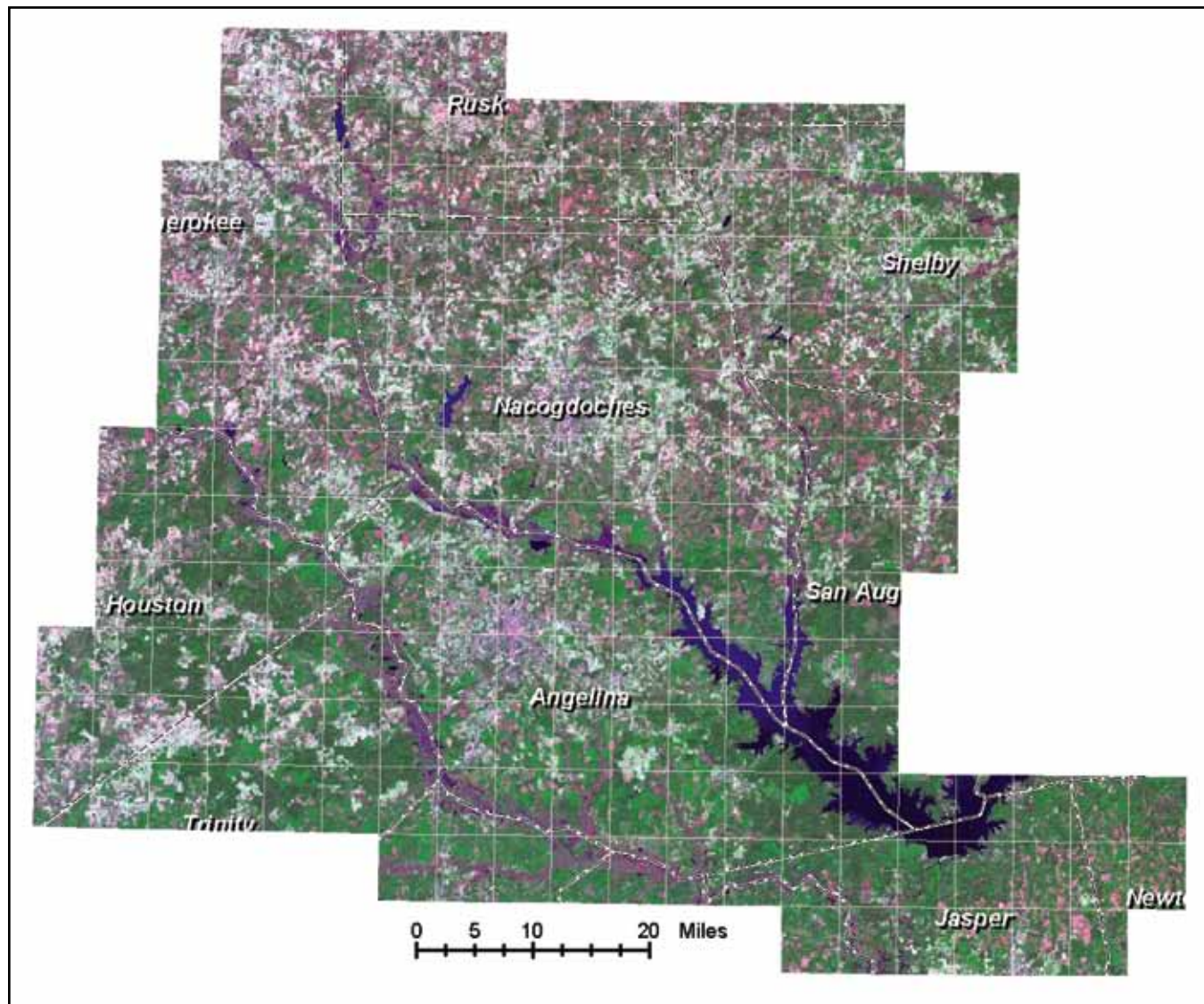


Figure 1—Location of the project area in east Texas. Boundaries of counties and 18,000-acre grid blocks are indicated in white.

more effectively, foresters need reliable means of predicting where infestations are most likely to occur. Once this capability is developed, areas where beetle-caused timber losses are likely can be identified and managed through long-range plans (Peterson 1984), silvicultural manipulations (Belanger and Malac 1980, Belanger and others 1993) or more responsive direct control tactics (Swain and Remion 1981), or all. Several practical stand hazard rating systems have been developed that utilize easily measured stand and site factors (basal area, tree age or height, growth rate in the last 5 years, land form, etc.) to ascertain susceptibility

to SPB at the stand level (Hicks and Mason 1982, Hicks and others 1980, Lorio and others 1982, Mason and others 1985). Identification of SPB hazard at the landscape level, however, has received much less attention.

A system for mapping SPB hazard at the landscape level using aerial photography has been developed and implemented by the Texas Forest Service (TFS) on an 18,000-acre grid (Billings and Bryant 1983, Billings and others 1985). The rating system uses conventional photo-interpretive methods to describe host presence, coverage, density, and site conditions within 30-acre photo plots.

Table 1—Independent variables used in the Billings and Bryant (1983) hazard model

Variable	Description
A	Nonhost (open land, hardwoods, water)
B	Young pine (less than 15 years of age)
C	Pine host, < 70 percent coverage, < 80 percent crown closure, other terrain
D	Pine host, < 70 percent coverage, < 80 percent crown closure, bottomland
E	Pine host, < 70 percent coverage, ≥ 80 percent crown closure, other terrain
F	Pine host, < 70 percent coverage, ≥ 80 percent crown closure, bottomland
G	Pine host, ≥ 70 percent coverage, < 80 percent crown closure, other terrain
H	Pine host, ≥ 70 percent coverage, < 80 percent crown closure, bottomland
I	Pine host, ≥ 70 percent coverage, ≥ 80 percent crown closure, other terrain
J	Pine host, ≥ 70 percent coverage, ≥ 80 percent crown closure, bottomland

Twenty circular photo plots, equally spaced in five rows and four columns, provide a 3-percent systematic sample of host conditions within each grid block.

An initial hazard map for east Texas based on 1981-83 aerial photography covered 656 grid blocks (11,808,000 acres) and was validated using subsequent SPB detection records (Billings and others 1985). In 2003-04, as part of the ongoing SPB prevention project, the east Texas hazard map was updated using 1996 color infrared photography. The updated map identified 16 grid blocks (2 percent) as extreme hazard, 92 (12 percent) as high hazard, 291 (37 percent) as moderate hazard, 280 (35 percent) as low hazard, and 117 (15 percent) as very low hazard.

Although the east Texas hazard maps have been valuable for predicting where SPB outbreaks are most likely to occur and for targeting prevention programs, their production process has three limitations that have prevented widespread adoption of this protocol. These limitations are:

- **Expense:** Creating a SPB hazard map across east Texas (14.3 million acres) requires high resolution color infrared imagery and procedures to digitize and orthorectify the imagery. Furthermore, photo interpretation must be performed by trained technicians. All three requirements are very costly.
- **Accuracy:** Though the aerial imagery is analyzed systematically, the manual interpretation process requires sampling and frequent judgment calls, both of which introduce inaccuracy.

- **Frequency:** Presently, the time and expense of aerial photo collection and interpretation limit the extent and update frequency of SPB hazard maps. If the process were relatively inexpensive and automated, hazard maps could be generated frequently over larger areas.

To address these limitations, the Texas Forest Service (TFS) and Forest One (now Lanworth, Inc.), with the support of the USDA Forest Service Southern Research Station, began a project to investigate the potential of satellite imagery and digital image processing methods to lower the costs and improve the accuracy of the operational east Texas SPB hazard maps. Here we report the principal results of this investigation in a pilot area of east Texas (Figure 1), specifically:

1. Development and integration of satellite and digital inputs into the Billings and Bryant model.
2. Accuracy assessment of the model inputs.
3. Validation of the model adaptation through comparison of satellite-derived SPB hazard maps to operational maps produced by TFS.
4. Revalidation of the model through comparison satellite-derived SPB hazard maps to known locations of SPB infestation.

Methodology

The Billings and Bryant model of SPB hazard takes the form of the discriminant function:

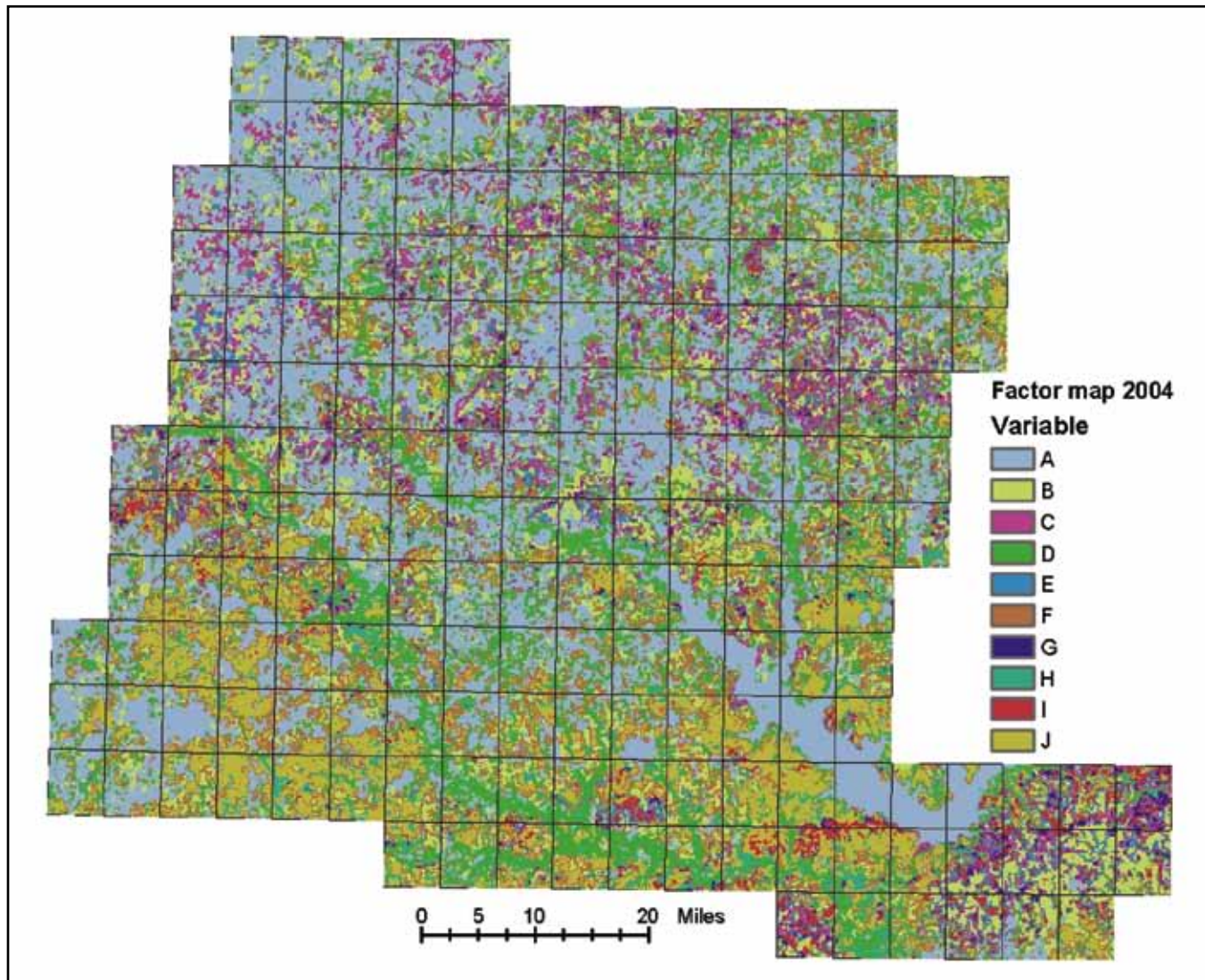


Figure 2—Distribution of hazard factor combinations (habitat types) across the project area (December 2004).

$$DS = -1.35 - 0.108(A) + 0.135(D) + 0.330(E) + 0.404(F) + 0.305(I) + 0.271(J)$$

where the values of the discriminating variables A, D, E, F, I, and J are the numbers of photo plots in a grid block that have the combination of site/stand factors listed in Table 1. Thus, if five photo plots in a given grid block fall on water or agricultural land (factor combination A) and seven others fall on dense, old pine (factor combination I), the values of variables A and I are 5 and 7, respectively. Because only 20 photo plots are analyzed per grid block, the possible value of each variable ranges from 0 to 20. Note, however, that the values of the discriminating variables will not necessarily

sum to 20 as not all site/stand factor combinations appear in the discriminant function.

The challenge of applying satellite data to the determination of SPB hazard at the grid block scale lies in replicating the process of photo interpretation used by TFS without violating the assumptions and conditions of the discriminant function.

The foundation of our approach to SPB hazard mapping is Forest One's Forest Age Map product, a raster map based in Landsat imagery in which each 28-m (30.6 yd) cell (pixel) is classified into one of four forest types (softwood, hardwood, mixed, nonforest) and in which all softwood

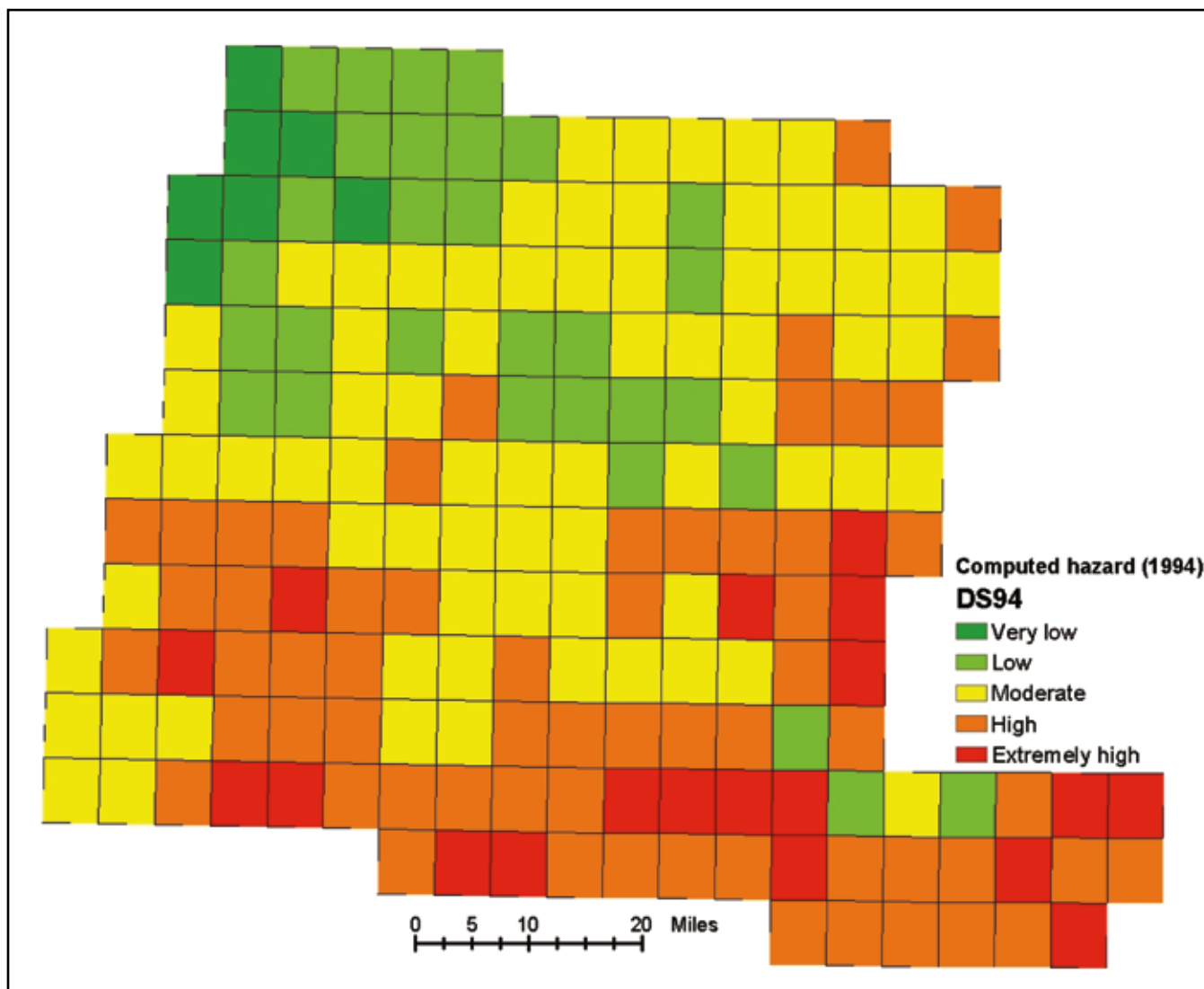


Figure 3—Grid block hazard ratings for 1994 derived from satellite and digital geospatial data.

pixels are further classified into 3-year age classes (e.g., 0 to 3 years, 7 to 10 years, etc.). To develop a SPB host map for the year 2004, Forest One recoded its Forest Age Map, classifying all hardwood, nonforest, and softwood younger than 15 years as nonhost and all softwood older than 15 years as host. To compare satellite-derived SPB hazard to existing TFS hazard maps from 1996 and SPB spot data from 1989 to 1993, Forest One also prepared Forest Age Maps and host maps for 1994 and 1990.

Because the TFS hazard rating protocol considers the percentage of host pine within a 30-acre (12.1 ha) photo plot rather than the predominant host type in a 28-m (30.6

yd) (12.1 ha) pixel, the host maps must be transformed to represent varying percentage of host pine across the study area. This was accomplished by recoding the host maps so that pixels classified as nonhost and young pine have a value of 0, and pixels classified as pine host have a value of 1. A 13 x 13 average filter was then passed over the recoded maps, replacing each pixel by the arithmetic mean of its neighborhood. A 13 x 13 matrix of 28-m (30.6 yd) pixels has an effective area of 32.7 acres (13.2 ha), and the resulting pixel value will therefore estimate the proportion of host pine in a photo plot-sized area centered on each pixel.

Table 2—Comparison of satellite (Forest One 1994) and photo-interpreted (TFS 1996) hazard class predictions. Agreement within 1 hazard class is 89 percent

Texas Forest Service hazard class	F1 hazard class					Total	
	Extreme	High	Moderate	Low	Very low		
	Extreme	4	1	0	4	0	5
	High	5	11	9	5	0	30
	Moderate	5	13	16	27	5	66
	Low	1	5	13	41	21	81
	Very low	0	0	0	7	1	8
Total	15	30	38	80	27	190	

	N	Percent
Agreement	73	38 percent
F1 underestimate (1 class)	58	31 percent
F1 overestimate (1 class)	38	20 percent
Disagreement	21	11 percent

As a further qualification of site/stand conditions, the TFS hazard rating protocol measures host density as a function of canopy closure in areas of each plot containing host pine. As a proxy for host density, we selected a vegetation index computed from the red, near-infrared, and midinfrared bands of Landsat TM imagery acquired in 1990, 1996, and 2004. The vegetation index, called NDVIc, uses the distinctively high reflectance of green vegetation in the near-infrared wavelengths relative to the red and middle-infrared wavelengths to map the relative distribution of green biomass across the project area. Furthermore, because soils tend to reflect strongly in the middle-infrared wavelengths, middle-infrared reflectance is negatively correlated with canopy closure. NDVIc is a unitless quantity that varies between -1 and 1. To restrict this measurement to areas of host pine, the host maps were used to assign null values to areas of young pine and nonhost. A 13 x 13 average filter was then passed over the NDVIc maps to estimate for each

pixel the average density of host pine in a photo plot-sized area centered on that pixel.

Current TFS protocol distinguishes between pine stands with less than 80 percent canopy closure and those with 80 percentage of closure or greater. To determine the actual relationship between NDVIc and percentage of canopy closure, we partitioned several 1-m color-infrared digital orthophoto quarter quadrangles from 1995 into canopy and noncanopy pixels. We then computed the proportion of 1-m (39.4 in) canopy pixels present within the area covered by each 28-m (30.6 yd) pixel from a 1994 Landsat image. The average NDVIc value of all Landsat pixels containing more than 80 percent crown elements (NDVIc = 0.425) was then selected as the threshold for 80 percent canopy closure.

The final site/stand factor considered under the TFS protocol is landform, expressed as bottomland or other terrain. To determine the landform most characteristic of the photo plot-sized area centered on each 28-m pixel within the

Table 3—Observed rates of SPB^a infestation (1991-93) by satellite-predicted hazard class (1990)

Hazard class	New spots (1991-93)	N Grid blocks	Infestation rate (spots/class/grid block)
Extreme	159	15	10.60
High	409	28	14.61
Moderate	652	58	11.24
Low	583	81	7.20
Very low	21	8	2.63

^a SPB = Southern pine beetle.

study area, the National Elevation Data set digital terrain model was resampled to match the grid resolution of the Forest Age Map. Percentage of slope was then computed for each grid cell, and a 13 x 13 average filter was passed over both the slope and elevation maps. Based on conversations with photo interpreters at TFS, bottomland was operationally defined as those areas having an average elevation less than 90 m (98.4 yd) and an average slope less than 3 percent.

Once all relevant site/site-stand factors were estimated using remotely sensed inputs, a series of rules was used to assign each pixel to 1 of the 10 factor combinations expected by the Billings and Bryant model (Figure 2). Once a site/stand factor combination was assigned to each 28-m grid cell, the percentage coverage of each factor combination was computed for each of the 190 grid blocks in the study area. This percentage value was then multiplied by 0.2 to scale percentage cover to the 0 to 20 range expected by the discriminant function. The discriminant score was then computed for each grid block for the years 1990, 1994, and 2004, and scores were assigned to hazard classes based on breakpoints used in the 2003-04 TFS update. The satellite-derived hazard map for 1994 is shown as Figure 3.

Accuracy of Model Inputs

To determine the accuracy of the satellite-derived inputs, we compared our maps of host type and coverage to reference data on managed pine stands of known age. The reference data were distributed widely across the project area and constitute a 5-percent sample by area. Analysis shows that Forest One's Forest Age Map classifies pine with at

least 80 percent accuracy and can distinguish between pine greater than 15 years and pine younger than 15 years with 82 percent accuracy. We also compared our measurements of crown closure and landform to operational photo-interpreted measurements made by TFS technicians and found similar levels of agreement. Our conclusion, therefore, is that the satellite-derived inputs into the model are sound.

Comparison of Satellite-Derived and TFS Hazard Maps

The principal result of our work so far has been SPB hazard maps for 1990, 1994, and 2004. The accuracy of the 2004 map is currently being assessed through comparison to operational measurements of hazard factors by TFS. The accuracy of the 1994 map has been assessed through comparison to the 1996 hazard map produced by TFS (Table 2).

The comparison shows that the 1994 map correctly predicted the hazard rating of 38 percent of the grid blocks. The maps predicted a further 51 percent of the grid blocks within 1 hazard rating class. Given some uncertainty about the accuracy of the TFS reference map, we propose to treat all agreements within 1 hazard class as correct predictions and offer 89 percent as the final accuracy of the satellite-derived hazard map. In general, the satellite-derived map tends to underpredict hazard ratings slightly. Forest One and TFS are working to explain and improve the correspondence between the two hazard rating systems.

Comparison of Satellite-Derived Maps and Historic SPB Infestation Data

As a further check on the accuracy of the satellite-derived hazard map and the validity of the underlying discriminant function, TFS supplied data on the location of SPB infestations within the study area during the years 1989-93. These data were organized as SPB infestation counts for each 15-arc second grid block within the study area. To compare these data to the 1990 satellite-derived hazard map, we selected all 15-arc second grid blocks that were infested in 1991, 1992, or 1993 but not in 1989 or 1990. This allowed us to create a map of new infestations since 1990. These infestations were presumably related to landscape conditions

represented by the 1990 satellite map rather than to lingering or renewed infestations from prior years.

To allow comparison of new infestations to satellite-predicted hazard class in 1990, we aggregated all new infestations to the 18,000-acre grid block level and summarized these counts by hazard class (Table 3). Because the area (number of grid blocks) of each hazard class is not constant over the study area, we divided the count of new infestations by the number of grid blocks in each hazard class that were either newly infested in 1991-93 or were not infested at all. This normalized measure was interpreted as the average infestation rate for each hazard class.

Our results reveal a qualitatively strong, positive correlation between the observed infestation rate and the model-predicted hazard class. The mean rate of SPB infestation declined from nearly 15 infestations/grid block in the high hazard class to fewer than 3 infestations per grid block in the very low hazard class. The only observed anomaly is that the infestation rate is slightly lower than expected for the extreme hazard class, perhaps indicating limited resolving power of either the satellite data or the underlying model. This anomaly is currently being investigated by Forest One and TFS.

Conclusions

From the investigations and analyses reported here, we conclude that satellite imagery, together with ancillary digital geospatial data and automated processing techniques, presents a powerful and cost-effective tool for operational mapping of SPB infestation hazard at the landscape scale.

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Previsual Detection of Two Conifer-Infesting Adelgid Species in North American Forests

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Abstract

The balsam woolly adelgid, *Adelges piceae*, and hemlock woolly adelgid, *A. tsugae* (Homoptera: Adelgidae), are invasive pests of coniferous forests in both the Eastern and Western United States. Balsam woolly adelgid is capable of attacking and killing native North American firs, with Fraser fir (*Abies fraseri* (Pursh) Poir.) in the East and subalpine fir (*A. lasiocarpa* (Hook.) Nutt.) in the West being particularly susceptible to infestation. Hemlock woolly adelgid is capable of infesting native hemlocks and is a serious pest in forests of the Eastern United States where it is causing significant mortality to both eastern (*Tsuga canadensis* (L.) Carr.) and Carolina hemlock (*T. caroliniana* Engelm.). Infestations by either of these insects may take several years to kill the host tree. Damage by hemlock woolly adelgid frequently causes needles to discolor from deep green to grayish green. Discoloration of needles is also one of the symptoms used to diagnose infestations of balsam woolly adelgid. Traditional methods for assessing damage by these adelgid species include field surveys and aerial detection surveys. However, because infestations frequently occur in remote locations and can take years to build up, stand damage may accrue prior to visual detection of the infestations. Branch-level, spectral data of the foliage from trees were collected for several categories of infestation. In the Western United States, data were collected from subalpine fir infested with balsam woolly adelgid in northern

Idaho. In the Eastern United States, data were collected from eastern hemlock in western North Carolina. Trees were sampled using a hand-held spectroradiometer. The measured radiance spectra were converted to percentage of reflectance and comparisons made between the infestation categories. Separation of the infestation levels occurred in a progressive pattern moving from noninfested to newly (or lightly) infested to heavily infested trees. Results suggest that previsual detection of this group of invasive insects may be possible with appropriate spatial and spectral sensor resolution.

Keywords: Host resistance, hyperspectral, insect detection, multispectral, remote sensing.

Invasive Adelgids in North American Conifers

Adelgids (Homoptera: Adelgidae) are small insects with piercing and sucking mouth parts. They have a white woolly covering that is secreted over the body. There are several native adelgid species within North America such as the Cooley spruce gall adelgid (*Adelges cooleyi*), and some of these can cause growth loss in trees or cause trees to reach economic injury levels under some conditions. However, the two adelgid species that are currently causing the most economic and ecological impacts within North America are the introduced balsam woolly adelgid (*A. piceae*) and the hemlock woolly adelgid (*A. tsugae*), both of which are established in both the Eastern and Western United States.

Balsam Woolly Adelgid: Hosts and Biology

Balsam woolly adelgid is native to the fir forests of central Europe and was introduced into the United States around 1900. The life cycle of the balsam woolly adelgid consists of the egg, three larval instars, and the adult (see Hain 1988 for a more thorough description). The only life stage capable of movement is the first instar larva (termed the crawler) that, upon locating a suitable feeding site, inserts its stylet into the bark and transforms (without molting) into a nonmobile phase, after which the insect is permanently attached to the host tree. As the female feeds, she secretes a dense woolly covering that ultimately covers the entire insect.

The crawler stage does not have wings, and between-tree dispersal is a passive process. The adult female produces as many as 248 eggs. These are oviposited within the woolly mass, which acts to protect all of the life stages except the crawler.

All of the true firs (*Abies*) that are native to North America show some degree of susceptibility to the balsam woolly adelgid (Mitchell 1966). The susceptibility ranges from slight for noble fir (*A. procera* Rehd.) and white fir (*A. concolor* (Gord. & Glend.) Lindl. ex Hildebr) to moderate for grand fir (*A. grandis* Dougl. ex D. Don) Lindl., cork-bark fir (*A. lasiocarpa* var. *arizonica* (Merriam) Lemm.), and Shasta red fir (*A. magnifica* var. *shastensis* Lemm.) to severe for subalpine fir (*A. lasiocarpa* (Hook.) Nutt.), Fraser fir (*A. fraseri* (Pursh) Poir.), balsam fir (*A. balsamea* (L.) Mill.), and Pacific silver fir (*A. amabilis* Dougl. ex Forbes). The insect is established on susceptible hosts in the Eastern and Western United States where it is responsible for significant levels of mortality in some stands. Prior studies suggest that there may be some connection between host monoterpenes and attack success by balsam woolly adelgid (Arthur and Hain 1987).

Hemlock Woolly Adelgid: Hosts and Biology

Hemlock woolly adelgid is native to Asia and was first reported in the Pacific Northwest in the 1920s. The adelgid was reported in Eastern North America in the 1950s and Connecticut in the 1980s. The insect is now present in many of the hemlock forests of the Eastern United States, where infestations frequently result in significant mortality to native hemlocks (Souto and others 1995). The hemlock woolly adelgid is a serious pest of Eastern hemlocks and represents a significant threat to the sustainability of native hemlocks (*Tsuga canadensis* (L.) Carr. and *T. caroliniana* Engelm) in the Eastern United States (McClure 1992). Whereas the adelgid is also established in the Western States, it does not appear to be a threat to the western hemlock species (*T. heterophylla* (Raf.) Sarg. and *T. mertensiana* (Bong.) Carr.) at the present time.

Hemlock woolly adelgid has two generations per year in much of its range in the Eastern United States. Only females are present, and the spring generation lays between

100 and 300 eggs. Upon hatching, the crawlers search for suitable feeding sites, insert their stylets and begin to feed. As with balsam woolly adelgid, crawlers become immobile once they settle and begin to feed. When the crawlers reach maturity, two types of adults can form. One type of adult has wings and dies as it searches for the alternate spruce host, which is not present in North America. The other is wingless and capable of laying eggs to produce the next generation.

Conifer Resistance to Insect Attack

There are several hypotheses regarding plant resistance to insect attack that involve the production and allocation of resources within the plant as they relate to the plant's resistance mechanisms. The carbon: nutrient balance hypothesis correlates the production of plant secondary metabolites that are important in determining the relative resistance/susceptibility of the plant with the ratio of carbon to other nutrients within the plant (see Herms and Mattson 1992). The growth differentiation balance hypothesis also views changes in the production and maintenance of plant secondary metabolites as a tradeoff owing to environmental constraints on growth and secondary metabolism (i.e., differentiation) (see Herms and Mattson 1992). The growth differentiation balance hypothesis predicts that under moderate stress, plant growth will be limited, and the production of secondary metabolites such as those important in insect resistance will increase.

Conifer Resistance to Insect Attack: Generalized Response Sequence

Conifer resistance to stem-invading insects has received much attention and involves a generalized, three-step sequence of wound cleansing, infection containment, and wound healing (Berryman 1972, Hain and others 1983). The first step of this response, wound cleansing, is characterized by the production and flow of constitutive resins. The second step of the resistance sequence, infection containment, can be described as a rapid necrosis of cells surrounding the infection site that is accompanied by the development of traumatic resin ducts and an increased concentration of monoterpenes and phenolics in the reaction zone (Cook

and Hain 1986, Raffa and Berryman 1982, Reid and others 1967). The accumulation of terpenes and phenolics in the reaction zone is also accompanied by a decrease in the level of soluble sugars in that zone (Cook and Hain 1986, Wong and Berryman 1977). Wound healing, or formation of wound periderm, is the final step of the resistance sequence. This isolates the wound from the rest of the tree. Wound periderm is located adjacent to the necrotic tissue and protects living tissue from the adverse effects of the dead cells in the necrotic zone surrounding the attack site(s) (Mullick 1977). The three-step resistance sequence requires an expenditure of energy by the tree, and there is typically a resulting change of color (fading) within the tree's foliage.

Fir Response to Stem Attack by the Balsam Woolly Adelgid—

The impact of balsam woolly adelgid infestation on North American firs has been studied extensively over the past several decades. Infestation by the adelgid results in anatomical and structural changes within host tissues that may be the result of salivary excretions from the insect's stylet during feeding. Physically, the xylem tissue of infested trees has higher concentrations of ray tissue (Mitchell 1967, Smith 1967), thickened cell walls, and shorter tracheids (Doerksen and Mitchell 1965). The tracheids have encrusted pit membranes that more closely resemble the pit membranes associated with heartwood (Puritch and Johnson 1971). There is a corresponding reduction in waterflow in infested trees (Mitchell 1967) that puts the tree into a state of physiological drought; this, in turn, reduces photosynthesis and respiration (Puritch 1973) and can ultimately result in tree death.

The damage to the host tree is related to both the size of the tree and the intensity of the infestation. Balsam woolly adelgid infestations in the crown of a tree usually result in gouting of the outer branches (characterized by node or bud swelling or both with a decrease in new growth of the stem and foliage) (Mitchell 1966). Over time, the crown thins, and the foliage fades in color. Balsam woolly adelgid infestations also occur on the stems of trees. In North America, these stem infestations usually kill native firs within 6 years (Hain 1988).

Hemlock Resistance to Attack by Hemlock Woolly Adelgid—

Once hemlock woolly adelgid settles onto a twig, the tree usually suffers needle loss and bud mortality, followed by branch and whole tree mortality (usually within 6 years) (McClure 1991, Shields and others 1995). Foliar chemistry appears to play some role in host susceptibility/resistance to hemlock woolly adelgid, with resistance being related to foliar levels of calcium, potassium (K), nitrogen (N), and phosphorous (Pontius and others, 2006). These authors suggest that higher levels of N and K in the foliage enhance host palatability and, thus, result in increases in the population levels of hemlock woolly adelgid. In addition, soil and foliar chemistry along with landscape position can be used to model hemlock susceptibility to Hemlock Woolly Adelgid (Pontius and others 2009). These hypothesized relationships between foliar chemistry and infestation could be important for early detection of hemlock woolly adelgid infestations because some foliar constituents such as chlorophyll, N, cellulose, and sugar can be accurately estimated using spectral data (Curran and others 2001).

As with other conifers, monoterpenes are major constituents of tree chemistry of hemlocks (i.e., Li and others 2001). These compounds may function in several ways to mediate the interaction between trees and herbivores, but one impact is that they are frequently toxic to attacking insects such as bark beetles (i.e., Cook and Hain 1988) or other arthropods such as spider mites (i.e., Cook 1992). It has been suggested that the monoterpene content of western hemlocks may function as a deterrent to hemlock woolly adelgid (Lagalante and Montgomery 2003). The authors suggest that elevated levels α -pinene, β -caryophyllene, and α -humulene may act as feeding deterrents against hemlock woolly adelgid, and that elevated levels of isobornyl acetate may attract the adelgid.

Importance of Previsual Detection

Minimizing the elapsed time between when a tree becomes infested with an insect and when that infestation is detected can increase the treatment options available to forest managers. Detection of an infestation prior to when the foliage begins to visibly fade should give managers more

time to respond. Active resistance mechanisms by a host tree to insect attack can be energy intensive to maintain and utilize. The decline that occurs within a host following infestation by adelgids may be categorized into various levels as characterized for hemlock infested with hemlock woolly adelgid (Pontius and others 2005) or balsam fir infested with balsam woolly adelgid (Luther and Carroll 1999). Changes in foliar chemistry that are related to tree stress can be manifested in measurable spectral changes within the foliage. Much of the literature with regard to another tree-killing insect, mountain pine beetle (*Dendroctonus ponderosae*), is reviewed by Wulder and others (2006). The review suggests that remotely sensed data is useful for detecting infestations of mountain pine beetle damage and that future experimental work be conducted at several spatial scales.

Spectral Data

Both the spatial resolution (i.e., pixel size) and spectral resolution (the width of the individual spectral wavebands over which plant response is measured) of spectral data, as well as the overall wavelength range examined (some sensors operate through the middle infrared region, some do not), can influence the ability to detect infested trees. Multispectral remotely sensed data types tend to have fewer, wider spectral wavebands and are operationally available from satellite platforms over a wide range of spatial resolution (< 1 to 30 m). Landscape-scale hyperspectral data are less widely available and have a large number of very narrow wavebands. Because most available data sets are acquired from aircraft platforms, these data tend to have spatial resolution on the order of 6 to 20 m. Handheld spectroradiometers with wavelength widths and numbers similar to hyperspectral sensors are often employed in the field and laboratory to study spectral response of canopy components.

Prior Attempts to Use Spectral Imagery to Detect or Delineate Adelgid Infestations

There have been several prior studies related to the detection and classification of trees infested with invasive adelgids. Luther and Carroll (1999) examined several

foliar indices for assessing stress in balsam fir using spectral reflectance data and reported that foliar reflectance decreased consistently with vigor. These authors conducted their work in the laboratory using a fixed-position spectroradiometer. Adelgid infestation was not specifically investigated, but infestation of fir with balsam woolly adelgid does result in tree stress (see Hain 1988). At the landscape scale, hemlock stands were similarly assessed and analyzed for health status using multispectral Landsat Thematic Mapper data (Bonneau and others 1999). The best overall accuracy for classifying stand health based on hemlock woolly adelgid infestation was obtained using the Modified Soil Adjusted Vegetation Index-2. Pontius and others (2005) used hyperspectral data to examine the abundance and early decline of hemlock infested with hemlock woolly adelgid. These authors suggest that wavelengths in the low end of the spectral range may be useful in assessing early stages of decline of hemlock infested with hemlock woolly adelgid. One purpose of our ongoing research is to determine if host decline resulting from infestation by invasive adelgids in multiple tree genera can be evaluated by using similar spectra among the host genera.

Comparison of Hyperspectral Data for the Previsual Detection of Balsam Woolly Adelgid and Hemlock Woolly Adelgid Infestations

Our studies have used hyperspectral data collected at the branch level. Spectral data were collected using a Geophysical Environmental Research Corp. 2600 handheld spectroradiometer with a spectral resolution of 1.5 nm from 350 nm to 1050 nm and a resolution of 11.5 nm from 1050 nm to 2500 nm. In the case of balsam woolly adelgid, we have concentrated on subalpine fir, the primary host of this insect in Idaho. Our studies of hemlock woolly adelgid have concentrated on its primary host in western North Carolina, *Tsuga canadensis*. For both insect-tree pairs, spectral data were collected from trees in various stages of infestation. Five branches were cut from each tree that was examined. Branches were cut from various heights and orientations throughout the canopy of the trees. The branches and foliage were placed on a flat black surface with negligible amounts of measurable radiation, and five measurements per tree

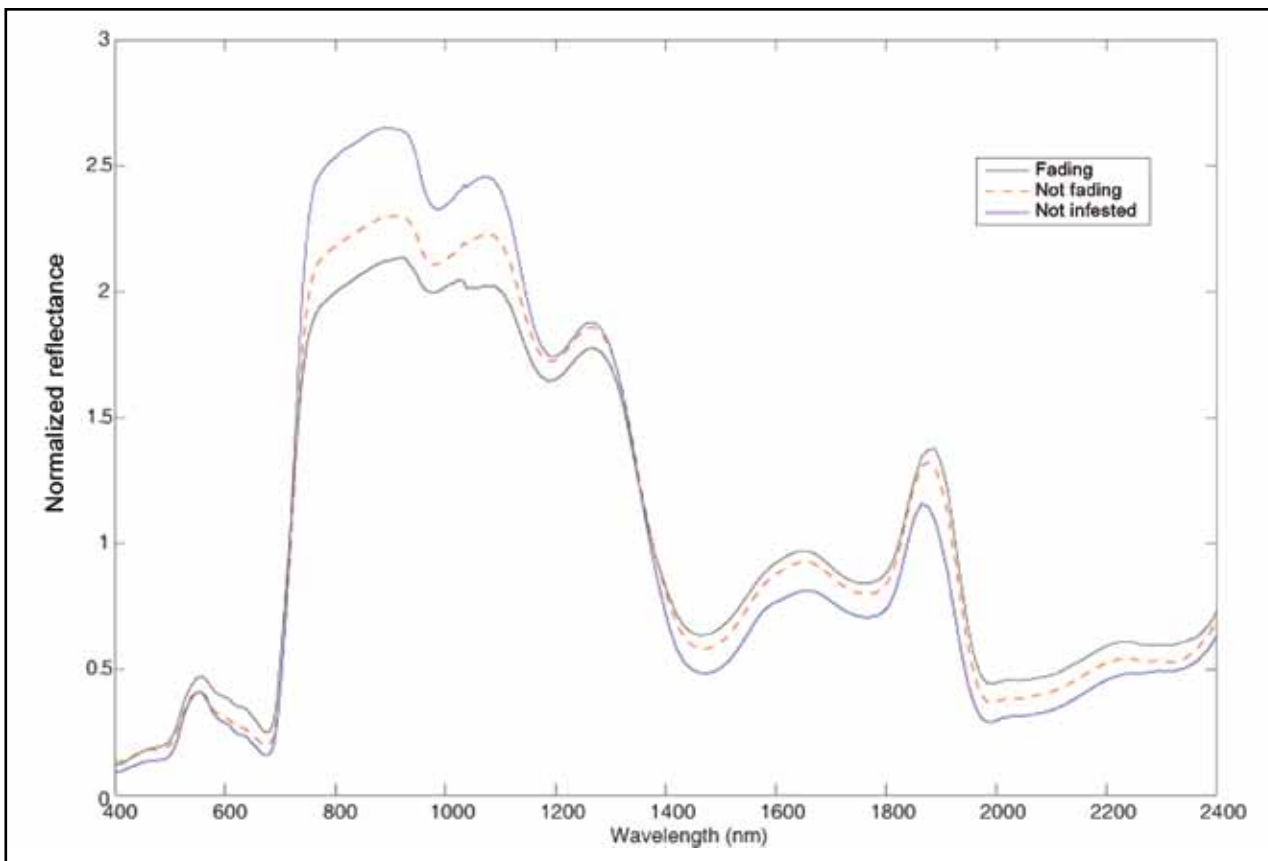


Figure 1—Spectral measurements of subalpine firs, *Abies lasiocarpa*, in Idaho with three levels of infestation with balsam woolly adelgid, *Adelges piceae*. The infestation categories are not infested = blue, infested but with no visible symptoms = red, and infested with visible symptoms = black.

were obtained in an iterative manner, with the foliage being rearranged between each measurement. The radiometer was placed at a height of approximately 50 cm above the branch samples, and measurements were made when the sun angle was within 10° of solar noon. The spectra for these five replicates of branch measurements were averaged to obtain a measure of each tree's reflectance properties. The data for each tree were smoothed using a weighted moving filter, and comparisons were made of the spectral response among infestation classes.

In Idaho, subalpine firs in three infestation categories were sampled. The categories included trees that had no current infestation with balsam woolly adelgid, trees that were infested with balsam woolly adelgid but had no apparent crown fading, and trees that were infested with balsam woolly adelgid and had visible signs of this infestation.

Using Analysis of Variance procedures and the SAS statistical analysis package, significant differences were found among the three infestation categories for some wavelength regions. Our results demonstrated a consistent response in the normalized spectral reflectance curve of subalpine fir, stressed by infestation of balsam woolly adelgid, across the reflectance spectrum shown in Figure 1. More specifically, there is an increased reflectance in the visible region of the reflectance curve (< 700 nm), decreased reflectance in the Near Infra Red plateau (centered around 1000 nm), and increased reflectance in the shortwave infrared region (beginning around 1450 nm) as visual decline becomes apparent. The overall changes in spectra are similar to those reported for other stresses in balsam fir (Luther and Carroll 1999). Multispectral aerial imagery (Landsat and SPOT data) was also collected for areas with active balsam woolly adelgid infestations. Because of the relatively

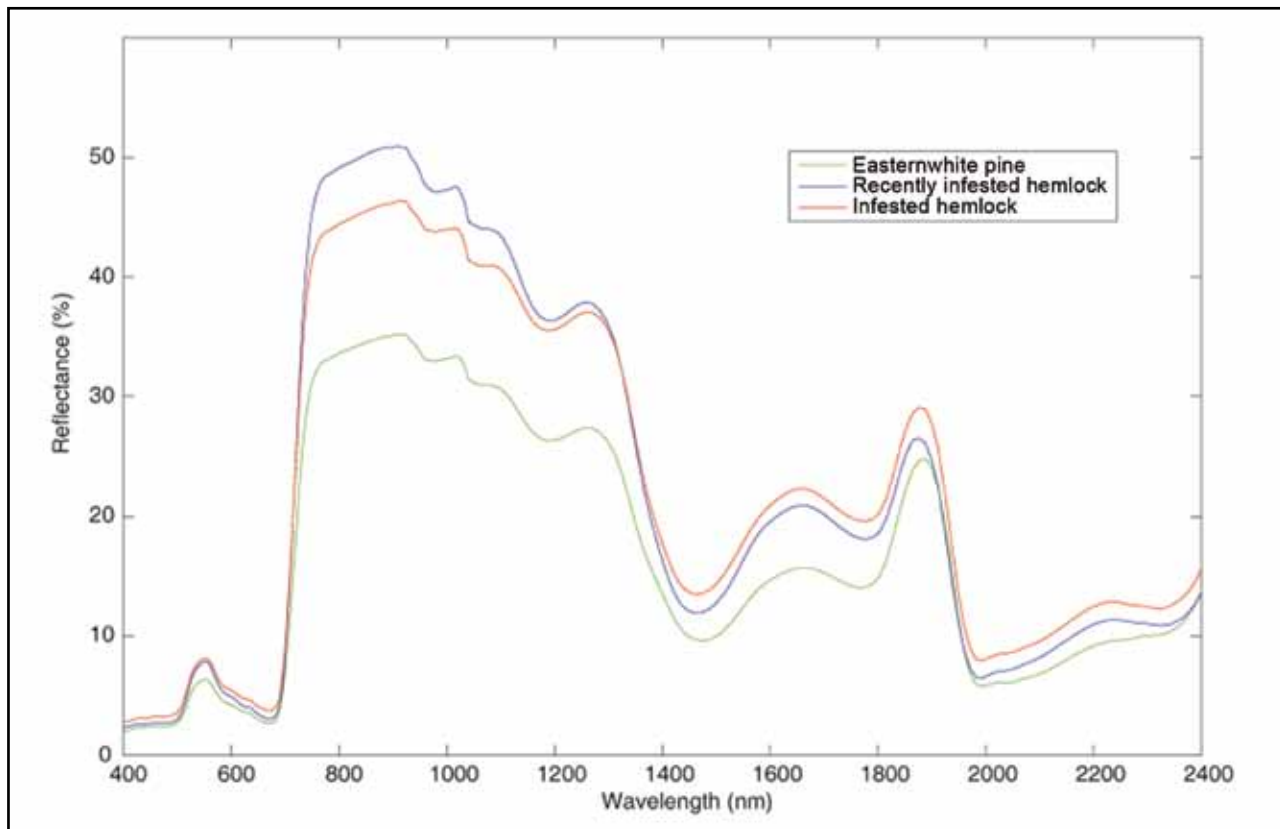


Figure 2—Spectral measurements of eastern white pine, *Pinus strobus*, and eastern hemlocks, *Tsuga canadensis*, in North Carolina with two levels of infestation with hemlock woolly adelgid, *Adelges tsugae*. The spectral categories are eastern white pine = green, recently infested but with no visible symptoms = blue, and infested with visible symptoms = red.

narrow canopy architecture of subalpine fir and the patchy distribution of the species in the areas of data collection, no conclusive results were obtained.

In North Carolina, hemlock trees that were recently infested (within the past year) or that had been infested for multiple years were sampled as was eastern white pine (*Pinus strobus* L.) (the only other conifer present within the stands that we sampled) during June of 2005. No uninfested stands of hemlock were found within the study areas. There were visible differences in the overall spectral measurements between the hemlocks that were recently infested with hemlock woolly adelgid and those that had been infested for a longer period of time (Figure 2). The spectral signature of eastern white pine, the only other conifer present within the stands that could be confused with the hemlocks, differed significantly from both categories of infested hemlock within the stands (Figure 2). The pattern

of decreased spectral values with increasing stress is similar to the decreases measured in subalpine fir infested with balsam woolly adelgid in the Near Infa Red plateau and increases again in the shortwave Infa Red region (Figure 1). The ability to distinguish declining hemlock at the branch level also supports the prior landscape-level investigations of Pontius and others (2005), but larger data sets from a variety of geographic locations are still needed.

Implications for Detection and Delineation of Forest Insect Infestations

The branch-level spectral data for both tree species infested with their specific invasive adelgids were both consistent and in general agreement with the shoot-level spectral changes of balsam fir under various stresses that were measured under laboratory conditions (Luther and Carroll 1999). The measurements were also in general agreement with the results of Pontius and others (2005) who examined

hemlock woolly adelgid at the landscape level. Therefore, the spectral changes that occur with stress are measurable at several scales. The combined results of these studies suggest that spectral data may aid in developing a tool for previsual detection and monitoring of forest decline associated with these adelgid species. However, limitations do exist. One of the primary limitations may be the ability to separate different stressing agents or factors.

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Estimating the Susceptibility to *Phytophthora alni* Globally Using Both Statistical Analyses and Expert Knowledge

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Abstract

Phytophthora alni subspecies *alni* Brasier and S.A. Kirk is a recently hybridized soil and waterborne pathogen causing root and collar rot of species of the genus *Alnus* spp. (alder). It has quickly spread throughout Europe via planting of infested nursery stock and irrigating fields with infested river water. Once introduced, the pathogen spreads naturally by means of streams, floods, and other drainage water. *Phytophthora alni* can also be passively transported with the bare-root nursery stock, as it is able to adhere to and infect fine roots of visually symptomless plants of alder and other tree species exposed to the pathogen.

We used a classification tree on 434 infested and healthy sample points to determine the required conditions for *P. alni* to successfully infest a nonflooded forest site. Sample points had been collected from 2003 through 2006, and a potential distribution surface was created for forested areas in Bavaria. A tenfold cross-validation accuracy of 78 percent was attained. To understand the potential hazard posed by *P. alni* elsewhere in the world, the rules from the Bavarian classification tree were applied along with additional expert knowledge in a multicriteria model to create a global susceptibility surface for *P. alni*.

Keywords: Alder, *Alnus*, classification tree, hazard, pathogen, *Phytophthora alni*, risk.

Introduction

Phytophthora alni Brasier and S.A. Kirk is a host-specific, highly aggressive soil and waterborne pathogen that causes root and collar rot of *Alnus* (alder) spp. All European alder species (i.e., black alder [*A. glutinosa* (L.) Gaertn.], gray alder [*A. incana* (L.) Moench], Italian alder [*A. cordata* (Loisel.) Duby], and green alder [*A. viridis*] (Chaix) DC.) and the North American red alder (*A. rubra* Bong.) are highly susceptible (Jung and Blaschke 2006, Gibbs and others 2003). The susceptibility of other North and South American and Asian alder species is currently unknown. *Phytophthora alni* was shown to be a recent interspecific hybrid between *P. cambivora* (Petri) Buisman and another species closely related to *P. fragariae* Hickman (Brasier and others 1995, 1999, 2004). There are three subspecies of *Phytophthora alni*, with markedly different aggressiveness to common alder (Brasier and others 2004, Brasier and Kirk 2001). The disease was first detected in 1993 in Southern Britain (Gibbs 1995) and has since been confirmed in 12 other European countries and across the United Kingdom (Figure 1) (Brasier and Jung 2003, 2006; Gibbs and others 1999, 2003; Jung and Blaschke 2004, 2006; Schumacher and others 2005; Streito and others 2002) (Orlikowski, L. Pers. comm., 2006. Pathologist, Research Institute of Pomology and Floriculture, Pomologiczna 18, 96-100 Skierniewice, Poland). Moreover, *P. alni* is likely present in Czech Republic, Spain, and Switzerland because typical symptoms and mortality of alders are reported from these countries. The disease occurs mainly along riverbanks, but also in orchard shelterbelts and forest plantations (Gibbs 1995; Gibbs and others 1999, 2003; Jung and Blaschke 2004; Streito and others 2002). Disease symptoms include abnormally small, sparse, and often yellowish foliage and crown dieback (Figure 2). Other symptoms are early and often excessive fructification and tongue-shaped necroses of the inner bark and cambium. Necroses can extend up to 3 m from the stem base and are marked by tarry or rusty spots on the surface of the outer bark (Figure 3) (Gibbs and

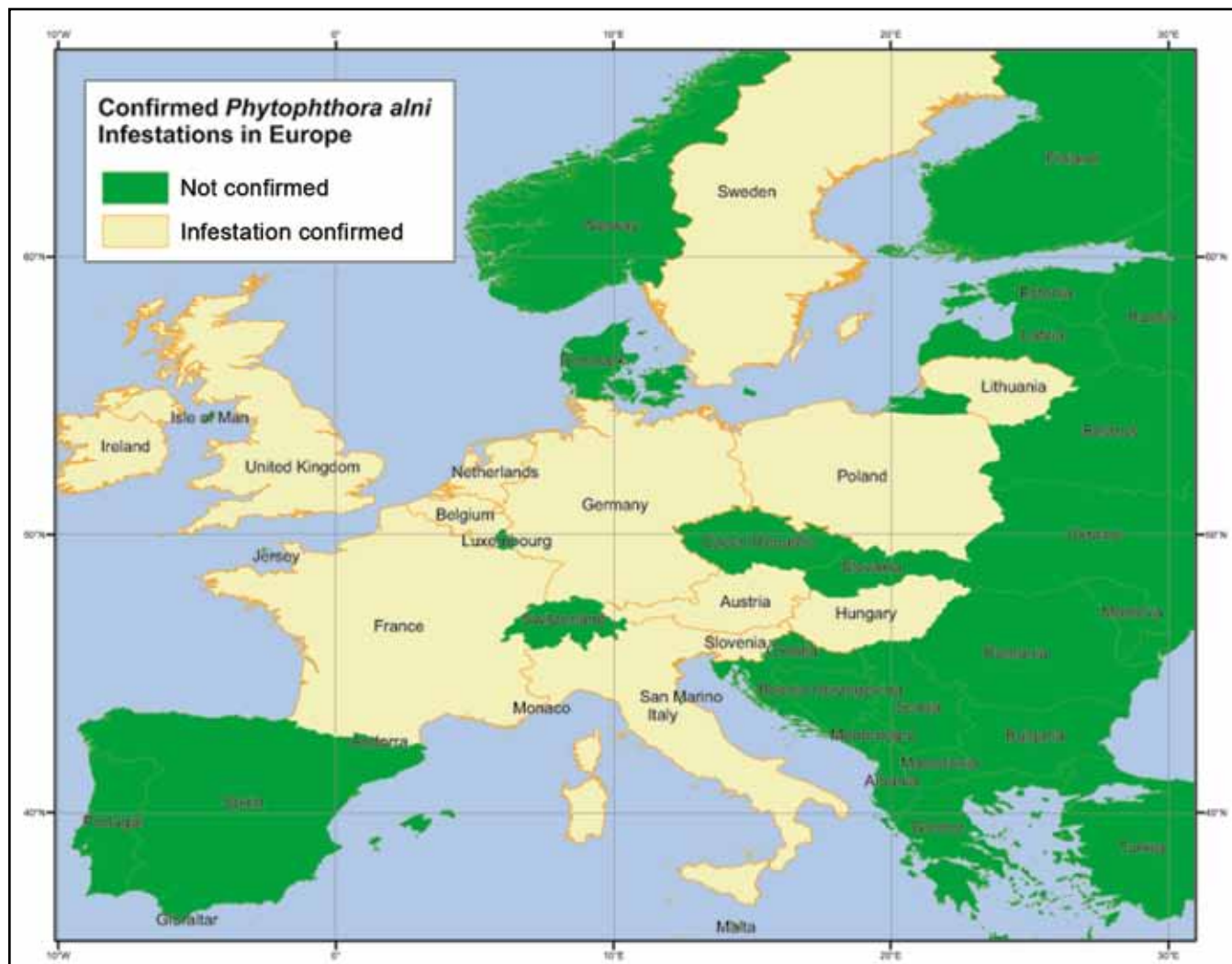


Figure 1—Current distribution of *Phytophthora alni* (yellow).

others 1999, 2003; Jung and Blaschke 2001, 2004) (http://www.baumkrankheiten.com/gallery/docs-en/alder_dieback/index.html). On riparian sites, *P. alni* has caused mortality as high as 70 percent in some locations. Disease incidences of 50 percent and high mortality rates were common (Gibbs and others 2003, Jung and Blaschke 2004, Streito and others 2002). The pathogen was shown to be widespread in alder nursery fields (Jung and Blaschke 2004, Schumacher and others 2005). Infected plants seldom showed symptoms, which made control efforts difficult.

A thorough investigation of disease pathways in Bavaria demonstrated that in most infested river systems, *P. alni* was introduced via infested young alder plantations

established on the riverbanks or on forest sites that drain into the rivers (Jung and Blaschke 2004). Once introduced, *P. alni* spreads downhill with water runoff and downstream with streams and floods. Many infected alders were planted in afforestations of former agricultural land and also on wet sites in woodlands to stabilize steep slopes and banks of white water rivers (Jung and Blaschke 2004). These plantings increased the risk of infestation of riparian sites with the increased length of the river and upstream tributaries.

The rapid proliferation of *P. alni* throughout Europe probably resulted from the increased importance of alders in afforestation activities on wet sites and on former agricultural lands. The combined anthropogenic impacts of



▲ Figure 2—Mature riparian stand of common alder (*Alnus glutinosa*) with sparse, chlorotic and small foliage and crown dieback owing to *Phytophthora alni* root and collar rot.

◀ Figure 3—Mature grey alder (*Alnus incana*), with collar rot caused by *Phytophthora alni*; typical tarry spots at the outer bark and tongue-shaped orange-brown necrosis of the inner bark.

outplanting infected nursery stock and utilizing contaminated river water to irrigate nursery fields were contributing factors (Brasier and Jung 2003, Gibbs and others 2003, Jung and Blaschke 2004). *Phytophthora alni* may also be passively transported on bare-root nursery stock because it is able to adhere to and infect the fine roots of alders as well as adhere to other nonhost tree species exposed to the pathogen.

As short-time control measures, coppicing of infected alder trees and stools is recommended along water courses (Gibbs 2003), but not in infested forest plantations (Jung and Blaschke 2006). Some survivors in highly infested common alder stands were shown to be less susceptible

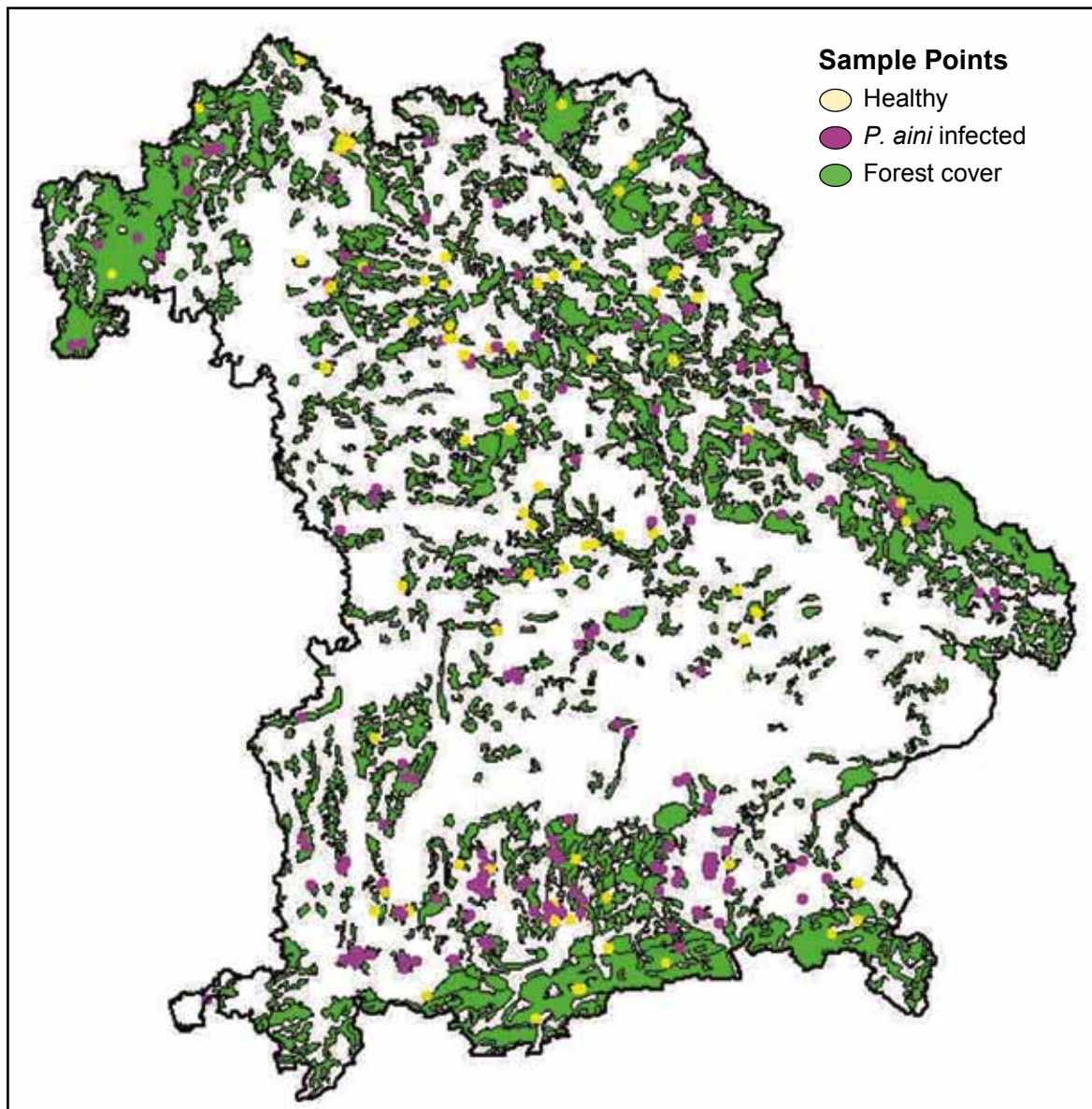


Figure 4—Study area of forested land cover (green) in Bavaria with *Phytophthora alni* infested (pink) and healthy (yellow) sample locations overlain.

to *P. alni* than declining trees, and in the long term, a resistance screening program may help to sustain alders as major components of riparian and swamp forests (Jung and Blaschke 2006).

The Exotic Forest Pest Web site, which is sponsored by the North American Forest Commission (NAFC 2006) lists *P. alni* as a high risk pest to North American forests for its potential to (1) adversely affect the economic trade of alder trees and (2) affect the environment; specifically by

changing forest composition, reducing wildlife food and habitat, increasing soil erosion, and changing soil composition owing to alder's nitrogen-fixing capabilities (Cree 1999).

An investigation of the conditions present at 434 sample locations in forested areas in Bavaria (Figure 4) was conducted using classification tree analyses and a tenfold cross validation to estimate the error. Classification and regression trees are a nonparametric iterative approach to compare all possible splits among the independent

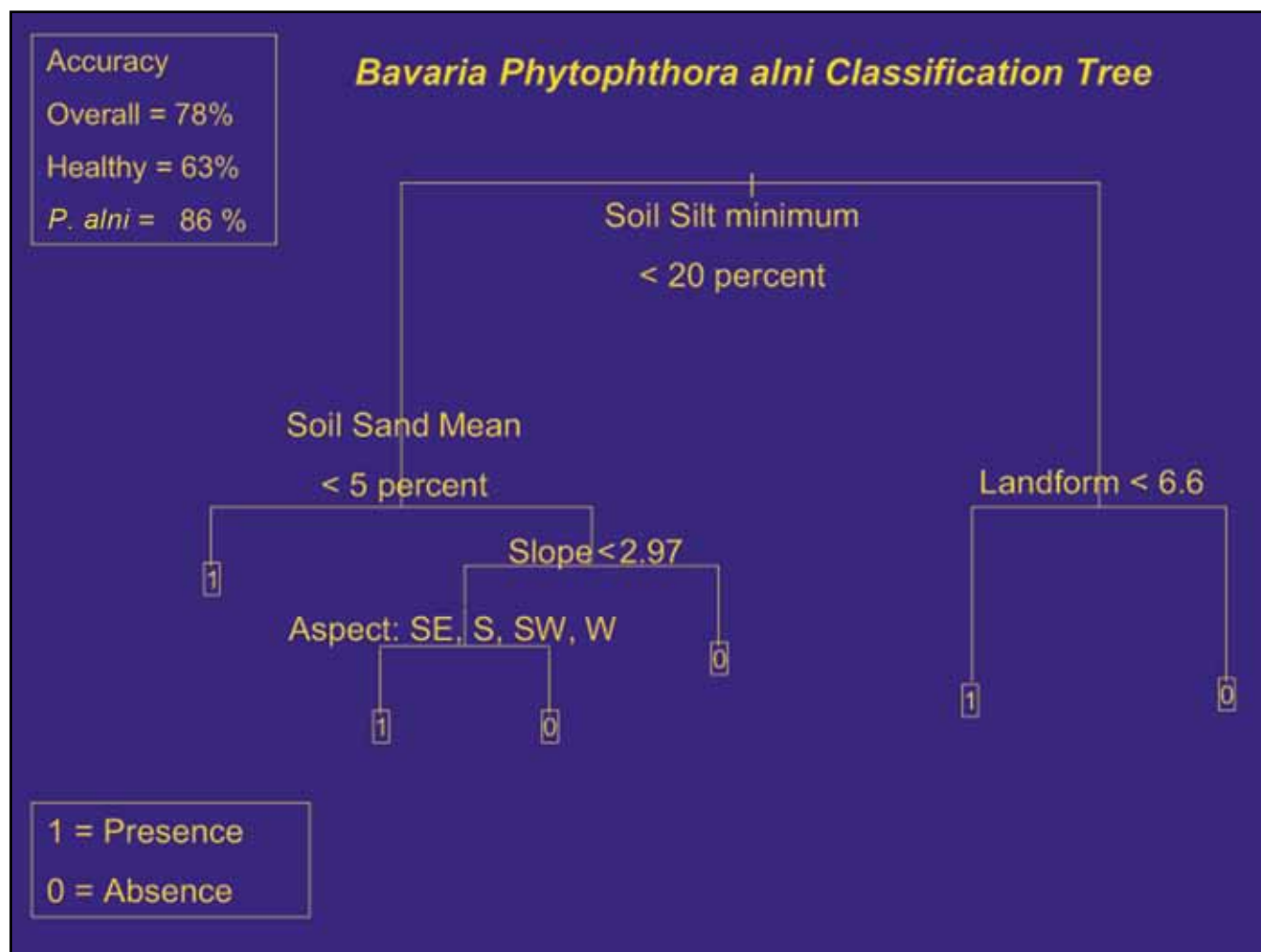


Figure 5—Classification tree model for presence and absence of *Phytophthora alni* in forested areas in Bavaria.

variables using a partitioning algorithm that maximizes the dissimilarities among groups. Classification trees are best used with binary data and regression trees with continuous data. Advantages of using decision trees such as classification and regression trees include the nonparametric nature of the model, ease of interpretation, and the robustness of the test (De'ath and Fabricius 2000). Decision trees have been successfully developed recently for (1) modeling landscape dynamics of the spread of *P. ramorum* (Kelly and Meentemeyer 2002), (2) mapping hemlocks via tree-based classification of satellite imagery and environmental data (Koch and others 2005), (3) predicting the presence and absence of lichen and past fires in Jalisco, Mexico (Reich and others 2005), (4) developing a spatial model for estimating fuel loads in the Black Hills, South Dakota (Reich and

others 2004), and 6) developing a methodology to predict oak wilt distribution in Minnesota and Texas (Downing and others 2007).

In this study, rules from the classification tree were used to create the potential distribution surface for Bavaria. Another potential distribution model, a Multicriteria model (Eastman 2001, Eastman and others 1995), was created for the globe using both the rules from the Bavarian classification tree, and additional parameters established with expert knowledge. Global susceptibility surfaces, such as the *P. alni* surface, may be used to illustrate the need for a pathway approach to regulate nursery stock and for host species resistance testing. Multicriteria models have been helpful to produce pest-risk maps for forested land in the United States (Krist and others, this volume).

Table 1—Bavarian independent variables used to produce a potential distribution for *Phytophthora alni* in Bavarian forested areas

A. Aspect: was derived via the DEM surface. (DEM: Digital Elevation Model) surface. The DEM surface was used to create ancillary data: slope, aspect, and landform. The DEM was created from the Shuttle Radar Topography Mission (SRTM) in February 2000. SRTM was a joint project between the National Geospatial-Intelligence Agency and the National Aeronautics and Space Administration. The DEM surface was obtained and distributed by the U.S. Geological Survey Earth Resources Observation and Science (EROS) Seamless Data Distribution System (http://seamless.usgs.gov]. The cell values of zero slope (flat) are assigned an aspect of -1. Therefore, the continuous aspect grid was reclassified to eight basic cardinal directions. Plus another class for zero slope.
B. Soil Fraction/Texture Percent: Soil fraction data were obtained as a soil polygon shapefile from the German Federal Institute for Geosciences and Natural Resources. Soil fraction percentage pertains to the volume amount by soil type (sand, silt, and clay) found in the soil sample. The percentage of all three soil fractions sums to 100 percent. To capture the variance of each soil type, three categories of soil fraction percentage were created for each soil type: (1) the minimum soil fraction value for each soil type, (2) the mean soil fraction value, and (3) the maximum soil fraction value. With three data sets included for each of the three soil types, there were a total of nine soil data sets used in the analysis.
C. Landform Index: Landform was derived via the DEM. Landform is independent of slope and created using a custom ArcView Avenue application. The application uses an irregular 3 by 3 kernel, where positive landform values indicate concavity, negative values indicate convexity, and a zero value indicates flat terrain (McNab 1989).
D. Normalized Difference Vegetation Index (NDVI): The NDVI was created from 250-m Moderate Resolution Imaging Spectroradiometer (MODIS) satellite data. The MODIS data was obtained from the Remote Sensing Applications Center (RSAC) in Salt Lake City, Utah. Personnel at RSAC downloaded an archived Bavarian MODIS image and performed the NDVI model. The values of NDVI relate to the relative greenness of the vegetation.
E. Slope: Slope was created using the “Derive Slope” function in the ArcView 3.3 Spatial Analyst extension coupled with the DEM surface. Slope units were degrees.

Methods

Between the spring of 2003 and the winter of 2006, a total of 307 *P. alni* infested and 127 healthy/noninfested alder tree locations were sampled in forested areas in Bavaria (Figure 4). Among the 307 infested sites, there were 232 points where alder trees had been planted, and 75 points where alders were naturally occurring. Of the 127 healthy sample points, 38 were planted and 89 had natural alder growth. A geographic information system sample point theme of the dependent variable was created containing all 434 sample point locations for analysis in the classification tree.

Thirteen independent variable raster data sets were used in the Bavarian classification tree analysis (Table 1). Specifically, these were twelve 93-m physiographic data sets including nine soil texture components (minimum, mean, and maximum percentage values for sand, silt, and clay

polygons), aspect, slope, and landform an index of concavity and convexity. The 13th data set was the Normalized Difference Vegetation Index (NDVI) calculated from the Moderate Resolution Imaging Spectroradiometer (MODIS) satellite imagery at 250 m. Numerical values were extracted from each of the independent variable grids at each of the sample locations from the sample point theme. These values were then used to compose the spatial information database that was exported to S-PLUS[®] statistical software (S-PLUS[®], Statistical Sciences 2000) for analysis. The independent variables selected by the classification tree were silt, sand, slope, aspects that were Southeast, South, Southwest, and West, and the landform index.

The default S-PLUS[®] validation technique, tenfold cross validation, was used to prune the tree to avoid overfitting the classification tree model to the Spatial Information Database. The tenfold cross validation was used, as it does not rely on an independent data set and can identify the optimum tree size for minimizing prediction errors.

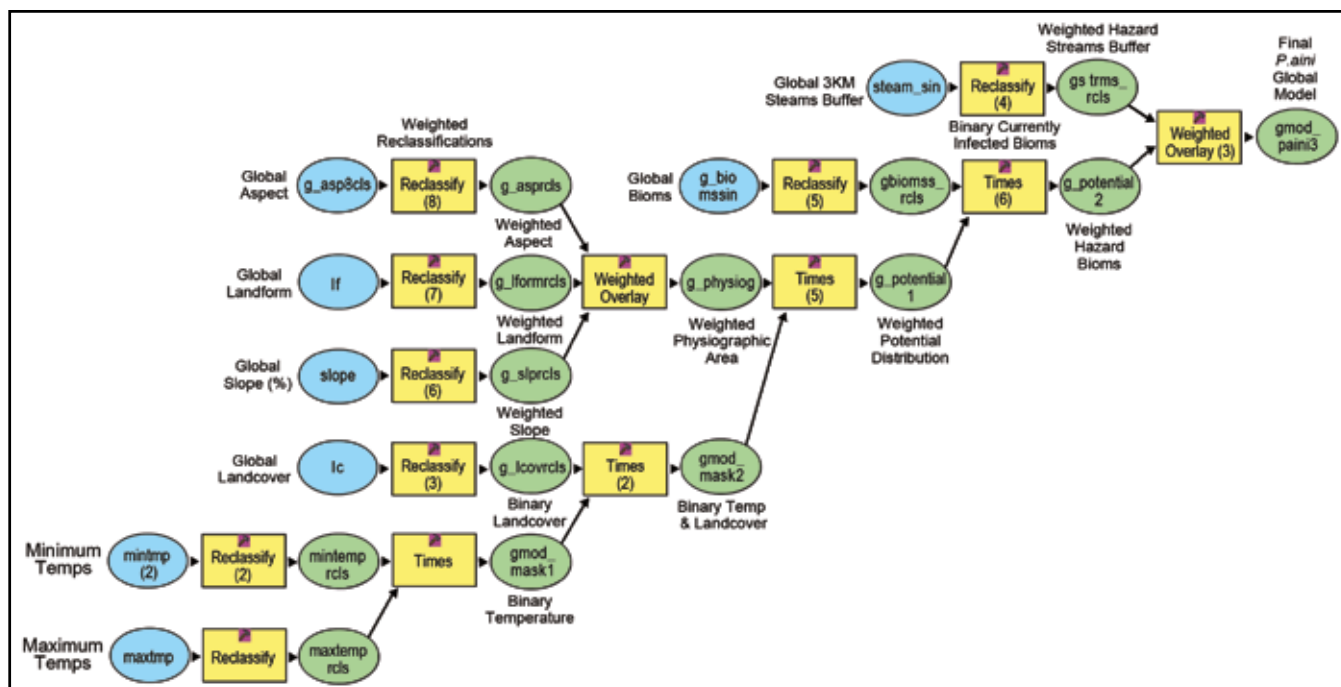


Figure 6—*Phytophthora alni* multicriteria model for the globe.

Based on the results of the classification tree analysis (Figure 5), conditional statements (CON statements; ESRI ArcView, 2000) were used to create a binary *P. alni* potential distribution (i.e., presence and absence) surface for the forested areas in Bavaria. The significant independent variables selected by the classification tree, as well as the decision tree rules (e.g., threshold values taken at the tree nodes), were the input for the CON statements.

Only three of the independent variables that were selected by the Bavarian classification tree were available globally. These were slope, aspect, and the landform index. To see how the rules would change given only the three independent variables, a second classification tree was developed for Bavaria using only those three data sets. This second model had limited utility as it overpredicted the presence of *P. alni*, predicting more than 90 percent of the study area to have *P. alni* present. Still, the rules from the second model did provide some additional insight regarding the broader range of conditions within which *P. alni* might be present. Therefore, the rules from both the original as well as from the second model were combined, along with additional expert knowledge, in a final multicriteria model to create a global susceptibility surface for *P. alni*.

To develop the multicriteria susceptibility model for the globe (Figure 6), the unique numerical values from each criterion had to be standardized. Therefore, each data set was reclassified using a hazard ranking of 0 to 10. The decision rules from both classification trees as well as additional expert knowledge were used as a guide in setting the hazard rankings.

Areas where alder and *P. alni* could not grow were eliminated from the global analysis by creating masks from climate and landcover data. To determine temperature thresholds for the climate mask, an investigation of *Alnus* species was conducted. It was determined from frost hardiness and heat/drought hardiness zones for all alder species that alder does not survive temperatures +/- 40 degrees Celsius.

In addition, lab results performed by Dr. Jung indicated that soil temperatures greater than 32 degrees Celsius prevent the survival of *P. alni*. Although soil temperature data is not available, a regression formula (Temperature MAX threshold value = (Soil Temperature MAX threshold value - intercept estimate)/Regression Coefficient Estimate), was applied to determine that 32 degrees Celsius equates to air temperatures of 34 degrees Celsius.

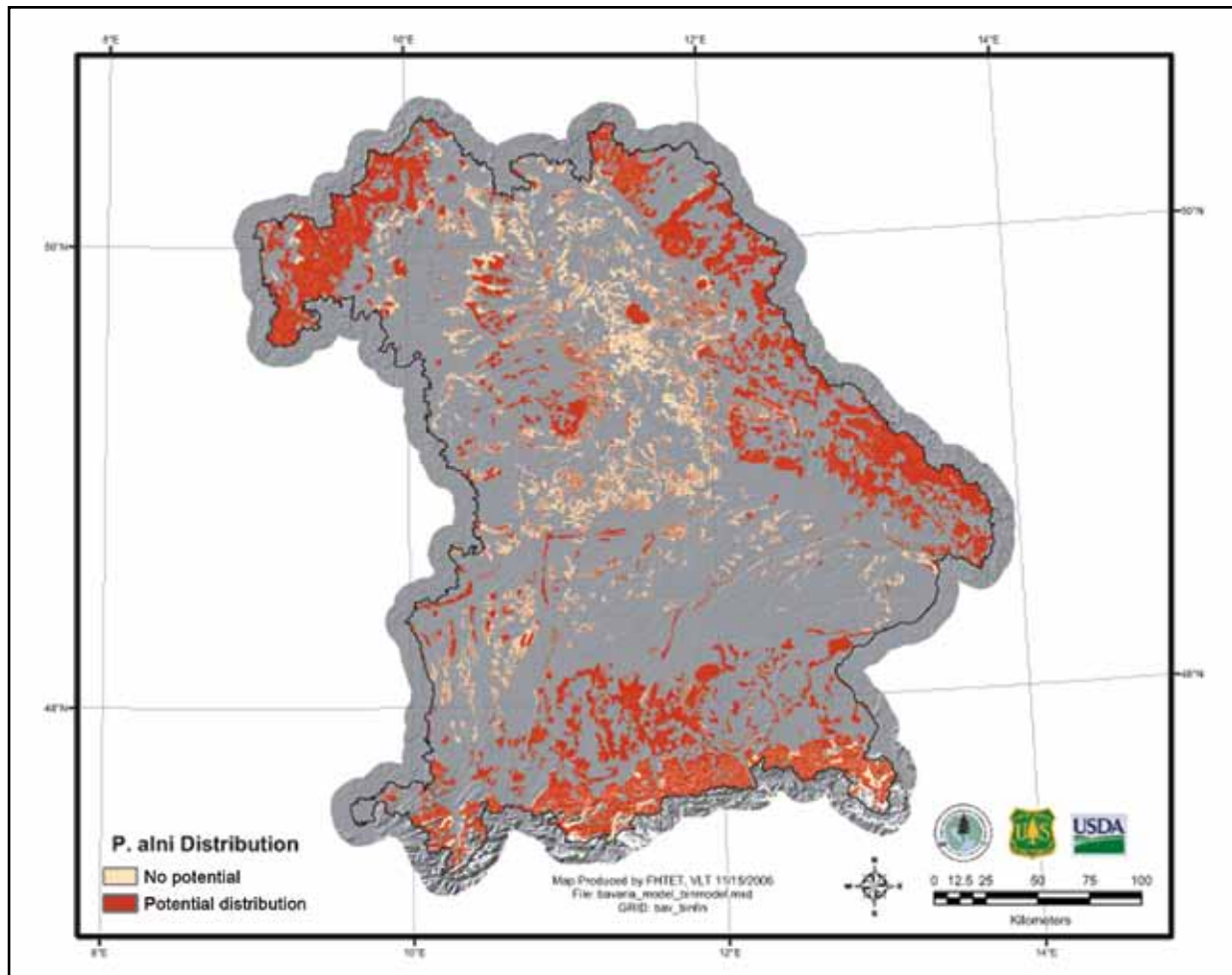


Figure 7—Potential distribution of *Phytophthora alni* (presence and absence) for all forested areas in Bavaria.

Consequently, areas with temperatures less than -40 degrees Celsius and greater than +34 degrees Celsius, as well as areas that could not support alder such as tundra, bare ground, and bodies of water, were removed from further analysis. The binary climate and landcover masks were combined by multiplying both surfaces together to create a combined binary temperature and landcover mask. The resulting mask was combined again in a weighted overlay with the reclassified criteria to produce a potential global distribution.

Because slope predicted most of the variability in the classification tree, it was weighted at 50 percent; aspect and the landform index were both weighted at 25 percent.

To produce the final global susceptibility surface, areas that were identified in the global distribution as having a

potential for a *P. alni* infestation were classified according to hazard. Biome and stream data were combined in an equal weighted overlay to assign a hazard ranking. The hazard ranking was based on each pixel's occurrence within ecological biomes similar to the biomes where *P. alni* presently occurs, as well as its proximity to streams. Thus, pixels within the selected biomes were assigned hazard rankings based on their proximity to streams. A set of three global stream buffers at distances of 1 km were used to assign the hazard rankings. Pixels that had the potential for infestation were given a high hazard ranking if they fell within 1 km of the stream. Those pixels between 1 and 2 km were assigned a medium potential hazard, and pixels between 2 and 3 km from the stream were assigned a low potential hazard. Pixels that were found greater than 3 km

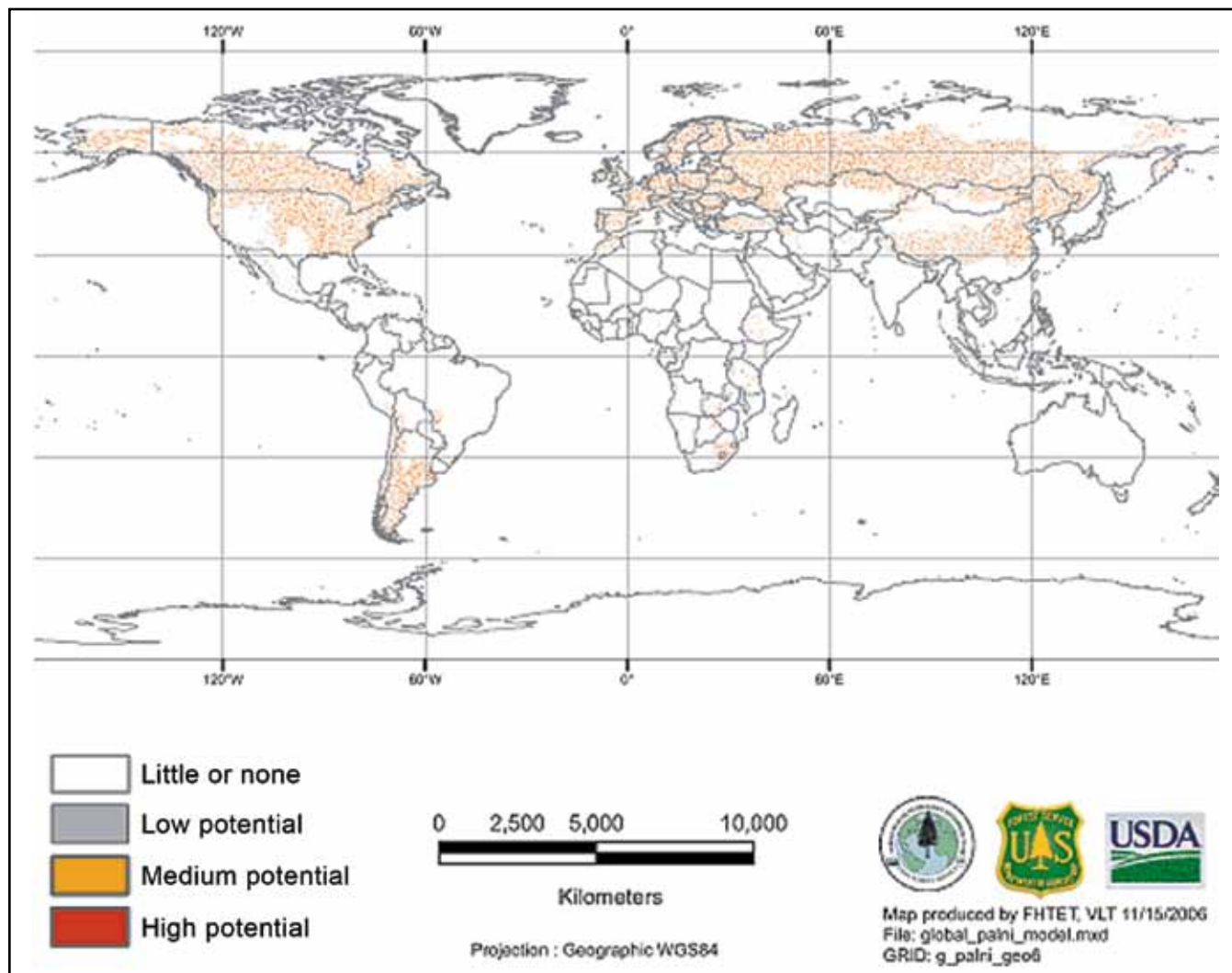


Figure 8—The final global susceptibility surface had 27, 835, 766 km² of suitable area where alder and *P. alni* could survive.

from a stream or outside the selected biomes were given a hazard ranking of little or no potential hazard.

Results

For the original Bavarian model: of the 19 620 forested km² assessed in Bavaria, approximately 14 015 km² (71.43 percent) were modeled to have a high potential for *P. alni* root and collar rot, and 5604 km² (28.56 percent) were modeled to have a high potential to remain healthy (Figure 7). Seven terminal end nodes were used and accounted for 78.34 percent of the variability. The tenfold cross validation gave a higher accuracy for predicting the 307 *P. alni* infested sites at 86 percent, and showed 63 percent accuracy in predicting

the 127 healthy sites. The independent variables important in predicting the presence or absence of *P. alni* were minimum silt fraction values less than 20 percent (range = 0 to 80 percent), mean sand fraction values less than 5 percent (range = 0 to 93 percent), slope less than 2.97 degrees (range = 0 to 30.74 degrees), aspects that were Southeast, South, Southwest, and West, and the landform index less than 6.6 (range = - 15.20 to + 21.60; < 0 = concave; 0 = totally flat; > 0 = convex) (Figure 5).

For the second Bavarian model, with only the three independent variable data sets that are available globally being used, 14 terminal end nodes accounted for 77.19 percent of the variability. The tenfold cross validation still

showed a higher accuracy for predicting the 307 infested sites at 92.51 percent, but only 40.16 percent accuracy in predicting the 127 healthy sites. As expected, all three of the independent variables were used to predict the presence or absence of *P. alni*, but, in this model, slopes less than 19.57 degrees had a higher probability for infestation, and the East aspect was selected in addition to the four aspects that had originally been selected. Also, landform indexes less than 6.2 had a higher probability of infestation.

The final global susceptibility surface had 27,835,766 km² of suitable area where alder and *P. alni* could survive (Figure 8). Of that area 1,482,487 km² (5.33 percent) were highly susceptible to *P. alni*; 3,930,660 km² (14.12 percent) had a medium susceptibility; 5,721,467 km² (20.55 percent) had a low susceptibility; and 16,701,152 km² (60.00 percent) had little or no susceptibility.

Discussion

The original Bavarian classification tree identified five ecological factors important in the distribution of *P. alni*. Where these factors occur together in the environment, the likelihood of infection is increased. Specifically, where silt minimum values are less than 20 percent, and sand mean values are less than 5 percent, the probability of a *P. alni* infection is high. When silt minimum values are less than 20 percent, and sand means are greater than 5 percent, the site is more likely to have *P. alni* infections if slopes are less than 2.97 degrees and have warmer aspects. Sites with a landform index measure of less than 6.6 (concave, flat or slightly convex) also have an increased probability of a *P. alni* infestation. These results make biological sense. Areas with poor drainage and warmer aspects provide an optimal environment for the pathogen to flourish, as will sites with fairly flat or concave physical structure. Conversely, areas with less clay and more silt or sand will drain better, as will sites with steeper slopes and convex landform. These types of sites will not provide a wet environment for this water-borne pathogen to form sporangia and release zoospores that are essential for the spread and infection of *P. alni*.

Not all of the five ecological factors identified as being important by the first Bavarian model for predicting the distribution of *P. alni* were available for the global model.

A second model for Bavaria, which utilized only the three data sets that were available globally, demonstrated the limitations of modeling invasive species at a global scale without appropriate data. The limitation most notable was the soil texture data because it was selected by the first Bavarian model as the most important variable for predicting the presence and absence of the soil-borne pathogen *P. alni*. In addition, data was not available for (1) forest species type (i.e., distribution of the individual alder species), and (2) susceptibility of North and South American and Asian alder species to *P. alni*. We addressed forest species type by keeping our analysis near and around streams and flood plains where most alders tend to grow. We also looked at the temperature range, eliminating areas with temperatures that were too cold or hot for alder and *P. alni* survival. Although we made compromises to work within the data limitations, this work emphasizes the need for quality spatial environmental data at the global scale.

Since planting infected nursery stock is one of the primary pathways by which *P. alni* has been spread, we were careful to consider the social or cultural habits in association with outplanting alder trees. Of particular concern was the outplanting of infected alder trees in respect to elevation. At higher elevations, alder trees are planted much less frequently than at lower elevations. Yet, it has been observed by the Jung that where *P. alni*-infected alder was planted at higher elevations, those sites have become infested and further contribute to infections downhill and downstream. Because alder was rarely planted at higher elevations, *P. alni* was much less prevalent on higher elevation sites. We therefore assumed that the model would be biased toward selecting elevation as an important variable for predicting presence and absence. Consequently, elevation was not used in the model.

A higher accuracy was attained for predicting the *P. alni*-infested sites than for predicting healthy sites. This is likely an outcome of having three times more infested than healthy sample locations. Had we sampled a greater number of locations for the healthy condition, it is likely that the accuracy for predicting healthy sites would improve.

Of the 127 healthy alder tree locations collected between 2003 and 2006, some sites may have changed in

status. Some of the sites that were not infested by 2006 may become infested in the future. These are problems one would expect in attempting to model a species that is unlikely to have been in existence before the 1980s (Gibbs and others 2003, Jung and Blaschke 2004) and has not yet completely expanded into its potential range. With no complete range map for *P. alni*, the Bavarian model provides managers worldwide with useful decision rules and a data mining tool for estimating the susceptibility of their resources to *P. alni*.

Because all of the applicable variables from the first Bavarian model are available in data sets for the United States, the extrapolation of the Bavarian model to forests in the United States should demonstrate the specific improvement that can be gained by applying the appropriate data sets identified by the Bavarian model.

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Assessing Insect-Induced Tree Mortality Across Large Areas With High-Resolution Aerial Photography in a Multistage Sample

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Abstract

In recent years, unprecedented tree mortality has occurred throughout the national forests owing to insect infestations and disease outbreaks. The magnitude and extent of mortality, coupled with the lack of routine monitoring in some areas, has made it difficult to assess the damage, associated ecological impact, and fire hazard in a timely and cost-effective manner. To aid forest managers in assessing the damage, a cost-effective multistage sampling method, using high-resolution digital aerial photography, was developed to estimate overall mortality across large areas. The method was tested within a 332,000-acre piñon/juniper woodland west of Flagstaff, Arizona, within the Kaibab National Forest. Piñon pine mortality caused by piñon ips bark beetles (*Ips confusus* (LeConte)) was assessed from high-resolution digital aerial imagery within percent-cover strata with the use of a digital dot grid. The sample revealed that dead trees covered 7.0 ± 0.3 percent of the study area. As a percentage of total tree cover, 20.0 ± 0.8 -percent mortality had occurred. The cost to obtain this estimate was approximately \$0.04 per acre.

Keywords: Dot grid, imagery, *Ips confusus*, pinyon, remote sensing, sample.

Introduction

Over the past century, stand density and fuel loading have increased in forests and rangelands throughout the United States, leading to a general decline in ecosystem health. The weakened condition of the forests and rangelands, coupled with drought stress in the Western United States, now place nearly 200 million acres of Federal forest and rangeland in the contiguous United States at risk of epidemic insect and disease outbreaks and catastrophic wildfires (USDA and USCI 2004). In many areas, devastating wildfires and unprecedented insect and disease outbreaks have already occurred.

To address the threat and impact of fire, insects, and diseases to the Nation's forests and rangelands, the U.S. Federal Government launched the Healthy Forests Initiative (HFI) in 2002 and enacted the Healthy Forests Restoration Act (HFRA) in 2003. Primary goals of the HFI and HFRA are to facilitate, expedite, and provide national guidance on hazardous-fuel reduction and ecosystem restoration. In areas where insects and diseases have already caused extensive tree mortality, the HFRA calls for accelerated information gathering on the impact of these mortality agents (USDA and USDI 2004). However, collecting information on the extent and severity of mortality in a timely and cost-effective manner can be very difficult because of the vast acreages that are affected, coupled with the lack of routine monitoring in some areas. In some cases, the extent of mortality is so great that even traditional assessment methods such as aerial sketch-mapping become impractical in terms of time and cost. An efficient and cost-effective alternative assessment method is needed.

One possible alternative is to sample rather than map mortality. Unlike aerial sketch-mapping, which produces a map, but no quantitative measure of mortality, a sample provides a quantitative measure of mortality, but no map. Not all sample designs are efficient or cost-effective, but some have the potential to be less costly and time consuming than traditional methods of assessing mortality. A multistage sample, for example, is one type of sample design

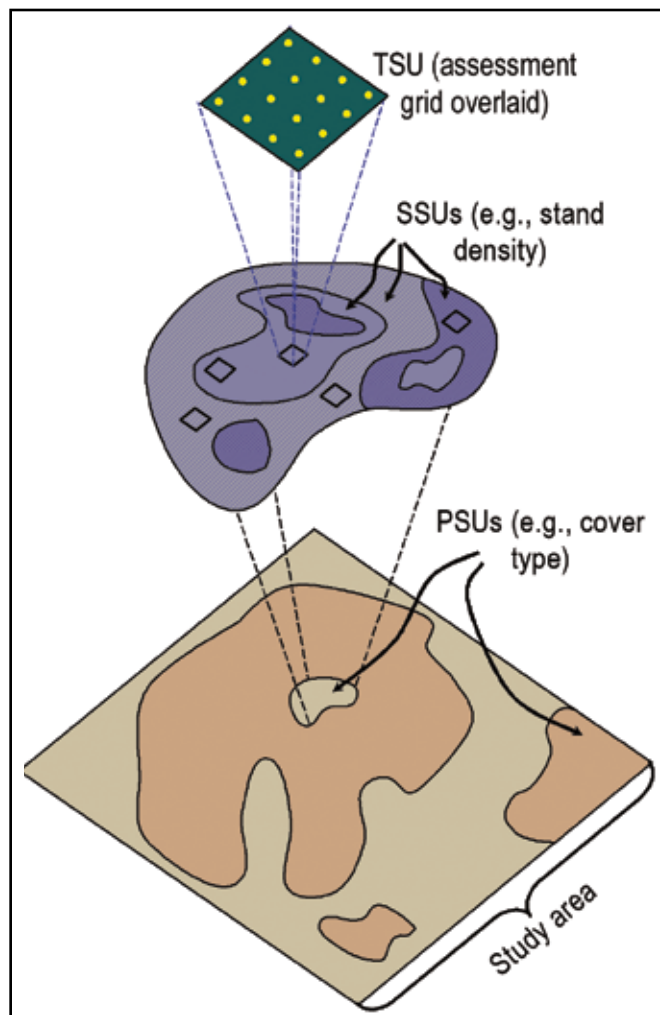


Figure 1—A common multistage sample design defines final sampling units in three consecutive stages: primary sampling units (PSUs), secondary sampling units (SSUs), and tertiary sampling units (TSUs).

that attempts to optimize sampling efficiency. Sampling efficiency is increased by constraining final sampling units to fall within only certain subsections or regions of a study area rather than distributing them across the entire area. Assessing groups or clusters of sample locations is generally easier and less time consuming than assessing widely dispersed sample locations.

In a multistage sample design, a study area is initially partitioned into coarse subunits, called primary sampling units (PSUs). A subset of PSUs is selected and subdivided into secondary sampling units (SSUs). This process of selecting and further subdividing sampling units may be repeated as often as necessary, but a three-stage design

is common (Figure 1). In the final stage of the sample, a subset of the previous stage's sampling units are selected and sampled or assessed. From these samples, the variable of interest can be estimated for the entire study area (Ciesla 2000, Schreuder and others 2004).

The dimensions and layout of sampling units can be arbitrary; however, it is generally advantageous to use strata correlated with the variable of interest as sampling units, particularly in the early stages of the design. Sampling within wisely chosen strata allows a multistage sample to take advantage of known sources of variability in the population and can greatly increase the precision of the estimate. When assessing tree mortality, possible strata might include cover type, stand density, elevation, slope, aspect, soil type, proximity to water, and others.

Multistage sample designs have been used previously to inventory timber, estimate tree mortality and volume loss caused by insects in conifer forests, and estimate tree mortality caused by diseases (Ciesla 2000, Langley 1971, Munson and others 1985, White and others 1983). Ciesla (2000) reviewed an assortment of aerial photography-based multistage forest inventories. Typically, these inventories used vegetation type or tree mortality or both as initial-stage strata. Mortality strata were obtained by aerial sketch-mapping or from complete coverage aerial photography. Subsequent stages in these surveys involved collecting aerial photography over plots within the PSU strata. In most of the surveys, the third stage consisted of further subdividing the SSUs into tertiary sampling units (TSUs). A subset of the TSUs was then ground sampled.

Although samples are less costly and time consuming than censuses, multistage samples based on aerial photography and manual photo interpretation can still be expensive and time consuming. Replacing traditional aerial photography in multistage samples with digital aerial and satellite imagery can potentially reduce survey time and related costs. The cost of digital imagery continues to decrease, and economical, yet powerful computers and image processing software can now automate some of the tasks traditionally done by hand. In addition, some field assessments can be greatly reduced by analyzing very high-resolution imagery acquired over the final sampling units.

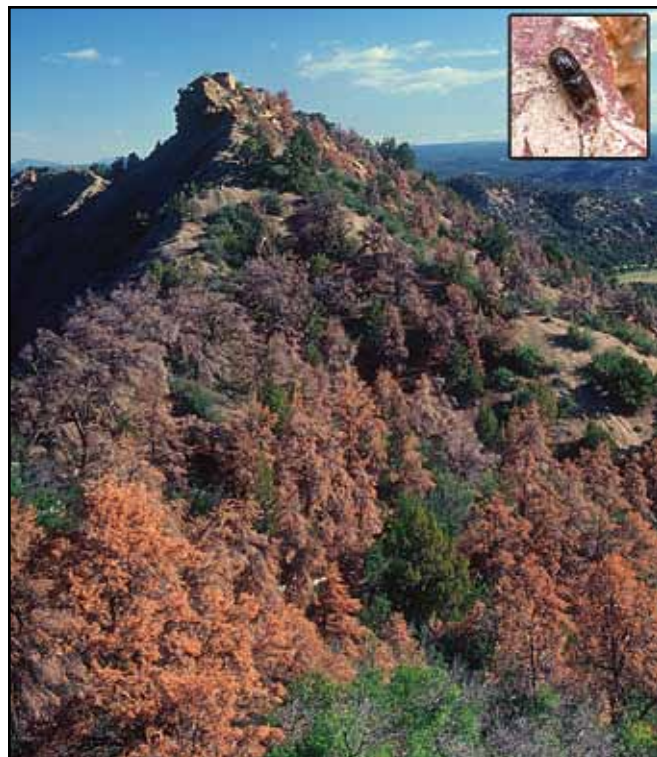


Figure 2—Piñon pine mortality caused by piñon ips beetles (see inset). (Photograph courtesy of William M. Ciesla, Forest Health Management International, Bugwood.org)

To aid forest managers in assessing the severity and extent of widespread tree mortality, the USDA Forest Service Remote Sensing Steering Committee sponsored a pilot study to develop a cost-effective, rapid, and statistically rigorous multistage sample design incorporating digital remotely sensed imagery to quantify tree mortality across large geographic areas. The method was evaluated in a piñon pine/juniper woodland.

Case Study—Assessing Piñon Pine Mortality

Beginning around 2002, extensive piñon pine (especially *Pinus edulis* Engelm. and *P. monophylla* Torr. & Frem.) mortality appeared throughout the Western United States. Dense stocking and sustained drought weakened the defenses of the pines, making them highly susceptible to attack by piñon ips bark beetles (*Ips confusus* (LeConte)) and other insects, as well as infection by black stain root disease (*Leptographium wagneri* (Kendrick) Wingfield) and other disease agents. Insect feeding and diseases

destroy and clog the conductive tissues of the trees, causing them to die. The piñon ips beetle, the most important mortality-causing agent of the recent mortality event, is a native species that typically plays an important role in maintaining healthy forests by removing stressed or injured trees. This process thins the forest and reduces competition for water, nutrients, and light. Healthy trees are generally unaffected by the beetles. However, the abundance of drought-weakened trees allowed beetle populations to explode, causing extensive mortality in stressed and healthy trees throughout the piñon/juniper range (Figure 2) (Keyes and Hebertson 2003, Negrón and Wilson 2003, Shaw and others 2005).

Because piñon/juniper woodlands are not routinely monitored, assessing the ecological impact of the mortality and the fire hazard it presented became very difficult. Therefore, a rapid and cost-effective multistage sample design incorporating digital aerial imagery was developed to assess tree mortality.

Methods

A multistage sample design was developed and evaluated in a study area located west of Flagstaff, Arizona, in and around the Williams Ranger District of the Kaibab National Forest. The footprint of a SPOT 5 satellite image (60 by 60 km) provided the approximate geographical boundary for the study (Figure 3). In this region, a mixture of piñon pine and juniper (*Juniperus* sp.) dominates lower elevations whereas the upper-elevation forests are predominantly ponderosa pine (*Pinus ponderosa* Dougl. ex Laws.) interspersed with aspens (*Populus tremuloides* Michx.) and higher-altitude conifers.

Stage 1—Piñon/Juniper Woodland Cover Type (PSUs)—

The first stage of the multistage sample design consisted of locating the piñon/juniper cover type within the study area to eliminate non-piñon/juniper areas from further analysis. Existing cover-type maps (i.e., from the Gap Analysis Program, the Rocky Mountain Resource Information System, the National Landover Data set, and other sources) were evaluated for this purpose. These data sets were deemed unsatisfactory for this study because changes in cover type from vegetation management activities were sometimes not

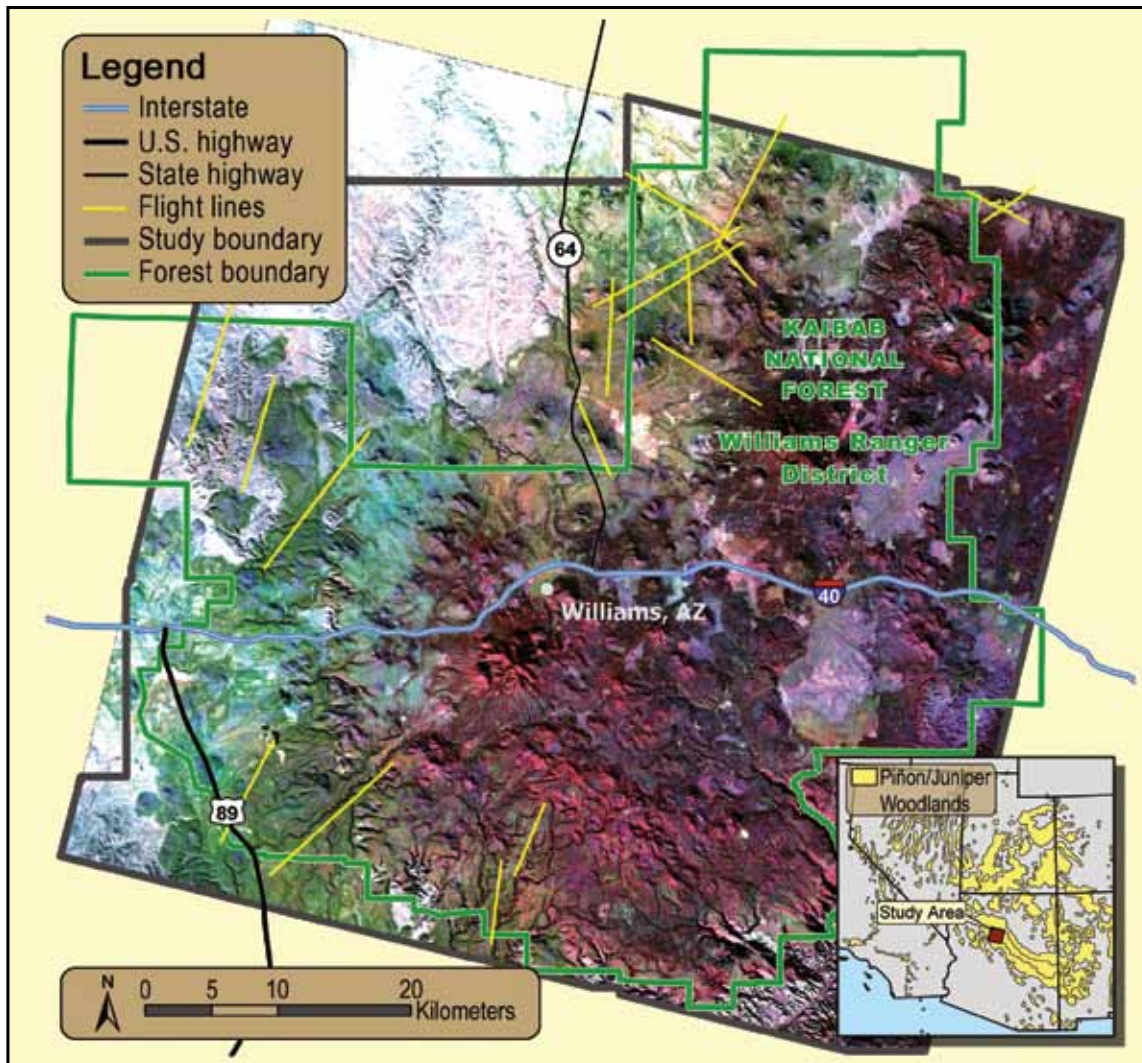


Figure 3—The pilot study area was located west of Flagstaff, Arizona, within the Williams Ranger District of the Kaibab National Forest. A SPOT 5 satellite imagery scene (acquired September 16, 2003)—displayed in false color infrared with topographic enhancement—defined the extent of the study area. Green and blue/green areas are piñon/juniper woodlands, whereas red areas are primarily ponderosa pine forests. Flight lines where airborne imagery was collected are shown as yellow lines.

reflected in these maps and some obvious fine-scale errors were also present. Therefore, a three-class vegetation map (piñon/juniper, ponderosa pine and other conifers, and meadow/bare ground/other) was developed for the study area using image segmentation and regression tree classification (Figure 4) (Hamilton and others 2004). This map formed the PSUs. For this study, the entire 332,000-acre piñon/juniper cover type was advanced to the second stage of the sample.

Stage 2—Percent-Cover Strata (SSUs)—

Negron and Wilson (2003) reported that the likelihood of piñon ips infestation increased with stand density in Arizona. To reduce sample variance, this known source of variability was incorporated into the sample design by stratifying the piñon/juniper vegetation class by percent cover. The percent-cover strata became the SSUs. Breaks for the percent-cover strata were based on the broad-level vegetation cover categories established by the USDA Forest Service Existing Vegetation Classification and Mapping

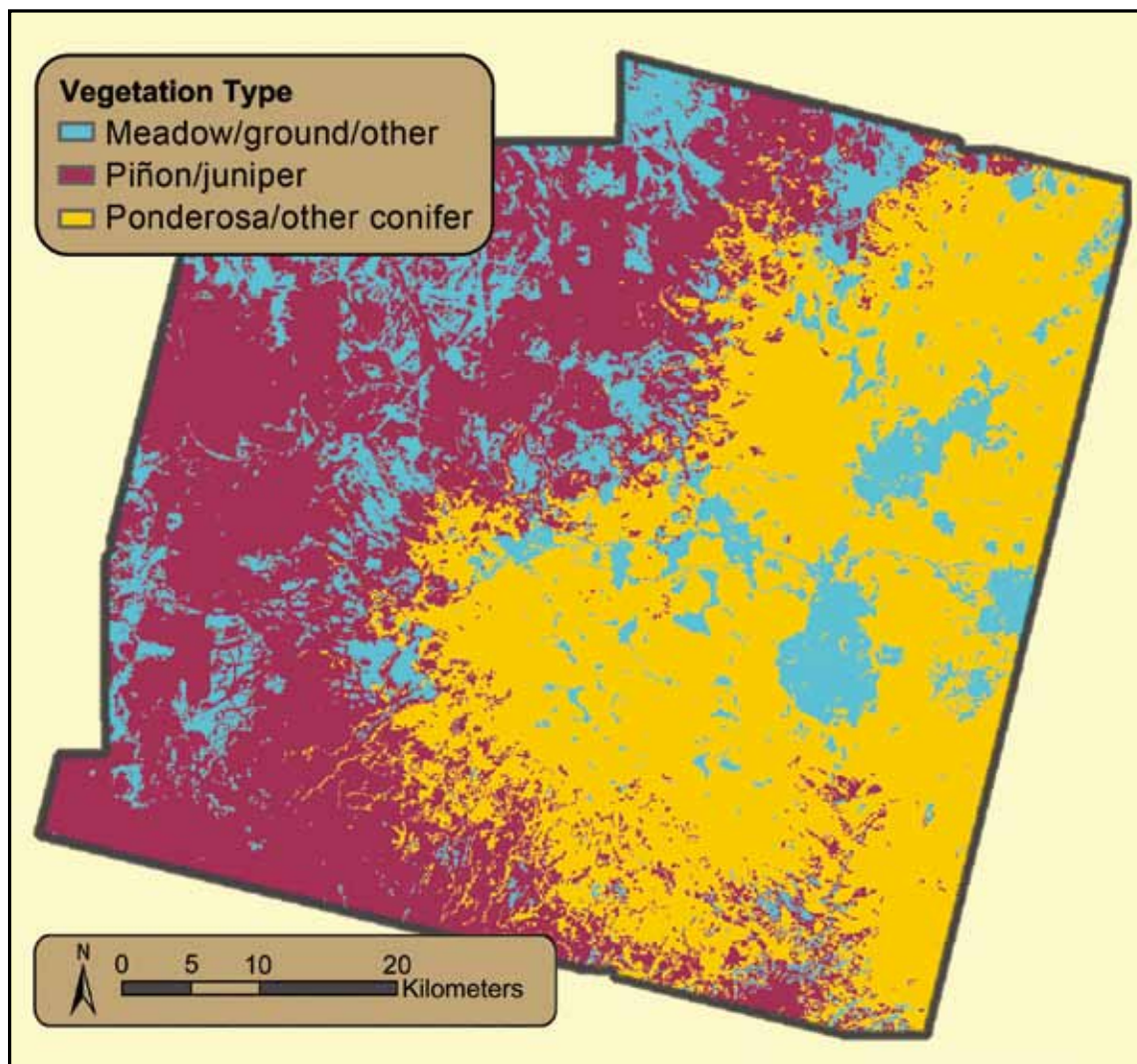


Figure 4—A vegetation-cover map, used to restrict the multistage sample to piñon/juniper woodlands, was created using regression tree classification. This is the first stage of the multistage sample.

Technical Guide (Brohman and Bryant 2005) (Table 1). The first vegetation-cover category (0 to 29.9 percent), was further subdivided into 0 to 9.9-percent and 10 to 29.9-percent categories to refine the SSUs.

Because no existing percent-cover maps were available for the study area, a map was created from 1992 digital orthophoto quadrangles (DOQs). The DOQs were not current, and we anticipated that the mapped percent cover would generally be lower than the actual percent cover. However, as the objective of this exercise was to stratify the study area (not to measure percent cover), relative differences in actual percent cover compared to mapped

percent cover would still allow the map to serve its purpose of stratifying the study area. The relative differences in percent cover are inconsequential so long as percent cover and, consequently, mortality, is more homogeneous within than between strata. To create the percent-cover map, a two-class classification (tree and nontree) was created from the DOQs in ERDAS Imagine. Then, for each image segmentation polygon (average size ≈ 2.5 acre) used in the vegetation classification, the percentage of the polygon occupied by mapped trees was calculated (Figure 5). No accuracy assessment was conducted for the percent-cover map. However, final sample results verified that percent

Table 1—Percent-cover ranges of SSU strata and strata acreages within the entire piñon/juniper region and the region of aerial imagery coverage are shown, including proportions by stratum of total study area and aerial imagery coverage for the two regions

Strata (percent-cover ranges)	Acreage and (proportion of total area)	
	Piñon/juniper region	Aerial imagery coverage
0-9.9 percent	36,224 (0.11)	739 (0.10)
10-29.9 percent	108,633 (0.33)	2,376 (0.32)
30-59.9 percent	140,162 (0.43)	3,582 (0.49)
60-100 percent	39,430 (0.12)	642 (0.09)
Total	324,449	7,339

SSU = Secondary Sampling Units.

cover and mortality were more homogeneous within than between strata.

Stage 3—Digital Aerial Imagery Plots (TSUs)—

Tertiary sampling units for this study consisted of 60-by-60-m digital aerial imagery plots. Plots were sampled for mortality using a digital dot grid. On March 30 and April 24, 2004, high-resolution digital camera (Kodak Pro Back 645C digital back attached to a Contax 645 medium format camera) imagery was acquired in 18 flight lines at various locations across the study area (Figure 3). The imagery was acquired with an 80 mm lens at an altitude of approximately 1375 m, producing 16-cm spatial-resolution imagery with a swath width of approximately 640 m. The imagery was orthorectified using OrthoBASE in ERDAS Imagine. Thirty TSU plots were randomly located within each percent-cover stratum (for a total of 120 plots) and the geographic boundaries of the aerial imagery. Although the percent-cover strata were not equal in size (Table 1), they were sampled equally to ensure a minimum number of samples in each stratum.

The plot size and dot density were chosen to optimize the accuracy and precision of the sample while minimizing the total number of dots per sample (i.e., minimizing costs). This was accomplished by sampling several areas from each percent-cover stratum using multiple dot grids, ranging in size from 30 by 30 m to 240 by 240 m with dot densities of 364, 648, or 1,012 dots per acre. A plot size of

60 by 60 m with dot density of 364 dots per acre (18 by 18 dots per plot) was considered optimal as the precision of the estimate increased only slightly with increased plot size or dot density or both. The dot grid was created from graphic elements in ArcGIS 8.3, allowing it to be moved easily from one plot to the next by dragging and dropping. Although this graphic dot grid worked well for this study, a new tool for ArcGIS 8.3 and 9.x, Digital Mylar—Image Sampler, was recently developed by the USDA Forest Service Remote Sensing Applications Center and now provides a more automated and user-friendly approach to dot grid sampling (USDA Forest Service 2005).

At each sample location, the dot grid was placed over the high-resolution imagery (Figure 6). The total number of dead-tree, live-tree, and ground hits from the dots was counted. First-year and later mortality were all grouped into the dead-tree category for this study. In this sampling procedure, piñon pines were not distinguished from junipers owing to the difficulty of distinguishing the two species on this imagery by photo interpretation. The proportion of each plot covered by dead trees, live trees, total tree cover (i.e., live plus dead trees), and other (typically bare ground) was calculated from the dot grids by tallying the hits of the individual variables and dividing by the total number of dots in the sample. Subsequently, mean values of these variables were calculated for each percent-cover stratum. The mean proportion of the entire piñon/juniper region (i.e., across all strata) covered by dead trees, live trees, total tree cover, and other was calculated from the estimates of each percent-cover stratum. These estimates were made using a standard weighted disproportionate-strata estimation equation, with the strata areas from Table 1 as weights (Equation 1) (Scheaffer and others 1990). Because relative proportions of strata areas for the entire piñon/juniper region were similar to those falling within the boundaries of the aerial imagery (Table 1), no adjustments were needed to extrapolate the estimate from the extent of the aerial imagery to the entire study area.

$$\hat{p}_{st} = \frac{1}{N} \sum_i (N_i \hat{p}_i), \quad (1)$$

where $\hat{p}_{st}$ is the mean across-strata proportion of land area covered by dead trees, live trees, total tree cover, or other;

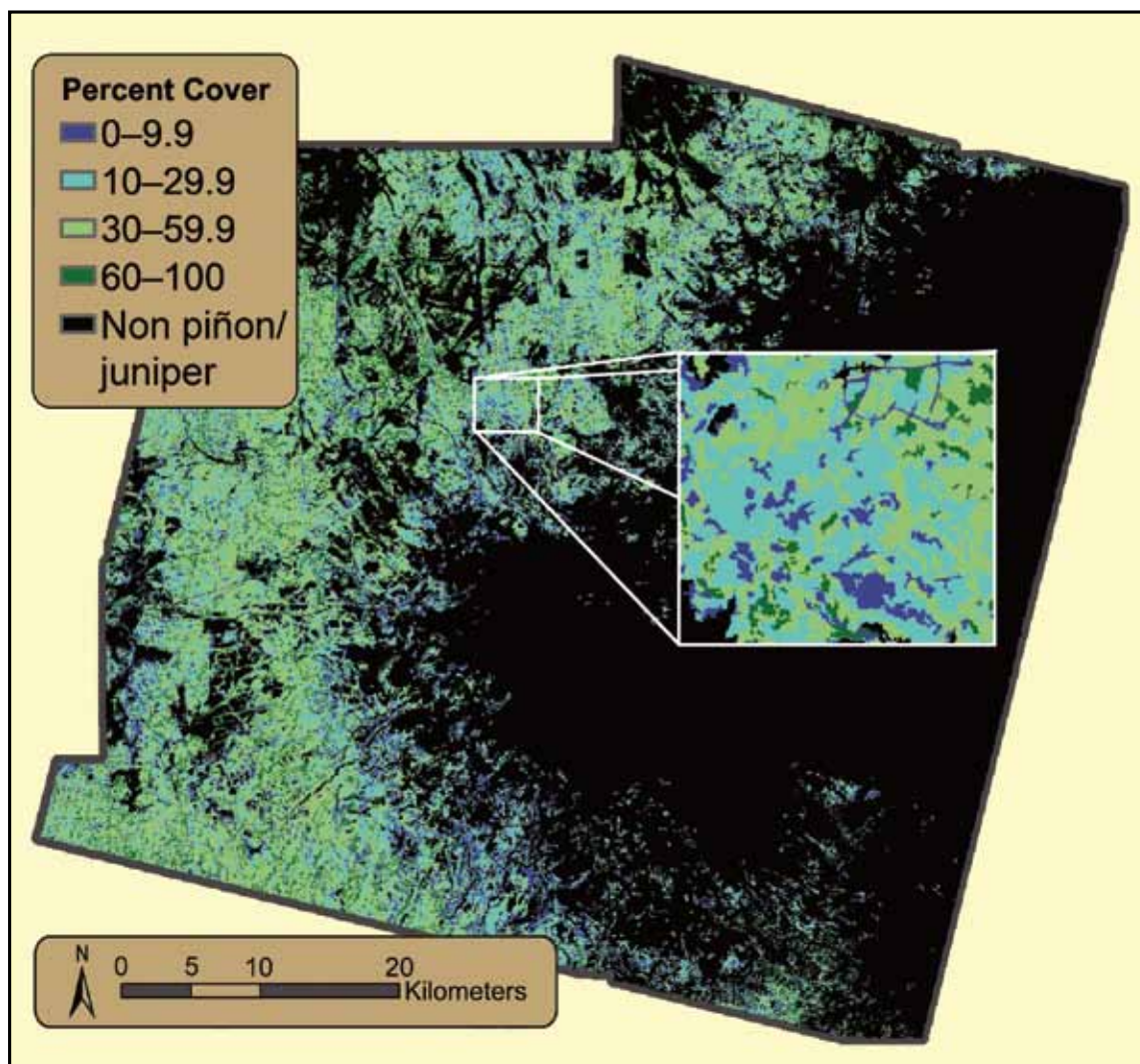


Figure 5—Percent-cover strata for the study area, derived from digital orthophoto quadrangles, were developed for the second stage of the multistage sample and formed the secondary sampling units.

$\hat{p}_i$ is the mean proportion of land area within each percent-cover stratum, indexed by i , covered by dead trees, live trees, total tree cover, or other; N is the total possible number of individual dot samples within the entire study area; and N_i is the weight of the i^{th} percent-cover stratum (i.e., the proportion of possible dot samples within the piñon/juniper forests of the i^{th} percent-cover stratum to the total possible number of dot samples within the entire piñon/juniper forested area).

Standard errors of the across-strata mean proportion estimates were also calculated (Equation 2). From these

results, the mean proportions of total tree cover that had died were calculated for each percent-cover stratum as well as across strata for the entire piñon/juniper population.

$$\hat{E}(\hat{p}_{st}) = 2 \sqrt{\frac{1}{N^2} \sum_i^n N_i^2 \left[\left(\frac{N_i - n_i}{N_i} \right) \left(\frac{\hat{p}_i(1 - \hat{p}_i)}{n_i - 1} \right) \right]} \quad (2)$$

where $\hat{E}(\hat{p}_{st})$ is the error of the mean across strata proportion of land area covered by dead trees, live trees, total tree cover, or other; n_i is the number of dot samples taken from each percent-cover stratum; and N_i , N , $\hat{p}_i$ are as described in equation 1.

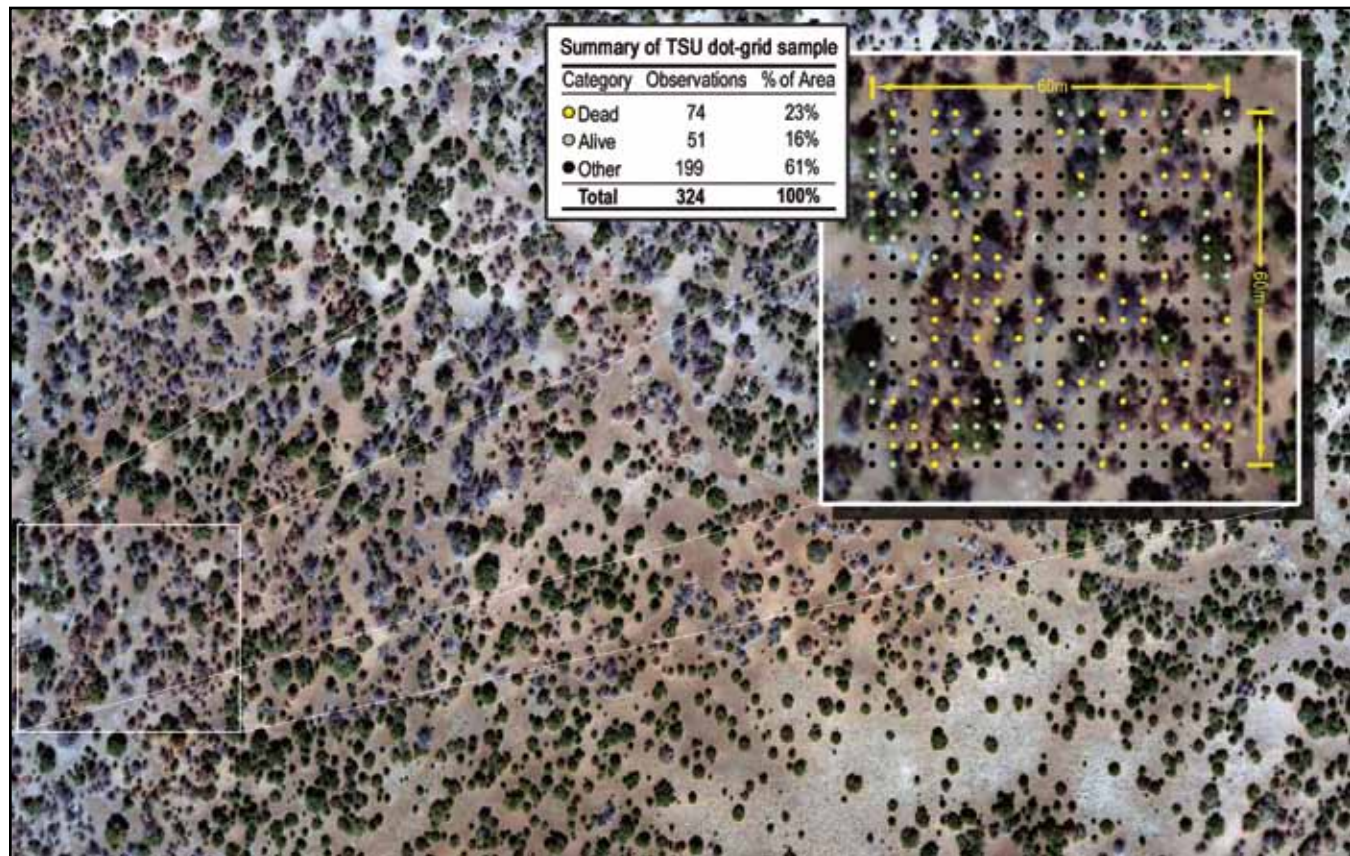


Figure 6—Natural-color, digital aerial imagery with dot-grid sample collected March 30, 2004, over an area of piñon mortality. The inset shows an example of a dot-grid sample of mortality within a 60 by 60 m (approximately 1 acre) tertiary sampling unit (TSU). Observations of dead trees, live trees, and other are color-coded and tallied for the site.

Results and Discussion

Estimates calculated for the entire piñon/juniper region of the study area (across strata) showed that dead tree canopies occupied 7 ± 0.3 percent of the total area as viewed from aerial imagery (Table 2). As a percentage of total tree cover, 20 ± 0.8 -percent mortality had occurred (Table 3).

Actual percentage of cover derived from the dot-grid samples often differed from the predefined percent-cover ranges of the SSUs. However, the stratification still served its purpose of reducing sample variance—percent cover and mortality were more homogeneous within than between strata. The results of this study generally support Negrón and Wilson's (2003) conclusion that mortality increases with percent cover (Table 3).

Costs to conduct this multistage sample included \$5,050 for materials and approximately 30 days of labor (Table 4). If labor were priced at \$300 per day, the cost per acre for the

332,000-acre study area would be slightly more than \$0.04.

These cost estimates do not include the cost of creating a vegetation-cover map to narrow the study region to only piñon/juniper woodlands. It is assumed that sufficiently accurate digital maps of vegetation cover are or will soon be available for most of the Nation.

The cost incurred in this study can serve as a reference point for similar studies in other areas. However, costs can vary greatly depending on location, imagery resolution, size of the study area, and other factors. The cost-per-unit area, for example, is scale dependent—decreasing as the size of the study area increases.

Conclusions

Forests and rangelands throughout the United States are at risk of severe insect and disease outbreaks and catastrophic wildfires. Epidemic insect infestations have already caused

Table 2—Mean proportions of land area comprised of various cover types for each stratum. For each stratum, the mean proportions ($\pm$ SE) of area (as viewed from aerial imagery) comprised of dead, live, and total tree cover, and ground are shown^a

Stratum (percent-cover range)	Mean proportion of stratum land area, by type of cover			
	Dead	Live	Total tree cover	Ground
0-9.9	0.021 $\pm$ 0.006	0.150 $\pm$ 0.018	0.171 $\pm$ 0.020	0.829 $\pm$ 0.020
10-29.9	0.059 $\pm$ 0.015	0.256 $\pm$ 0.014	0.315 $\pm$ 0.018	0.685 $\pm$ 0.018
30-59.9	0.090 $\pm$ 0.016	0.319 $\pm$ 0.018	0.409 $\pm$ 0.016	0.591 $\pm$ 0.016
60-100	0.082 $\pm$ 0.017	0.336 $\pm$ 0.021	0.418 $\pm$ 0.018	0.582 $\pm$ 0.018
All strata	0.071 $\pm$ 0.003	0.281 $\pm$ 0.005	0.352 $\pm$ 0.006	0.648 $\pm$ 0.006

^a Weighted mean proportions ($\pm$ SE) for the entire piñon/juniper study area (i.e., across strata) are also provided.

Table 3—For each percent-cover stratum, the mean proportion ($\pm$ SE) of total tree cover that has died, and mean weighted proportion ($\pm$ SE) across all strata are provided

Stratum (percent-cover range)	Mean proportion of total tree cover that is dead
0-9.9	0.125 $\pm$ 0.031
10-29.9	0.186 $\pm$ 0.033
30-59.9	0.221 $\pm$ 0.033
60-100	0.197 $\pm$ 0.033
All strata	0.196 $\pm$ 0.008

Table 4—Approximate material costs and labor for an experienced analyst to conduct a multistage sample to estimate piñon mortality in the Williams Ranger District

Expense item	Labor (days)	Cost
Materials		\$5,050
Orthorectification of aerial imagery	9	
Percent-cover stratification	9.5	
Dot-grid sampling	10.5	
Calculating results	1	
Total	30	\$5,050

extensive mortality in many of the Nation's forests and rangelands. The severity and large extent of mortality, coupled with the lack of routine monitoring in some areas, has made it difficult to assess the damage, ecological impact, and fire hazard in a timely and cost-effective manner.

The multistage sampling method used in this study offers a statistically rigorous approach to estimate tree mortality across large geographic areas, increase sampling efficiency, and provide a confidence interval about the mean. By using a digital dot-grid to assess mortality from high-resolution aerial imagery, the need for time consuming and costly field assessments was eliminated. The use of imagery also allowed samples to be drawn from remote areas that would have been prohibitively time consuming to sample in the field. This method was tested over a 332,000-acre piñon/juniper woodland west of Flagstaff, Arizona, but the technique is applicable to many other forest types

impacted by insects and diseases. The multistage sample estimated that, as of spring 2004, dead-tree canopies occupied approximately 7 percent of the area within the piñon/juniper woodlands of the Williams Ranger District. Relative to the total tree cover, approximately 20-percent mortality was sustained. For the piñon/juniper study area, the cost to estimate mortality was approximately \$0.04 per acre. In other areas, the cost per unit area to conduct a similar study could vary greatly depending on the location, size of the study area, and imagery requirements.

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Modeling Potential Movements of the Emerald Ash Borer: the Model Framework

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Abstract

The emerald ash borer (EAB, *Agrilus planipennis* Fairmaire) is threatening to decimate native ashes (*Fraxinus* spp.) across North America and, so far, has devastated ash populations across sections of Michigan, Ohio, Indiana, and Ontario. We are attempting to develop a computer model that will predict EAB future movement by adapting a model developed for the potential movement of tree species over a century of climate change. We have two model variants, an insect-flight model and an insect-ride model to assess potential movement.

The models require spatial estimates of EAB abundance and ash abundance. The EAB abundance map shows a zone of initial infestation in the western suburbs of Detroit, with ash trees first dying about 1998. The fine-scale (270-m cells) ash basal area maps show highly variable values, but woodlots often have very high levels of ash. At the coarse scale (20-km cells) for the Eastern United States, available ash is high throughout the northern part of the country.

With the flight model, probability of movement is dependent on EAB abundance in the source cells, the quantity of ash in the target cells, and the distances between them. With the insect-ride model, we used geographic information system data to weight factors related to potential human-assisted movements of EAB-infested ash wood or just hitchhiking insects. We are developing a gravity model

that considers traffic volumes and routes between EAB source areas and various distances to campgrounds.

Preliminary results from a test strip through northern Ohio show (1) the insect-flight model creates a relative probability of colonization that decreases quickly from the EAB range boundary edge; and (2) the insect-ride model provides occasions for long-distance transport via human-aided dispersals.

Keywords: Ash, dispersal, emerald ash borer, invasive, Ohio.

Introduction

The emerald ash borer (EAB), *Agrilus planipennis* Fairmaire (Coleoptera: Buprestidae), poses a serious threat to all ash trees in North America. The larvae feed on phloem, producing galleries that eventually kill large trees in 3 to 4 years and small trees in as little as 1 year (Poland and McCullough 2006). A native of Northeastern China, Korea, Japan, Mongolia, Taiwan, and Eastern Russia, the species was first identified in the United States near Detroit, Michigan, in July 2002 (Haack 2006). The borer was probably imported into Michigan in the early 1990s via infested ash crating or pallets (Herms and others 2004).

The impact of EAB may be enormous. An estimated 8 billion ash trees exist in the United States, comprising roughly 7.5 percent of the volume of hardwood sawtimber, 14 percent of the urban leaf area (as estimated across eight U.S. cities), and with a value exceeding \$300 billion (Poland and McCullough 2006).

Research into the spatial distribution of the host ash species helps us better understand the resource at risk and the potential for EAB spread. The USDA Forest Service's Forest Inventory and Analysis (FIA) units continually conduct inventories across more than 100,000 plots in the Eastern United States (Miles and others 2001). This invaluable data source provides the information critical to the assessment of the ash resource, including the work reported here. For a detailed look, we rely on 30-m Landsat data that have been classified into forest types with associated ground samples to calculate ash content.

There are a variety of approaches for using computer models to predict the risk and spread of invasive insects (e.g., Rykiel and others 1988, Sharov and Liebhold 1998, Sharov and others 1997, Sturtevant and others 2004, Turchin 2003). To summarize, modeling insect movement is a complicated venture, especially in heterogeneous landscapes. BenDor and Metcalf (2006) and BenDor and others (2006) have initiated a dynamic modeling approach to learn more about EAB spread and possible mechanisms to retard it. These authors also call for high resolution data on the ash resource and human-assisted components such as campgrounds to move this work forward. In this paper, we present a slightly different approach using higher resolution databases.

The objectives of this work are to (1) evaluate the ash quantity across the Eastern United States at a coarse level and in Ohio at a fine scale; (2) assess EAB spread and rate of spread through the region so far; and (3) begin to model future spread through the two modeling approaches of insects flying and insects riding with humans.

Methods

Distribution of Ash

Fraxinus (ash) is the only genus that EAB has attacked in North America. There is no observed host expansion. Consequently, we mapped ash availability as a resource for EAB spread, at a coarse resolution for the Eastern United States (20- by 20-km cell) and at a fine resolution for Ohio (30- by 30-m cell size).

Coarse-Level Analysis of Ash for the Eastern United States—

We used FIA data (Miles and others 2001) to determine ash distribution and abundance for 9,782 cells over the Eastern United States. We created these data sets for a climate change atlas (Iverson and others 1999) and have made these data available (Prasad and Iverson 2003). We summed the data for four species of ash that comprise the vast majority of ash in the Eastern United States: *Fraxinus americana* L. (white ash), *F. pennsylvanica* Marsh. (green ash), *F. nigra* Marsh. (black ash), and *F. quadrangulata* Michx. (blue ash). This effort produced maps of relative availability of rural

ash (FIA does not sample urban areas well) to the EAB. The methodology used data from the 100,000+ forested plots to calculate the basal area (BA) of ash per plot, then calculated average BA of ash per unit area based on all the plots within the 20- by 20-km cell. One drawback of this methodology is that in some cells with small quantities of forest, no FIA plots were established so that the average ash BA is calculated as zero for that cell. In reality (as seen in the fine-scale analysis), every cell in the Midwest has at least some forest (and, most likely, some ash). Another drawback is that the urban forest resource is underrepresented in the FIA data. (There is an ongoing effort within our group to conduct surveys to better understand the forest resource in certain urban areas.)

The next step was to create a map of percentage of forest cover by 20- by 20-km-grid cell. We acquired classified 30 m Landsat TM-interpreted data from Riitters and others (2002). The data were reclassified into forest or nonforest and tallied by 20- by 20-km cell to yield an estimated percentage of forest cover for each cell.

Data from the average BA map was multiplied by the percentage of forest cover per 20- by 20-km cell, providing an estimate of the total availability of the ash resource to the invasive species, in thousands of square feet of BA per 20 by 20 km.

Fine-Scale Analysis of Ash in Ohio—

For a detailed estimate of ash resource availability in Ohio, we combined estimates of ash BA per FIA plot with a Landsat TM-based classification of forest types. We acquired the landcover data from the Ohio Gap Analysis Program (GAP) (Ramirez and others 2005). This data set contains vegetation classes based on leaf-on and leaf-off imagery from 1999 to 2002 (Ramirez and others 2005), at 30-m resolution. Vegetation classes conform to the NatureServe's Ecological Classification system (Comer and others 2003). We attempted to establish ash percentages for 28 classes, including the following classes important in northwestern Ohio: row crop, open water, low density urban, high density urban, urban forested, grassland, evergreen forest, North-Central Interior Dry Oak Forest and Woodland (CES202.047), North-Central Interior Floodplain (CES202.694), North-Central Interior Wet

Flatwoods (CES202.700), and North-Central Oak Barrens (CES202.727).

To estimate ash BA in each landcover type, 2,298 FIA plots were overlaid on the Ohio GAP data set by Elizabeth LaPoint, FIA geographic information system (GIS) Service Center. Due to the restriction on releasing FIA plot coordinates, Ms. LaPoint performed the overlay and reported the average ash BA per class. However, only a portion of the plot coordinates (ca 40 percent) were global positioning system (GPS) corrected so that locational error is present in the overlay (which caused, for example, some plots to appear in open water). For certain classes, including the urban classes, row crop, and oak barrens, we believed, based on number of FIA plots available and ancillary information, the data reported for southern Michigan by McFarlane and others (2005) better represented the quantity of ash present in the types. So, the McFarlane and others data were substituted for the FIA estimates.

Finally, an EAB habitat availability map was prepared by applying the estimated BA of ash per class. To prepare a smoother map, which was needed for the modeling effort, we calculated the mean ash BA per 270-m pixel (a 9 by 9 cell average). This product was used for mapping and modeling, which requires estimates of ash quantities per cell (see “Modeling Spread in Ohio”).

Mapping Estimated EAB Spread, 1998–2005

To model potential future spread and assess observed spread rates, a preliminary map of historical spread was created. For this, multiple data sets and GIS manipulations were used, and for the most part, represent the spread of visible damage to ash trees rather than the initial infestation of EAB. The Michigan Department of Natural Resources (DNR) mapped pest outbreaks, which estimated the range of EAB-damaged ash in 2002 and 2003. These were accepted as accurate. We also used data from interstate highway exit surveys for 2003 and 2004, with 10 ash trees per exit tallied for death/life (irrespective of what killed the ash) by Smitley (2005) and Smitley and others (2005). These data were useful to show where ash death first occurred in the region. A Michigan ash damage survey from September 2004 also was used (Michigan State University, 2006) along

with the actual locations and density of extent of known EAB locations as of December 2005, obtained from the Cooperative Emerald Ash Borer Project (2006). In addition, multiple dates of these national EAB positive maps were acquired to detect additional finds temporally. Finally, our own field work on ash tree assessment in northern Ohio and southern Michigan during the summers of 2004 and 2005 yielded additional spatial information, particularly on ash not yet visually affected by EAB. Again, most of the data were based on visual observations of damage, which can be delayed for several years after EAB colonization in otherwise healthy, larger ash trees. It should also be noted that girdled detection trees were used in Michigan in 2004 and 2005, which increased detection capability over that available from visual inspection alone. Thus, this new survey method may have artificially expanded the estimated front in 2004 and 2005 over what would have been detected with only observable symptoms. It definitely expanded the number of outliers detected outside the main front.

Known EAB locations were inputted into ArcGIS (GRID - focalsum commands), and four EAB abundance classes were derived based on the number of EAB positives recorded within three spheres of influence around each point: 1, 2.5, and 5 km. Each 270-m cell within the study area was assigned EAB abundance class 0 (no EAB positives within 5 km), 1 (1 to 5 positives within 5 km), 2 (1 to 3 positives within 2.5 km or 6 to 10 positives within 5 km), or 3 (1 to 38 positives within 1 km, or 4 to 100 positives within 2.5 km, or 11 to 110 positives within 5 km). The resulting map presented the best summary of EAB concentration areas as of December 2005.

Armed with all the above data sets overlaid in ArcGIS, we manually drew estimated boundaries of the EAB-damaged ash front for 2005, 2004, 2001, 2000, 1999, and 1998. Lines were drawn to encompass the higher EAB density zones, the mortality estimates, and the nearby new finds for each year. For 2002 and 2003, we used the pest maps from Michigan DNR mentioned above. Although EAB probably entered the United States prior to 1998 and was likely present in these trees prior to that date, we started with 1998 as the first year to estimate the visual damage front based primarily on the Smitley (2005) data. These

data indicated that ash tree mortality was already quite high in that zone by 2003, which coincided with our assumption that it takes about 6 years for a cell to reach peak infestation (see “Gravity Model Scenarios”). Subsequent studies of tree rings in the initial zone of infestation have indicated that initial death of ash trees occurred in 1997 (Nathan Siegert. Personal communication, 2006. Department of Forestry, Michigan State University, East Lansing, MI 48824). The final map shows estimated limits of the front by year for 1998–2005.

Modeling Spread in Ohio

Most of the data collected in the preceding sections was a prerequisite for efforts to model the spread of the EAB. We have worked some years on a SHIFT model, designed to estimate the potential migration of trees under the northward climatic pressure (Iverson and others 1999, 2004a, 2004b; Schwartz and others 2001). This model was adapted to work for the spread of EAB. The fundamental basis of this model is a spread model that is driven by existing local density of infestations, ash BA, and the distance of habitat patches to known and modeled infestations. This basic model required modification based on the idea that EAB spread is facilitated through human activities (insect-ride model).

The formula SHIFT uses to calculate the probability of an unoccupied cell becoming colonized during each generation is:

$$P_{\text{colonization}, i} = HQ_i \left(\sum HQ_j \times F_j \times (C/D_{i,j}^X) \right)$$

where $P_{\text{colonization}, i}$ is the probability of unoccupied cell i being colonized by at least one individual and surviving into reproductive status; HQ_i and HQ_j are habitat quality scalars for unoccupied cell i and occupied cell j , respectively, that are based on the basal area of ash in each 270-m cell; F_j , an abundance scalar (0 to 1), is related to the current estimated abundance of EAB in the occupied cell j ; and $D_{i,j}$ is the distance between unoccupied cell i and an occupied cell j .

The colonization probability for each unoccupied cell, a value between 0 and 1, is summed across all occupied cells at each generation. Thus, an unoccupied cell very close to numerous occupied cells may end up with a colonization

probability greater than 1.0. These cells are modeled as colonized. For cells with summed colonization probabilities less than one, a random number less than 1.0 is chosen, and all cells with a probability of colonization that exceeds the random number are colonized in that model step. Those “newly colonized” cells then contribute to the colonization probability of unoccupied cells in the next model time step. The value of C , a rate constant, is derived independently through trial runs to achieve a migration rate of approximately 20 km per year under high ash BA and moderate EAB abundance. The value of X , or dispersal exponent, determines the rate at which dispersal declines with distance. Being in the denominator, this decreases colonization with distance as an inverse power function. Further discussion on the dispersal function can be found in Schwartz and others (2001).

Insect-Flight Model—

With the insect-flight model, we use the modified SHIFT model to advance the front based on the current front location, the abundance of EAB behind the front, and the quantity of ash ahead of the front. The model runs at a 270-m cell size, and based on the known progression of EAB densities and ash mortality in outlier zones, we assume an 11-year cycle for EAB initial infestation to death of all ash trees in the cell. EAB abundance in the cell was assumed to form a modified bell-shaped curve, with maximum abundance (multiplier = 1) in years 6, 7, and 8; a 0.6 multiplier in years 5 and 9; a 0.14 multiplier in year 4; a 0.011 multiplier in year 3; a 0.0003 multiplier in years 2 and 10; and a 0.0001 multiplier in years 1 and 11. The assumptions for this curve include a slow EAB population increase for the first few years after colonization, followed by peak infestation for 3 years starting with year 6, followed by a rapid decline as all the ash trees in the cell die off in years 9 to 11. The fine-scale ash BA for Ohio was normalized to 0 to 100 and also used as a multiplier. The 11-year cycle may be a liberal assumption on how fast the EAB infestations can grow, as there is some evidence that it may take as long as 10 years for populations to peak (rather than the 6 we assumed). For each cell, the program calculates the probability of new colonization, based on a small probability that the insect

will fly from an occupied cell to an unoccupied cell, for all surrounding cells within a specified search window (40 km in this case). Once selected for colonization, the cell starts the 11-year cycle of EAB increasing and then decreasing as ash dies out.

Insect-Ride Model—

To develop the insect-ride model, we used GIS data to weight factors related to potential human-assisted movements of EAB-infested ash wood or just hitchhiking insects: roads, urban areas, various wood products industries, population density, and campgrounds. Each of these five factors was converted into weighting layers that became multipliers for the ash BA component of the insect-ride model. That is, the increase in probability of EAB infestation by the insect-ride factors is made manifest by increasing the amount of ash available in those cells. Thus, if no ash exists in the cell, it matters not whether there is an escaped EAB from one of the human-assisted vectors, but if there is a large ash component, an escaped EAB could quickly find a place to colonize.

To register the increased probability of insects riding on windshields, radiators, or otherwise attached to vehicles moving down the road, we assigned weights to two widths of major road corridors. We used the U.S. Geological Survey major roads data and created buffers of 1 and 2 km, with a scoring of 10 for 0 to 1 km and 5 for 1 to 2 km distance from the roads.

For urban areas, where there is much more vehicular density and opportunity for EAB transport, we assigned values of 7 if the urban center size was less than the median size and 10 if greater than the median. We therefore assume larger cities will have greater chance of EAB infestation via human movement. Data were acquired for the State of Ohio urban centers from the Department of Transportation Office of Technical Services (Ohio Department of Transportation 2006).

Related to the urban areas, weighting is the population density scoring by zip code. This factor creates a wall-to-wall scoring and distinguishes rural from more urbanized areas. Data were acquired from the U.S. Census Bureau, which included population estimates for 2001 by zip code

area. Population densities were divided into six classes with scoring as follows: 1 = 1 to 100 people/km²; 2 = 101 to 200; 4 = 201 to 800; 6 = 801 to 2,000; 8 = 2,001 to 4,000; 10 = 4,001 to 16,582.

Wood products industries also have been responsible for some EAB movement, so a scheme was developed to weight buffers around individual businesses dealing in wood products. We performed an analysis of potential industries carrying wood products, based on the listing of SIC codes from Dunn and Bradstreet. We scored each industry for likelihood of EAB getting to the site and emerging based on our estimate of the amount and status of ash used in the industry: 0 = none; 2 = small likelihood; 4 = somewhat likely; 6 = higher likelihood. For example, forest nurseries and wood pallet industries scored a 6, whereas manufacturers of decorative woodwork or wooden desks scored a 4 (mostly used kiln-dried wood), and manufacturers of pressed logs of sawdust or woodchips scored a 2. Movement of material from nurseries historically has been a source for several infestations, which are not accounted for in this model. Presumably, this source has been slowed recently via quarantine regulation. Next, buffer distances around the businesses were created based on the number of employees (surrogate for size or volume of wood) working at the facility. For 1 to 10 employees, the buffer of 0 to 1 km scored 8, and the 1 to 2 km buffer scored 3; for 11 to 50 employees, the buffer of 0 to 1.5 km scored 9, and the 1.5 to 3 km buffer scored 4; and if the facility had more than 50 employees, the 0 to 2 km buffer scored 10, and the 2 to 4 km buffer scored 5. Because facilities could be within each other's buffer space, scores were added, and the maximum score over the study area was 22.

Finally, campgrounds were considered likely destinations of human-assisted EAB transport, primarily through the (mostly illegal) movement of firewood. The general public is the primary vector, so it is much more difficult (relative to industry vectors) to achieve education, regulation, and enforcement goals related to stopping EAB spread. Campgrounds were treated in two ways: through the weighting scheme described here and the gravity model described in the next section. Campground locations were acquired from Dunn & Bradstreet (unpublished data

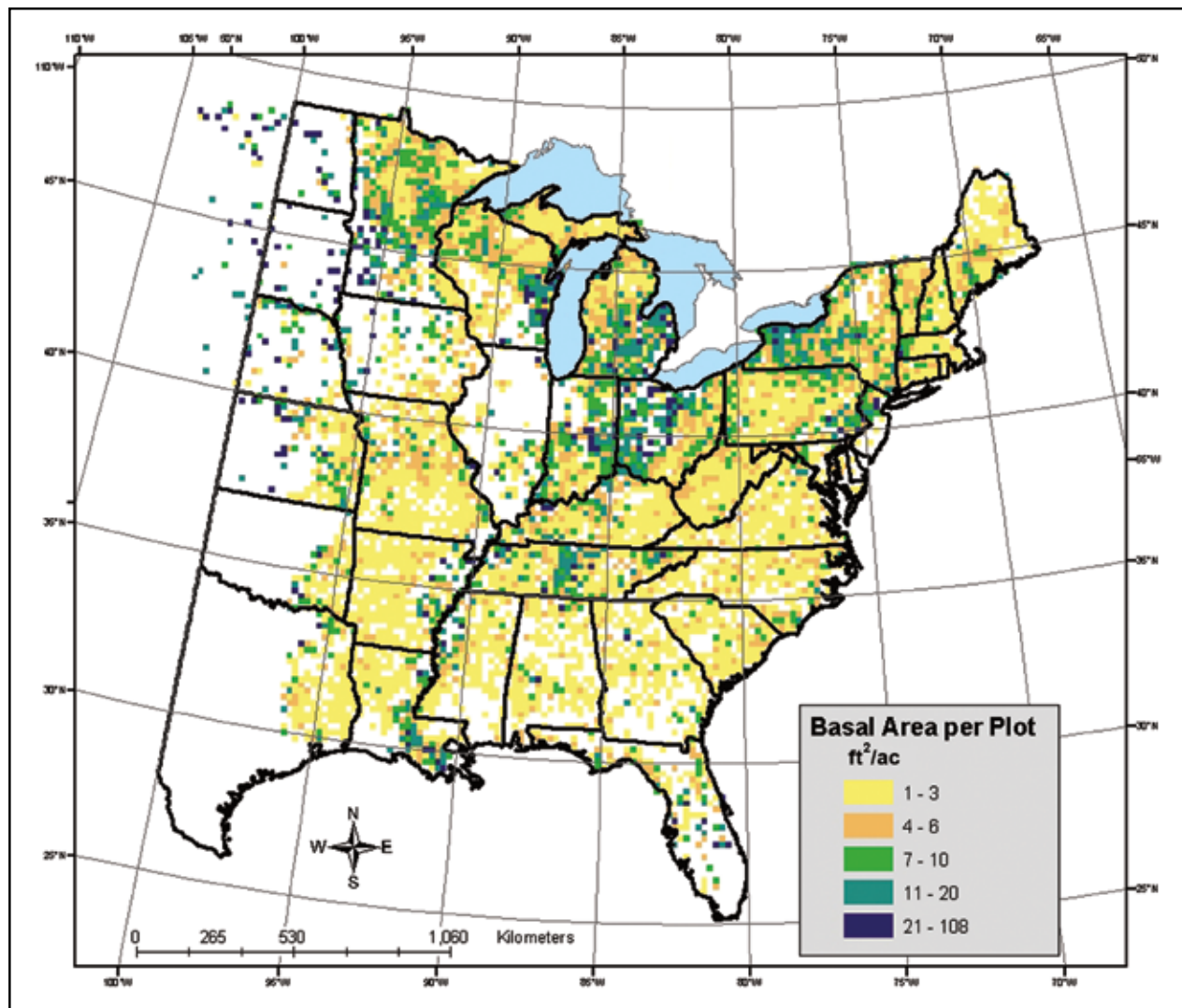


Figure 1—Ash basal area (square feet per acre for 20- by 20-km cells) across the Eastern United States, from Forest Inventory and Analysis data for white, green, black, and blue ash.

purchased by Iverson) and the AAA Travel and Insurance Company (unpublished data provided to Bossenbroek). Similar to that described for wood products industries, we base the weighting on both distance (from the camp headquarters) and number of campsites. For campgrounds with less than 50 campsites, the buffer of 0 to 0.5 km scored 10, and the buffer of 0.5 to 1 km scored 5; for 51 to 200 campsites, the equivalent buffers were 0 to 1 (10 points) and 1 to 2 km (5 points); for 201 to 400 campsites, buffers were 0 to 1.5 and 1.5 to 3 km; for 401 to 600 campsites, buffers were 0 to 2 km and 2 to 4 km; and for more than 600 campsites, buffers were 0 to 2.5 km and 2.5 to 5 km.

Gravity Model Scenarios—

In the second approach used with campgrounds, we are developing a gravity model (Bossenbroek and others 2001) that considers traffic volumes and routes between EAB source areas and various distances to campgrounds (Muirhead and others 2006). Muirhead and others (2006) presented an initial model predicting human-mediated dispersal of the EAB through the movement of campfire wood. Given the rapid spread of the EAB and a need for a quick response, simple models based on simple assumptions, such as developed by Muirhead and others (2006), are an essential step. One of the goals of this project is to incorporate

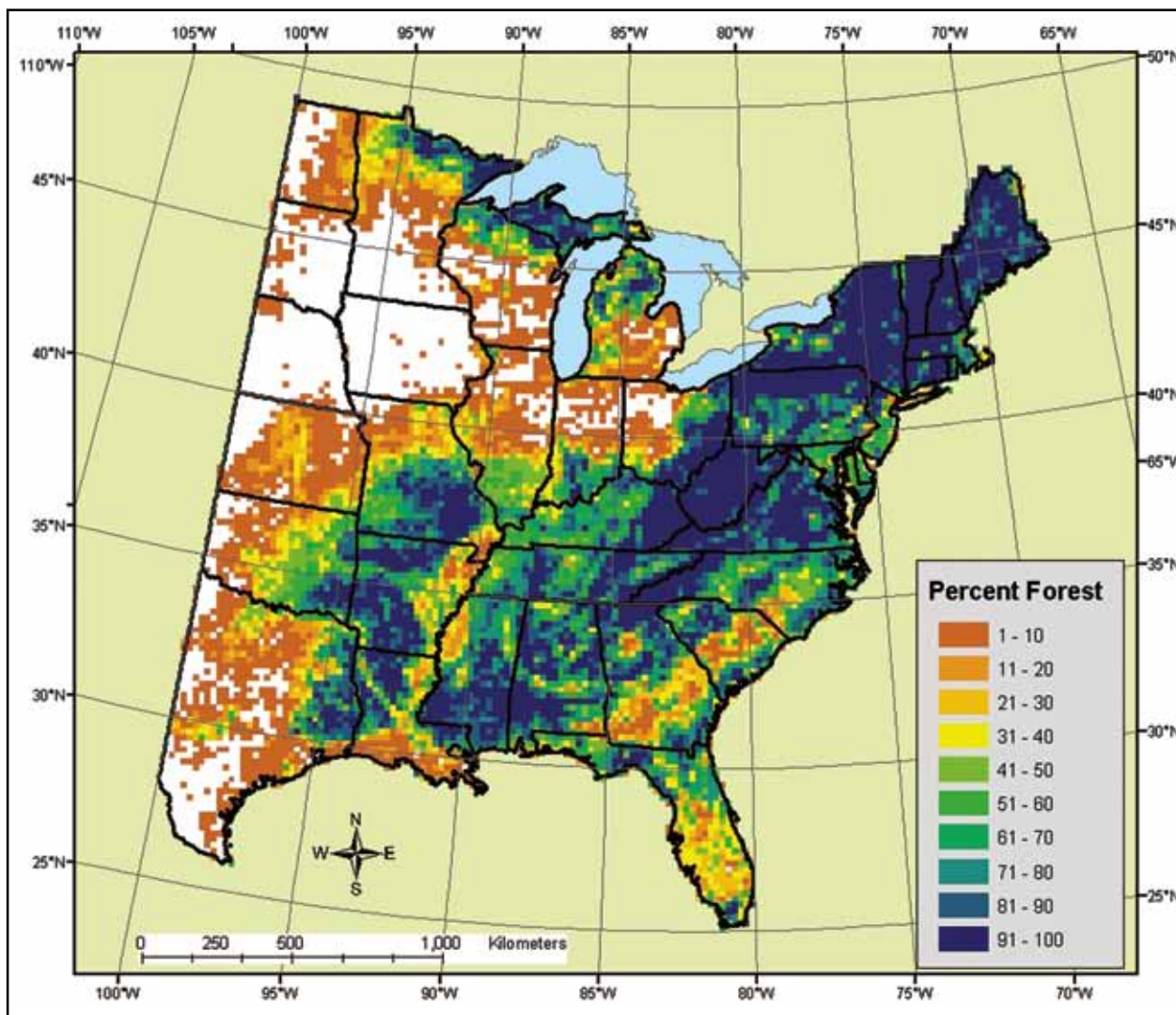


Figure 2—Percentage of forest cover per 20- by 20-km cell, based on the forest/nonforest classification of all the classified 30-m pixels within each cell.

more detail into the models of long-distance dispersal of the EAB. Empirical data on the use of campgrounds, i.e., reservation data, is only available for public campgrounds; thus to incorporate private campgrounds, a modeling framework is necessary. Here we develop a gravity model for Ohio to predict the relative number of campers traveling from EAB infested areas to the campgrounds of Ohio.

Gravity models calculate the number of individuals, (e.g., campers) who travel from location i to destination j , (e.g., a campground), T_{ij} , as estimated as

$$T_{ij} = A_i O_i W_j c_y \quad (1)$$

where, A_i is a scalar for location i (see below), O_i is the number of people at location i , W_j is the attractiveness of location j , c_{ij} is the distance from location i to location j , and α is a distance coefficient, or distance-decay parameter, which defines how much of a deterrent distance is to interaction. A_i is estimated via

$$A_i = 1 / \sum_{j=1}^N W_j c_y, \quad (2)$$

where N represents the total number of destinations, and j represents each destination in the study region. A production-constrained gravity model of the movement of firewood thus requires information on the number of campers,

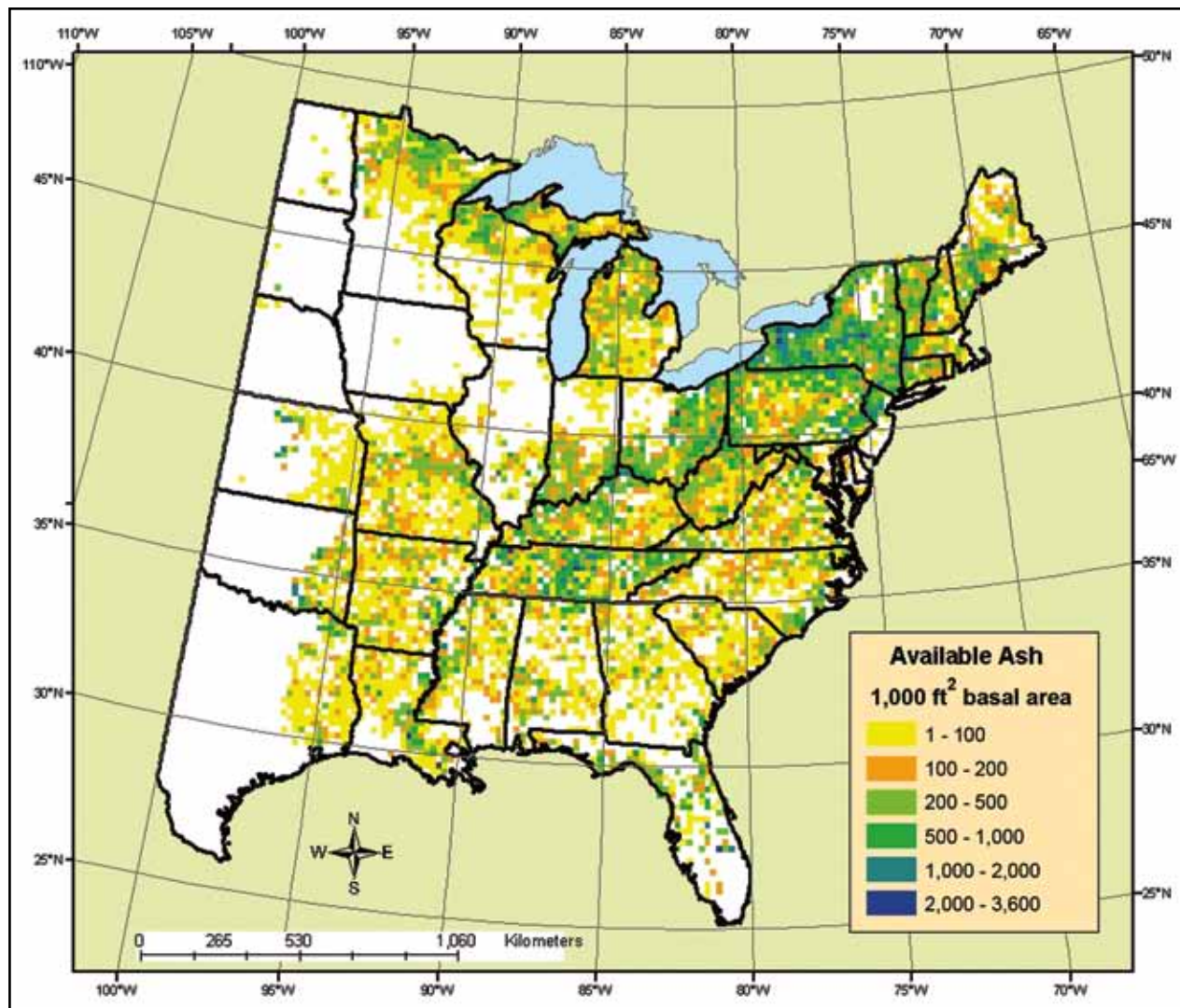


Figure 3—Amount of ash available (square foot basal area of ash per 20- by 20-km cell) to the emerald ash borer. It is the product of Figures 1 (basal area of ash) and 2 (percent forest).

the residency of the campers, the location of potential destinations (i.e., campgrounds), the attractiveness of those destinations, and the distribution of the EAB (i.e., source locations). The spatial resolution of our gravity model is based on ZIP code regions for the residency of campers and the point locations of campgrounds.

Based on data from Dunn & Bradstreet and the AAA Travel and Insurance Company, we identified the location of 241 public and private campgrounds in Ohio. For a measure of attractiveness for each campground (w) we initially are using the number of camp sites at each location. Other factors, such as proximity to boating, fishing, and hiking,

are likely to influence the attractiveness of individual campgrounds, but these data are unavailable on a regional and consistent basis. The distance between a ZIP code and a campground (c) was calculated as the road network distance between these locations. For simplification, the road network is based on all roads with either a State or Federal designation and excludes local roads. The point of origin for each ZIP code was determined as the road location nearest the centroid of the ZIP code region. Likewise, for each campground, the point location was determined as the point on the nearest road to the campground. The result of the gravity model is a prediction of the number of campers that

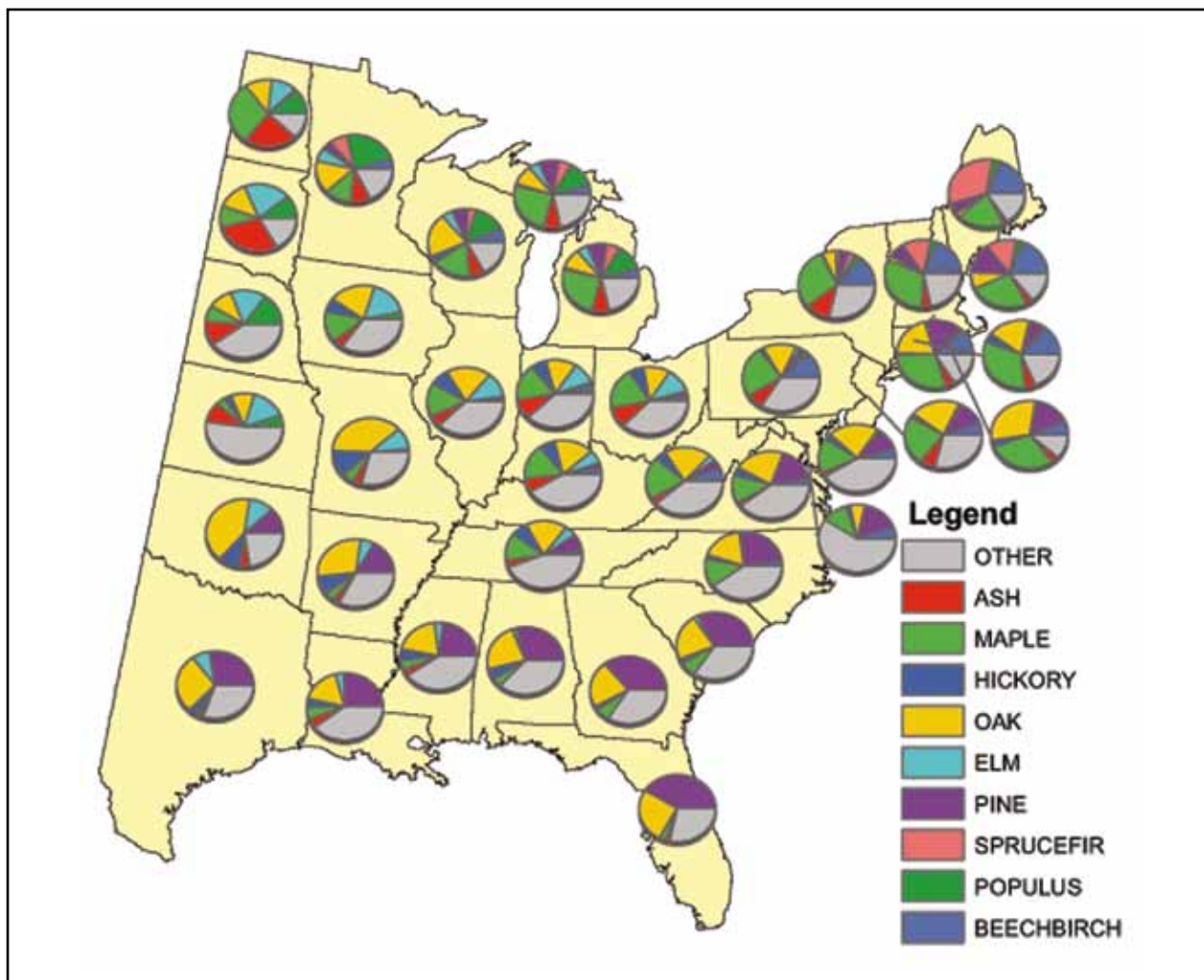


Figure 4—Proportion of various genera of trees, based on basal area as calculated from data depicted in Figure 3. The proportions of ash are highest in the Northern States.

travel from an area of EAB infestation to each particular campground.

To estimate the distance coefficient (α), we compared our gravity model with reservation data obtained from the Ohio Division of Parks and Recreation for 58 state parks. These records contained the number of reservations for each campground summed by ZIP code of the camper's residence. We used sum of squares to measure goodness-of-fit between model predictions and the observed data. To identify the best-fit model, the value of α was systematically assessed over a range from 0.1 to 10. By fitting the model to the reservation data for Ohio state parks, we assume that campers using private and public campgrounds behave in

the same manner, i.e., distance and attraction affect their travel decisions in the same manner.

Once the gravity model was parameterized, we used the estimated distance coefficient value to determine the expected number of campers that would travel to all 241 campgrounds within Ohio. We reported the percentage of campers coming from EAB-infested ZIP codes (as of 2003) traveling to each campground in Ohio to give a relative estimate of risk.

Results and Discussion

This project is a work in progress, and consequently, results presented in sections 3.1 and 3.2 could change pending new

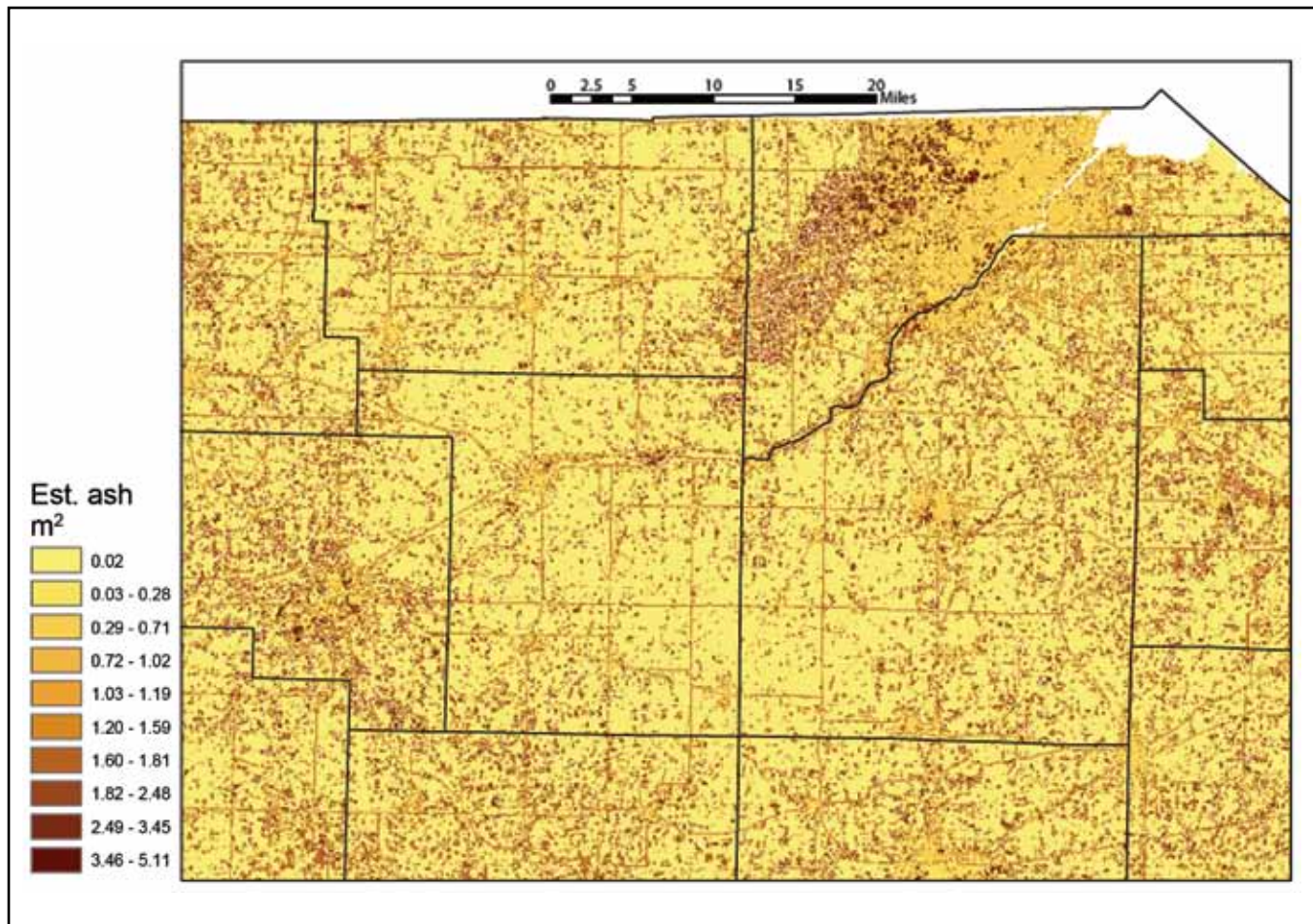


Figure 5—Estimated distribution of ash for a portion of northwest Ohio, calculated at a resolution of 270 m (but not resolved that finely for this map). Data are based on classified 30-m Landsat imagery and Forest Inventory and Analysis plot data for the entire State.

data or analysis or both. Results reported in “Modeling Spread in Ohio” are very preliminary.

Distribution of Ash

Analysis of the distribution of ash at two scales showed two facts: there is a lot of ash available to the insect, and it is distributed throughout the Eastern United States. Consequently, the EAB threat is real for most communities and rural locations throughout the region.

Coarse-Level Analysis of Ash for Eastern United States—

The map of ash BA (including white, green, black, and blue ash) per unit area of forest shows there is a great deal of ash in the woodlots and small forests common within the current range of the EAB (southern Michigan, northern Ohio,

northeastern Indiana) (Figure 1). However, the amount of forest in that zone is limited (Figure 2), so the total available ash is less compared to the more forested regions (Figure 3). Of major concern is the large amount of ash available just south of Lake Erie (northeast Ohio, northwest Pennsylvania) and Lake Huron (western New York). The western edge of this zone is just now being reached by the EAB.

These maps show a high level of ash availability in the zones surrounding the borer’s current range, indicating a difficult control task ahead.

Figure 4 shows a map with the proportions of various genera of trees in each State of the Eastern United States. Ash comprises a significant proportion of basal area across the Northern States, but is less prevalent in the Southeastern States.

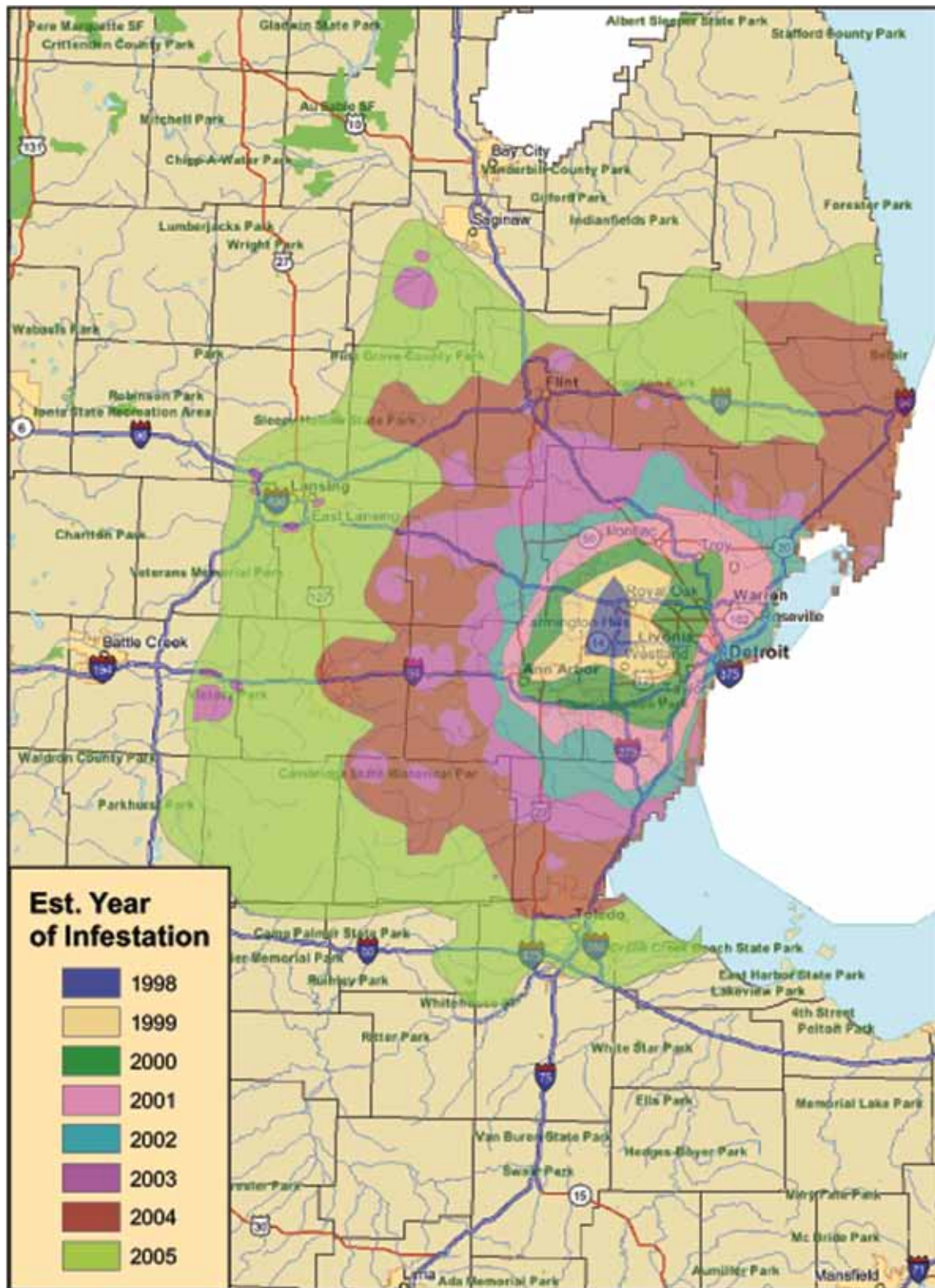


Figure 6—Estimated emerald ash borer front spread by year, 1998-2005, as estimated from a variety of data.

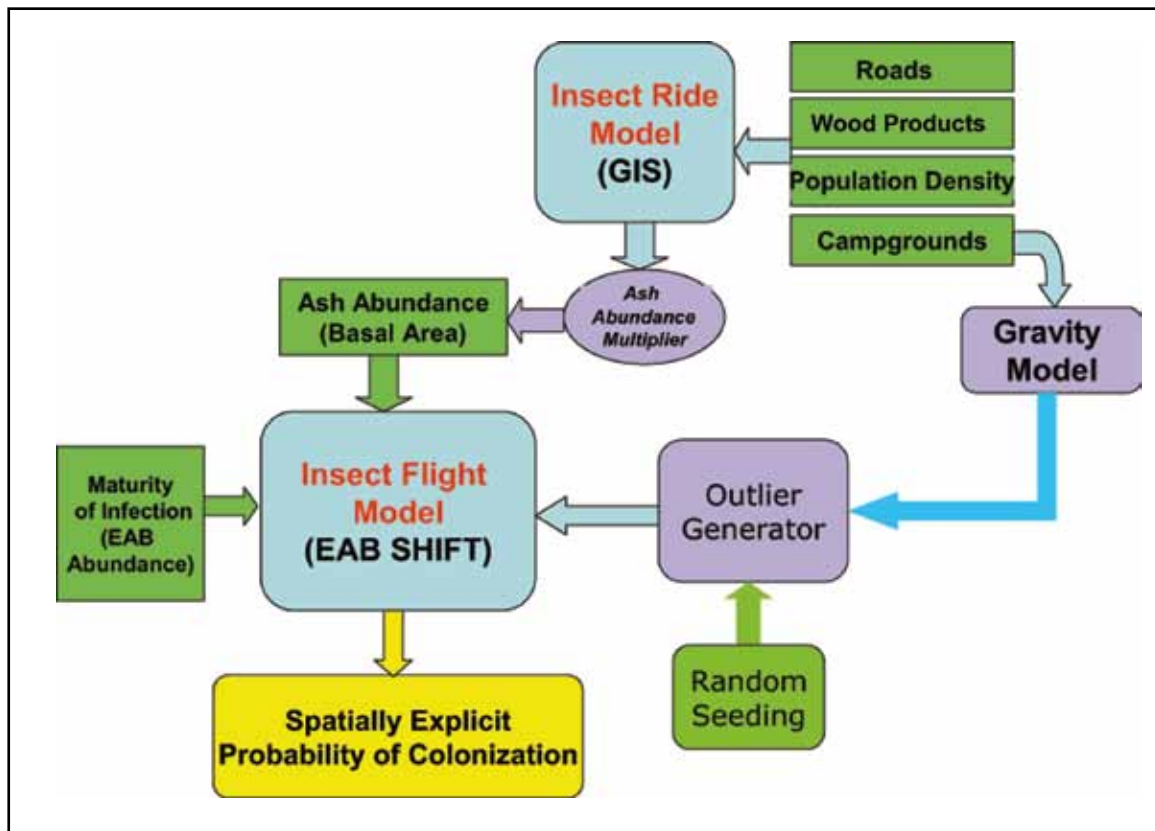


Figure 7—Modeling framework showing two major components—the insect flight and insect ride models. The insect ride model considers both a geographic information system weighting scheme and the gravity model of attractiveness of campgrounds for receiving emerald ash borer (EAB) from the core zone in southern Michigan. The insect flight model considers the observed rate of spread of 20 km/yr, the emerald ash borer (EAB) abundance, and the abundance of food for EAB.

Fine-Scale Analysis of Ash for Ohio—

The fine-scale analysis for Ohio, using 30-m data and plot information, shows an estimate of the urban and riparian zones with levels of ash (BA) (Figure 5). Most of the area shown in Figure 5 is agricultural land, but ash is maintained in the landscape even in these croplands along roadsides, ditches, and small wetlands. There are also numerous woodlots, many of which contain high proportions of ash.

Mapping Estimated EAB Spread, 1998–2005

The map of estimated EAB front locations was required for two reasons: (1) to create a baseline from which our spread modeling will commence; and (2) to estimate the average historical rate of spread that will help calibrate the model. The resulting map (Figure 6) shows expansion from a core area in western Detroit, with substantial concentric

movement each year. Using these data, and assuming a start date of 1998, we calculated an average spread rate of approximately 20 km/yr for the years 1998 through 2005. This expansion rate is much faster than the field and laboratory dispersal (flight) studies that have been presented thus far of 1 km/yr (McCullough and others 2005) to 4.8 km/yr (Taylor and others 2005), respectively. Clearly, much of the historical movement of the front, as we detected it here, is hastened by shorter human-assisted movements, and the two mechanisms (flight vs. ride) cannot be clearly distinguished from each other in the real world.

Modeling Spread in Ohio

We present a modeling framework that considers both the insect- and the human-controlled dispersal mechanisms (Figure 7). Though we have not completed this work, we have some preliminary results, which are presented here.

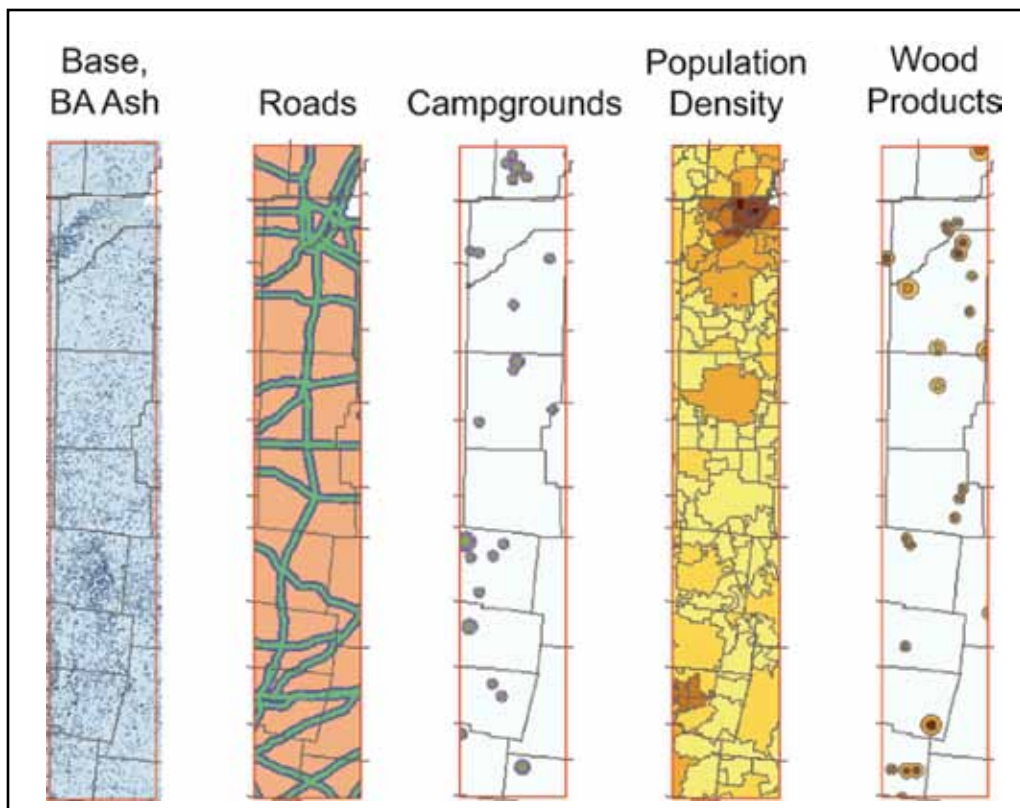


Figure 8—Insect-ride components. Small map strips of Ohio stretching south from Toledo to Columbus, showing more detail of relative weights of the area for basal area of ash, roads, campgrounds, population density, and wood products industries.

Insect-Ride Model—

When we include the five factors of human-assisted dispersal, all of the land is affected to some degree (Figure 8). These factors together modify the environment for susceptibility for EAB invasion in our model by supplying a multiplier to the quantity of ash available to the EAB. In our example section of Ohio, we see that the largest multipliers will be in the densely populated centers, especially where there are wood products industries and roads nearby. We have yet to experiment with various weighting schemes among the five factors. For example, we plan to incorporate relatively more influence of campgrounds, probably via the gravity model.

Gravity Model—

In evaluating Ohio campgrounds, we demonstrate the influence of proximity to the core area of EAB presence. Figure 9 shows that the higher scores (larger symbols) are at campgrounds with more campsites (=more attractiveness),

with more traffic, and that are closer to the core area of EAB infestation in southern Michigan. The areas around these larger symbols are potential areas that should be monitored with detection trees and visual inspections, as new outliers may emerge near these zones.

Insect-Flight Model—

The EAB Shift model produces an estimate of relative probability of colonization away from the already occupied zones. Figure 10 shows the preliminary results of a test strip from Toledo to Columbus, Ohio (same strip as shown in Figure 8). The relative probability of colonization decreases quickly from the EAB range boundary edge (Figure 10, top strip). When we add the influence of single factor weights (e.g., roads, campgrounds, population density, and wood products industries), there are some minor variations that align with the weights in the preliminary output (Figure 10). We emphasize that this example is only to show the kinds of outputs we are pursuing and that the testing and calibration

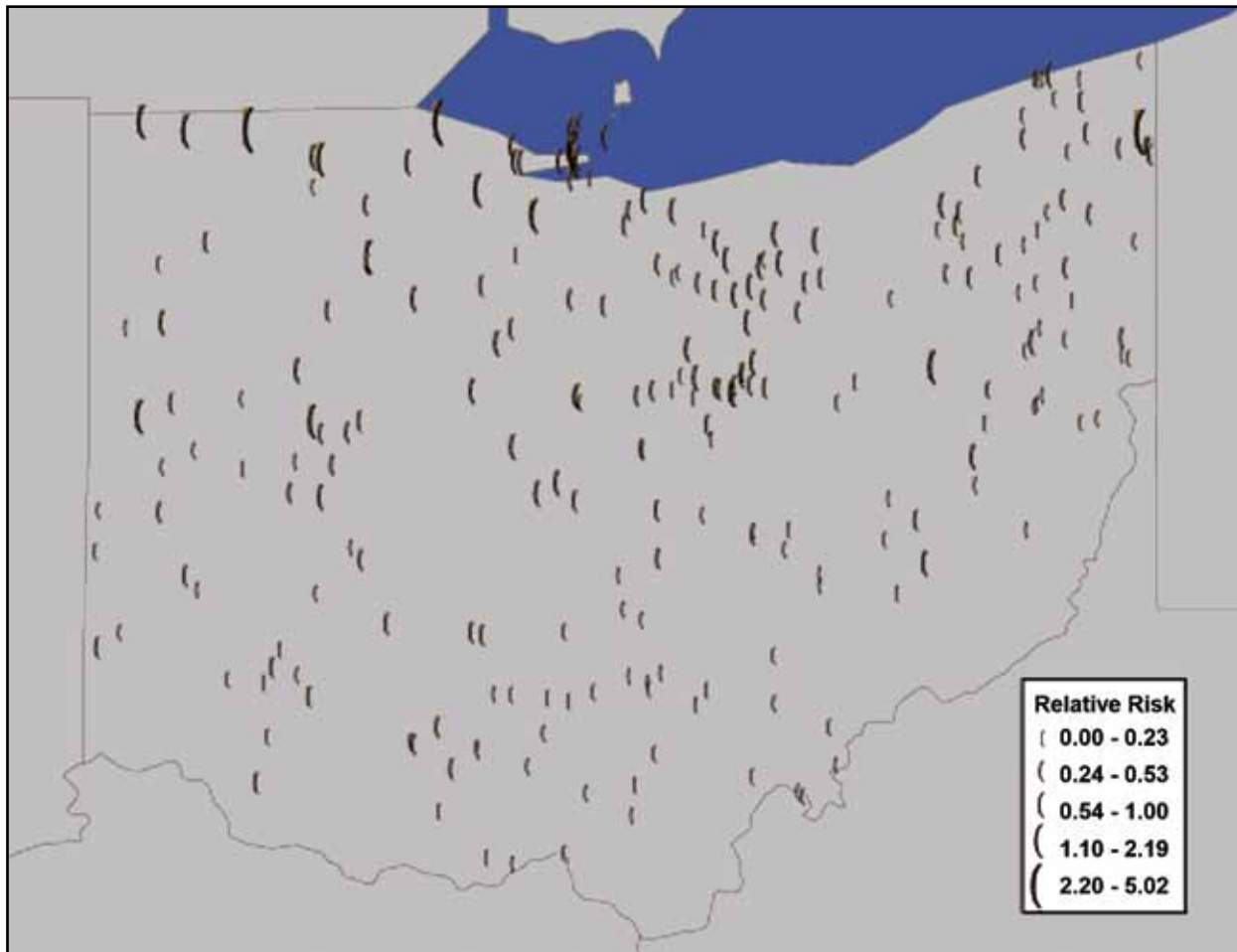


Figure 9—Ohio campground scores with gravity model. Larger marks represent increasingly higher scores of relative potential for emerald ash borer (EAB) invasion owing to higher attractiveness or travel or both from the EAB-infested core area in southern Michigan.

is still in progress. We also have begun to incorporate an outlier seed generator, which depends partly on a random generator and partly on the weighting scheme of the insect-ride components.

Conclusions

The results on assessment of the ash resource, estimates of past spread of EAB, and preliminary efforts to create a model of spread leave us with the following conclusions:

- There is a great deal of ash resource in the Eastern United States, especially in the northern half of the region. For many States, ash makes up a sizeable portion of the total BA.
- As of spring 2006, the front border of the current EAB infestation is just now reaching the

areas with the largest amount of available ash, e.g., in northeast Ohio, northwest Pennsylvania, and western New York.

- Although much of the current expanding range of EAB in northwest Ohio and northeast Indiana is dominated by agriculture, our high-resolution analysis shows plenty of ash exists for EAB expansion in this zone in small woodlots, riparian woods, small wetlands, and miscellaneous parcels bordering the agricultural fields.
- The map of our estimate of the expansion of the front from 1998 to 2005 shows a fairly consistent pattern of roughly 20 km/yr. This rate

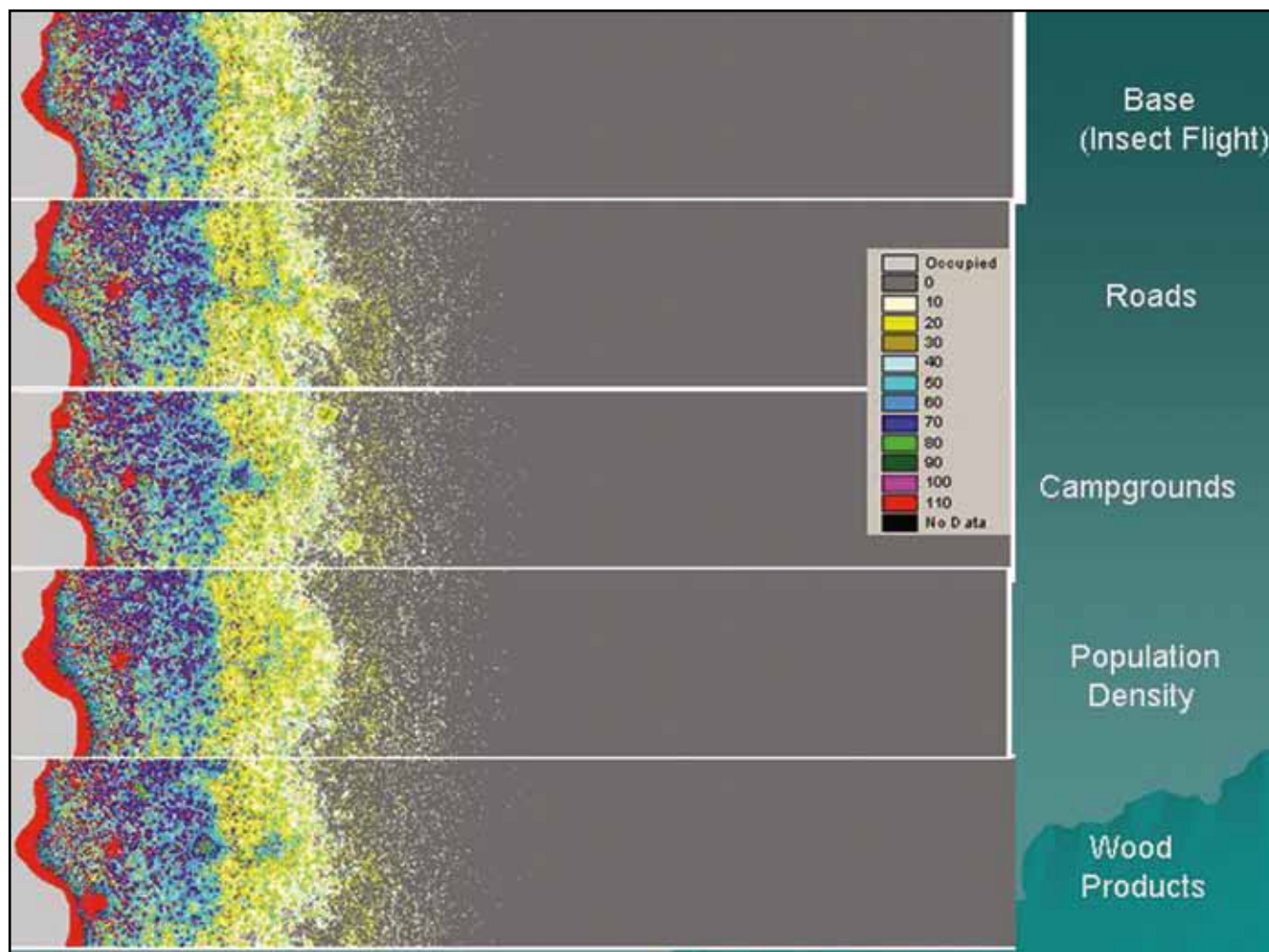


Figure 10—Combined model outputs using single-factor weights. These maps show preliminary results on the Toledo (left side) to Columbus (right side) test strip of the emerald ash borer Shift base run, and then with single-factor multipliers for roads, campgrounds, population density, and wood products industries. Numbers correspond to relative colonization probabilities after 11 years of runs. It does not include random or directed outlier generation that may occur through the insect-ride or gravity modeling components.

of expansion would necessarily have to include both the biological dispersal capacity of the insect and some short-distance movement assisted by humans (e.g., on or in vehicles, plant material, wood material, etc.).

- The components of the insect-ride model (roads, campgrounds, wood products industries, population density, and urban centers) have been acquired and processed to create a weighting scheme based on various factors, including buffer distances and number of people involved in the endeavor. When combined, every 270-m

pixel in the study area has been scored for its likelihood of enhancing EAB spread.

- The gravity model yielded a relative scoring of potential EAB invasion among campgrounds based on traffic from the core EAB zone and attractiveness of the campgrounds.
- Preliminary test results of movement of the front from the EAB shift model shows the probability of colonization diminishes quickly away from the front, and that the insect-ride components modify those results through the multiplier effects.

- We hope to use these data along with GIS and modeling tools to better understand the potential rate of spread, which could inform management decisions that will hopefully slow the spread of this destructive pest.

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Risk Analysis and Guidelines for Harvest Activities in Wisconsin Oak Timberlands to Minimize Oak Wilt Threat

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Abstract

Oaks (*Quercus* spp.) are an important species group in the forests of Wisconsin. The State's timberland typed as oak-hickory forest was estimated at 2.9 million acres in 1996. Growing stock volume for red oak was estimated at 2.4 billion cubic feet, whereas select white oak volume was estimated to be 927 million cubic feet. Oak wilt, the oak disease of greatest concern in Wisconsin, is widespread in the lower two-thirds of the State. Harvest activities in oak stands may result in introduction of the disease agent, *Ceratocystis fagacearum* (Bretz), into the stand or promote intensification of the disease within the stands or both. A risk-rating system based on scientific- and experience-based knowledge was used to develop a statewide system for oak wilt risk analysis. Guidelines for timber harvest activities in oak stands were then developed based on results of the risk analysis. The analysis and recommendations have been published (<http://www.dnr.wi.gov/forestry/fh/oakWilt/guidelines.asp> [Date accessed: July 8, 2010]) in three different formats. The formats include a pdf version of decision-trees with accompanying tables, a simple spreadsheet application allowing the user to obtain specific guidelines based on his/her response to five questions about the stand and timing under consideration, and an interactive online format derived from the spreadsheet version. The query page of the interactive formats is linked to a concealed table containing the risk analysis and recommendation matrix. The tool provides consistent, statewide guidelines for harvest activities that will, when applied, minimize spread and reduce the biological and economic impacts of oak wilt to Wisconsin's oak timberlands. The rule-based, expert-driven system approach used to develop these guidelines could be

used to assess risk and develop large-scale management guidelines for other established forest pathogens.

Keywords: *Ceratocystis fagacearum*, oak wilt, *Quercus* spp., risk analysis, timber harvest guidelines.

Introduction

Oak Forests of Wisconsin

Oaks (*Quercus* spp.) are a dominant component of the extensive oak-hickory forests of the Central U.S.A. (Leopold and others 1998). In Wisconsin, timberland typed as oak-hickory forest was estimated at 2.9 million acres in 1996 (Schmidt 1997). Growing stock volume for red oak (section Lobatae) was estimated at 2.4 billion cubic feet, whereas white oak (section *Quercus*) volume was estimated to be 927 million cubic feet (Schmidt 1997).

Oak Wilt – Primary Disease of Concern

Oak wilt, the oak disease of greatest concern in Wisconsin, occurs in 51 of the State's 70 counties (http://www.na.fs.fed.us/fhp/ow/maps/ow_dist_fs.shtm [Date accessed: July 8, 2010]). Thousands of oaks in woodland and urban settings succumb to the disease every year. The causal fungus, *Ceratocystis fagacearum* (Bretz), is spread from diseased to healthy oaks belowground through functional root grafts or aboveground by insect vectors (Tainter and Baker 1996). Species of the sap beetle family (Coleoptera: Nitidulidae) are considered the primary vectors in Wisconsin. New disease centers are established when *C. fagacearum*-contaminated beetles visit fresh xylem-penetrating wounds (e.g., axe blazes, logging wounds, branch-pruning wounds) on healthy oaks and successfully inoculate them with propagules of the fungus (Gibbs et al. 1980, Juzwik and others 2004). Stump surfaces created by tree felling and wounds to branches, stems, and roots by heavy equipment or adjacent falling trees are avenues for infection during timber stand improvement or harvesting activities. In a timber sale unit near Waube Lake, Wisconsin, many new infection centers occurred over a large area following a May 2001 timber harvest (M. Mielke 2006. Plant pathologist, Northeastern Area State and Private Forestry, USDA Forest Service).

Felling of diseased oaks adjacent to healthy oaks can lead to intensification of the disease within stands if root connections exist. Slow movement of the pathogen through grafted roots of healthy trees felled within 50 feet of a diseased tree explained the sporadic appearance of oak wilt in subsequent years at the edge of clear-felled areas (Yount 1955).

Need for Statewide Guidelines

The Wisconsin Department of Natural Resources (DNR) identified the need to develop consistent, statewide guidelines for timing harvest activities in oak timberland in order to minimize potential for oak wilt introduction or spread or both in existing and future stands where oak regeneration is the management objective. A committee of government, industrial, and consulting foresters was formed to develop such guidelines. Both scientific and experience-based knowledge of the oak wilt host – pathogen system were the basis of the guidelines. The approach used to (1) analyze the risk and the potential for introduction and spread of oak wilt in stands targeted for harvest, and (2) develop guidelines for timing harvest are described in this paper.

Approach

Risk Assessment

Risk refers to the chance of injury or loss defined as a measure of the probability and severity of an adverse effect to health, property, the environment, or other things of value (North American Forest Commission 2004). Our risk analysis includes (1) the assessment of risk posed by the oak wilt pathogen to oak timberland scheduled for harvest and regeneration to oak, and (2) recommendations for minimizing frequency of pathogen introduction to and spread within such stands. A rule-based, expert-driven model, such as that used for pest risk assessment in the Exotic Forest Pests (ExFor) system (North American Forest Commission 2004), was adapted for this analysis. This approach falls under the umbrella term of multicriteria decision analysis, which seeks to take multiple criteria into account when groups explore decisions that matter, e.g., natural resource management decisions (Mendoza and Martin 2006). Two criteria were evaluated within the risk assessment process.

Criterion 1: Risk of *Ceratocystis fagacearum* introduction to the stand [between-stand spread] or for initiation of new centers within the stand [within-stand spread] by insect vectors—

Statements were developed for this criterion that considered two factors: (1) time of year during which harvest activities would occur (resulting in fresh wounds suitable for infection), and (2) proximity of existing oak wilt centers in other locations to the stand in which harvest activities would occur. A risk rating, ranging from very low to very high, was then assigned to each of the possible combinations of time and proximity. The risk values were determined through a group consensus process after review of pertinent scientific literature and of each individual's experience working with the disease.

Criterion 2: Risk of *C. fagacearum* belowground spread within an oak stand following pathogen establishment—

Statements for this criterion included three factors (i.e., stand conditions): (1) density of oaks, (2) general topographic relief, and (3) general soil type in the stand to be harvested. Each of these factors is known, either through scientific studies or experiential knowledge or both, to influence the frequency and the distance over which intraspecific root grafting occurs. Two or more levels were selected for each factor. Basal area (square feet per acre) levels for describing red oak species composition and density were less than 15, between 15 and 35, and greater than 35. The general levels for topographic relief were (a) flat to rolling terrain, and (b) steep hills with deep valleys terrain. Soil type was divided into light textured (sandy, loamy sand, and sandy loam) and heavier textured (all other types depicted in classic soil texture triangle). A risk rating, ranging from very low to very high was then assigned to each of all possible combinations of statements by factor. The risk values were determined by a group consensus process.

Overall risk: combined risk rating for the two criteria—

The ratings for each criterion were then used to generate the overall risk of oak wilt's threat to the stand of interest following a timber harvesting event. The overall rating, ranging from very low to very high, was assigned to each stand scenario based on the combination of introduction and

Table 1—Combined risk ratings for Criterion 1 factors - proximity of oak wilt centers to stand under consideration and proposed timing of harvest activities

Proximity of oak wilt centers to stand of interest		Proposed timing of harvest activities	Risk rating^a
Within county?^b	Within stand?		
No	No	Spring to early summer	M
No	No	Summer to early fall	L
No	No	Mid-fall through winter	VL
Yes	No	Spring to early summer	VH
Yes	No	Summer to early fall	M
Yes	No	Mid-fall through winter	VL
Yes	Yes	Spring to early summer	AP
Yes	Yes	Summer to early fall	AP
Yes	Yes	Mid-fall through winter	AP

^a Explanation of ratings: VH (very high), M (moderate), L (low), VL (very low), AP (oak wilt already present in the stand).

^b Includes stands occurring in oak-wilt-free counties, but within 6 miles of oak-wilt-affected counties.

root graft spread factors. As before, the risk values were determined through a group consensus process.

Timber Harvesting Guidelines

Timber harvest guidelines for minimizing the initiation of new infection centers and subsequent tree loss from spread within stands were developed based on results of the risk assessment. The risk rating for each stand condition scenario was considered and harvest recommendations determined through a group consensus process.

Display of Risk Analysis Results and Guidelines

Three methods were used to display results of the risk analysis. For the first method, graphical decision-trees were constructed, and associated tables were developed for harvest guidelines for three proximity levels (i.e., no oak wilt in county, oak wilt in county but not in stand, and oak wilt in the stand [not shown]). This output was used in the development of the electronic displays. Initially, the risk analysis and associated harvest guidelines were combined in a simple electronic spreadsheet. The spreadsheet features a front query page that allows the user to obtain risk ratings and recommendations for specific stand scenarios. The query page is linked to a concealed table containing the risk analysis and recommendation matrix. Later, an interactive, Web version of the spreadsheet product was developed for online use.

Results

Risk Analysis Results with Scientific Knowledge Basis

The combined risk ratings for Criterion 1 (“Criterion 1: Risk of *Ceratocystis fagacearum* Introduction to the Stand [Between-Stand Spread] or for Initiation of New Centers within the Stand [Within-Stand Spread] by Insect Vectors”) are shown in Table 1. The risk of overland pathogen transmission by sap beetles was considered to increase as proximity to an existing oak wilt center decreased. The existing centers would be the source from which inoculum-laden beetles would originate, assuming oak wilt mats were formed on recently wilted red oaks in that originating center. Menges and Loucks (1984) and Shelstad and others (1991) found higher efficiencies of vector spread over short distances (e.g., ≤ 300 m); longer distance spread occurs very infrequently and on a random basis. Although the number of new centers occurring at greater distances is small, over time they can have a significant influence on distribution of oak wilt within the total forest area (Shelstad and others 1991). Timber harvest activities would result in wounding of residual oaks in shelter wood cut situations or create stump surfaces of removed healthy oaks or both. Such xylem-exposing cuts are attractive to dispersing sap beetles. The risk of pathogen transmission to such wounds by certain sap beetle species is high during the spring months, low from

Table 2—Combined risk ratings for Criterion 2 factors – density of oaks, topographic relief, and general soil type

Density of oaks ^a (ft ² /acre)	Topographic relief	Soil category ^b (texture)	Risk rating ^c
< 15	Flat – rolling	Light	L
15 – 35	Flat – rolling	Light	H
> 35	Flat – rolling	Light	VH
< 15	Flat – rolling	Heavy	L
15 – 35	Flat – rolling	Heavy	M
> 35	Flat – rolling	Heavy	H
< 15	Hills & valleys	Light	L
15 – 35	Hills & valleys	Light	H
> 35	Hills & valleys	Light	H
< 15	Hills & valleys	Heavy	VL
15 – 35	Hills & valleys	Heavy	M
> 35	Hills & valleys	Heavy	M

^a Density of oaks measured as basal area.^b Light texture includes sandy, loamy sand, sandy loam, sandy clay loam, and loam; Heavy texture includes sandy clay, clay, clay loam, silt, silt loam, silty clay loam, and clay loam. Based on classic soil texture triangle.^c Explanation of ratings: VH (very high), H (high), M (moderate), L (low), and VL (very low).**Table 3—Stand condition scenarios for which overall risk ratings were high (H) to very high (VH), where oak wilt is not yet present in the stand of interest but occurs elsewhere in the same county or in a second county that is less than 6 miles from the first**

Timing for harvest	Oak density ^a (ft ² /acre)	Topographic relief	Soil category ^b (texture)	Overall risk rating
Spring to early summer	>35	Flat - rolling	Light	VH
Spring to early summer	> 35	Hills & valleys	Light	H
Spring to early summer	> 35	Flat - rolling	Heavy	H
Spring to early summer	> 35	Hills & valleys	Heavy	H
Spring to early summer	15 – 35	Flat - rolling	Light	H
Spring to early summer	15 – 35	Hills & valleys	Light	H
Spring to early summer	15 – 35	Flat - rolling	Heavy	H
Spring to early summer	15 – 35	Hills & valleys	Heavy	H

^a Density of oaks measured as basal area.^b Light texture includes sandy, loamy sand, sandy loam, sandy clay loam and loam; heavy texture includes sandy clay, clay, clay loam, silt, silt loam, silty clay loam, and clay loam. Based on classic soil texture triangle.

midsummer to early fall, and none during the late fall and winter (Ambourn and others 2005, French and Juzwik 1999, Juzwik and others 2006).

The combined risk ratings for Criterion 2 (“Criterion 2: Risk of *C. fagacearum* Belowground Spread within an Oak Stand Following Pathogen Establishment”) are shown in Table 2. Frequencies of root graft spread increase with

increasingly lighter textured soils, e.g., from silt loam to sands (Menges 1978). Furthermore, frequency of root graft transmission is highest for stands with > 60 percent red oak density (Menges and Loucks 1984). Lastly, oak wilt is very common in areas of low topographic relief in portions of Iowa, Michigan, Minnesota, and Wisconsin (e.g., Albers 2001, Menges and Loucks 1984). In areas with obvious

Table 4—Summary of management guidelines for timing of timber harvest activities based on oak wilt risk analysis results

Stand proximity to oak wilt centers	Guidelines by timing of timber harvest activities		
	Spring – early summer	Summer – early fall	Fall – winter
Not in county or within 6 miles of county with oak wilt and not in stand	No restrictions April 1 - July 15 (south) ^a and April 15 - July 15 (north). (12) ^b	No restrictions July 16 - September 30. (12)	No restrictions October 1 - March 31 (south) and April 14 (north). (12)
In county or within 6 miles of a county with oak wilt, but not in stand	May cut between April 1 - July 15 (south) and April 15 - July 15 (north) IF new stumps are treated. (4) Do not harvest or conduct activities that may wound oaks April 1 - July 15 (south) and April 15 - July 15 (north). (8)	No restrictions July 16 - September 30. (12) 15	No restrictions October 1 - March 31 (south) and April 14 (north). (12)
In county & in stand	First consider owner interest in oak wilt control; otherwise, no restrictions April 1 - July 15 (south) and April 15 - July 15 (north) if new stumps are treated. (4) First consider owner interest in oak wilt control; otherwise, do not harvest or conduct activities that may wound oaks April 1 - July 15 (south) and April 15 - July 15 (north). (8)	First consider owner interest in oak wilt control; otherwise, no restrictions July 16 - September 30. (12)	First consider owner interest in oak wilt control; otherwise, no restrictions October 1 - March 31 (south) and April 14 (north). (12)

^a South denotes stands located south of the tension zone in Wisconsin; north denotes stands located north of the tension zone. Wisconsin's tension zone is a border between northern and southern floristic provinces (Curtis 1959). Data on average monthly temperatures and flight of oak wilt insect vectors support use of different risk dates for these portions of the State.

^b Twelve scenarios are possible for each timing-proximity combination. Number of scenarios to which the particular guideline applies is stated in parentheses.

topographic relief, oak wilt is most common on upper slopes and ridge tops (Anderson and Anderson 1963, Bowen and Merrill 1982, Cones and True 1967).

Each of the 108 stand condition scenarios described by combinations of the five factors was assessed for overall risk of oak wilt occurrence based on the individual criterion

ratings. Overall risk ratings were high to very high for eight stand-condition scenarios where oak wilt was not known to be present in the stand (Table 3). Overall risk rating of very low, however, was often determined by a late fall–winter timing for the harvest.

To obtain oak wilt risk rating and associated guidelines, answer the following questions:	Codes	Enter your responses in the grey boxes
1) Is oak wilt present? Choose one of the responses below:		
Oak wilt not present in the county.	NN	
Oak wilt present in the county, but not known to be in the stand.	YN	
Oak wilt present in the county and the stand	YY	
2) What time of the year do you propose cutting? Choose one of the responses below: April 1 to July 15 - south of tension zone. April 15 to July 15 - north of tension zone. July 16 to Sept. 30 statewide. Oct. 1 to March 31 if south of tension zone. Oct. 1 to April 14 if north of tension zone. Note: If uncertain about tension zone location, please see Figure 2 (to right).	SPS SPS SEF FWN FWN	
3) What is the basal area (BA) of red oak in the stand? Choose one of the responses below: 15 ft ² /acre or less 15 to 35 ft ² /acre greater than 35 ft ² /acre	L M H	
4) What is the general topography of the stand? Choose one of the responses below: Flat to rolling land or 0 - 12% slope. Hilly with valleys or > 12% slope.	FR HV	
5) What is the general soil texture of the stand? Choose one of the responses below: Light textured (e.g., sandy, loamy sand, sandy loam, sandy clay loam, and loam) or Heavy textured (sandy clay, clay, clay loam, silt, silt loam, silty clay loam, and clay loam)	L H	
Note: The light and heavy textured soils were described by using the soil triangle, please see Figure 3 (to right).		

Figure 1. Oak wilt present in shaded counties (2006 distribution).

Figure 2 (A & B).
Tension zone dividing the two climate regions in Wisconsin.

A. Tension Zone

B. Climate region by county

Figure 3.
Soil texture triangle of various combinations of sand, silt and clay.

Figure 1—Query section of user page of spreadsheet application for oak wilt risk rating and associated management guidelines. Shaded boxes are where user enters information.

Timber Harvest Guidelines

Preventive measures developed for minimizing initiation of new oak wilt infection centers and the potential for future tree losses owing to oak wilt in regenerated stands were described in three brief statements: (1) Do not harvest or conduct activities that may wound oaks, (2) Harvesting may be conducted if stumps are treated, and (3) No restrictions. The first two measures largely apply to stand harvest activities being considered for spring and early summer. A summary of the stand/harvesting scenarios associated with each of the preventive recommendations when categorized by timing and proximity factors is presented in Table 4. For timber stands where oak wilt centers already exist

and harvest of and regeneration to oak are planned, the guidelines include some further considerations. Specifically, foresters are advised to first consider the landowner’s tolerance for future tree losses to oak wilt in the regenerated stands. Disease control actions, such as stump extraction or soil trenching, could be valuable for greatly reducing the carryover of oak wilt into the future stand.

Risk Analysis and Guidelines Tool Formats

Three different formats of the risk analysis results and the harvest guidelines were developed for end users. A hard-copy, decision-tree format (filename: oakwiltguide031507.

Figure 2—Response section of user page in spreadsheet application
Response section of user page of spreadsheet application for oak wilt risk rating and associated management guidelines.
Shaded boxes are where varying responses are displayed.

RISK RATING RESULTS AND ASSOCIATED GUIDELINES		
Risk Rating Results: (rating scale to right)		
Risk of introducing oak wilt to the stand.		
Risk of spread and impact in the future oak stand.		
Combined risk for oak wilt introduction and impact in the future oak stand.		
Harvest Guidelines (for the conditions of the stand):		
Information on other time periods:		
Further Considerations:		

Risk Rating Scale:

VH - Risk is very high
H - Risk is high
M - Risk is moderate
L - Risk is low
VL - Risk is very low

AP - Oak wilt is already present in stand
CON - Spread and impact of oak wilt will continue
INCR - Overall impact of oak wilt will increase

Figure 2—Response section of user page of spreadsheet application for oak wilt risk rating and associated management guidelines. Shaded boxes are where varying responses are displayed.

pdf) with accompanying tables is available from the Wisconsin DNR Web site (<http://dnr.wi.gov/forestry/fh/oakWilt/guidelines.asp>). The electronic spreadsheet version of the results (filename: oakwiltguide031507.xls) is also available from the same site. The interactive, online format was adapted from the spreadsheet version. The front user page of the spreadsheet-based tool (Figure 1) and of the online tool requires the user to input conditions of the stand being considered for harvest. The questions asked of the user include (1) Is oak wilt present? (2) What time of the year do you propose cutting? (3) What is the basal area of red oak in the stand? (4) What is the general topography of the stand? and (5) What is the general soil texture of the stand? The user selects a response from the multiple-choice answers offered for each question. The spreadsheet application then selects and displays the appropriate ratings and

recommendations for the conditions described by the user (Figure 2), as does the online version.

Discussion

The rule-based, expert-driven model used in an exotic pest risk analysis context (North American Forestry Commission 2004) was adapted for use in assessing risk of oak wilt introduction to and potential for subsequent spread within oak timberland based on spatial, temporal, and site factors. Such a system may be useful for analyzing spread and impact risks in the management of other significant forest diseases. The Wisconsin DNR plans to use the same approach to analyze risk and develop guidelines for reducing spread of *Heterobasidion annosum* in pine forests of the State. The model was also considered for use in modifying existing guidelines for managing oak wilt in urban and

periurban forests of Wisconsin. Existing guidelines were, however, considered sufficiently robust and did not warrant such an effort.

The success of our approach relied on a collaborative planning and decisionmaking environment. The participatory method sought and obtained the involvement of multiple experts, stakeholders, and end users. The committee responsible for developing the criteria, conducting the risk analysis, formulating guidelines appropriate to risk ratings, and reviewing the prerelease product met for 4 hours on each of 4 days. Solicitation of stakeholder and user response and suggestions to the proposed system occurred over a 7-month time period through presentations and subsequent comment sessions held at numerous meetings, e.g., the Wisconsin Chapter of the Society of American Foresters' annual meeting and the Wisconsin Woodland Owners' Association annual meeting.

Several research questions were raised during the exercise of developing criteria, conducting the risk analyses, and developing guidelines appropriate for the assigned risks. The need for observed frequency or estimated probability for successful overland transmission of the oak wilt fungus between mid-July and early October is being addressed in a 3-year study initiated in summer 2006. Questions were also raised about the ultimate quantitative impact of oak wilt introduced during shelter wood preparatory cuts or clearcutting on future oak stocking in stands regenerated on dry and dry-mesic sites. On the basis of results of a West Virginia study (Tyron and others 1983), we hypothesize that the impact would be low in areas where regeneration is mostly of seedling origin. However, where coppice or stump-sprout regeneration predominates, the ultimate impact of oak wilt on stand stocking would likely be higher. A long-term study is needed to address these questions. New knowledge or previously overlooked scientific knowledge pertinent to our risk assessment system will be considered in future revisions of the product.

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Modeling Current Climate Conditions for Forest Pest Risk Assessment

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Abstract

Current information on broad-scale climatic conditions is essential for assessing potential distribution of forest pests. At present, sophisticated spatial interpolation approaches such as the Parameter-elevation Regressions on Independent Slopes Model (PRISM) are used to create high-resolution climatic data sets. Unfortunately, these data sets are based on 30-year normals and rarely incorporate up-to-date data. Furthermore, because they are constructed on a monthly rather than a daily time step, they do not directly measure simultaneous occurrence of multiple climatic conditions (e.g., days in the past year with appropriate temperature and adequate precipitation). Yet, the actual number of days—especially consecutive days—where multiple conditions are met could be significant for pest dispersal or establishment. For the sudden oak death pathogen (*Phytophthora ramorum*), we used National Oceanic and Atmospheric Administration daily weather station data to create current, national-scale grids depicting co-occurrence of multiple climatic conditions.

For each station, we constructed two count-based variables: the total number of days and the greatest number of consecutive days in a year where the station met several conditions (temperature, rain/fog, relative humidity). We then employed gradient plus inverse distance squared (GIDS) interpolation to generate grids (4-km² resolution) of these variables for 5 years (2000–2004). The GIDS technique weights standard inverse distance squared interpolation using coefficients based on geographic location (x, y) and a spatial covariate such as elevation. Using these variables, we determined the GIDS coefficients for each output grid cell via Poisson regression on the 30 closest

stations. We also performed model selection to ensure only significant variables contributed to the GIDS coefficients.

We compared the GIDS approach to cokriging and detrended kriging using cross-validation and found similar accuracies among all three interpolation methods. We also compared the output grids to maps assembled from the PRISM data depicting the probability all conditions were met in a given year. As expected, we found differences in areas highlighted as suitable for *P. ramorum* establishment by the two methods. We suggest that using current weather data and calculating the variable of interest directly will provide more practical information for mapping forest pest risk.

Keywords: Climate, forest pests, GIDS, *Phytophthora ramorum*, risk, spatial interpolation.

Introduction

Forest pest risk assessments detail the nature and severity of threats posed to particular forest species and ecosystems by insects, pathogens, or other organisms (Andersen and others 2004a). With respect to nonindigenous forest pests, risk can be categorized or quantified based on a combination of factors: the potential for the pest to become established, the potential for it to spread following introduction, the potential to cause economic damage, or the potential to cause environmental harm (NAFC 2004). A commonly desired product of such assessments is a map depicting the threat posed by introduction or establishment of a forest pest throughout a geographic area of interest (Andersen and others 2004a). These maps can facilitate early detection and response procedures, providing a template for the design of regulatory programs and detection surveys. If a pest has already been established in one part of the geographic area of interest, threat assessment maps are used to help set control priorities for other geographic areas that are at high risk of invasion (Andersen and others 2004b).

Importance and Availability of Climate Information

Forest pest risk maps are typically assembled by combining spatial data from three principal subject areas: host species

distribution, pathways of pest movement, and key environmental factors (Bartell and Nair 2004). Climatic attributes such as temperature and moisture strongly shape pest behavior, affecting survival, reproductive rate, and in many cases, the ability to spread at a continental scale. Thus, climatic data provide an important coarse filter for forest pest risk analyses. Regularly gridded climate maps covering the entire geographic area of interest are typically required for analytical purposes. Such maps may be constructed by spatial interpolation of weather station data. These data are readily available for much of the United States, dating back several decades, from the National Oceanic and Atmospheric Administration (NOAA) National Climatic Data Center (NCDC).

Spatial Interpolation of Climatic Variables—

A wide array of spatial interpolation algorithms (e.g., geostatistical, regression, spline, inverse distance weighting) have been used to construct broad spatial-scale climatic data sets from weather station data (Daly 2006, Mardikis and others 2005, Nalder and Wein 1998, Price and others 2000, Xia and others 2000). Most currently accepted methods acknowledge that terrain is a significant factor governing climate at all but the broadest scales, and they use elevation measurements to represent terrain and adjust climatic variable values accordingly (Daly 2006). One well-received interpolation approach is the Parameter-elevation Regressions on Independent Slopes Model (PRISM). Initially developed to generate precipitation maps for the Pacific Northwest (Daly and others 1994), the approach has since been applied to create maps of temperature, relative humidity, snowfall, growing-degree days, and many other variables (Daly and others 2000). In particular, the PRISM approach was applied to generate most of the maps in the recent version of the Climate Atlas of the United States (Plantico and others 2002), as well as similar products for Canada and China (Daly and others 2000). The PRISM approach is a knowledge-based system integrating a local climate-elevation regression with other algorithmic components: station weighting, topographic facets, coastal proximity, and a two-layer atmosphere (Daly and others 2002). When initially tested on precipitation in the Pacific Northwest, the PRISM approach outperformed

other interpolation methods in comparative analyses (Daly and others 1994).

Limitations of Existing Interpolated Climatic Data Sets—

There are several limitations of PRISM-derived or similar data sets with respect to their use for forest pest risk maps. First, most national-scale climatic data sets are calculated as normals, meaning an average of the variable of interest across a window of time, typically a 30-year period. For example, most data sets in the recent version of the Climate Atlas of the United States are based on inputs from 1961 through 1990 (Plantico and others 2002). Current weather data are not incorporated into the maps, so any pest risk map constructed from them will not include current events—and the accompanying variability—that may be relevant to an assessment of immediate risk.

Second, there are related issues of cost and data format. The Climate Atlas contains polygonal maps for a large number of potentially relevant climatic normals but does not include the regularly gridded data from which the maps are derived. These polygonal maps have limited attribute resolution, with the range of the original gridded data typically compressed into nine or fewer classes. Monthly gridded maps of a few variables—precipitation amount, mean minimum temperature, mean maximum temperature, and mean dewpoint—are available for public download from the PRISM group at Oregon State University (<http://www.ocs.orst.edu/prism/>). [Date accessed unknown]. Notably, these maps are fairly current (finalized maps are available from 1997 through mid-2006), and the database is regularly updated, but it does not include many climatic variables that might be of interest for forest pest risk assessment (e.g., relative humidity, number of days above freezing, or number of days with measurable precipitation). Regularly gridded data of these and other (30-year normal) variables, derived using the PRISM method, are available, but at substantial cost (from the Climate Source: <http://www.climatesource.com/>). [Date accessed unknown].

Third, most available climatic spatial data sets, whether derived using PRISM or other methods, are monthly or annual summaries depicting mean or extreme values over the time period. For some forest pests, the short-term,

even daily status of multiple weather conditions may be relevant to the pest's growth, persistence, or invasiveness. Fungal pathogens are particularly affected by the interaction of temperature and moisture availability. For example, the pathogen that causes late blight of potato (*Phytophthora infestans*) develops best at cool temperatures during extended periods of wet weather, as do many other *Phytophthora* species (Davidson and others 2002, Harvell and others 2002, Marshall-Farrar and others 1998). The interaction of climatic variables can also be important for some insect pests (Harrington and others 2001, Peacock and others 2006). Nevertheless, although there has been some effort to create maps of daily precipitation and temperature at a broad scale (Hunter and Meentemeyer 2005), there has been little attention paid to the co-occurrence of multiple weather conditions favorable to pest persistence and spread. Daily weather data allow the counting of how often, and for how long, variables meet certain threshold values. Creation of broad-scale maps from data derived in this manner may require a different spatial interpolation approach than that used for continuously distributed variables (van de Kastele and others 2005).

Objectives

Given the limitations of existing climatic data sets, we explored the use of NCDC daily weather station data for the United States as an alternate source for maps relevant to forest pest risk assessments. We had three basic objectives: (1) spatially interpolate annual counts of the number of days with co-occurrence of multiple climatic variables relevant to the growth and spread of a specific forest pest—the pathogen that causes sudden oak death (*P. ramorum*); (2) identify a spatial interpolation method appropriate for count-based data and compare it to some common geostatistical approaches; and (3) assess the utility of the derived maps for depicting risk.

Case Study Species: *Phytophthora ramorum*

Phytophthora ramorum was first recognized in the United States in 1994 and was likely introduced via international trade of commercial plants (Ivors and others 2006). Since its

introduction, the pathogen has infected western live and red oaks in coastal forests of California and Oregon, sometimes causing mortality greater than 40 percent (Garbelotto and others 2001, 2003). In addition, *P. ramorum* infects dozens of commercial shrub host species that can yield large numbers of aurally dispersed spores (Davidson and Shaw 2003, Davidson and others 2002, Tooley and others 2004). Many of these shrubs (e.g., rhododendrons, azaleas, camellias) are sold as nursery stock (Garbelotto and others 2001, Tooley and others 2004). In the past few years, wholesale nurseries on the west coast have unknowingly shipped infected plants to retail and wholesale outlets in roughly 40 States (Stokstad 2004), although surveys have not detected the pathogen in natural forests outside California and Oregon.

A large portion of the Eastern United States is considered at high risk for establishment of *P. ramorum* if it is introduced into forested areas. Much of the concern has to do with climatic conditions believed to be favorable for the pathogen. Growth, sporulation, and infection are all affected by moisture and temperature. Optimal temperatures for *P. ramorum* growth, based on laboratory analysis, appear to be between 64.4 °F and 71.6 °F (Werres and others 2001), but some growth occurs across a wider temperature range (up to at least 80 °F). Peak sporangia formation appears to occur at 59 to 68 °F (Davidson and others 2005). Persistent moisture on foliage is considered critical to spread. Laboratory inoculation trials on California bay laurel (*Umbellularia californica* (Hook. & Arn.) Nutt.), a major source of *P. ramorum* spores in California, suggest 9 to 12 hours of free moisture on leaf surfaces under appropriate temperatures are necessary for significant leaf infection (Garbelotto and others 2003). Further studies suggest that at least 24 to 48 hours of generally wet conditions are necessary for sporulation, with infection requiring additional time (Davidson and Shaw 2003, Davidson and others 2002, Rizzo and Garbelotto 2003). Fog and high relative humidity may be important for spread of aerial *Phytophthora* species within forest stands (Werres 2003), as high air moisture can keep leaf surfaces wet and enable spore production. Nevertheless, despite regular summer fog in California, *P. ramorum* sporulation and infection seem to be restricted to the winter-spring rainy season (Rizzo and others 2005).

Table 1—Number of NCDC weather stations used in interpolations

Year	Number of stations
2000	4,310
2001	4,258
2002	4,302
2003	4,144
2004	3,926

NCDC = National Climatic Data Center.

Isolated rains during otherwise dry summer months do not appear to facilitate spore production or dispersal (Davidson and others 2002). Ultimately, it is unknown how the pathogen's behavior on the west coast will translate to the Eastern United States, where warm season and cool season precipitation are similar (Akin 1991).

Methods

We downloaded 5 years (2000 to 2004) of daily surface data from the NCDC online climate data clearinghouse (<http://cdo.ncdc.noaa.gov/CDO/dataproduct>. [Date accessed unknown]). The downloaded data included dozens of climate variables recorded for more than 19,000 stations nationwide. We processed the data to extract four variables: total precipitation, minimum and maximum temperature, and relative humidity. For each station, we tallied (1) the total number of days and (2) the longest number of consecutive days in a given year that met the following conditions: maximum temperature greater than 60 °F, minimum temperature less than 80 °F, and at least a trace amount of precipitation or relative humidity of greater than 85 percent. These threshold values were selected to reflect current knowledge about the climatic conditions favorable for *P. ramorum* survival and spread.

We recorded the latitude, longitude, and elevation values for each weather station from an associated data set. We dropped any stations that fell outside the conterminous United States and any stations with more than 30 days of missing data for any variable in a given year. This filtering process reduced the number of usable stations (Table 1), but still yielded consistent national coverage. For stations missing 1 to 30 days of data, we normalized the total-day and consecutive-day count values by dividing them by the

proportion of days in the year for which data were available and then rounding to the closest integer.

Gradient Plus Inverse Distance Squared Interpolation

We interpolated gridded maps of the conterminous United States for both the total-day and consecutive-day variables using a gradient plus inverse distance squared (GIDS) approach. This statistical method was first proposed as a way to interpolate climatic data on a broad spatial scale as input for plant growth models (Nalder and Wein 1998). The GIDS technique combines multiple linear regression with inverse distance weighting interpolation, and like other recently developed interpolation techniques, incorporates elevation as a covariate. For a given unmeasured location k and climatic variable Z , an ordinary least squares regression is performed using the N closest neighboring locations to calculate coefficients (C_x , C_y , and C_e) representing x , y , and elevation gradients: $Z = a + C_x X + C_y Y + C_e E + \varepsilon$, where a is the intercept and ε is error. Then, the basic GIDS formula is

$$Z_k = \frac{\sum_{i=1}^N \frac{Z_i + C_x(X_k - X_i) + C_y(Y_k - Y_i) + C_e(E_k - E_i)}{d_i^2}}{\sum_{i=1}^N \frac{1}{d_i^2}}$$

where Z_k = the predicted value at an unmeasured location k , Z_i = the measured value at location i , X = the x -coordinate for the specified location, Y = the y -coordinate, E_i = the elevation value, and d_i = the distance from measured location i to Z (Nalder and Wein 1998).

Nalder and Wein (1998) compared GIDS with several other methods for interpolating monthly normals of precipitation and temperature in the Canadian boreal forest region. The tested methods included inverse distance squared weighting, nearest neighbor interpolation, ordinary kriging, universal kriging, co-kriging, and detrended kriging. Based on cross-validation using a held-out subset of the data, the GIDS method resulted in the lowest mean absolute errors (MAE), which averaged 0.5 °C for temperature and 3.6 mm, or 11 percent, for monthly precipitation. Price and

others (2000) compared the GIDS method with thin-plate moving splines and noted that GIDS, as an inverse distance approach, may have greater occurrence of extreme errors. However, they also noted its transparency and ease of use.

Modification of GIDS for a Count-Based Variable—

The ordinary least squares regression implemented in the GIDS approach is intended for continuous, normally distributed variables. Because each of our variables of interest was a count, with large values being rare, we instead performed Poisson regression (Neter and others 1996). For each location of interest, we fitted a Poisson regression model, based on the 30 closest neighboring weather stations, using a maximum likelihood approach. We acknowledged that all three gradient variables (x, y, and elevation) could prove insignificant for a given prediction location and its closest measured neighbors. As a result, we evaluated a sequence of the full and all possible reduced models for statistical significance:

$$\log(Z) = a + C_x X + C_y Y + C_e E + \varepsilon,$$

$$\log(Z) = a + C_x X + C_e E + \varepsilon,$$

$$\log(Z) = a + C_y Y + C_e E + \varepsilon,$$

$$\log(Z) = a + C_x X + C_y Y + \varepsilon,$$

$$\log(Z) = a + C_x X + \varepsilon,$$

$$\log(Z) = a + C_y Y + \varepsilon,$$

$$\log(Z) = a + C_e E + \varepsilon.$$

For each prediction location, we tested all seven regression models using the 30 closest stations and identified those models in which all variables were significant. In cases where more than one of the models had all significant variables, we identified the one that yielded the smallest value for Akaike's Information Criterion (AIC). If the best-performing model was not the full Poisson regression model, then the coefficient(s) for any insignificant variable(s) were set to zero in the GIDS equation. If none of the tested models proved to have significant variables, then the GIDS interpolation reverted to inverse distance squared weighting (i.e., all variable coefficients were set to zero).

Interpolation Using GIDS—

We implemented the Poisson-based GIDS formulation in a script written for R statistical software (R Core Development Team 2006), which we then used to interpolate values

for cells covering the conterminous United States. We created a regular grid (with x, y, and elevation values) for the country by resampling an 8100-m² resolution digital elevation model (DEM) generated from U.S. Geological Survey data to 4-km² cells using a nearest neighbor method. Notably, this is the same spatial resolution used in most of the data sets that are publicly downloadable from the PRISM Group as well as the data sets available for purchase from the Climate Source (see "Limitations of Existing Interpolated Climatic Data Sets"). For each 4-km² cell, we determined the 30 closest NCDC weather stations using three-dimensional Euclidean distance measured from the cell's centroid. We rounded the GIDS-predicted value for each grid cell to the nearest integer.

Evaluation

For comparison to the GIDS-derived total-day and consecutive-day count maps, we created gridded maps for 2000 to 2004 using two spatial interpolation methods available through the ArcGIS Geostatistical Analyst extension (Johnston and others 2003). First, we performed cokriging on the count data using elevation as a covariate. Second, we performed detrended kriging, where we removed a second-order trend from the data and then performed ordinary kriging on the residuals. For both methods, we fit a spherical semivariogram model to the input data, calculating the model parameters (nugget, range, and sill) using a weighted least squares approach (Cressie 1993). As with the GIDS maps, we generated a predicted value for each 4-km² cell based on the 30 closest NCDC stations, and rounded the predicted value to the nearest integer.

We compared the accuracy of the three methods via station-by-station cross-validation. Using each interpolation method, we derived a predicted total-day and consecutive-day value for each station based on its 30 closest neighbors. We calculated errors by subtracting the actual observed counts for each station from the interpolated values. We then calculated three mean error measures: mean error (ME) indicates bias (positive = over-prediction, negative = underprediction); mean absolute error (MAE) indicates the magnitude of error regardless of sign; and root mean square error (RMSE) is sensitive to outliers and can be used to

Table 2—Interpolation method comparison for total-day variable^a

	Interpolation Method	2000	2001	2002	2003	2004
Mean	Observed	62.94	61.77	58.77	64.43	70.32
	GIDS ^b	63.44	62.21	59.23	64.93	70.86
	Cokriging	62.97	61.81	58.89	64.64	70.36
	Detrended kriging	62.98	61.79	58.79	64.41	70.37
RMSE	GIDS	13.52	13.13	12.82	13.42	14.27
	Cokriging	13.07	13.07	13.19	14.35	14.00
	Detrended kriging	13.22	12.85	12.52	13.39	14.09
MAE	GIDS	10.39	9.97	9.61	10.14	10.81
	Cokriging	10.10	10.03	9.94	10.92	10.59
	Detrended kriging	10.24	9.82	9.44	10.18	10.66
ME	GIDS	0.508	0.438	0.464	0.500	0.532
	Cokriging	0.033	0.037	0.114	0.216	0.029
	Detrended kriging	0.046	0.018	0.025	-0.018	0.040

^a Cross-validation results for each interpolation method based on five annual data sets. Errors calculated as observed values minus the predicted values; see text for interpretation of root mean square error (RMSE), mean absolute error (MAE), and mean error (ME).

^b GID = gradient plus inverse distance squared.

assess the magnitude of extreme errors (Daly 2006, Nalder and Wein 1998).

To provide a basic visual reference, we used 30-year normal PRISM-derived data sets to construct U.S. maps depicting the total number of days and longest string of consecutive days when weather conditions are typically favorable for *P. ramorum*. We started with 12 monthly grids depicting the number of wet days (i.e., the number of days with precipitation) throughout the conterminous United States. For each monthly wet-days grid, we masked out any cells where temperatures did not fall within the 60 to 80 °F range at some time during the month. Using map algebra, we added the 12 monthly grids together to develop a total-day count for each grid cell in our output map. The consecutive-day count map was, by necessity, more approximately constructed. First, we standardized values in each of the masked monthly grids by converting the number of wet days to a proportion (number of wet days / total number of days in the month) and then multiplying this proportion by 30. Then, using map algebra, we recorded the maximum standardized monthly value for each cell in our output map. This approximated the range of values in the GIDS-derived consecutive-day maps. Nonetheless, because

we used monthly rather than daily data to build the PRISM-derived maps, any comparison to the GIDS-derived maps must be done with care.

Results

In terms of cross-validation errors, the three spatial interpolation methods performed similarly for both the total-day and consecutive-day count variables (Tables 2 and 3). The GIDS approach, as suggested by the ME values as well as the actual versus the predicted means, tended to over-predict slightly more than the other two techniques. The RMSE results indicate that, for some years, the GIDS approach yielded a few more extreme errors, although GIDS had a lower RMSE than cokriging for the total-day variable in 2002 and 2003, as well as a lower MAE in 2001, 2002, and 2003. In general, error differences among the three were not substantial, with MAE consistently holding at approximately 16 percent of the total-day mean value and 25 percent of the consecutive-day mean value for all three techniques.

The GIDS-derived maps for the two count variables (Figures 1 and 2) most obviously show a great deal of annual variability. For the consecutive-day variable, the

Table 3—Interpolation method comparison for consecutive-day variable^a

	Interpolation method	2000	2001	2002	2003	2004
Mean	Observed	5.73	5.72	5.21	6.03	6.25
	GIDS	5.78	5.77	5.25	6.08	6.30
	Cokriging	5.74	5.73	5.21	6.04	6.25
	Detrended kriging	5.73	5.74	5.20	6.03	6.25
RMSE	GIDS ^b	2.14	1.98	1.89	2.15	2.35
	Cokriging	2.10	1.97	1.83	2.14	2.30
	Detrended kriging	2.09	1.98	1.85	2.15	2.30
MAE	GIDS	1.46	1.41	1.28	1.51	1.62
	Cokriging	1.44	1.41	1.24	1.51	1.59
	Detrended kriging	1.45	1.42	1.25	1.51	1.60
ME	GIDS	0.053	0.052	0.039	0.048	0.051
	Cokriging	0.009	0.009	-0.001	0.006	-0.002
	Detrended kriging	0.005	0.018	-0.007	0.003	-0.001

^a Cross-validation results for each interpolation method based on five annual data sets. Errors calculated as observed values minus the predicted values; see text for interpretation of root mean square error (RMSE), mean absolute error (MAE), and mean error (ME).

^b GID = gradient plus inverse distance squared.

Eastern United States generally tended to have higher values than the Western United States, with parts of the Appalachian Mountain region and States along the Gulf of Mexico typically exhibiting high values. However, the extent and spatial distribution of the highest-value area fluctuated substantially year to year. The total-day maps exhibited a similar spatial pattern, but more clearly highlighting some relatively high-value areas in the southern and central Rocky Mountains. Perhaps unsurprisingly, the patterns of the GIDS-derived maps were quite different than the patterns depicted in the PRISM-derived maps.

Discussion

Four main points of emphasis emerge from the results. First, for the tested data sets, the interpolation method did not significantly influence the resulting error. There are several possible explanations for this. Foremost, although the GIDS approach may be technically more appropriate than geostatistical approaches for count-based variables, the Poisson model may not have been a good fit for these data, or the data may have been approximately normal enough to remove any advantage of a Poisson-based process over

geostatistical approaches. Furthermore, among weighted-average interpolation approaches—a category that includes GIDS—kriging is often the best unbiased predictor for data that are not normally distributed (Johnston and others 2003). Another count-oriented approach—Poisson kriging—has recently emerged in health geography and ecological literature, and this may be a promising future direction for count-based spatial interpolation (Goovaerts 2005, Monestiez and others 2006). In the meantime, GIDS has a number of positive characteristics. It violates fewer assumptions than geostatistical approaches—in particular, the assumption of second-order stationarity (Cressie 1993). Furthermore, the GIDS approach is transparent and easily implemented. To use more complex approaches, particularly PRISM, requires estimation of numerous parameters, so a certain degree of subjectivity is involved. The GIDS approach can easily accommodate covariates besides elevation, and, in fact, could easily be adapted for multiple covariates in order to refine the results. Finally, the GIDS approach has been implemented in R code (R Core Development Team 2006), and as such is an open source resource that may be more readily available than GIS-based interpolation approaches.

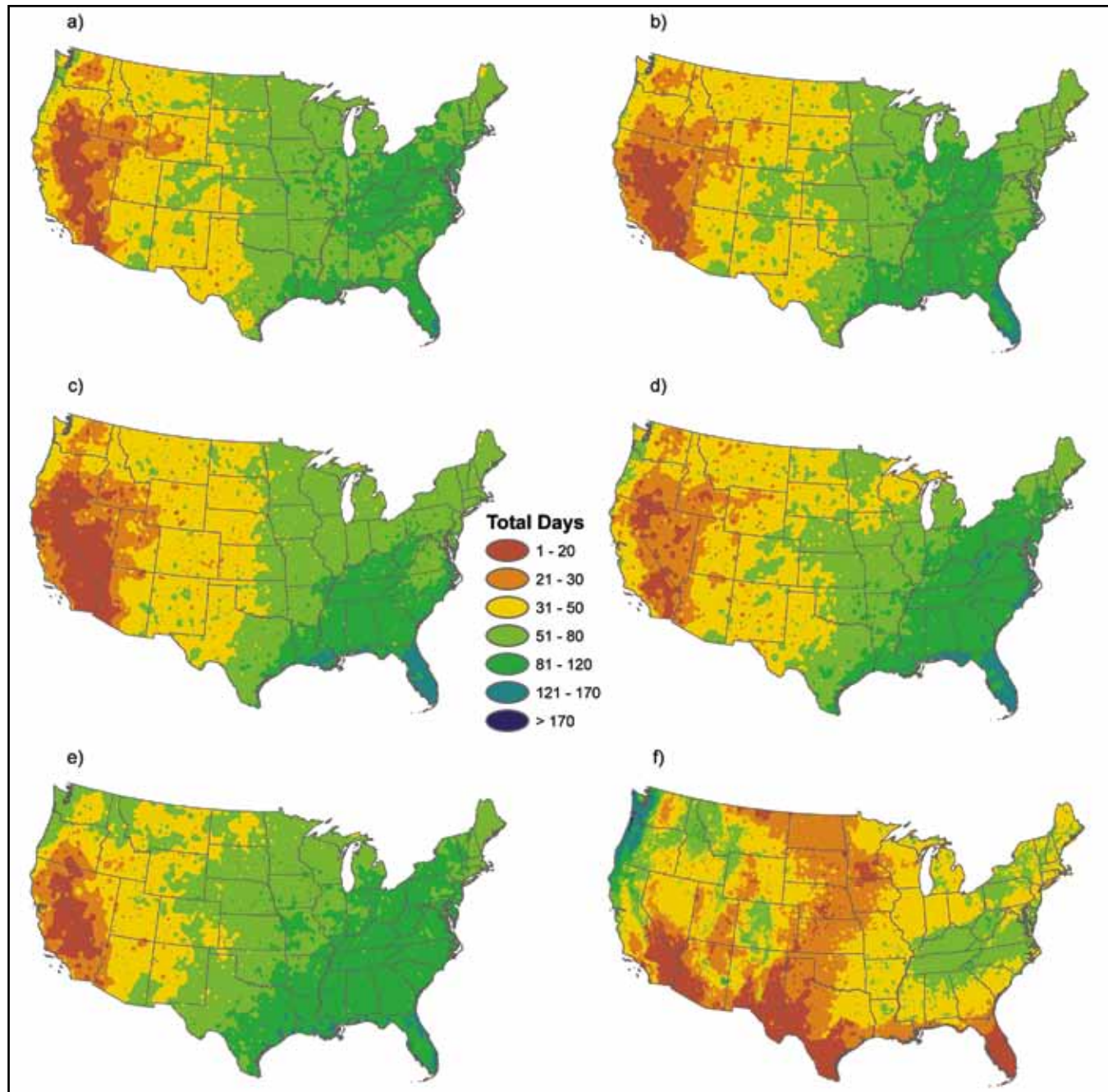


Figure 1—Annual maps of the total number of days with weather conditions favorable for *Phytophthora ramorum*, interpolated using the gradient plus inverse distance squared method: (a) 2000, (b) 2001, (c) 2002, (d) 2003, and (e) 2004; (f) for visual comparison, a total-day map approximated using monthly Parameter-elevation Regressions Independent Slopes Model.

Second, the interpolations of the two-count variables appear to have an acceptable degree of error. The distribution of cross-validation errors for the GIDS interpolations are revealing in this regard. For the consecutive-day variable, across all 5 years, only 25 percent of values were

exactly predicted, but nearly two-thirds of predicted values were within 1 day of the observed value. For the total-day variable, only 4 percent of values were exactly predicted, but nearly 50 percent were within 5 days and greater than 75 percent were within 10 days. This should be adequate for

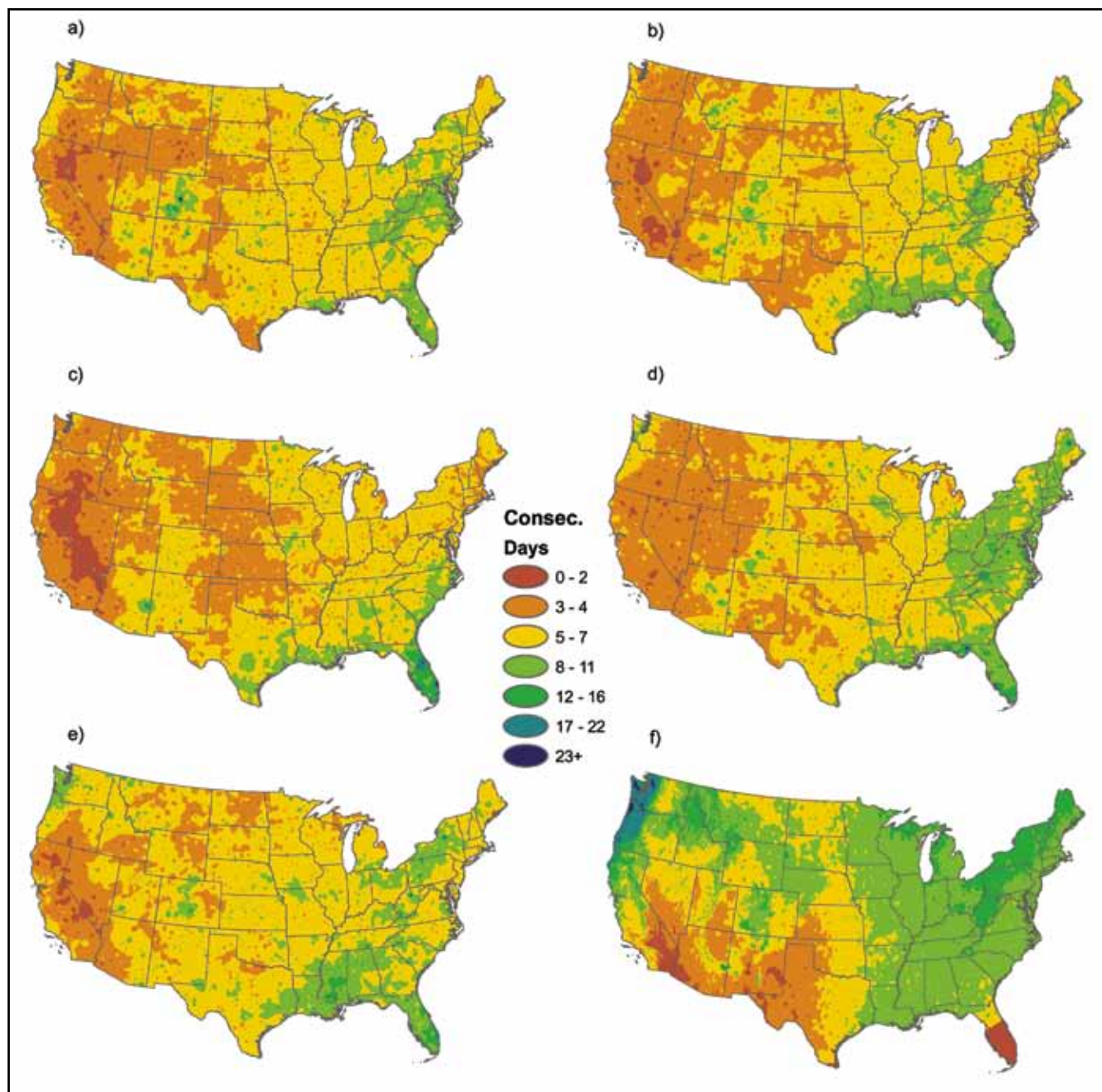


Figure 2—Annual maps of the longest string of consecutive days with weather conditions favorable for *Phytophthora ramorum*, interpolated using the gradient plus inverse distance squared method: (a) 2000, (b) 2001, (c) 2002, (d) 2003, and (e) 2004; (f) for visual comparison, a total-day map approximated using monthly Parameter-elevation Rrgressions Independent Slopes Model

broad-scale ranking of areas according to their relative risk based on climatic and weather conditions.

The third and perhaps more important point is that the information provided by the constructed annual count maps is substantially different from results that can be captured

using monthly climatic data sets based on 30-year normals. For *P. ramorum* and other currently emerging threats, it may be advantageous to identify areas that have exhibited favorable conditions in a given year and determine whether, for example, the pathogen was positively detected at any

nurseries in those areas during that time period. In fact, this suggests a need for a regularly updated database, and the GIDS method may be one way to generate a regularly updated data set from the NCDC data. Recent annual maps can be used in conjunction with 30-year normal data to create a strong picture of current risk.

Fourth, if the count-based variables we calculated are reasonable representations of the level of favorable climatic conditions for *P. ramorum*, then this suggests that large portions of the Eastern United States—perhaps more than originally estimated—have periods during each year where they may be especially susceptible to infection. Because climate and weather may not be severely limiting factors, detailed analyses of potential pathways and potential host species distribution may be in order for much of the Eastern United States.

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A Multicriteria Framework for Producing Local, Regional, and National Insect and Disease Risk Maps

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Abstract

The construction of the 2006 National Insect and Disease Risk Map, compiled by the USDA Forest Service, State and Private Forestry Area, Forest Health Protection Unit, resulted in the development of a GIS-based, multicriteria approach for insect and disease risk mapping that can account for regional variations in forest health concerns and threats. This risk mapping framework, used by all nine Forest Service regions and 49 States, provides a consistent, repeatable, transparent process through which interactive spatial and temporal risk assessments can be conducted at various levels to aid in decisionmaking. The national framework was designed to be highly iterative, using input from a wide range of sources including subject area experts. The framework consists of a five-step process: (1) identify agents of concern (insects and diseases) and target-host species; (2) identify, rank, and weight criteria that determine the susceptibility (potential for introduction and establishment) and vulnerability (potential for tree mortality to occur if an agent is established) to each agent; (3) standardize criteria values, and combine the resultant maps using a series of weighted overlays; (4) convert modeled values for each agent to predicted basal area (BA) loss over a 15-year period; and (5) identify regions at risk of encountering a 25-percent or greater loss of total basal area in the next 15 years. This potentially interactive threshold was set by the National Risk Map Oversight team for the national risk map product.

The National Insect and Disease Risk Map resulted in the integration into a national map of 186 forest insect and disease models, individually run and assembled on a central server located at the Forest Health Technology Enterprise Team (FHTET) in Fort Collins, Colorado. The national

framework also enables local knowledge and data to be entered into models, allowing for quick, large-scale assessments. The development of this national framework is described here.

Keywords: Forest health monitoring, GIS, insect and disease risk, multicriteria modeling, NIDRM, risk map.

Introduction

Ensuring the health of America's forests requires the analysis, understanding, and management of complex and interrelated natural resources. Increasing human-use pressures, a continual threat from native and exotic insects and diseases (USDA FS 2005), and more complex management policies make natural resource management demanding. To accurately assess where and how forest resources are being impacted, resource managers and policymakers require information beyond tabular summaries. In turn, this requirement has created an increasing need for spatial-based, decision-support systems that can quickly summarize a wide range of tabular and geographic information. Such systems provide resource managers with the information they need to make clear, informed choices and efficiently allocate human and financial resources. Therefore, integrated and comprehensive approaches that use technologies, such as geographic information systems (GIS), with their ability to analyze a large number of spatial variables concurrently, are becoming increasingly important for the protection and management of our Nation's forest resources (Ciesla 2000, McRoberts and others 2006, Mowrer 1992, Reynolds 1999, Stein and others 2005).

The primary goal in the development of the 2006 National Insect and Disease Risk Map (NIDRM) is the creation of a national communications tool that will provide policymakers, USDA officials, and Federal and State land managers with a periodic, strategic assessment for risk of tree mortality from major insects and diseases. NIDRM is an integration of 186 individual risk models constructed within a common, consistent, GIS-based, multicriteria framework that accommodates regional variations in

current and future forest health conditions, knowledge, and data availability. The 2006 NIDRM was created through a modeling process that is repeatable and transparent, and through which interactive spatial and temporal risk assessments can be conducted at various scales to aid in the allocation of resources for forest health management. This process is intended to increase the utilization of forest health risk maps within and outside the National Forest System.

The production of the 2006 risk map has been a highly collaborative process, coordinated by the USDA Forest Service, State and Private Forestry Area, Forest Health Monitoring Program (FHM). Entomologists and pathologists from all States and every FHM region were invited to take part in the process of developing the NIDRM. Teams were created with forest health and GIS specialists from the Forest Service, State agencies, and academia to oversee and assist in model development. Even though the goal of the authors is to describe in this paper the GIS framework developed for the construction of NIDRM and to briefly demonstrate how this process can be used to conduct assessments at multiple spatial scales, the authors want to emphasize the importance of a team approach that ensures participation from local resource managers.

The Assessment Framework

Defining Risk

The definitions for “risk” and “hazard” in forest pest management can be confusing and contradictory. Rather than reconcile the various definitions of risk and hazard we use the following construct.

When assessing risk as it relates to forest health, risk is often composed of two parts: the probability of a forest being attacked (susceptibility) and the probability of resulting tree mortality (vulnerability) (Mott 1963). Characterizing the spatially explicit probability of insect and disease activity requires spatially explicit quantitative data. However, because such data are often lacking at regional, national, and local levels, we define risk as the potential for harm owing to exposure to an agent(s). Also, we draw the distinction between susceptibility and vulnerability (Mott 1963), but in the context of potential rather than probability.

Our threshold value for mapping risk is defined as the expectation that, without remediation, over the next 15 years 25 percent or more of standing live basal area (BA) in trees greater than 1 inch in diameter will die owing to insects and diseases. The threshold value for mapping insect and disease risk is independent of the GIS framework discussed in the remainder of this paper. Therefore, the framework can support any threshold.

A Conceptual Overview of a National Risk Assessment Framework

Figure 1 provides a conceptual overview of the risk-assessment process discussed here. The modeler first indicates whether the forest pest under study is endemic or not. If a pest is established throughout a region, then the potential or source for actualized harm is assumed to be equal everywhere, and all host material is susceptible. In such cases, susceptibility assessments are not required. If a mechanism or data set exists that addresses varying pest densities in time and space, we can accommodate those densities within our framework. However, few national data sets for pest density exist, and we assume presence or absence in our modeling scenarios. A vulnerability model, which determines the likelihood and extent to which trees will be harmed by the pest of concern within the defined time frame of 15 years, is required to complete a risk assessment in a case where a pest is already established.

When considering forest pests, either nonnative exotics that have not been established or cyclic native pests whose outbreaks occur sporadically about the landscape, the modeler must first construct a model of pest potential or susceptibility. Susceptibility is based on the biological availability of a host and the potential for introduction and establishment of a forest pest within a predefined time frame (in this case, 15 years). With a susceptibility model constructed, the next step is to determine whether a forest pest will always kill its host.

Generally, once established, some risk agents, such as sudden oak death (*Phytophthora ramorum*) and chestnut blight (*Cryphonectria parasitica*), harm their hosts in the same manner throughout the landscape, regardless of existing site and stand conditions. This applies to exotics and

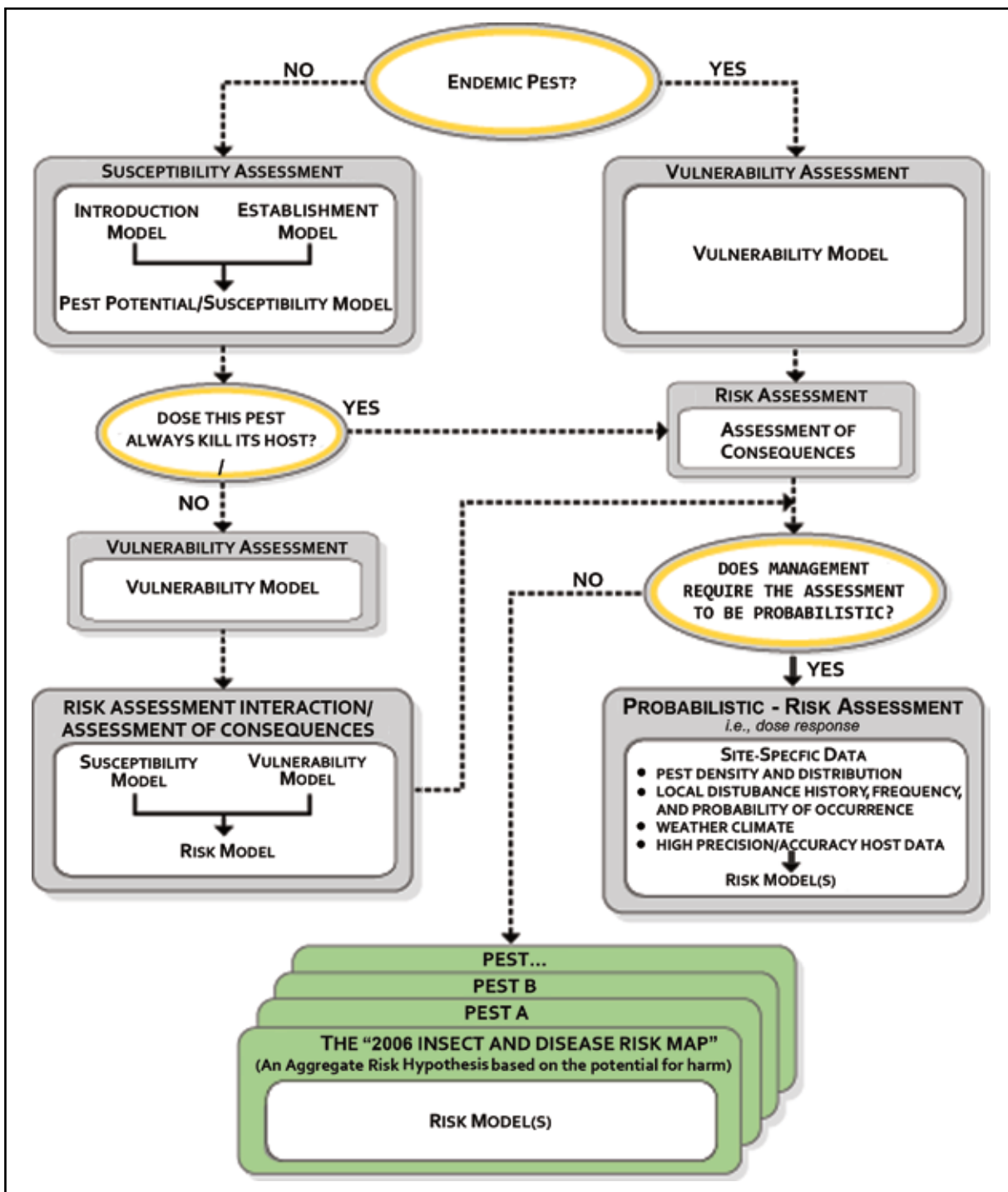


Figure 1—Risk assessment framework provides a conceptual overview of the multicriteria risk assessment framework used to construct the 2006 National Insect and Disease Risk Map (NIDRM).

native species, alike, although some exotic pests, such as gypsy moth (*Lymantria dispar*), produce mortality rates that can differ greatly depending on site and stand

conditions. If a vulnerability assessment is not required, i.e., pest effects are not site-dependent, then the susceptibility model can be used for the final risk assessment.

In some cases, as in the case of gypsy moth where both susceptibility and vulnerability models are run, the interaction of these models creates the risk assessment. The degree to which either vulnerability or susceptibility always results in harm determines how much influence each model has on the final outcome of the risk assessment. Think of the interaction between susceptibility and vulnerability as being on a continuum, whereupon agents such as gypsy moth are at or near the middle where both susceptibility and vulnerability receive equal influence, and risk agents such as sudden oak death and mountain pine beetle (*Dendroctonus ponderosae*) require only vulnerability assessments. All forest pests fall somewhere on this continuum. When a risk assessment is in hand, estimates of potential BA loss over the next 15 years can be derived.

A GIS-Based Multicriteria National Risk Assessment Framework: A Five-Step Process

The risk assessment framework used to construct NIDRM is best explained using a hypothetical example, particularly in steps 2 through 5. (A real world example is not used here, so the reader is free to focus on the process rather than the correctness of the example.) It should be noted that the modeling process presented is not limited to regional or national-level work; rather, it is designed to be usable at any scale. This is illustrated in the latter part of this paper. However, the accuracy of the model outputs depends on knowledge about forest pest behavior, the degree of informed personal judgment of the model developers, and the spatial accuracy and precision of the data driving the models.

Because of its availability from State and Federal agencies, ease of use, and relative stability, ESRI ArcView 3.x Spatial Analyst 2.x ModelBuilder was selected as the software for the multicriteria framework. In addition, previous familiarity with ESRI Spatial Analyst 2.x among GIS specialists greatly reduced the rollout time of the risk assessment framework. Other commercial software that supports multicriteria modeling includes IDRISI and ESRI ArcGIS 9.x ModelBuilder. IDRISI (Eastman 2001, Eastman and others 1995) has a very comprehensive set of multicriteria modeling tools, but is not widely used in the Forest Service. In addition, the Ecosystem Management Decision

Support (EMDS) (Reynolds 1999), an ESRI ArcGIS extension at version 3.0, was developed within the Forest Service to support local and regional decisionmaking and to provide a framework for conducting knowledge-based ecological assessments.

Step 1: Identify Risk Agents and Host Species—

Often, forest pest distributions are limited to specific climatic or biophysical regimes or both. In addition, pest behavior and population dynamics often differ by geographic area and must be modeled differently to accommodate local and regional conditions. It is possible within the NIDRM framework to account for this variation by constructing multiple models for an individual forest pest. In order to better capture this natural variation and to prevent models from differing along political boundaries, models were constrained to the extents of Bailey's (2004) ecoregions. Because ecoregions capture broad climatic and biophysical patterns, they provide a more realistic base map on which to delineate differences in forest-pest models.

For much of the remaining discussion, we will use the following hypothetical example: risk agent X is a nonendemic pest that attacks aspen in the central Rocky Mountains. The amount of aspen mortality occurring in infested trees varies according to site conditions. Because of this, risk assessments for agent X require the construction of both a susceptibility and vulnerability model (Figure 1).

Step 2: Identify, Rank, and Weight Criteria—

After risk agents and host species are identified, the criteria (factors and constraints) that determine both the potential for risk-agent establishment and host vulnerability for potential mortality must be identified. For the risk assessment framework presented here, we define:

- **Susceptibility** as the potential (rather than probability) for introduction and establishment, over a 15-year period, of a forest pest within the range of a tree species.
- **Vulnerability** as the potential (rather than probability) for mortality of a tree species at a maximum realizable mortality rate over a 15-year period if a forest pest were to become established.

Table 1—Sample weights for vulnerability to risk agent X

Criteria	Weight
BA	65 percent
QMD	22 percent
Soil Dry/Wet	13 percent

Weights always sum to 100 percent and represent the relative importance of each criterion.

Regions that are both very susceptible to a pest attack and highly vulnerable to its effects (as where many trees that are weakened or stressed are present) are the most likely to experience the maximum realizable mortality rate—an estimation of the largest likely mortality loss for a risk agent over a 15-year period. Regardless of how vulnerable trees are at any given location, they will not experience mortality from a risk agent if these regions are not susceptible to attack. In other words, under some circumstances, susceptibility can act as a constraint. Constraints are criteria that must be met for susceptibility and vulnerability potential to occur at any given location. For our hypothetical risk agent X, we have assigned the maximum realizable mortality rate of 100 percent, meaning that risk agent X is always lethal to aspen when all criteria for susceptibility and vulnerability are met. With risk agent X isolated, we can now identify a set of criteria for susceptibility and vulnerability.

Factors for susceptibility are:

1. Distance to known infestations
2. Average annual minimum extreme temperature
3. Aspen host presence, with the latter criterion acting as a constraint.

Factors for vulnerability are:

1. Aspen BA
2. Aspen quadratic mean diameter (QMD)
3. Soil dryness/wetness

Although risk agent X requires both a susceptibility and vulnerability model for its risk assessment, recall that risk assessments for some pests require only one or the other.

Many forest pests invariably cause tree mortality whereas others only contribute to their demise. Stressors often work in concert to cause mortality. Modeling pest

complexes that may work together to cause mortality requires a layer (or layers) representing the distribution and intensity of each insect and disease that contributes to the mortality in any given area. Depending on whether the additional agent(s) contributes to an increased risk of establishment or mortality or both, this layer is then used as a criterion in the susceptibility or vulnerability models of the primary risk agent. Owing to the lack of understanding of the interactive effects among multiple stressors, we treat interactions as additive. The exception to this is where we use one pest risk model to constrain the other. If the interactive effects of risk agents are understood, we can model them under this current framework.

The relative importance of each criterion, or rank, for determining whether an area, or pixel, has the potential to be susceptible and vulnerable to a risk agent is entered into a pairwise comparison matrix. A pairwise comparison matrix is a robust method for assessing the comparative importance of factors (Eastman 2001, Eastman and others 1995, Saaty 1977). It is particularly useful when attempting to derive weight evaluations for multiple criteria under many considerations. Every possible pairing of factors must be identified and entered into the matrix, ordering the criteria most important to least important. The matrix is used to generate a set of weights representing the relative importance of each criterion (Table 1). The resultant weights, expressed as percentage influence, must sum to 1 (or 100 percent) and are used to combine criteria values within a weighted overlay (see step 3). Separate matrices are generated for both susceptibility and vulnerability. A matrix is not needed if only a single criterion is present.

Prior to entering values into the pairwise matrix, comparisons must be made between criteria using a 10-point continuous rating scale (Table 2) modified by Krist (2001, 2006) from the 9-point scale Eastman uses in the IDRISI software (Eastman 2001, Eastman and others 1995). Rankings in Table 2 represent the relative importance of each criterion. For example, QMD is moderately less important than BA for determining the vulnerability potential to risk agent X; therefore, QMD receives a value of one-third in the comparison matrix whereas BA receives a value of 1. Soil

Risk Model Worksheet - Interior West

Risk Agent(s): Risk Agent X Host(s): Aspen

Model Extent: Central Rocky Mountains Max Percent Mortality: 100%

Susceptibility

Rank/Weight	Criterion	Risk Begins (a)	Risk Peaks (b)	Risk Decreases (c)	Risk Ends (d)	Curve	Rank	Weight
1 50%								
Criteria 1	Dist. To Known Infestations (km)	0	0	0	200	Linear	1	75%
Criteria 2	Avg. Minimum Extreme Temperature (F)	-20	0	0	0	Linear	1/3	25%
Criteria 3								
Criteria 4								
Criteria 5								
Criteria 6								
Criteria 7								
Criteria 8								
Criteria 9								
Criteria 10								

Vulnerability

Rank/Weight	Criterion	Risk Begins (a)	Risk Peaks (b)	Risk Decreases (c)	Risk Ends (d)	Curve	Rank	Weight
1 50%								
Criteria 1	Aspen BA (sq. ft./acre)	20	120	120	120	Linear	1	65%
Criteria 2	Aspen QMD (in.)	0	10	10	10	Linear	1/3	22%
Criteria 3	Soil Dryness/Wetness	20	20	20	50	Linear	1/5	13%
Criteria 4								
Criteria 5								
Criteria 6								
Criteria 7								
Criteria 8								
Criteria 9								
Criteria 10								

Constraints Aspen host must be present. **Comments**

Citations 2, 8, 15 **Model Certainty** 4 - Expert Opinion

Figure 2—The spreadsheet template allows staff to document models, rank criteria, and calculate weights. Values shown are for the risk agent X example.

Table 2—10-point continuous rating scale

Description	Comparison rating
Most important	1
	1/2
Moderately less	1/3
	1/4
Strongly less	1/5
	1/6
Very strongly less	1/7
	1/8
Extremely less	1/9
	1/10
Unsuitable	N/A

dryness/wetness is of strongly less importance to BA and is assigned a value of one-fifth.

The 10-point rating system enables forest health specialists to select the most important factor(s) and compare

the remaining criteria to it (them). All these criteria have a positive influence and contribute to potential. The negative impacts of a criterion can be accounted for by reversing the rankings for the criterion values, thus turning a negative relationship into a positive one (one that contributes to potential).

We simplified the workload of regional forest health specialists constructing models by enabling the spreadsheet (Figure 2) used to collect model information to simulate the comparison matrix developed by Saaty (1977) and to calculate weights automatically. Having the ability to calculate weights automatically enables the user to see changes in weights immediately as ranks are adjusted. The weights in the spreadsheet are calculated by first summing all the rank values and then dividing each rank by that sum.

In addition to weighting risk agent criteria, weights must be assigned to susceptibility and vulnerability, based on their importance in determining the potential for a tree species to experience the maximum realizable mortality rate from the pest of concern in the next 15 years. These weights are used to combine the resultant susceptibility and vulnerability models in the final risk assessment (see step 3). In the case of risk agent X, equal weight (50 percent) was given to both susceptibility and vulnerability. Remember that susceptibility acts as a constraint in the final risk assessment; areas with no possibility of being susceptible in the next 15 years are not at risk.

Criteria, rankings, and weights for risk agent models can be selected in a number of ways. Ideally, if data on the distribution and intensity of a risk agent exist, statistical analyses may be performed in the hopes of identifying relationships between risk agent activity and forest and biophysical attributes, as represented in GIS layers. If such a relationship is found, the strength of the correlation can be used to determine weights. Unfortunately, this data-driven or literature/research-based approach is not always possible because, in many instances, data on risk agent distributions, intensity, and behavior are either inadequate or incomplete or both. In such instances, modelers must rely on informed professional judgment or expert opinion or both when selecting criteria and weights.

The information collected for steps 1 and 2, including the basis for a model or model certainty, appears in Figure 2.

Step 3: Standardize Criteria Values and Combine the Resultant Maps—

With the risk agent criteria and their corresponding GIS layers identified and weights generated for susceptibility and vulnerability, the factor values must be standardized, based on a common evaluation scale. Standardization allows for the comparison of criteria with differing values, such as BA, with units of square feet per acre and QMD, with units in inches. An evaluation scale from 0 to 10, with 0 representing little or no potential, and 10 representing the highest potential, was chosen. For example, in the case of risk agent X, higher stocked stands are more vulnerable;

Table 3—Rankings for BA (basal area) using linear and sigmoidal memberships

BA	Rank (linear)	Rank (sigmoidal)
ft^2/ac		
20-30	0	0
30-40	1	0
40-50	2	0.3
50-60	3	1.2
60-70	4	2.5
70-80	5	4.1
80-90	6	5.9
90-100	7	7.5
100-110	8	8.8
110-120	9	9.7
>120	10	10

Fuzzy memberships enable values to be reclassified or recoded in a variety of ways to capture relationships between insect and disease behavior and criteria. This example demonstrates the effects of two commonly used memberships on BA, with 10 representing the highest potential for risk.

therefore, areas with stocking levels approaching 120 square feet or more are given a value of 10 (Table 3). Potential in areas with less than 20 square feet of aspen is very low and is assigned a 0. Assigning a 0 to a criterion does not eliminate the possibility for risk to occur. If other criteria have values greater than 0, potential is still possible at a location.

Regions without aspen are constrained. Constraints do not need to be standardized, but they do restrict modeled potential to specific areas, such as regions containing aspen or areas within a certain distance of current risk agent infestations. Because of the restrictions, the potential for risk can be precluded in some areas, regardless of the strength of the other criteria.

For simplicity, and due to the frequent lack of precise data layers or models or both at the national level, an integer scale from 0 to 10 was chosen. At a finer resolution and with more precise data, an extended standard scale may be of more use (e.g., an integer scale of 0 to 100 would capture a wider range of variation in the data).

A standard scale may be applied to a GIS data set through a manual recoding of the criterion values. This is particularly easy to do when data have sharp boundaries and discrete classes. However, this is not an easy task when the transition from criterion values with potential to values without potential is gradual. Consider the BA criterion in our example. Potential for vulnerability to pest X ends or

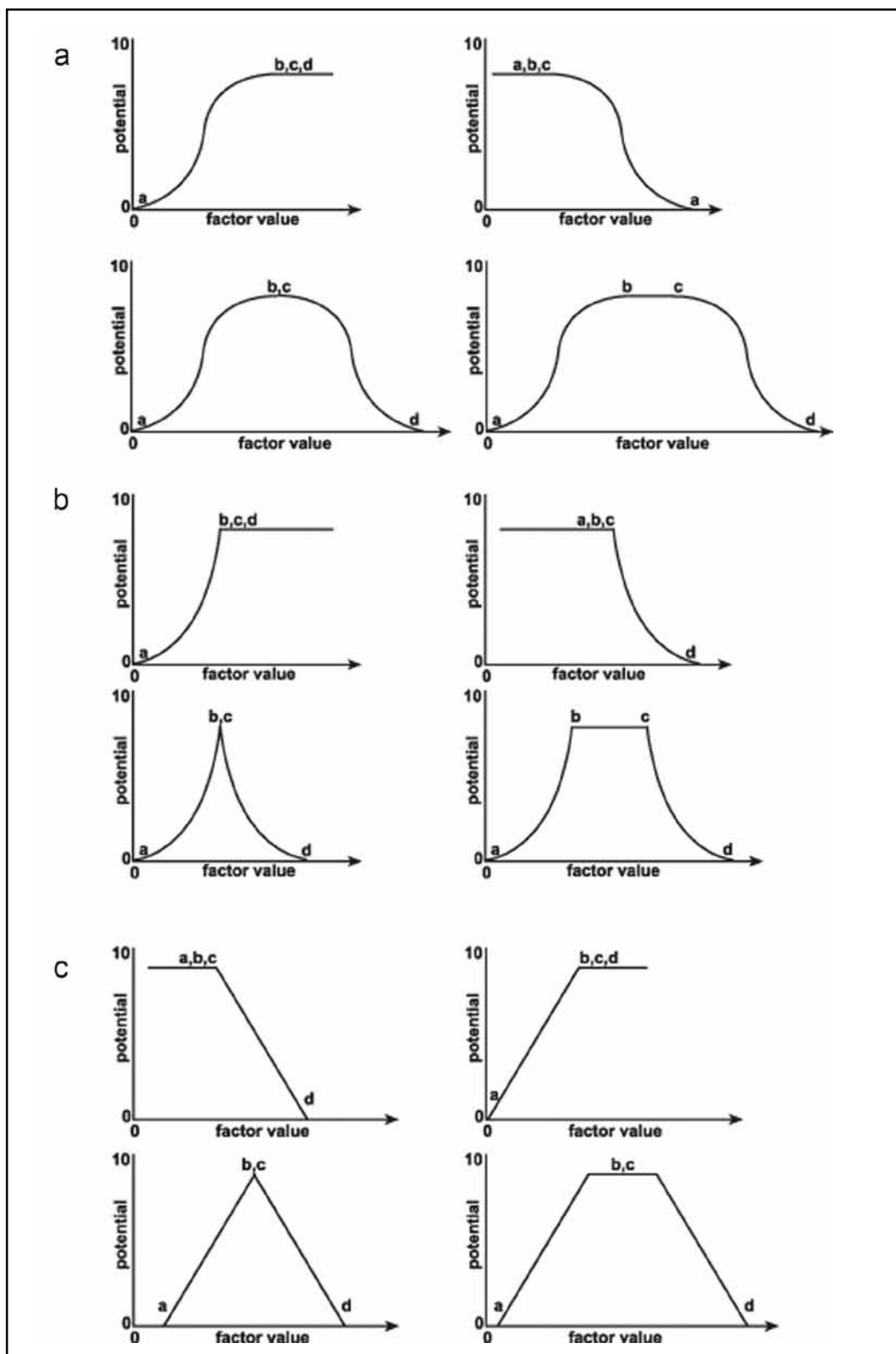


Figure 3—Available fuzzy memberships include: (a) sigmoidal functions, (b) J-functions, and (c) linear functions. Custom functions can be developed, based on the relationship of the data to potential risk.

is very low when stocking levels are at or below 20 square feet and increases gradually until stocking levels reach 120 square feet. Rather than manually break down this range into 10 discrete classes, a fuzzy membership set (Eastman 2001, Eastman and others 1995) can be used to stretch or assign a continuous set of values, automatically. Another advantage to selecting a fuzzy membership is that the data can be assigned standardized values in differing ways, depending on how potential varies within a criterion. The most common method of assigning values based on fuzzy membership is a simple linear stretch in which potential increases linearly. However, if potential rises gradually and then accelerates and tapers off, a sigmoidal function may capture the natural variation more precisely.

Twelve fuzzy membership functions were used in the construction of NIDRM models, including linear, sigmoidal, and J-shaped functions (Figure 3). Table 3 shows the difference between the resultant values of a linear and sigmoidal stretch, the most common memberships used during the production of NIDRM. The letters (inflection points) on the graphs in Figure 3 represent where risk potential (a) begins, (b) peaks or reaches its highest, (c) begins to decrease (though this may or may not happen), and (d) ends or no longer changes (levels off). In the risk agent X example, a is set to 20 square feet of BA whereas b, c, and d are set to 120 square feet. Notice the letters and descriptions on the column headers of the spreadsheet in Figure 2 and the Curve column, where a fuzzy membership (curve) is chosen for each criterion.

ESRI ArcView 3.x Spatial Analyst 2.x ModelBuilder does not contain a routine that will automatically recode or stretch criterion values according to a fuzzy membership, so a worksheet was developed to calculate break points that could then be manually entered into the RECLASS module of ESRI ModelBuilder (Figure 4), effectively dividing the values into 11 classes. Values for each risk agent's criteria were manipulated in Model Builder in this way. Figure 5 illustrates what a set of standardized GIS layers would look like for the central Rocky Mountains using risk agent X as an example.

With weights generated and values standardized for each criterion, all criteria can be combined in a series of

weighted overlays representing susceptibility, vulnerability, and the final risk-agent-mortality assessment. Factors are combined within a weighted overlay or weighted linear combination by multiplying the factor weight by each criterion value, followed by a summation of the results (Saatty 1977):

$$P = \sum w_i x_i$$

where:

P = potential for susceptibility, vulnerability, and risk

W_i = weight criterion i

X_i = criterion score of factor i

Figure 6 illustrates how sample values from risk agent X are combined in a series of three weighted overlays. The output from each weighted overlay has a value from 0 to 10, the same as the standard evaluation scale used for each criterion (Figure 5). The higher the value, the greater the likelihood or potential for a tree species to be susceptible or vulnerable to a risk agent. The greater the value from the weighted overlay of the resultant susceptibility and vulnerability maps, the greater the likelihood or potential for a tree species to experience mortality over the next 15 years.

ESRI ModelBuilder provides a weighted overlay module in which criteria can be entered, weighted, and formally ranked. Figure 7 shows the Model Builder weighted overlay for the risk agent X vulnerability model.

Step 4: Convert Modeled Values to an Estimate of BA Loss—

Using a standardized scale from 0 to 10 allows for the easy conversion of risk potential to estimates of BA loss. Recall that when all criteria are met for susceptibility and vulnerability within a particular area or pixel, the host species within that area is likely to experience the maximum realizable mortality rate over the next 15 years. Based on this assumption, a pixel in our agent X risk assessment receiving a value of 10 would be assigned a 100-percent mortality rate, whereas a pixel with a value of 5 would receive a 50-percent mortality rate. Once pixels have been assigned their mortality percentages based on their mortality potential, this layer is multiplied by a surface representing host BA to produce loss estimates for a tree species. For example, a stand with 100 BA of aspen and a simulated

Risk/Mortality Scaling Tool

To obtain eleven class values (for risk values, mortality thresholds), enter the risk begins and risk peaks values. Equal interval classes will be calculated.

	Input Value	Classes	Scaled Value
Risk Begins (0):	20	20	0
		30	1
		40	2
		50	3
		60	4
		70	5
		80	6
		90	7
		100	8
		110	9
Risk Peaks (10):	120	120	10

Reclassification

Classification Type: **Classification Table**

Define the classification table

The input values are grouped into classes. The Class Start Value and Class End Value define the range for each class; Labels are used in the input theme's legend. The New Class Value is assigned to cells in the output theme; To insert or delete a class, highlight it, then click the Insert Class or Delete Class button. To apply a classification method, click the Classify button.

	Class Start Value	Class End Value	New Class Value	Label
▶	0	1	1	0 - 1
	1	20	2	1 - 20
	20	30	3	20 - 30
	30	40	4	30 - 40
	40	50	5	40 - 50
	50	60	6	50 - 60
	60	70	7	60 - 70
	70	80	8	70 - 80
	80	90	9	80 - 90
	90	100	10	90 - 100
	100	110	11	100 - 110

Figure 4—Criteria-ranking tool and RECLASSIFICATION module in ModelBuilder are shown. The criteria-ranking tool divides factor values into classes that can be entered into ModelBuilder using the RECLASSIFICATION module.

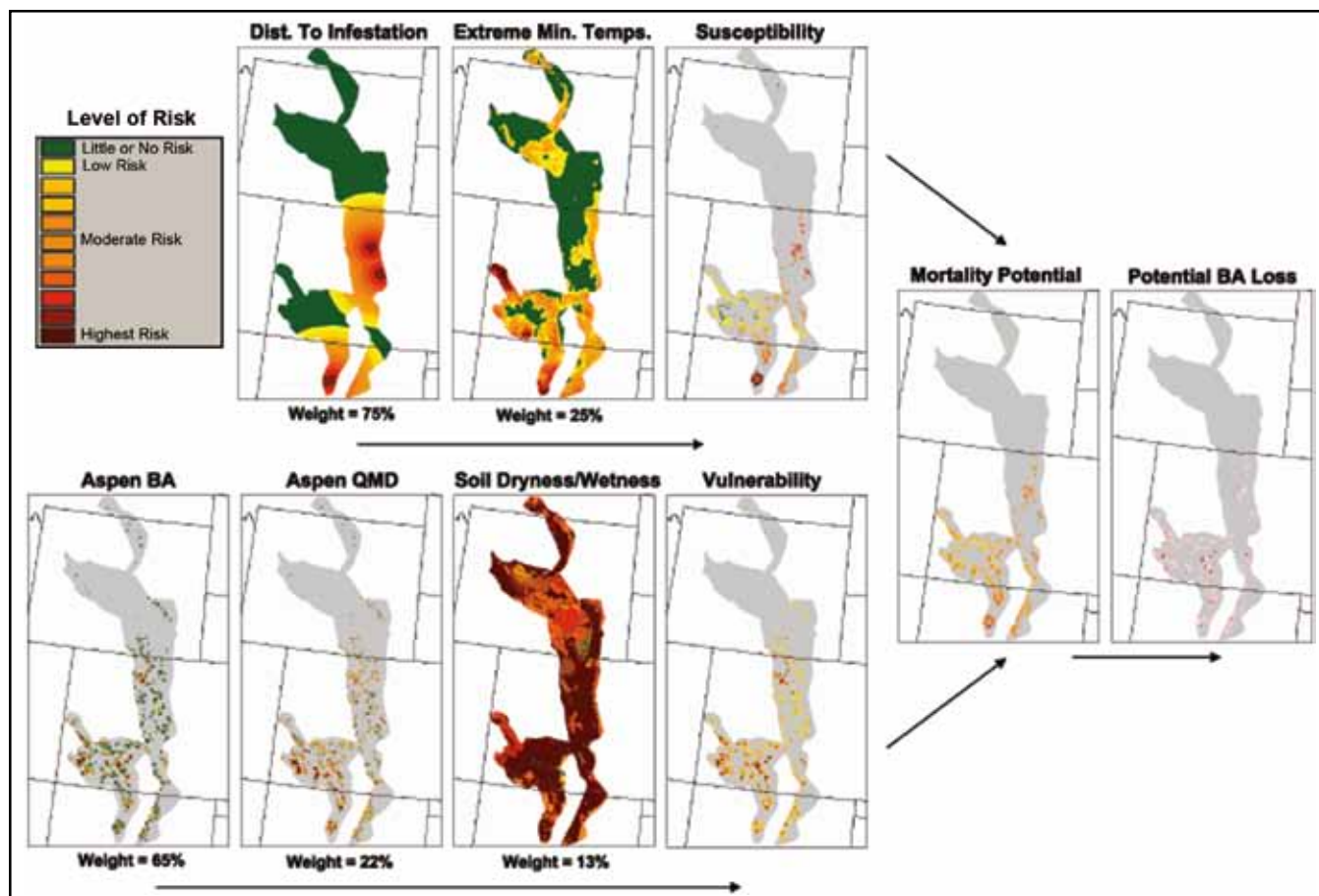


Figure 5—Sample criteria maps and resultant model outputs are shown below. All criteria used in the risk agent X model are displayed with their corresponding weights. Notice how each layer affects the outcome. Arrows show the direction of flow during model construction.

mortality potential for risk agent X of 5 could lose 50 square feet of basal area (50 percent $\times$ 100 BA) in the next 15 years. Remember, model results do not guarantee that mortality will occur at any given location; rather they suggest the **potential** for loss. Figure 5 shows the map of BA loss for Risk agent X.

In cases where multiple pests are acting on a single forest species, the resultant BA losses cannot be added up to calculate total BA losses. For example, if one agent attacks a resource and has the potential to kill 75 percent of the trees and another agent attacks the same resource in the same area and may kill 75 percent of the trees, it is incorrect to say that up to 150 percent of the trees may be killed. Under the simplifying assumption that mortality agents act independently (a common assumption in the development of the

NIDRM), mortality from multiple agents is calculated as:

$$D = 1 - (1-p_1)(1-p_2)(1-p_3)\dots (1-p_n)$$

where:

D = total proportionate mortality

p1 = proportionate mortality caused by agent 1

n = nth agent

In the example above where two agents may each cause 75-percent mortality, 94 percent of the total BA would be lost in that pixel when total losses are calculated using the simplifying assumption:

$$D = 1 - (1-p_1)(1-p_2)$$

$$D = 1 - (1-0.75)(1-0.75)$$

$$D = 0.9375 \text{ or } 94 \text{ percent}$$

We realize this mechanism does not address complex interactions between various pests. The body of literature

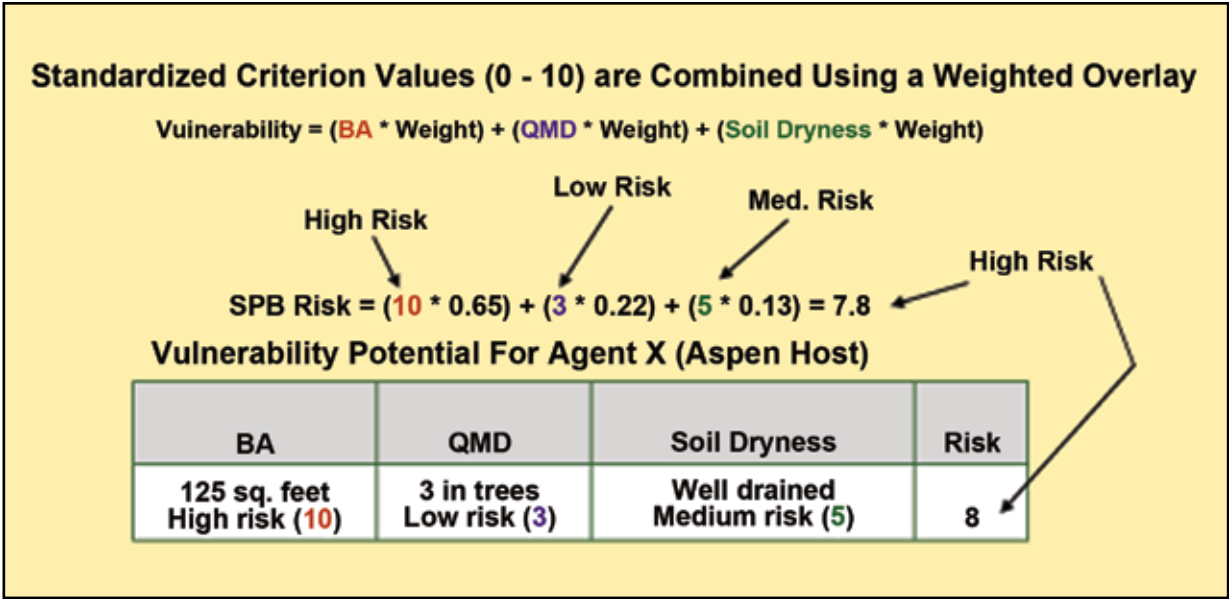


Figure 6—Weighted overlay for the risk agent X vulnerability model is displayed. Notice how much influence the relatively high weight for BA has on the final model outcome.

regarding pest interactions is limited, and cumulative impacts are not understood well enough so that complex interactions can be modeled with confidence.

Step 5: Identify Regions at Risk—

To calculate the percentage of total BA that might be lost in each pixel, estimates of potential 15-year BA loss compiled for all risk agents in step 4 were divided by a surface representing total BA. Pixels where the total loss exceeded or met 25 percent of the total BA were flagged for the national composite risk map (NIDRM). Because the original percentages are available, different threshold values for risk can be defined and mapped. Risk owing to individual pest species by host also were provided.

Modeling at Multiple Scales/Resolutions

Once a model has been constructed in ESRI ArcView 3.x Spatial Analyst 2.x ModelBuilder, models can be rerun at multiple scales. The GIS layers can be swapped in and out of ModelBuilder with little or no modification when standard measurement units exist across scales. As additional data become available at finer scales, supplementary criteria can be added to the model. Figure 8 shows the same southern pine beetle (*Dendroctonus frontalis*) (Thatcher and

Barry 1982) model run at three different resolutions (1 km, 250 m, and 30 m). The 30-m-resolution model includes an additional criterion depicting forest stand connectivity.

Discussion/Conclusions

The 2006 national risk assessment employed 186 risk-agent models representing over 50 risk agents acting on 61 tree species or species groups, with all models assembled into a national composite (NIDRM) (Figure 9). Given the nature of our assignment to construct a national, 15-year assessment of forest health risk from insects and diseases, we believe that NIDRM is successful because it is:

1. Based on an integrating technology. NIDRM represents the collection and integration of multiple risk models developed through an iterative, hands-on process by local forest health specialists. The risk assessment frame-work presented in this paper is able to integrate outputs from a wide range of models and is implemented through software that gives forest health specialists direct access to GIS models.
2. Transparent and repeatable. The 2006 modeling framework provides a consistent, repeatable, transparent process to conduct risk assessments.

Weighted Overlay

Evaluation Scale **Overlay Table**

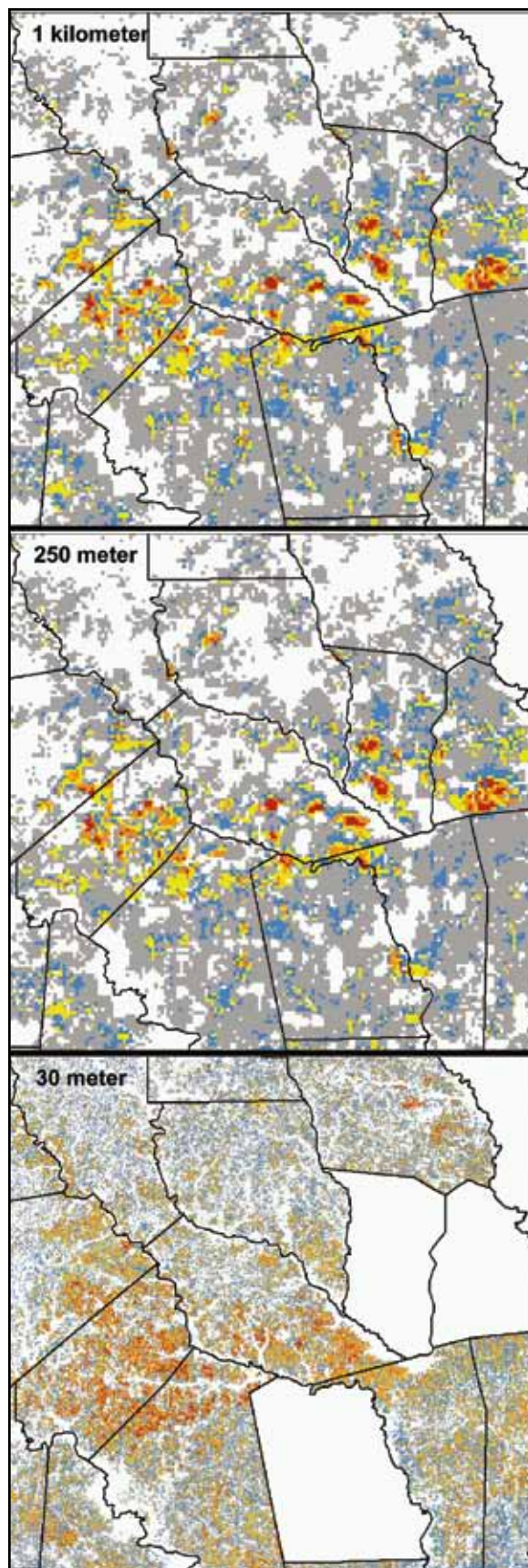
Define the weighted overlay table

Specify the Percent (%) Influence for each theme and a Scale Value for each input Field value. Scale Values will be multiplied by the % Influence value before they are added to other themes. To edit a % Influence value, click on it and type a new one. To edit a Scale Value, click on it, then use the dropdown list or type a value. Cells with a Restricted value are not added to other themes and retain the Restricted value in the output theme. To add a new input theme, click the Add Theme button. To delete an input theme, click on its name, then click the Delete Theme button.

Input Theme	% Inf	Input Field	Input Label	Scale Value
Aspen BA Rcl.	65	Value		
		1	0 - 1	Restricted
		2	1 - 20	0
		3	20 - 30	0
		4	30 - 40	1
		5	40 - 50	2
		6	50 - 60	3
		7	60 - 70	4
		8	70 - 80	5
		9	80 - 90	6
		10	90 - 100	7
		11	100 - 110	8
		12	110 - 120	9
		13	120 - 9999	10
		NODATA	No Data	Restricted
Aspen QMD Rcl.	22	Value		
		1	0 - 1	0
		2	1 - 2	1
		3	2 - 3	2
		4	3 - 4	3
		5	4 - 5	4
		6	5 - 6	5
		7	6 - 7	6
		8	7 - 8	7
		9	8 - 9	8
		10	9 - 10	9
		11	10 - 99	10
		NODATA	No Data	Restricted
Soil Dryness/Wetness Rcl.	13	Value		
		1	0 - 20	10
		2	20 - 23	9

Sum of influences (must equal 100%)

Figure 7—The ModelBuilder weighted overlay module provides a user friendly interface in which criteria can be combined in a multicriteria model. This module also provides a means of documenting model information and can be used to rerun a model using various weights and ranks.



Within this framework, forest health specialists are able to determine why an area is at risk, what the source data are, and how the model(s) for that region were constructed, thus documenting any models composing NIDRM. This type of framework also enables shortcomings in data and models to be identified and can be used to prioritize future research and data development.

3. Interactive and scalable. The framework is interactive enough to support sensitivity analysis while allowing risk assessments to be conducted at various spatial and temporal scales. Sensitivity analysis ensures that models can be adjusted according to local knowledge or as additional data and models become available or both. Scalability enables subject area experts to conduct local and regional assessments using an identical framework. This continuity ensures that national products do not conflict with local knowledge.
4. Efficacious. Efficiency, precision, accuracy, and usability must be considered when developing a framework. A national risk-map product with potentially hundreds of models behind it not only requires a highly efficient modeling process, it must be able to capture the information and variation within each individual model. With a wide range of audiences, including both subject area experts and private citizens, the risk map framework is able to produce detailed model documentation and results that are easy to interpret.
5. Comparable across geographic regions. The 2006 modeling framework has resulted in a standard modeling process that provides a level playing field for every region being examined as part of NIDRM. This ensures that regional comparisons can be made. Without standardization, NIDRM would be little more than a federation of maps with little or

Figure 8—Southern pine beetle risk models are shown in the following: each map depicts the potential for southern pine beetle mortality in a portion of east Texas. Red represents extreme risk, orange, high, yellow, medium, and blue, low.

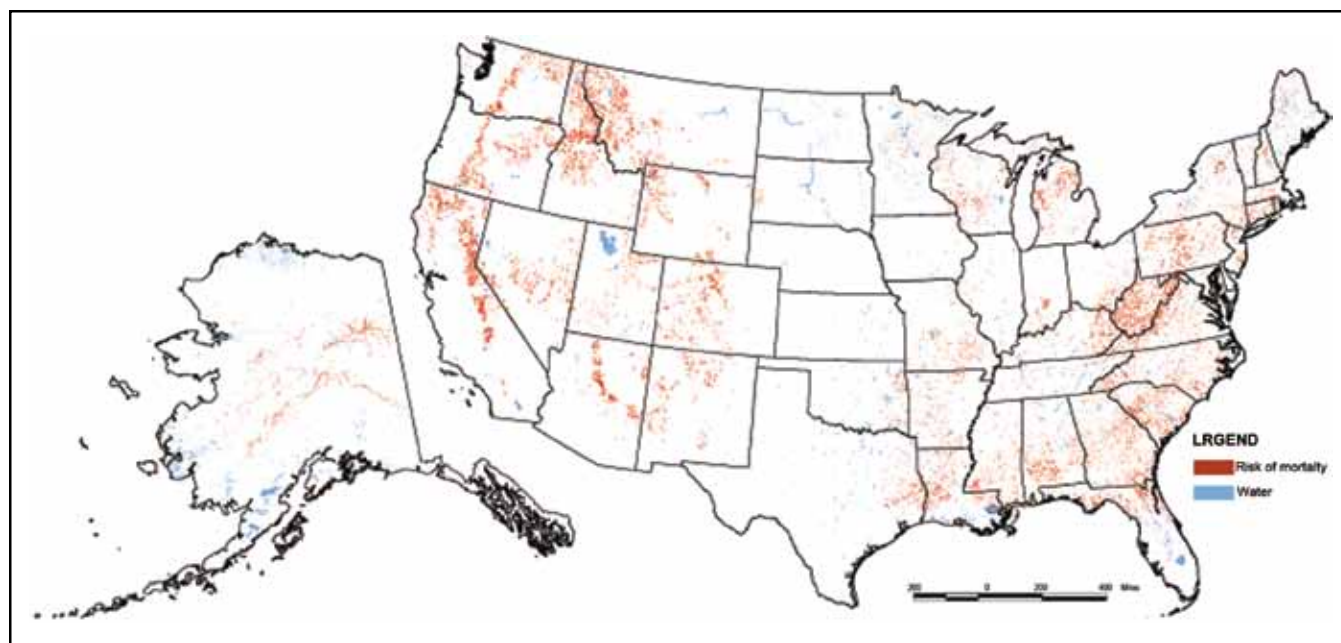


Figure 9—2006 National Insect and Disease Risk Map (NIDRM) for the United States characterizes the potential for significant losses (at or above 25 percent) of total forest basal area (BA), spatially.

no consistency between them, making regional comparisons and national summaries impossible.

Although the framework described in this paper was developed around modeling potential risk of tree mortality from insects and diseases, the process can be used for a wide range of other applications including estimating potential for wildlife and forest habitat (Krist 2001, 2005).

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Uncertainty Estimation for Map-Based Analyses

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Abstract

Traditionally, natural resource managers have asked the question, “How much?” and have received sample-based estimates of resource totals or means. Increasingly, however, the same managers are now asking the additional question, “Where?” and are expecting spatially explicit answers in the form of maps. Recent development of natural resource databases, access to satellite imagery, development of image classification techniques, and availability of geographic information systems has facilitated construction and analysis of the required maps. Unfortunately, methods for estimating the uncertainty associated with map-based analyses are generally not known, particularly when the analyses require combining maps. A variety of uncertainty methods is illustrated for map-based analyses motivated by the threat of the emerald ash borer (*Agrilus planipennis* Fairmaire) to the ash tree (*Fraxinus* spp.) resource in southeastern Michigan. The analyses focus on estimating the uncertainty in forest/nonforest maps constructed using forest inventory data and satellite imagery, ash tree distribution maps constructed using forest inventory data, and estimates of the total number of ash trees for a selected region obtained by intersecting the two maps. A crucial conclusion of the study is that spatial correlation, an often ignored component of uncertainty analyses, made the greatest contribution to uncertainty in the estimate of the total number of ash trees.

Keywords: Emerald ash borer, forest classification, forest inventory, logistic model, Monte Carlo simulation, spatial correlation.

Introduction

The emerald ash borer (*Agrilus planipennis* Fairmaire, Coleoptera: Buprestidae) (EAB) is a wood-boring beetle

native to Asia that was initially discovered in the United States in June 2002. It most likely entered the country in solid wood packing material such as crates and pallets and was transported to Detroit, Michigan, at least 10 years before it was discovered in 2002 (Cappaert and others 2005, Herms and others 2004). Ash trees are the only known host, and damage is the result of larval activity. Once eggs hatch, larvae bore into the cambium and begin to feed on and produce galleries in the phloem and outer sapwood. Larval feeding disrupts the translocation of water and nutrients and eventually girdles the tree. Tree mortality occurs within 1 to 3 years, depending on severity of the infestation (Haack and others 2002, McCullough and Katovich 2004). All of Michigan’s native ash species (*Fraxinus* spp.) and planted cultivars are susceptible (Cappaert and others 2005). Since 2002, southeastern Michigan has lost an estimated 15 million ash trees due to EAB (Cappaert and others 2005).

The natural rate of EAB dispersal is estimated to be less than 1 km per year in low-density sites. Natural dispersal has been enhanced by human transportation of infested firewood, ash logs, and nursery stock. This artificial spread of EAB has initiated the majority of outlier infestations (Cappaert and others 2005). Continued spread outside of the core zone increases the threat to ash across the United States.

The objective of the study was twofold: (1) to illustrate methods of uncertainty analysis, and (2) to investigate the contributions of spatial correlation to the uncertainty of areal estimates. Although the study was motivated by the need to predict the magnitude of potential loss of ash trees owing to the EAB, the biological context for this study was simply a medium to address the uncertainty objectives; no results from this study regarding the EAB or its effects should be construed as definitive.

Methods

The motivating problem for the study was to calculate an estimate, A_{total} , of the total number of ash trees in a region of southeastern Michigan (Figure 1) that is susceptible to infestation by the EAB. The estimation approach entails intersecting two 30-m by 30-m-resolution spatial layers or maps, one depicting the spatial distribution of forest land

and the other depicting the spatial distribution of ash trees on forest land. The technical objective was to estimate the uncertainty of A_{total} for a selected region. The layer depicting the distribution of forest land was derived from a forest probability layer constructed using forest inventory plot observations, Landsat Thematic Mapper (TM) satellite imagery, and a logistic regression model. The ash tree distribution layer was constructed using the same forest inventory plot observations and a spatial interpolation technique. Uncertainty in A_{total} was estimated using an analytical technique for estimating the covariances of the logistic regression model parameters, a sample-based technique for estimating the uncertainty in interpolated ash tree counts per hectare, and Monte Carlo techniques for generating forest/nonforest maps, and for combining the components of uncertainty.

Data

The study area is wholly contained in Landsat scene path 20, row 30 (Figure 1), for which three dates of Landsat TM/ETM+ imagery were obtained: May 2002, July 2003, and October 2000. Preliminary analyses indicated that normalized difference vegetation index (NDVI) (Rouse and others 1973) and the tassell cap (TC) transformations (brightness, greenness, and wetness) (Crist and Cicone 1984, Kauth and Thomas 1976) were superior to both the spectral band data and principal component transformations with respect to predicting forest attributes. Thus, 12 satellite image-based predictor variables were used—NDVI and the three TC transformations for each of the three image dates. Mapping units for all analyses consisted of the 30-m by 30-m TM pixels.

The Forest Inventory and Analysis (FIA) program of the USDA Forest Service has established field plot centers in permanent locations using a sampling design that is assumed to produce a random, equal probability sample (Bechtold and Patterson 2005, McRoberts and others 2005). The plot array has been divided into five nonoverlapping, interpenetrating panels, and measurement of all plots in one panel is completed before measurement of plots in the next panel is initiated. Panels in the study area are selected on a

5-year rotating basis. Over a complete measurement cycle, the Federal base sampling intensity is approximately one plot per 2400 ha. The State of Michigan provided additional funding to triple the sampling intensity to approximately one plot per 800 ha. In general, locations of forested or previously forested plots were determined using global positioning system receivers, whereas locations of nonforested plots were determined using aerial imagery and digitization methods. Each plot consists of four 7.31-m-radius circular subplots. The subplots are configured as a central subplot and three peripheral subplots with centers located at a distance of 36.58 m and azimuths of 0°, 120°, and 240° from the center of the central subplot. Data for 2,995 FIA plots or 11,980 subplots with centers in the selected TM scene and measured between 2000 and 2004 were available for the study. For each subplot, the proportion of the subplot area that qualified as forest land was determined from field crew observations. The FIA program requirements for forest land are at least 0.4 ha, with at least 10-percent stocking, at least 36.58 m crown-to-crown width, and forest land use. For each subplot, the number of observed ash trees with diameters at breast height of at least 12.7 cm was scaled to a count-per-hectare basis. The ash tree count per hectare for the i^{th} subplot is denoted A_i° where the superscript denotes a subplot observation and is distinguished from the ash tree count per hectare for the i^{th} pixel, which is denoted A_i .

The spatial configuration of the FIA subplots with centers separated by 36.58 m and the 30-m by 30-m spatial resolution of the TM/ETM+ imagery permits individual subplots to be associated with individual image pixels. The subplot area of 167.87 m² is an approximately 19-percent sample of the 900 m² pixel area.

Areal Estimation

The estimate, A_{total} , for a region was calculated in three steps: (1) generate a 30-m by 30-m-resolution forest/non-forest map, (2) construct a 30-m by 30-m ash tree count-per-hectare layer; for each pixel, estimate the number of ash trees as the product of the ash tree count per hectare and the 0.09-ha pixel area, and (3) estimate A_{total} as the sum over forest pixels from step 1 of pixel-level estimates of the number of ash trees from step 2. Thus, two layers are

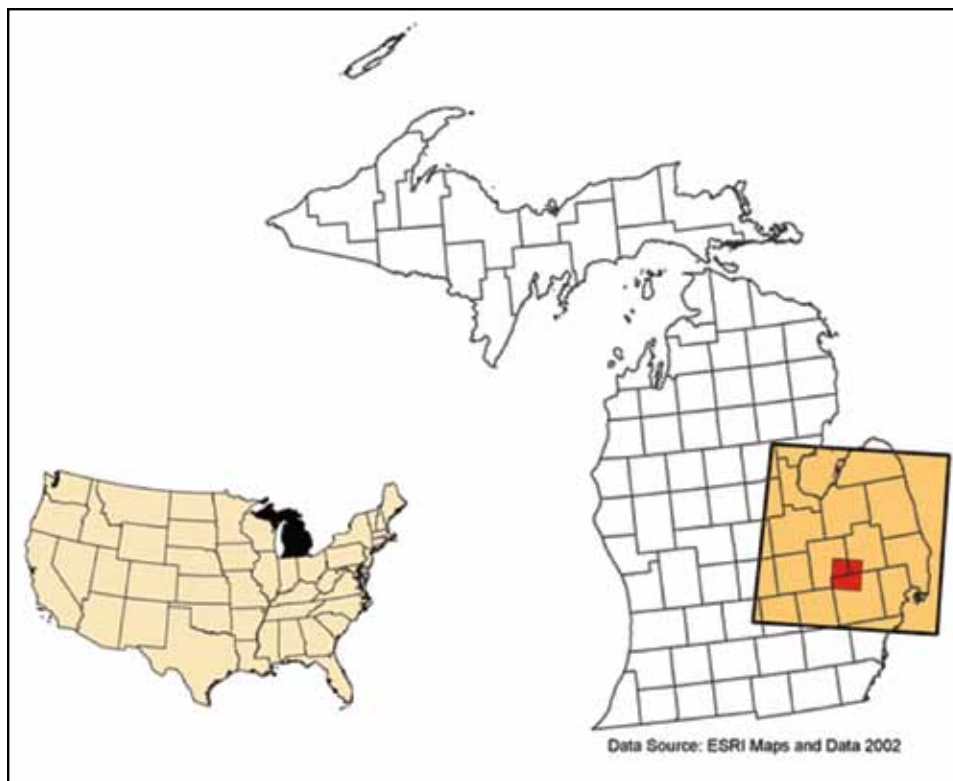


Figure 1—State of Michigan, U.S.A. with Landsat scene path 20, row 30 (large yellow box) and 30-km by 30-km study area (small red box). (Base map courtesy of ESRI)

required, a forest/non-forest layer and an ash tree count-per-hectare layer. Both layers were constructed specifically for this study so their pixel-level uncertainties would be known. Two sets of analyses were conducted. First, forest/nonforest, ash tree count per hectare, and the two associated uncertainty maps were constructed for a 30-km by 30-km study area in the selected TM scene (Figure 1). Second, uncertainty analyses for A_{total} were conducted for a smaller 2-km by 2-km portion of the larger study area (Figures 2 and 3). The restriction of the latter analyses to the smaller area was due to technological constraints as is noted in a later section.

Forest/Nonforest Layer

Because satellite image pixels with different ground covers often have similar spectral signatures, assignment of classes to individual pixels is often probability based. A layer depicting the probability of forest was constructed using a logistic regression model (McRoberts 2006),

$$E(p_i)^{j=1} = f(X_i; \beta) = \frac{\exp\left(\beta_0 + \sum_{j=1}^{12} \beta_j x_{ij}\right)}{1 + \exp\left(\beta_0 + \sum_{j=1}^{12} \beta_j x_{ij}\right)}, \quad (1)$$

where p_i is the probability of forest for the i^{th} pixel, X_i is the vector of the 12 spectral transformations for the i^{th} pixel with x_{ij} being the j^{th} element, the β s are parameters to be estimated, $\exp(\cdot)$ is the exponential function, and $E(\cdot)$ denotes statistical expectation. When estimating the parameters of (1), only data for the 7,920 completely forested or completely nonforested subplots were used. The covariance matrix for the vector of parameter estimates was estimated analytically as,

$$\text{Var}(\hat{\beta}) = (Z' V_{\epsilon}^{-1} Z)^{-1}, \quad (2)$$

where the elements of the matrix Z are defined as,

$$Z_{ij} = \frac{\partial f(x_i; \hat{\beta})}{\partial \beta_j},$$

the elements of V_e are defined as,

$$v_{ij} = \sqrt{\hat{p}_i(1 - \hat{p}_i)} \sqrt{\hat{p}_j(1 - \hat{p}_j)} \hat{\rho}_{ij},$$

and $\hat{\rho}_{ij}$ is the spatial correlation among the standardized residuals estimated using a variogram (Gumpertz and others 2000).

The most probable forest/nonforest classification of the imagery is constructed by comparing the probability, $\hat{p}$, from (1) for each pixel to 0.5: if $\hat{p} \geq 0.5$, the pixel is classified as forest and assigned a numerical value of 1; if $\hat{p} < 0.5$, the pixel is classified as nonforest and assigned a numerical value of 0. However, because the assignment of forest or nonforest to pixels is based on probabilities, there is uncertainty as to whether this procedure correctly assigns forest or nonforest to individual pixels. However, forest/nonforest realizations that reflect the uncertainty in the classification were obtained using a 4-step procedure designated procedure A:

- A1. Using the procedure of Kennedy and Gentle (1980: 228-231), generate a vector of random numbers from a multivariate Gaussian distribution with mean 0 and covariance $\hat{V}\hat{a}r(\hat{\beta})$ from (2); add these random numbers to the logistic regression model parameter estimates to obtain simulated parameter estimates.
- A2. Using the simulated parameter estimates from step A1 with (1), calculate a probability, $\hat{p}$, of forest for each pixel.
- A3. For each pixel, generate a random number, r , from a uniform [0, 1] distribution; if $r \leq \hat{p}$ the pixel is designated forest with a numerical value of 1; if $r > \hat{p}$ the pixel is designated non-forest with a numerical value of 0.
- A4. Repeat steps A1-A3 a large number of times and calculate the mean and variance of the numerical values assigned to each pixel.

The variance of the forest/nonforest classifications for each pixel is a measure of the uncertainty of the pixel's classification.

Ash Tree Distribution Layer

Because the ash tree distribution layer was for forest land, only data for the 1,953 FIA forest subplots were used in its

construction. Using these plot data, an empirical variogram was constructed, and an exponential variogram model was fit to the data (Zhang and Goodchild 2002). An interpolated surface was constructed for which the ash tree count per hectare, A_i , for the i^{th} pixel was estimated as,

$$\hat{A}_i = \sum_{j=1}^{1953} \frac{w_{ij}}{W_i} A_j^o \quad (3a)$$

where

$$w_{ij} = \begin{cases} \frac{1}{v_{ij}} - \frac{1}{\hat{\lambda}} & \text{if } \frac{v_{ij}}{\hat{\lambda}} \leq 0.95 \\ 0 & \text{if } \frac{v_{ij}}{\hat{\lambda}} > 0.95 \end{cases}, \quad (3b)$$

v_{ij} is the predicted covariance from the variogram model corresponding to the distance, d_{ij} , between the i^{th} pixel and the j^{th} plot; $\hat{\lambda}$ is the estimate of the variogram sill; and

$$W_i = \sum_{j=1}^{1953} w_{ij}, \quad (3c)$$

The variance of $\hat{A}_i$ was estimated as,

$$\hat{V}\hat{a}r(\hat{A}_i) = \sum_{j=1}^{1953} \left(\frac{w_{ij}}{W_i} \right)^2 (A_j^o - \hat{A}_i)^2, \quad (4)$$

and is a measure of the uncertainty of the estimate, $\hat{A}_i$. This approach to constructing the ash tree count per hectare was selected to illustrate a sample-based method for estimating variances. Other approaches such as kriging could have been used to construct the ash tree count-per-hectare layer. Realizations of the ash tree count-per-hectare distribution were obtained by selecting for each pixel a random number from a normal distribution with mean 0 and variance, $\hat{V}\hat{a}r(\hat{A}_i)$ from (4), and adding the number to $\hat{A}_i$ from (3a).

Uncertainty Estimation

Uncertainty in the areal estimate, A_{total} , is due to contributions from four sources: (1) uncertainty in the logistic regression model parameter estimates, (2) uncertainty in the pixel-level forest/nonforest classifications, given a set of parameter estimates, (3) uncertainty in the interpolated pixel-level ash tree count-per-hectare estimates, $\hat{A}$, and (4) spatial correlation in forest/nonforest and ash tree observations not accommodated in the logistic regression model predictions and the ash tree count-per-hectare interpolated estimates.

The spatial correlation contribution to uncertainty in A_{total} results from two phenomena. First, forest areas tend to be clustered rather than independently randomly distributed throughout the landscape. Thus, to mimic natural conditions, forest/nonforest realizations generated from the logistic regression model predictions of forest probability should exhibit clustering comparable to that observed among the FIA subplot observations of forest and nonforest. This feature requires that the random numbers used to generate the forest/nonforest realization in step A3 be drawn from a correlated uniform [0,1] distribution. Second, the errors obtained as the differences between $\hat{A}$ and $\hat{A}$ were expected to be spatially correlated; i.e., if an interpolation, $\hat{A}$, overestimates its true value, other interpolations in close spatial proximity would be expected to overestimate their true values also. However, for this investigation, the range of spatial correlation for the interpolation errors was only slightly more than the 30-m pixel width, was regarded as negligible, and was ignored.

To generate random numbers from an appropriately correlated uniform [0,1] distribution as required to accommodate spatial correlation, an 8-step procedure designated procedure B was used:

B1. Construct an empirical variogram,

$$\hat{\gamma} = \frac{1}{2|n(d)|} \sum_{(F_i, F_j) \in n(d)} (F_i - F_j)^2$$

where F is the numerical designation for forest or nonforest subplot observations, and $n(d)$ denotes a collection of pairs, (F_i, F_j) , whose Euclidean distances in geographic space are within a given neighborhood of d .

B2. Fit an exponential variogram,

$$\hat{\gamma}(d) = \hat{\alpha}_1 [1 - \exp(-\hat{\alpha}_2 d)] \quad (5)$$

to the empirical variogram from step B1, where the estimate of the range of spatial correlation is

$$\frac{1}{\ln(0.05)} ;$$

$$\hat{\alpha}_2$$

B3. Construct a spatial correlation matrix by assigning to each pixel pair, (i, j) , a correlation, ρ_{ij} , calculated as,

$$\rho_{ij} = \exp(\gamma d_{ij}) ,$$

where d_{ij} is the distance between the i^{th} and j^{th} pixel centers and, initially, $\gamma = \hat{\alpha}_2$ from step B2.

B4. Generate a vector of random numbers, one for each pixel, from a multivariate Gaussian distribution with the correlation structure constructed in step B3 using the technique described by Kennedy and Gentle (1980: 228–231).

B5. Convert the Gaussian random numbers from step B4 to Gaussian cumulative frequencies, resulting in a correlated uniform [0, 1] distribution.

B6. Generate a forest/nonforest realization using procedure A with the correlated uniform [0, 1] distribution from step B5.

B7. Construct an empirical variogram of the forest/nonforest realization from step B5; fit an exponential variogram model; and estimate the range of spatial correlation as in step B2.

B8. Repeat steps B3 through B7, adjusting the γ parameter in step B3 each iteration until the range of spatial correlation from step B7 is close to that obtained in step B2.

The exponential variogram model was used in step B2 because of its simplicity and the adequacy of the fit to the data. Construction of the multivariate Gaussian distribution in step B4 requires the Cholesky decomposition of a covariance matrix corresponding to the correlation matrix constructed in step B3. For a square region, n pixels on a side, the correlation matrix will be n^2 by n^2 . Thus, the 30-km by 30-km study area, which has 1,000 TM pixels on a side, would require decomposition of a 10^6 by 10^6 matrix. To accommodate personal computer space and processing limitations, analyses involving spatial correlations were constrained to a 2-km by 2-km region, which is approximately 67 pixels on a side, requires decomposition of a 4,489 by 4,489 matrix, and comprises approximately 400 ha (Figures 2 and 3).

Uncertainty in A_{total} for the 2-km by 2-km region was estimated using a 6-step Monte Carlo simulation procedure designated procedure C.

C1. Generate random numbers from a multivariate Gaussian distribution with mean 0 and variance matrix, $\text{Var}(\beta)$, from (2); add these random numbers to the logistic regression model parameter estimates to obtain simulated parameter estimates; calculate the probability, $\hat{p}$, of forest for each pixel using the simulated parameter estimates with (1).

C2. For each pixel, generate a random number, r , from a correlated uniform $[0,1]$ distribution using procedure B; if $r \leq \hat{p}$, designate the pixel as forest; if $r > \hat{p}$, designate the pixel as nonforest.

C3. Calculate the total forest area, F_{total} , as the product of the number of forest pixels from step C2 and the 0.09-ha pixel area.

C4. For each pixel, generate a random number from a normal distribution with mean 0 and variance, $\text{Var}(\hat{A}_{\text{total}})$ from (4); add the random number to the interpolated estimate of ash tree count per hectare, $\hat{A}$, to obtain a simulated ash tree count per hectare; multiply the simulated ash tree count per hectare and the 0.09-ha pixel area to obtain a simulated ash tree count for the pixel.

C5. Estimate A_{total} as the sum of the simulated ash tree counts from step C4 for forest pixels from step C2.

C6. Repeat steps C1 through C5 a large number of times; calculate the mean and variance of F_{total} and A_{total} over all repetitions; estimate the uncertainties in F_{total} and A_{total} as $\text{Var}(F_{\text{total}})$ and $\text{Var}(A_{\text{total}})$, respectively.

In procedure C, the contribution of uncertainty due to the logistic regression model parameter estimates may be excluded by not adding uncertainty in step C1; the contribution of uncertainty in the classification, given the parameter estimates, may be excluded by skipping step C2 and comparing the probabilities generated in step C1 to 0.5 using procedure A; the contribution of uncertainty due to spatial correlation may be excluded by generating an uncorrelated uniform $[0,1]$ distribution in step C2; and the contribution of uncertainty due to the interpolated ash tree counts per ha may be excluded by not adding uncertainty in step C4. The magnitude of the contributions of individual sources

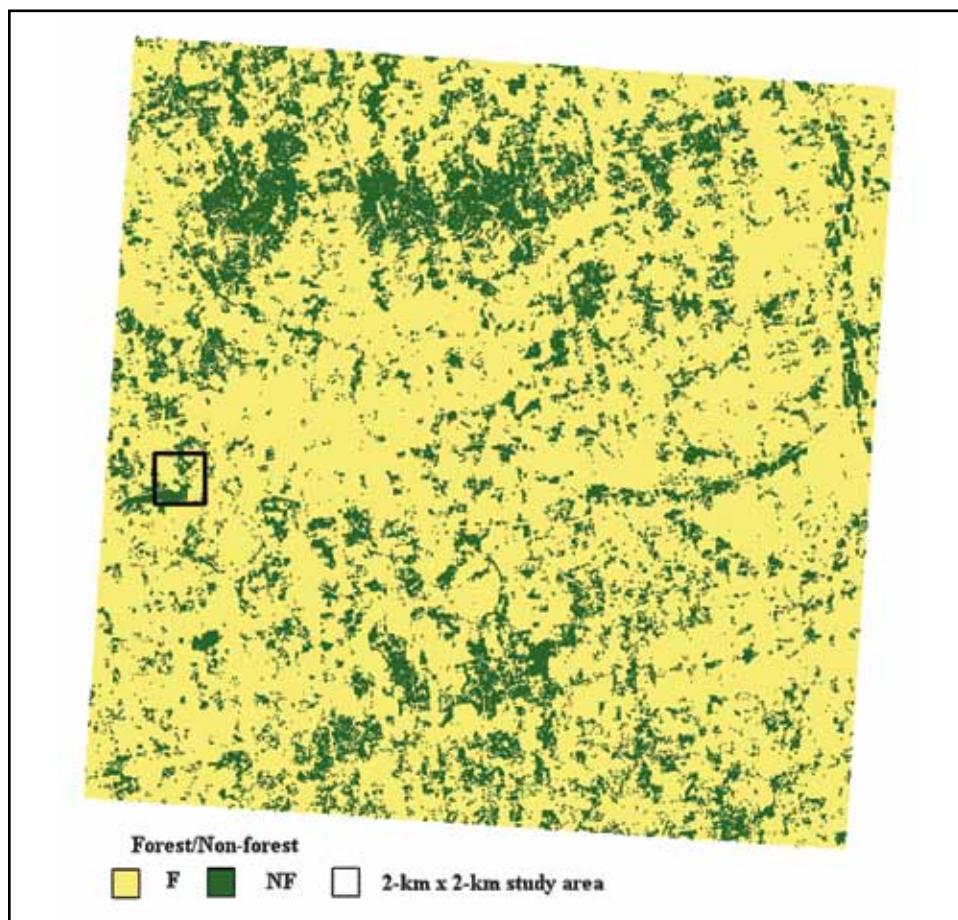


Figure 2a—Forest/nonforest classification for 30-km by 30-km study area.

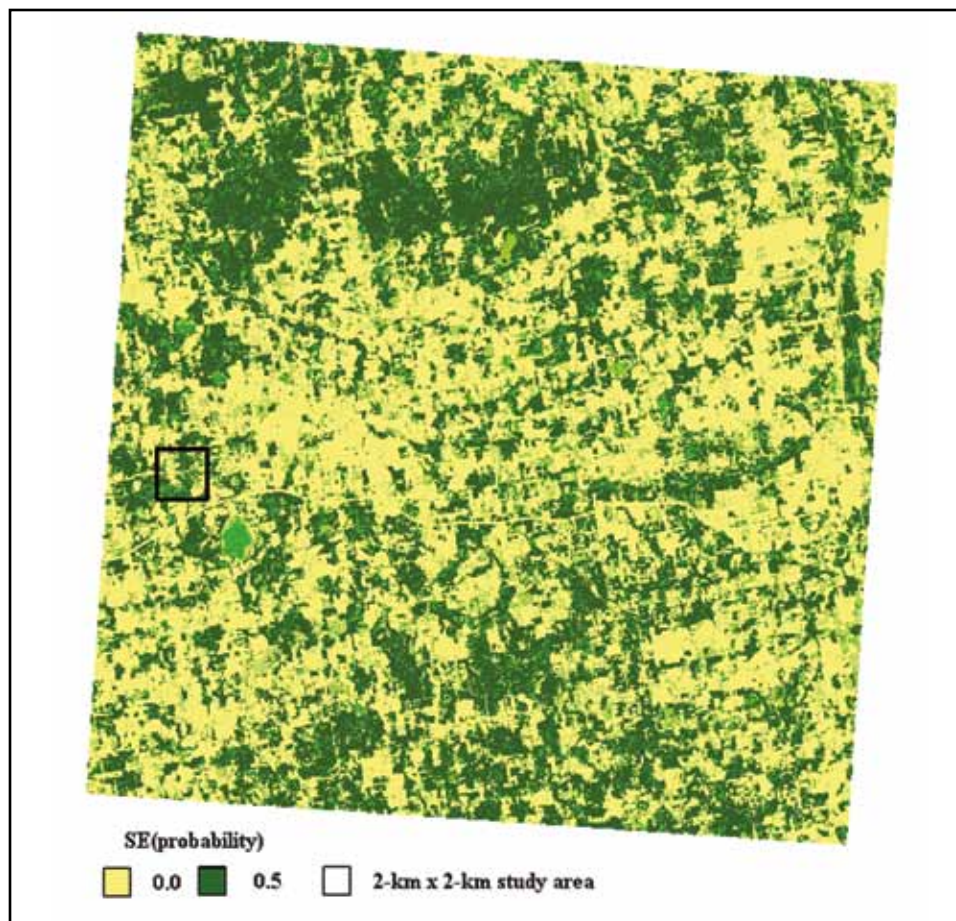


Figure 2b—Standard errors of pixel forest/nonforest classifications 30-km by 30-km study area.

of uncertainty may be estimated by considering $\hat{V}ar(F_{total})$ and $\hat{V}ar(A_{total})$ obtained by including contributions from all sources individually and in combinations.

Results

The analysis of Monte Carlo results indicated that all estimates stabilized to within less than 1 percent by 25,000 simulations. Therefore, 25,000 simulations were used when applying procedure C to estimate the contributions of the various sources of uncertainty.

The forest/nonforest maps constructed using logistic regression model predictions produced realistic spatial distributions, although no independent accuracy assessment was conducted (Figure 2a). However, considerable detail was revealed in the uncertainty map; e.g., the field structure, road networks, and highway on/off ramps (Figure 2b).

Considerably less detail was revealed in the ash tree counter-per-hectare map, but this result was expected because of the fewer FIA plots available and the more continuous nature of the layer (Figure 3a). As expected, the greatest uncertainty in the latter map occurred in the same locations as the greatest estimated values (Figure 3b). This phenomenon is common to biological analyses.

The estimates obtained using procedure C revealed in dramatic fashion that the source of uncertainty making the greatest contribution to uncertainties in the estimates of both F_{total} and A_{total} was spatial correlation in the realizations of the forest/nonforest maps (Table 1). The magnitude of this effect is highlighted by noting that when uncertainty from this source was included, 95-percent confidence intervals for both F_{total} and A_{total} included or were close to including 0. The contribution of the uncertainty in the

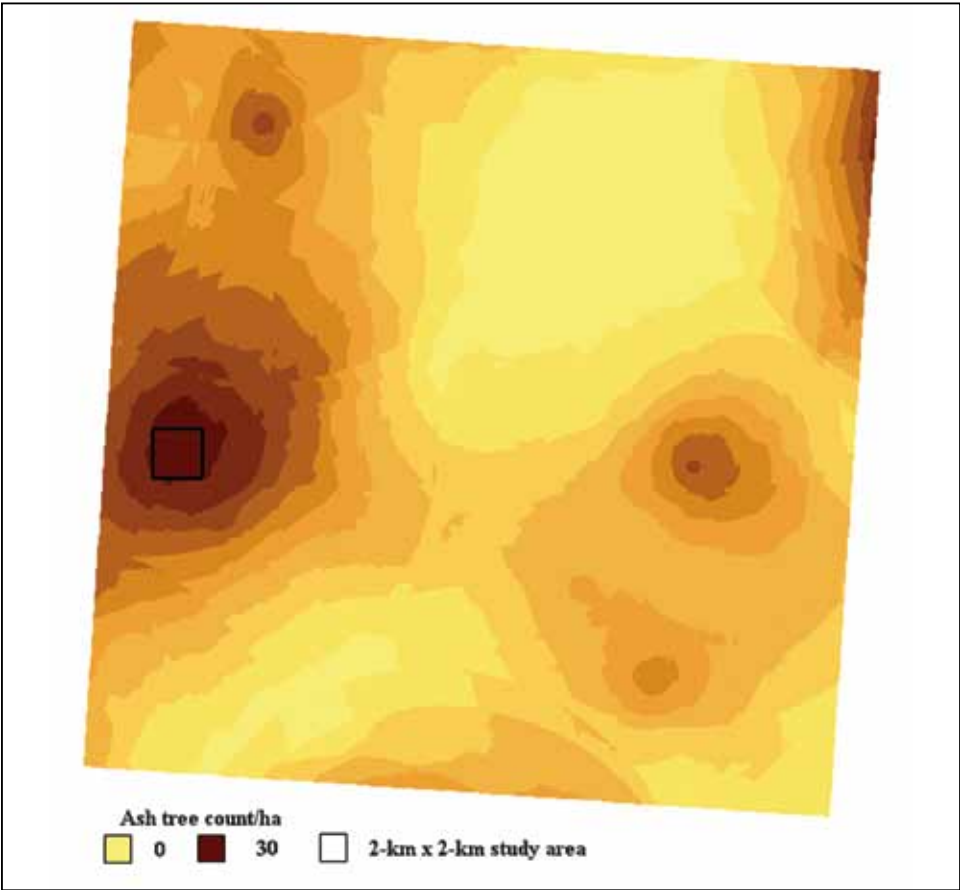


Figure 3a—Ash tree count/ha interpolations for 30-km x 30-km study area.

Table 1—Monte Carlo simulation estimates from procedure C

Source of uncertainty				Estimates			
Model parameter estimates		Spatial correlation	Ash tree count/ha interpolation	F _{total}		A _{total}	
	Classification			Mean	SE ^a	Mean	SE ^a
				-----ha-----		-----count-----	
X		113.40		2860.11			
	X			110.93	25.37	2801.15	643.23
	X			104.22	1.87	2637.11	47.61
X	X	X		104.04	55.32	2541.56	1180.34
	X	X		105.58	60.89	2633.00	1549.91
			X	113.40		2860.55	47.57
X	X		X	105.53	18.97	2670.79	484.46
X	X	X	X	106.25	61.92	2690.07	1576.47

^a SE is standard error and is calculated as the square root of the variance of the mean.

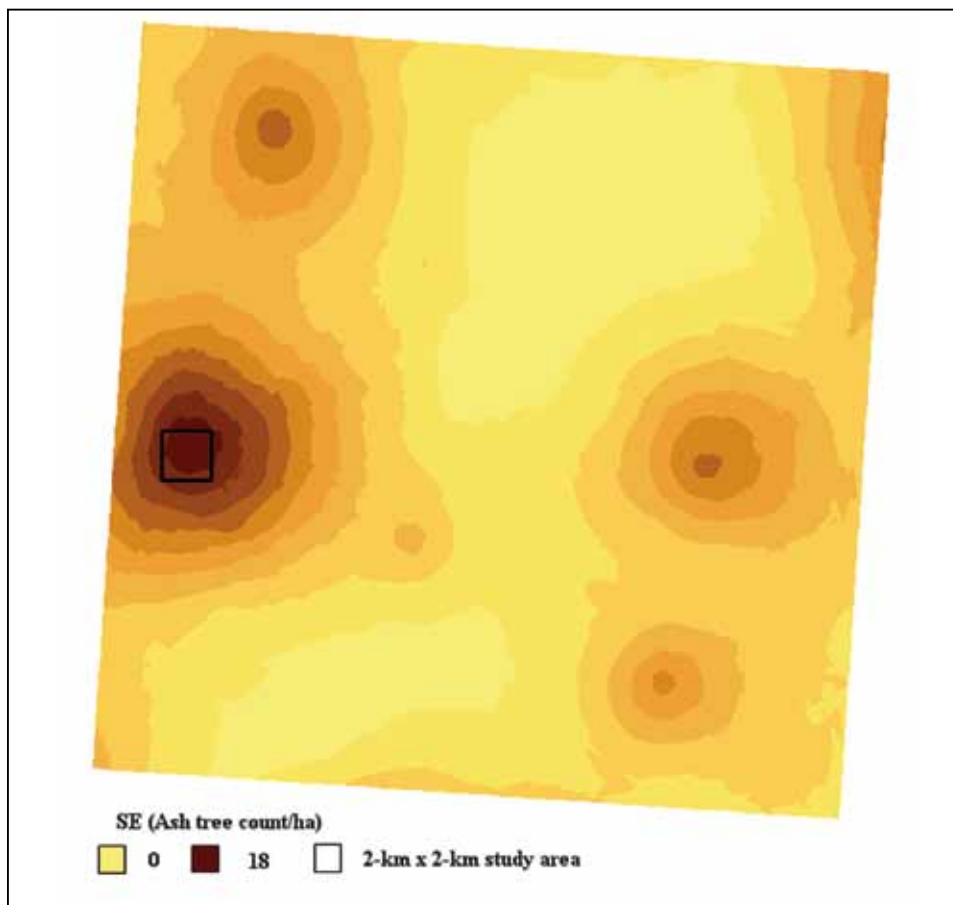


Figure 3b—Standard error of pixel ash tree count-per-hectare interpolations for 30-km by 30-km study area.

underlying ash tree count-per-hectare layer to $\text{Var}(\hat{A}_{\text{total}})$ was much less than was the contribution from the uncertainty in the forest/nonforest layer.

Conclusions

Three conclusions may be drawn from this study. First, spatial correlation is a crucial contributor to uncertainty in map analyses that aggregate results over multiple mapping units. Ignoring this contribution inevitably leads to underestimates of variances and unwarranted statistical confidence in estimates. Unfortunately, the importance of this source of uncertainty is generally not known to those who conduct map-based analyses, and techniques for estimating its effects are generally unfamiliar. The second conclusion is that greater attention in the literature and academic environments should be given to uncertainty estimation. Third,

estimation of uncertainty is not trivial, either conceptually or from a technical perspective. The necessity of decomposing very large matrices limits the size of regions that can be analyzed without high-speed computing facilities.

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An Aquatic Multiscale Assessment and Planning Framework Approach—Forest Plan Revision Case Study

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Abstract

The Aquatic Multiscale Assessment and Planning Framework is a Web-based decision-support tool developed to assist aquatic practitioners in managing fisheries and watershed information. This tool, or framework, was designed to assist resource assessments and planning efforts from the broad scale to the fine scale, to document procedures, and to link directly to relevant research. The framework is a hierarchical, hyperlinked template that is readily updateable. For aquatic resources in a planning area, such as those occupied by salmonid fishes, the framework produces tabular and spatial displays of (1) current habitat conditions and distributions, (2) desired future conditions, (3) risks and threats to the species concerned, (4) analysis approaches, (5) a conservation and restoration strategy, and (6) a monitoring, inventory, and research strategy. The framework also provides a logical system for developing, tracking, and documenting aquatic information at various spatial scales (subwatershed to basin) and can hyperlink management questions and data to best available science and procedures. For example, different analysis approaches (e.g., extinction risk matrices, influence diagrams, probabilistic networks) along with supporting science or case studies are directly linked and downloadable from the framework. The framework helps define and display information assumptions and gaps and is transparent and defensible.

Keywords: Aquatic population information, cutthroat trout, decision-support tool, multiscale assessments, multiscale planning, salmonid fish extinction risks, watershed information.

Introduction

The Aquatic Multiscale Assessment and Planning framework is a Web-based decision-support tool developed to facilitate conservation and restoration planning for aquatic species influenced by national forest management: (http://www.fs.fed.us/rm/boise/research/techtrans/projects/multiscale_home.shtml). This tool, or framework, was designed to support resource assessments and planning efforts from the broad scale to the fine scale, to document procedures, and to hyperlink directly to relevant research. The framework was originally developed in collaboration between U.S. Forest Service Northern (R1) and Intermountain (R4) Regions and the Rocky Mountain Research Station (Boise, ID) to help aquatic biologists organize data and prioritize management actions for the restoration of native trout populations. Though not specifically designed for national forest plan revision, the framework lends itself well to this purpose, and forest planners have used all or parts of the framework to craft various planning components (e.g., species viability assessments, desired conditions, aquatic conservation strategies) required under the 1982 Planning Rule or the new 2005 Planning Rule (36CFR219, <http://www.fs.fed.us/emc/nfma/index.htm>).

The framework provides a logical template for developing, tracking, and documenting aquatic population and watershed information. It summarizes data at various spatial scales, distinguishes quantitative from qualitative data, and is transparent and defensible. The framework hyperlinks data and management options to procedures, best available science, and case studies and spotlights assumptions made in the analysis process, as well as data gaps.

The framework template follows modern principles of conservation and restoration for aquatic ecosystems, addressing (1) ecological patterns and processes that contribute to persistence of aquatic ecosystems and species (Naiman and others 2000, Rieman and others 2006, Thurow and others 1997), (2) natural variability, ecosystem and species diversity, and population resilience and resistance

to disturbance (Angermeier 1997, Rieman and others 2006), (3) habitats needed for all life stages of aquatic species (McElhany and others 2000); (4) genetic diversity of populations (Allendorf and Leary 1988, Likens and Graham 1988, Rieman and others 1993, Soule 1987); and (5) habitat connectivity for dispersal and migration (Dunham and Rieman 1999, McElhany and others 2000, Rieman and Dunham 2000, Rieman and others 1993, Sheldon 1998).

The framework is designed to support U.S. Forest Service regional species status overviews, subbasin assessments, watershed analyses, cumulative effects assessments, National Environmental Policy Act analyses, and consultation. Spatially explicit outputs are used to define and display risks and threats associated with fish, fish habitats, and watershed conditions. Broad-scale summaries provide context for fine-scale projects to help prioritize management actions for addressing risks and threats. The transparent design helps step down data and priorities for field unit verification and implementation.

The framework provides a hierarchical approach for summarizing available fisheries information at various spatial and watershed scales. All the geographic scales of a drainage system function together to create and maintain aquatic habitats (Wissmar 1997). The subwatershed (6th field Hydrologic Unit Code [HUC], sensu Maxwell and others 1994) is often synonymous with local fish populations or their life stages, risks and threats, or project-level management action assessments or all. Subwatersheds that support self-sustaining populations, i.e., strongholds, act as sources for populations that bolster weaker populations or recolonize vacant habitats. Aquatic data used for national forest land management plans are usually summarized at the subwatershed scale.

In order to determine how habitat conditions are distributed across a larger geographic area, subwatershed information is aggregated up to the subbasin scale (4th field HUC). The subbasin is the primary broad-scale summary unit for addressing salmonid fish extinction risks. The subbasin acts as a terminal aquatic environment and is the spatial scale where metapopulations, or interacting groups

of two or more local populations, operate as a hedge against extinction (Lee and others 1997, Rieman and others 1993). Thus, a multiscale approach allows for broader interpretations of current conditions in terms of salmonid population dynamics. In addition, aggregating data up to larger scales such as the basin (3rd field HUC) provides context for sub-basin assessments and national forest plans.

The Six-Step Framework Template

The framework process is organized into six steps (Figure 1). For aquatic resources in the planning area, following these steps produces tabular and spatial displays of (1) current conditions and distributions of populations or habitats, (2) desired future conditions, (3) risks and threats to the species concerned, (4) analysis approaches, (5) a conservation and restoration strategy, and (6) a monitoring, inventory, and research strategy.

The steps are in a logical order of priority for completing habitat and population assessments and providing base data for strategic planning. Framework steps 1 through 3 can be worked on concurrently but should be completed prior to moving onto step 4. Information needed for each step is gleaned from numerous sources, including published research, broad-scale assessments, and conservation strategies (e.g., FEMAT 1993, Pacific Anadromous Fish Strategy [PACFISH] 1995, Inland Fish Strategy [INFISH] 1995, Lee and others 1997), and forest-level inventory and monitoring. Information used can be a combination of quantitative and qualitative data. For each step, the Web-based framework provides hyperlinks to data-collection protocols used, relevant research and case studies, assessment data, and planning products.

The following is a case study from the Bridger-Teton National Forest in Wyoming, which used the framework and its six steps to structure a conservation and restoration strategy for Bonneville cutthroat trout (*Oncorhynchus clarki utah*) as a part of the forest plan revision process. The forest used the framework to assemble data on the cutthroat, its habitat, stream-riparian ecosystems, and watersheds (Figure 1), with a focus on the Central Bear River subbasin.

Step 1—Current Condition, Status, and Distribution of Native Trout Populations and Associated Stream-Riparian Habitats

Step 1 (Figure 1) provides the environmental baseline or current condition for cutthroat trout in the Central Bear River subbasin and is assumed to reflect natural disturbances and the effectiveness of the Forest's current land use plan direction and past management actions.

Step 1(A) displays the current distribution of cutthroat trout populations across their assumed historical distribution in Central Bear River subbasin. This information was based on stream survey data collected by aquatic researchers as well as State, tribal, and forest biologists and compiled at the subwatershed level.

The status of the trout population in each subwatershed was coded as strong, depressed, or absent (includes extirpated) based on criteria developed for assessing interior Columbia River basin fish populations (Lee and others 1997). For example, subwatersheds were coded as having strong populations if the subwatershed had all the following conditions:

- Fish presence had been verified within the last 10 years using standard sampling methods.
- Major life history stages (e.g., resident, migratory) that historically occurred in the subwatershed were still present.
- Fish numbers were stable or increasing, and the population was no less than half of its historical size or density.
- The population or metapopulation was at least 5,000 individuals or 500 adults; if the population size was based on a population that extended outside the subwatershed, the subwatershed was an important core area for this larger population.

In addition, current conditions of stream-riparian habitats were evaluated for each subwatershed. Qualitative approximations of subwatershed conditions were obtained from Inland West Water Initiative (IWWI) assessments (on file with Rocky Mountain Research Station, Boise Aquatic Sciences Lab, Idaho Water Center, Suite 401, 322 East Front St., Boise, Idaho 83702) and plotted in step 1(B). The Bridger-Teton National Forest chose to display water quality

integrity, but other indicators of watershed condition could have been used.

Data for populations and subwatershed conditions were organized in standardized Excel format. The data were readily displayed and reviewed by creating geographic information system (GIS) spatial data layers both by subwatershed and subbasin and creating maps using ArcGIS® tools.

In forest plan revisions, information developed in step 1 could be incorporated into the comprehensive evaluation report (CER), a constituent of the planning set of documents, which provides current conditions and analyses for the Forest Plan.

Step 2—Desired Future Condition of Native Trout Populations and Associated Stream-Riparian Habitats

Desired future conditions for fish populations and watershed conditions were generated by forest biologists familiar with the subbasin. They used the Conservation Rules of Thumb summarized in earlier broad-scale aquatic assessments (FEMAT 1993, INFISH 1995, Lee and others 1997, PACFISH 1995):

- The greater the population size, the greater the chance of persistence.
- Population recovery potential is greater the closer you are to a source or strong population.
- Preserving genetic and phenotypic diversity requires maintaining populations throughout a wide geographic range in a variety of habitats.
- Project assessments and planning will have to address habitat disruption and population responses at both the local and regional scales.

In addition, the biologists considered key aquatic habitat characteristics, such as water quality/quantity, channel integrity, and riparian vegetation, and visualized those future states that would allow persistence and sustainability of aquatic populations. They recognized that ecologically healthy watersheds are maintained by natural disturbances (e.g., fire, landslides, and debris torrents, channel migration) that create spatial heterogeneity and temporal variability to the physical components of the system (Naiman 1992) and

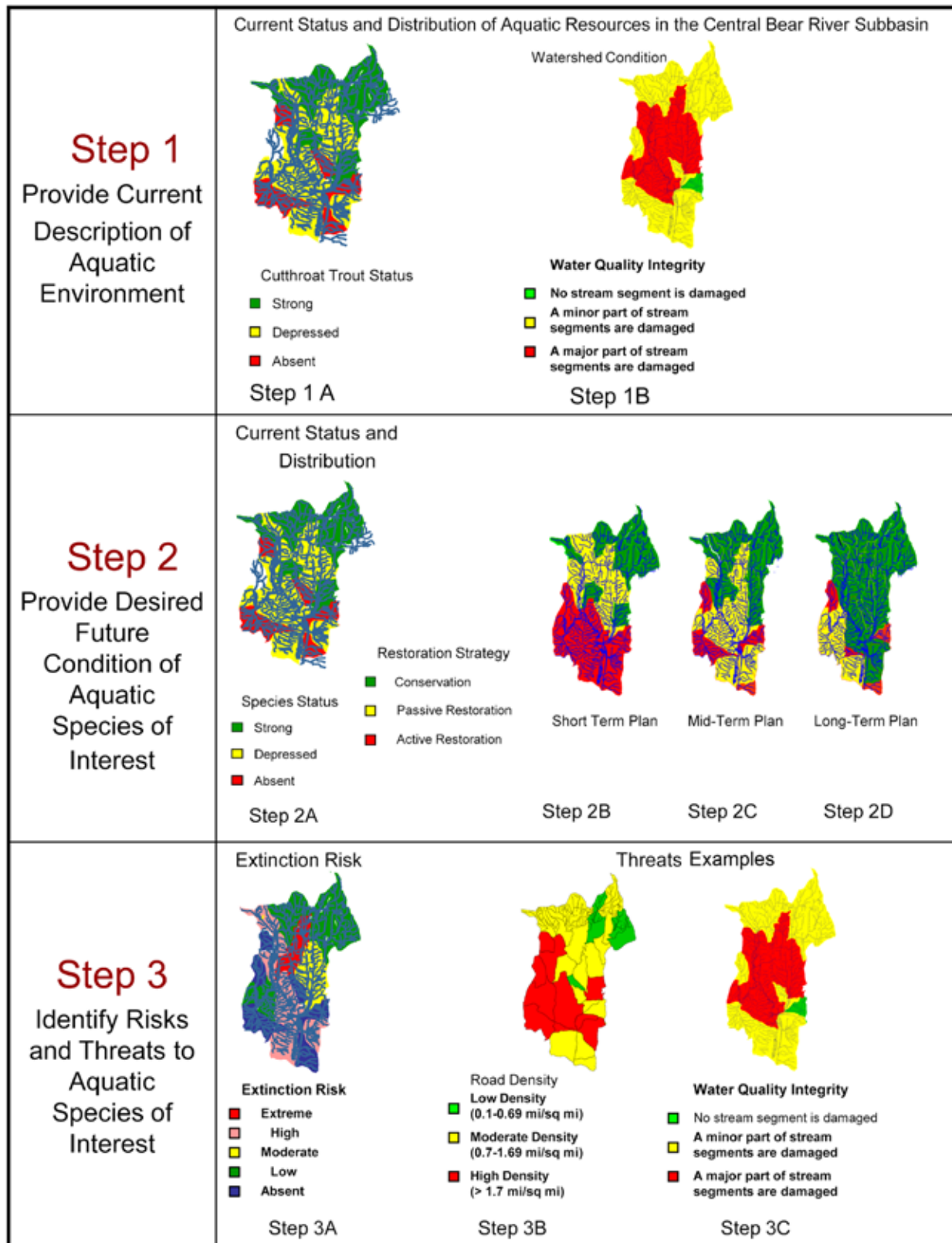
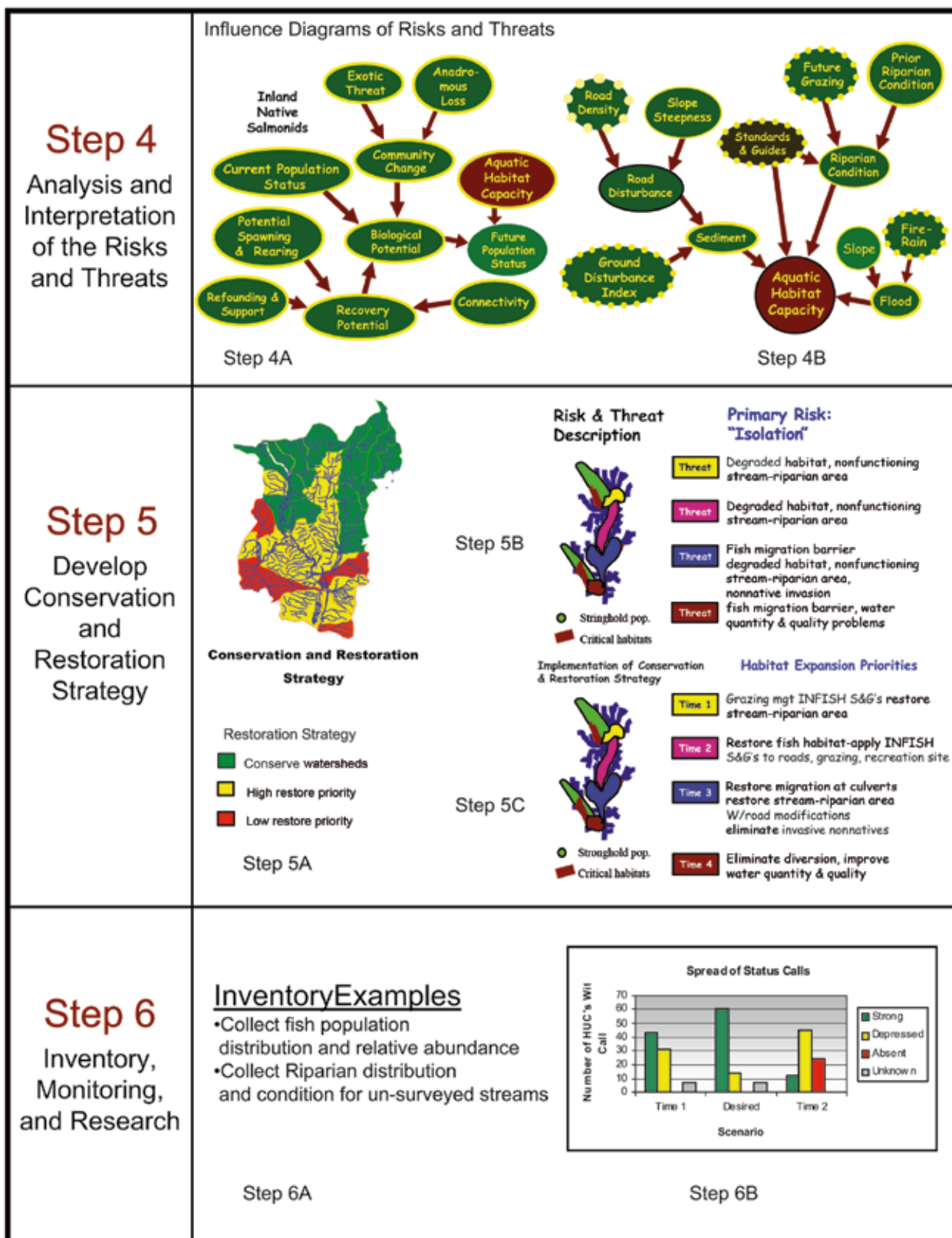


Figure 1—The multiscale framework steps and example products. The schematic displays example outputs specific to the individual steps (six) within the framework tied to the document step-by-step description.



that management consistent with natural variation should lead to more diverse, resilient, and productive biological systems (Rieman and others 2006).

Step 2 (B-D) displays desired future conditions over a period of time for cutthroat trout in the subbasin, showing fish population status and distribution at short-, mid- and long-term intervals. The long-term desired condition (Step 2(D)) would be expected to represent a healthy, self-sustaining metapopulation occupying ≥ 50 percent of its historical range along with strong population characteristics and high-quality habitats (FEMAT 1993, Lee and others 1997).

Outputs from Step 2 would be consistent with the desired condition component of the plan in the forest plan revision process, which is to describe how management activities will cause desired natural resource conditions to be obtained.

Step 3—Risks and Threats to Native Trout Populations and Associated Stream-Riparian Habitats

Extinction risks for salmonids are influenced by complex and interacting factors that are often difficult, if not impossible, to identify and measure. Despite this difficulty, understanding the nature of the extinction process and the characteristics of local fish populations and habitats can lead to management prescriptions that minimize risks. Relative extinction risks include deterministic, stochastic, and genetic factors at several spatial and temporal scales (Rieman and others 1993). The risk portion of step 3(A-C) was generated for cutthroat trout by agency biologists through application of an extinction risks matrix (Rieman and others 1993) that matched population characteristics, such as size, isolation, and survival rates, with a variety of environmental and demographic parameters. The outcome was an estimate of relative risk of extinction (low to extreme) for trout in a given reach, subwatershed, or across a subbasin. For the Central Bear River subbasin, step 3(A) maps extinction risks in subwatersheds based on trout population size and isolation.

The persistence of trout populations in a subwatershed can be threatened by land use practices such as grazing, fire suppression, mining operations, recreational activities, road

construction, dams and water diversions, and introduction of invasive species. Threats to fish populations or their habitat may be direct, as in the alteration of riparian and channel structure by overgrazing, or indirect, such as when high road density in the uplands impacts water quality and timing of flows. For the Central Bear River subbasin, biologists spatially displayed possible threats to cutthroat populations by mapping road densities (step 3(B)), and water quality integrity (step 3(C)). Aquatic specialists then evaluated the threats and identified those that Forest Service management or cooperative partnership actions could influence.

In the national forest plan revision process, risks and threats to native fish species on the subwatershed level would be included in the CER.

Step 4—Analysis and Interpretation of the Risks and Threats That Influence Native Trout Populations and Associated Stream-Riparian Habitats

In step 4, national forest biologists analyzed risks and threats to populations and their habitats at the subwatershed level, identifying those risks and threats most urgently to be considered and addressed by management. Influence diagrams, or cause-and-effect diagrams (step 4 (A, B)) were used to display the analysis thought process. These unambiguous diagrams helped biologists, managers, scientists, and interested publics visualize, discuss, and review all the factors and pathways that potentially impacted population and watershed condition. Forest biologists created an influence diagram for risks and threats to trout population persistence (step 4 (A)) and a separate influence diagram for risks and threats to habitat (step 4 (B)). The step 4 (A) diagram included elements such as migration barriers and presence of invasive species. To predict the future status of cutthroat trout populations in the subbasin, both influence diagrams were incorporated into a probabilistic network (in this case, a Bayesian belief network [Lee and Rieman 1997, Rieman and others 2001]) and run under different management scenarios. For example, the biologists found that if riparian road densities were high in a subwatershed where invasive species (brook trout *Salvelinus fontinalis*) were present, the future population status for cutthroat

was depressed. In this way, the forests could test different management alternatives and predict their impacts on trout population status at both the subwatershed and the subbasin scale (step 4(C)).

The framework facilitates risks and threats analysis by providing hyperlinks to potential analysis tools and current conservation literature. For example, links connect the user to the trout extinction risk matrix (Rieman and others 1993), effects of Federal land management alternatives on population trends (Rieman and others 2001), population viability assessment of salmonids using probabilistic networks (Lee and Rieman 1997), and conservation assessments for various fish species.

For forest plan revision, step 4 risks and threats analyses can be incorporated into the CER and thereafter provide background information for crafting the desired conditions and objective components of the plan. Step 4 is useful in the planning process in that it provides a transparent view of the spatial and temporal interaction of biological, physical, chemical, and social processes at work in the planning area. Implications of these interactions are used as the basis for development of the conservation and restoration strategies outlined in the next step of the framework, below.

Step 5—Conservation and Restoration Strategies to Address Risks and Threats to Native Fish Populations and Associated Stream-Riparian Habitats

In step 5, the forests used information gained in steps 1 through 4 to design a conservation and restoration strategy that could maintain and restore aquatic ecosystems at subwatershed through subbasin scales. A network of subwatersheds (green polygons in step 5(A)) was selected to serve as strongholds to conserve cutthroat trout or high-quality habitat or both. These conservation watersheds were intended to provide refugia across the landscape, areas where management focuses on protection and maintenance of quality aquatic habitat and strong native fish populations. Habitats that support the most productive, diverse, and otherwise critical populations provide the best opportunities for ensuring short-term persistence and provide an essential

nucleus for rehabilitating more complete networks in the future (Rieman and others 2000).

In addition, restored watersheds, with assigned priorities of low to high, were essential to delineate because protection of fish strongholds alone would not be sufficient to sustain populations (Franklin 1993). By restoring watershed processes that create and maintain habitats across stream networks, the long-term productivity of aquatic and riparian ecosystems can be ensured (Rieman and others 2000, Thurow and others 1997). Prioritizing the restoration in subwatersheds considers the extent of habitat degradation and how much natural diversity and ecological functioning still exists. For example, high-priority watersheds for restoration are those with limited loss of function and condition and that will likely serve as the next generation of strongholds for fish and water quality. Once the watersheds are prioritized for restoration, the type of restoration action implemented can be active (fence construction, bank stabilization, willow planting) or passive. Passive restoration relies on the implementation of guidelines, design criteria (i.e., Forest Service Manual and Handbook direction) and best management practices to maintain watershed processes and habitat conditions. Because passive restoration primarily maintains current conditions, active restoration is often needed to move a degraded system toward recovery.

Planning out the timing and sequence of restoration actions is aided by using displays generated by the framework. Step 5(B, C) illustrates the conservation and restoration strategy development process used by the Central Bear subbasin forests for a hypothetical watershed. This watershed has two subwatersheds with stronghold populations of cutthroat trout (step 5(A)). The primary risk to these two populations is isolation caused by threats such as degraded riparian areas and fish migration barriers. Through risk and threat analysis in step 4, the biologists determine the relative importance of the threats to population persistence and prioritize the timing and spatial sequence of restoration actions. The restoration strategy for the watershed can then be mapped (step 5(C)).

In the forest plan revision process, step 5 conservation and restoration strategies could be incorporated into various plan components. Conservation and restoration strategies

could appear in “Guidelines and Special Areas” within the forest plan components and could be used as a foundation of the monitoring program in the plan set of documents.

Step 6—Inventory, Monitoring, and Research Strategies

Inventory and research strategies fill data gaps and validate assumptions. If information required for the framework (e.g., population status, habitat condition) is incomplete or outdated, inventories or research may be needed to obtain the data (step 6(A)). Monitoring strategies are used to track the progress of conservation and restoration actions. Three types of monitoring generally apply to species assessments and forest planning: implementation, effectiveness, and validation. Implementation or compliance monitoring ensures that management actions were implemented as planned, such as moving livestock before utilization exceeds a critical level. Effectiveness monitoring assesses the progress of management actions in attainment of desired conditions. Validation monitoring corroborates the assumptions made during evaluation and analysis and is important for determining if restoration activities result in desired population and habitat conditions.

Forest biologists in the Upper Salmon subbasin chose the PACFISH/INFISH Biological Opinion Effectiveness Monitoring Program (Kershner and others 2004) to measure and evaluate the effectiveness of their actions and best management practices in restoring and maintaining the structure and function of trout habitats at multiple landscape scales. In addition, periodic fish sampling surveys in specific subwatersheds monitored population responses to restoration activities, and results were aggregated across the subbasin to ascertain trends toward or away from desired condition for the planning area (step 6(B)). In our example, step 6(B) shows a movement away from the desired condition (Time 1) toward a less desired condition (Time 2). Research needs that emerged from the framework process included developing a decision-support strategy for barrier removal and brook trout control.

For national forest plan revision, the framework’s inventory, monitoring, and research strategies would slip

seamlessly into the monitoring program portion of the planning set of documents.

Conclusions

The Aquatic Multiscale Assessment and Planning Framework, as a Web-based decision-support tool, will facilitate conservation and restoration planning for aquatic species and watersheds influenced by national forest management. Through a six-step process that organizes information into tabular and spatial (GIS) formats, the framework helps aquatic practitioners visualize and analyze complex fisheries and watershed data, from the site-specific project level up through national forest and regional spatial scales. The framework provides a one-stop shopping site for maintaining and updating data and analysis procedures.

For aquatic resources, such as cutthroat trout populations and their habitat in a particular planning area, the framework produces tabular and spatial displays of (1) current status and distribution of populations or habitats, (2) desired future conditions, (3) risks and threats to the species concerned, (4) analysis approaches, (5) a conservation and restoration strategy, and (6) a monitoring, inventory, and research strategy. The framework also provides a logical system for developing, tracking, and documenting aquatic information. It summarizes available information at various spatial scales (subwatershed to basin) and can hyperlink the user to best available science resources and supporting information. For example, different analysis approaches (e.g., extinction risk matrices, influence diagrams, probabilistic networks) along with scientific papers or case studies are directly linked and downloadable from the framework. The framework helps define and display information assumptions and gaps and is transparent and defensible.

This transparent and defensible design will assist in clarifying decision outcomes and the elements and pathways used to support them. The ability to clearly track the background data and thought processes that go into controversial land use planning and decision documents may reduce misunderstandings and mistrust in stakeholders and enable timely and effective restorative actions that will benefit fishes and other aquatic organisms.

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A Landscape-Scale Remote Sensing/GIS Tool to Assess Eastern Hemlock Vulnerability to Hemlock Woolly Adelgid-Induced Decline

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Abstract

The hemlock woolly adelgid (*Adelges tsugae* Annand) (HWA) is an invasive insect pest that is causing widespread mortality of eastern hemlock (*Tsuga canadensis* (L.) Carr.). However, some stands remain living more than a decade after infestation. The ability to target management efforts in locations where hemlock is most likely to tolerate prolonged HWA infestation is critical to successful integrated pest-management programs. Here, we build a landscape-scale hemlock risk model for the Catskills region of New York based on coverage like slope and aspect derived from a traditional Digital Elevation Model (DEM). We also show that additional data layers derived from hyperspectral sensors such as NASA's Airborne Visible InfraRed Imaging Spectrometer (AVIRIS) can provide critical information for geographic information system (GIS) modeling. The initial landscape-only model was able to predict the rate of overall decline following HWA infestation for 21 plots across the Northeast with $R^2 = 0.35$, $p = 0.027$. Adding foliar N concentration to our model improved results to $R^2 = 0.79$, $p = 0.0009$. An AVIRIS-derived hemlock abundance coverage was then used to define the hemlock resource and its relative vulnerability to rapid decline. These results indicate that the inclusion of both landscape and chemical variables is critical to predicting hemlock vulnerability to HWA, and that landscape-scale modeling in a GIS platform is possible with the addition of hyperspectral remote sensing coverages. Whereas the resulting risk map covers only the Catskills region of New York, the relationships established here should be applicable to HWA infestation across the

range of eastern hemlock, providing a basis for forest land management agencies to make informed management decisions.

Keywords: AVIRIS, *Adelges tsugae*, forest health, invasive insect pest, susceptibility.

Introduction

Insect pests and pathogens represent the largest and most pervasive agents of natural disturbance in North American forests, with potentially significant economic, aesthetic, and ecological consequences for northern forest ecosystems (Ayres and Lombardero 2000). In order for land managers to make successful management decisions for mitigation and treatment activities, they must know the location and extent of the host resource as well as the anticipated risk of host mortality. Traditionally, land managers have relied on plot-based field sampling efforts to supply this information. Although this is useful, a comprehensive, landscape-scale, spatially continuous coverage of the resource and its vulnerability is needed to fully assess the potential impacts on the forest resource and to devise successful management strategies.

Recently, remote sensing technologies have greatly increased the amount and quality of information that is available for landscape-scale ecological risk modeling. This information includes abundance maps for individual tree species, detailed forest decline assessments (including previsual symptoms), and foliar chemical concentrations (Foody 2002; Martin and Aber 1997; Ollinger and others 2002; Plourde and others 2007; Pontius and others 2005, 2006; Smith and others 2002). Such remote-sensing-based products, combined in a geographic information system (GIS) platform with traditional topographic-based data layers, have expanded the tools available for risk modeling. They provide the potential to greatly enhance our ability to create spatially continuous, landscape-scale models of ecosystem function and response to disturbance.

Hemlock woolly adelgid (*Adelges tsugae* Annand) is an invasive insect pest that is causing widespread mortality

of eastern hemlock (*Tsuga canadensis* (L.) Carr.). Current rates of spread into uninfested areas are estimated at 10 to 15 miles per year, and all indications are that HWA will penetrate the entire range of eastern hemlock (McClure 1995a). Because HWA may infest all hemlock stands eventually, susceptibility assessments (assessments of likelihood of infestation) are not necessarily informative for long-term, landscape-scale risk modeling. However, there is evidence of differing hemlock vulnerability (ability to tolerate prolonged HWA infestation). Many infested hemlock have shown minimal resistance to *A. tsugae* and little chance for recovery (McClure 1995b). However, some stands remain living more than a decade after infestation (Pontius and others 2006). Indeed, two adjacent hemlock stands can often respond very differently to attack, with differences commonly attributed to topographic characteristics such as landscape position, slope, and aspect (Bonneau and others 1997, Hunter 1993, Orwig and others 2002, Royle and Lathrop 1999). Greenhouse studies have shown that the presence of HWA alone did not cause the death of hemlock seedlings, and that it is the combined stress of drought and infestation that ultimately leads to mortality (Sivaramakrishnan and Berlyn 1999). In the field, reduced growth rates were associated with infested trees on ridgetop and southwestern facing sites, but not with those on well-watered sites (Sivaramakrishnan and Berlyn 1999). Pontius and others (2006) found that several site factors could be used to predict hemlock decline across the Northeast. The severest hemlock decline was associated with markedly low growing-season precipitation levels, southern and western exposures, and ridgetop/side-slope positions.

All of these landscape variables are in some way related to potential soil-moisture content, indicating that water availability may be an additional stressor, accelerating decline in the drought-sensitive eastern hemlock (Bonneau and others 1997, Orwig and Foster 1999, Orwig and others 2002, Young and others 1999). Orwig and others (2002) concluded that although the duration of infestation primarily controls the intensity of hemlock decline and mortality, stands on xeric aspects succumb most rapidly.

Although significant, these landscape variables typically explain only a small portion of the overall variation in hemlock decline. Adding foliar chemistry to site factors at the plot level, Pontius and others (2006) predicted an 11-class decline rating with 98 percent one-class tolerance accuracy on an independent validation set, indicating that foliar chemistry may also play an important role in HWA dynamics and hemlock decline. Herbivory is often positively correlated with foliar nitrogen concentrations, with low nitrogen-limiting insect populations (McClure 1980, Schowalter and others 1986, White 1984). Nitrogen can be particularly limiting to insects because there is a large difference between the nitrogen concentration of plants (around 2 percent dry weight) and that of insects (approaching 7 percent) (Dale 1988). This link between nitrogen and aphid success has been documented for a variety of host species (Carrow and Betts 1973, Douglas 1993, Koritsas and Garsed 1985, McClure 1980).

For relatively immobile insects such as HWA, the nutritive quality of forage becomes even more important. McClure (1991, 1992) found that N fertilization resulted in increased relative growth rate, survivorship, and fecundity of HWA, thus increasing hemlock vulnerability and reducing the effectiveness of implanted and injected pesticides. Regionally, Pontius and others (2006) also found that foliar N concentration was the strongest correlate with HWA infestation, with higher N consistently associated with higher HWA population levels.

Although useful in identifying key factors in the hemlock decline complex, such plot-level and greenhouse-based studies do little to assist land managers in making critical planning and treatment decisions for their forests. Here, we apply a plot-level empirical model from field-based observations of hemlock decline across the Northeast to a landscape-scale GIS model for hemlock woolly adelgid risk assessment. Because of the unique ecological niche occupied by hemlock stands, it is important to identify the stands that are most likely to tolerate HWA infestations so that hemlock can be preserved as a component of forest habitats in the region. At the same time, stands likely to suffer high rates of mortality can be evaluated for integrated pest management activities or conversion to other species.

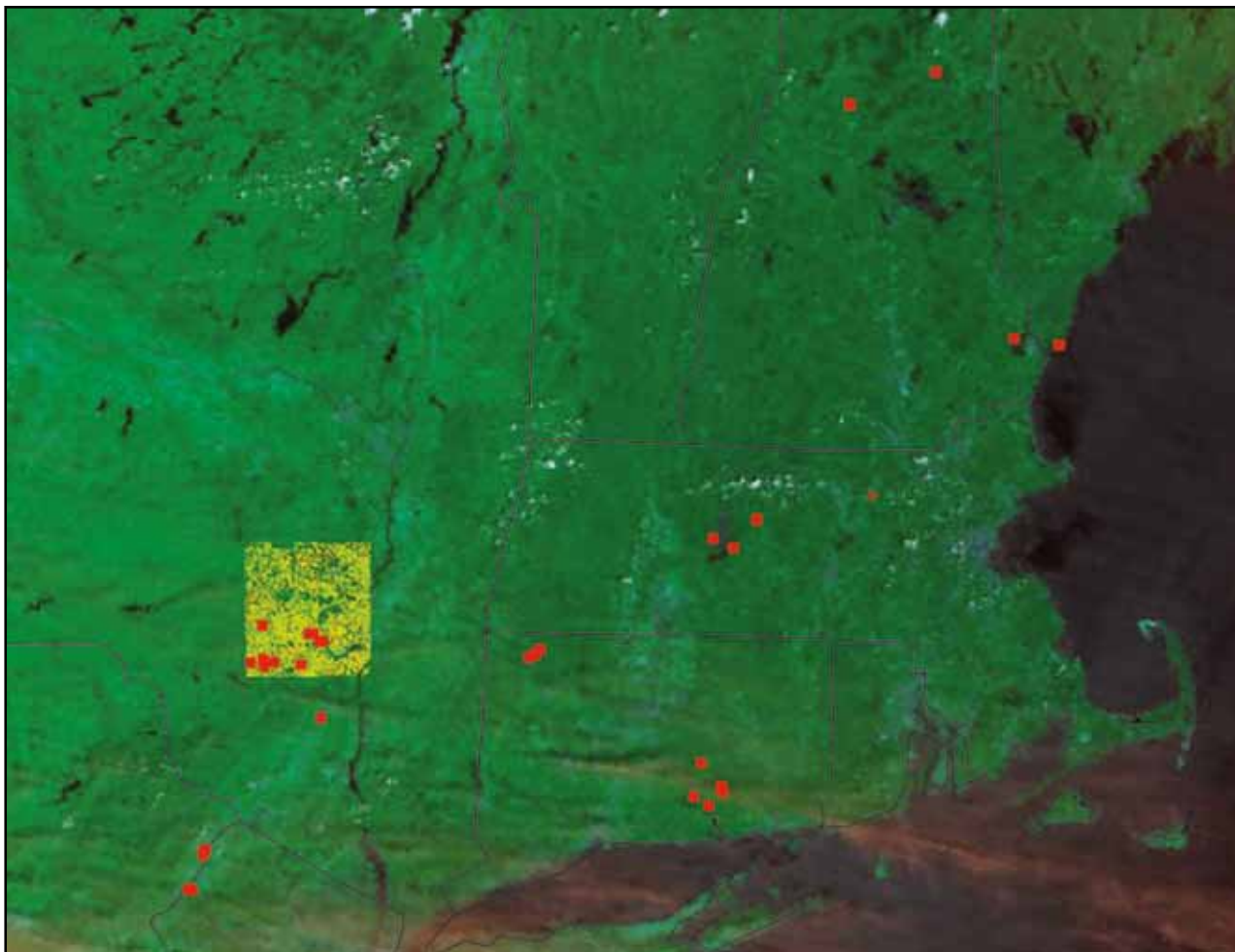


Figure 1—The study area spanned the full range of hemlock health and infestation histories over 11 states in the Northeast. Of the 48 total plots, 27 were concentrated in the Catskills region of New York, where hyperspectral coverages were available for GIS modeling.

Objectives

The goal of this work was to build a data-driven, empirical decline model for hemlock vulnerability to HWA that could be applied to a spatially continuous GIS model. Here we considered variables that were identified in previous research to be linked to HWA/hemlock decline and that were available in landscape-scale, wall-to-wall coverages. Using plot-level data from a regional set of infested hemlock stands, our specific objectives were to:

1. Develop and evaluate data-driven, quantitative linear models to predict the average rate of decline following HWA infestation based on (1) only landscape variables and (2) landscape variables plus foliar N concentrations.

2. Use the most effective model to incorporate key variables into a GIS model to map relative risk to hemlock on a landscape scale in the Catskills region of New York.

Methods

Plots in mature hemlock (where hemlock occupies >50 percent of the canopy) were established across a wide range of hemlock health, HWA infestation levels, site characteristics, and stand demographics. This included 48 sites characterizing the extremes of hemlock resistance and vulnerability to HWA from Pennsylvania to Maine (Figure 1). At each plot, a minimum of five hemlocks was sampled yearly between 2001 and 2005. Whereas statistical models were based on

Table 1—Decline ratings

Class	Overall decline status	New growth	Canopy transparency	Fine twig dieback	Live crown
-----Percent-----					
0	Perfect health	> 98	0 to 3	0	> 97
1	Very healthy	96 to 98	2 to 5	-	91 to 96
2	Healthy	94 to 96	4 to 7	5	78 to 90
3	Pre-visual decline	86 to 94	6 to 9		64 to 77
4	Decline first visible	69 to 86	8 to 12	10	52 to 63
5	Early decline	48 to 69	11 to 18	15	42 to 51
6	Moderate decline	23 to 48	17 to 26	20 to 30	30 to 41
7	Severe decline	4 to 23	25 to 40	35 to 45	20 to 29
8	Extremely unhealthy	2 to 4	39 to 64	50 to 60	10 to 19
9	Death imminent	1 to 2	63 to 67	65 to 80	1 to 10
10	Dead	0	> 67	85 to 100	0

Overall summary decline ratings were calculated by averaging the class assignment for each measured variable according to Table 1. These class assignments were then averaged to summarize overall decline status.

plot-level data, development of a landscape-scale GIS risk coverage was limited to the Catskills region of New York where hyperspectral imagery was available.

Field and Laboratory Methods

Each plot was sampled and evaluated yearly for a suite of decline symptoms, foliar chemistry, and HWA infestation levels. Site, stand, climate, and soil physical and chemical characteristics by genetic horizon were added for all plots by the final year of the study. Here we considered only those variables that would be available as digital landscape-scale coverages. This included slope, aspect, and landscape position derived from a digital elevation model (DEM). Additional data layers of foliar N (Martin and Aber 1997, Smith and others 2003), and percentage hemlock basal area (Pontius and others 2005) derived from hyperspectral sensors were also considered for inclusion in the second model.

Rather than rely on a nominal variable of vulnerable/tolerant for model development, a continuous overall decline value was calculated for all trees. This overall decline value was assessed using methods specifically designed to quantify the various, sequential symptoms that follow *A. tsugae* infestation. This included the percentage of terminal branchlets with new growth, percentage transparency (quantified using a concave spherical densiometer

(Pontius and others 2002), percentage fine twig dieback, and live crown ratio (USDA Forest Service 1997).

Because the goal was to calculate one summary variable to describe hemlock decline, we designed a method to normalize and then average all measured health variables into one value. Individual measurements for each variable were first normalized and rescaled to a 0 to 10 category value based on the quantile distribution cut-offs from a database of over 1,000 northeastern hemlock measurements (Table 1). These new normalized category values were then averaged to determine one summary decline rating that best described overall tree status (a continuous variable where 0 = perfect health, 10 = dead). These summary values were averaged over all trees for each plot yearly for the duration of the study.

The number of years since infestation was determined based on the first year that any HWA was witnessed on any hemlock within the plot. This is often first noticed at very low infestation levels (less than 5 percent infestation), typically well before the first year that HWA populations reach outbreak levels. In order to maximize accuracy of initial infestation dates, plots were selected from stands where monitoring and sampling have been conducted yearly by State, Federal, local, or private groups. We then calculated the change in overall plot-level decline from initial infestation to the current year to determine the dependent variable:

average yearly decline since infestation. This continuous output variable represents how rapidly health deteriorates and allows for flexibility in interpretation, depending on the needs of the user.

In addition to decline variables, the five canopy dominant hemlocks on each plot were sampled yearly between 2001 and 2005 for foliar chemical analyses. Needles were dried at 70 °C and ground to pass a 1-mm-mesh screen. A NIRSystems spectrophotometer was used to measure foliar nitrogen (N) concentrations (Bolster and others 1996). Dried and ground foliage was digested using a microwave-assisted, acid digestion procedure (US EPA 1995) and analyzed for calcium, potassium, magnesium, manganese, and phosphorus using an inductively coupled plasma spectrometer. Plot-level average chemistry was used from all years to compare to decline rates.

Slope and aspect were measured at plot center for each plot. Local physiography was assessed based on methods presented in Bailey and others (2004), where plots are assigned an ordinal classification based on landscape position as it relates to nutrient and moisture retention (streambeds and flats = 1, benches and toe slopes = 2, gentle midslopes = 3, moderate midslopes = 4, severe upper slopes = 5, and summit and shoulder positions = 6).

Predictive Model Calibration

Data from plots that are known to have been infested for at least 4 years were used to calibrate a linear, mixed stepwise model of average decline since infestation. In a mixed platform, forward and backward steps are enlisted. The most significant terms are entered first. Then any variables that become insignificant as the model becomes more complex are removed. It continues removing terms until the remaining terms are significant, when it changes back to the forward direction. Variables were retained under our mixed stepwise platform if the p-value was less than 0.1 and the variance inflation factor was below 2.0 (identifies potential autocorrelation between variables). The final model was then used to link key variables in a landscape-scale GIS model using ESRI® ArcGIS version 9.1.

GIS Data Layers Used for Mapping Risk in the Catskills

Hemlock vulnerability to HWA is complex and linked to multiple site, climatic, stand, and chemical factors (Pontius and others 2006). Here, we were limited to those variables for which raster-GIS coverages are available for inclusion in a landscape-scale risk assessment of the Catskills. This included the following.

Topographic Features—

Topographic variables such as slope, aspect, and landscape position can easily be derived from DEMs using the 3D Analyst available in ArcToolbox (ESRI® ArcMap v.9.1). Here we used a digital raster DEM with 10-m resolution from the National Elevation Data set (NED) assembled by the U.S. Geological Survey (USGS). NED is designed to provide national elevation data in a seamless form with a consistent datum, elevation unit, and projection for the conterminous United States. These can be downloaded at no charge from <http://seamless.usgs.gov>. The selected GIS-available topographic variables included aspect (calculated as the degrees from southwest), slope (degrees), and physiographic landscape position (classes of 1 to 6 representing the least to most xeric landscape positions following Bailey and others [2004]).

Foliar Chemistry—

The researchers involved in this project have used National Aeronautic and Space Administration's (NASA's) AVIRIS instrument to predict foliar concentrations of N, lignin, and cellulose in forested areas of New England (Martin and Aber 1997) with a high degree of accuracy. Using these same methods, we have developed a map of foliar N from 2001 AVIRIS imagery for the Catskills region of New York. Such coverages can be used to inform palatability or defensive chemical-based relationships related to risk assessment.

Hemlock Species Abundance Coverage—

Existing maps of hemlock abundance in the Catskills region were available from previous work (Pontius and others 2005). Using 2001 AVIRIS imagery, Mixture Tuned Matched Filtering in ENVI (v.4.2, ©Research Systems, Inc.

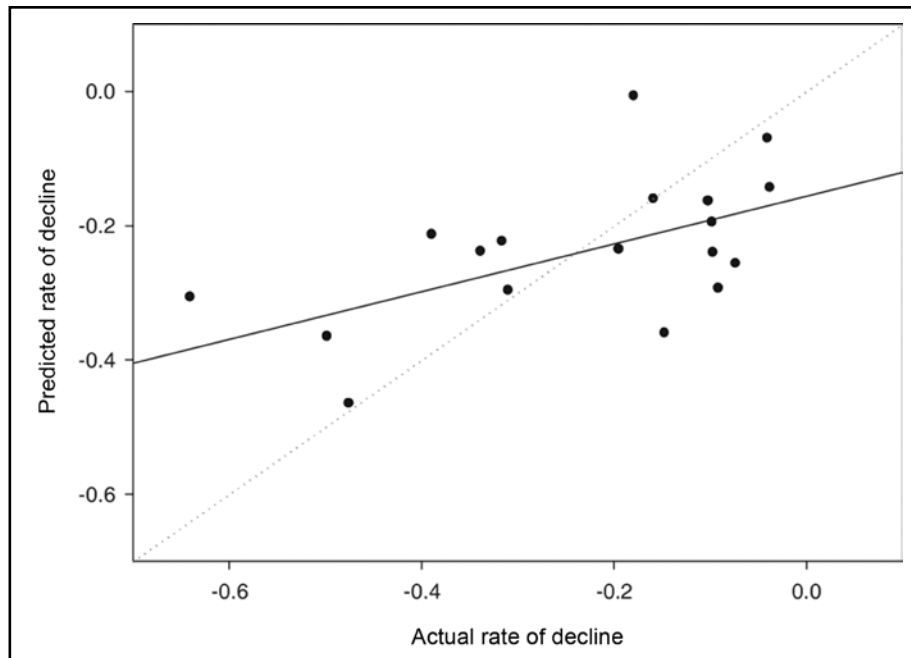


Figure 2—A stepwise linear regression model of landscape variables was only able to produce a predictive decline model that accounted for slightly more than a third of the overall variability witnessed in decline rates following infestation ($R^2 = 0.35$, $RMSE = 0.19$, $p = 0.027$). Variables included slope, aspect, and physiography, all discernable from digital elevation models in ArcToolbox.

2005) was used to unmix spectra and quantify the hemlock signature contribution to each pixel. The availability of this distribution coverage allowed us to isolate only those areas dominated by hemlock (greater than 40 percent hemlock basal area) for the final risk coverage.

Risk Coverage for the Catskills

The results of this model highlight which environmental variables are significant in determining hemlock decline rates and the nature of those relationships. Whereas the ultimate goal is to apply the actual quantitative predictive model based on modeled parameter estimates for each variable, final published estimates of foliar nitrogen for the region are still being finalized. To account for this inability to use direct parameter estimates for this version of the risk coverage, coverages for each of the significant predictive variables were scaled to a continuous value from 0 to 10 based on the nature of the modeled relationships. The Spatial Analyst available in ArcToolbox (ESRI® ArcMap v.9.1) was then used to add together the rescaled pixel values

from each significant variable, resulting in a risk map that highlights the cumulative effect of all significant variables in terms of relative hemlock vulnerability.

Results and Discussion

The final three-term predictive model included aspect and slope (Equation 1). This model accounted for a little over a third of the variability in decline rates for the 21-plot calibration set with a $p = 0.02$, $R^2 = 0.35$, $R^2_{adjusted} = 0.27$ and $RMSE = 0.19$ (Figure 2). A PRESS statistic for jack-knifed residuals of 1.04 indicates that, on average, if each plot was left out of the calibration and retained individually for validation, the average error would equal approximately 0.23 for the 0 to 10 scale.

$$\text{Decline\_rate} = -0.154 + (\text{aspect} \times 0.002) - (\text{slope} \times 0.118)$$

Equation 1. A landscape-variables-only model selected aspect (calculated as degrees from south) and slope (degrees) to predict the rate of decline expected following HWA infestation. This model accounted for 35 percent of the variability in the calibration set.

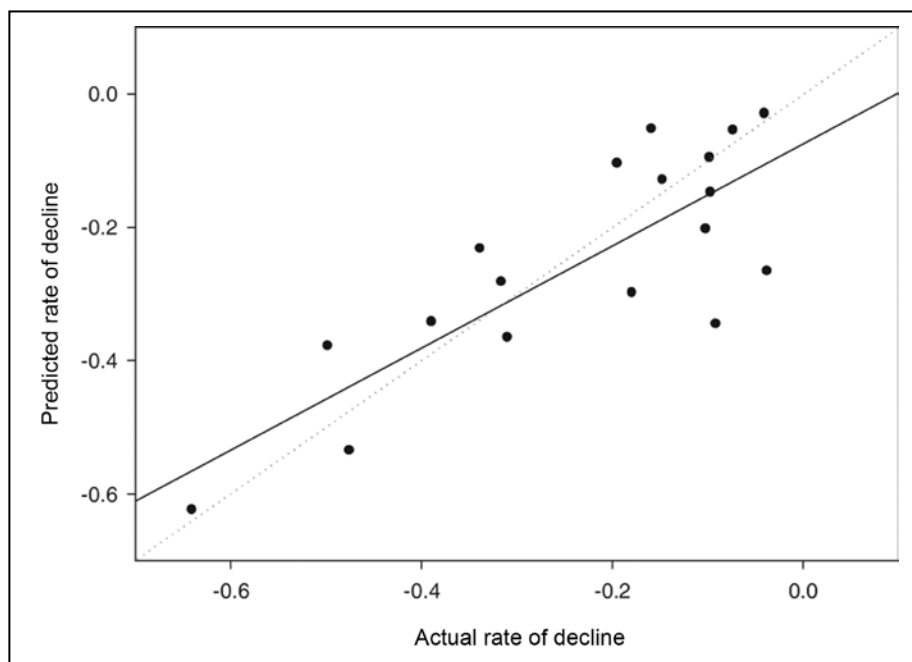


Figure 3—A stepwise linear regression model that added nitrogen as an option to the landscape-variable-only model was able to produce a predictive decline model that accounted for over two-thirds of the overall variability witnessed in decline rates following infestation ($R^2 = 0.79$, $RMSE = 0.13$, $p = 0.0009$). Nitrogen coverages were possible as additions to the GIS model from 2001 AVIRIS-derived hyperspectral coverages of the Catskills region.

When we added foliar nitrogen concentrations to the mixed stepwise linear regression, model accuracy improved significantly. The resulting model based on slope, aspect, physiography, and foliar nitrogen concentration produced a $p = 0.0009$, $R^2 = 0.79$, $R^2_{adjusted} = 0.69$, and $RMSE = 0.13$ (Figure 3). Jack-knifed residuals resulted in a $PRESS$ statistic of 0.55, or an average error of approximately 0.16 on the 0 to 10 decline rating scale.

$$\begin{aligned} \text{Decline\_rate} = & 0.643 - (\text{aspect} \times 0.0003) - (\text{slope} \times 0.158) + \\ & (\text{physiography} \times 0.049) - (\text{foliarN} \times 0.425) + \\ & ([\text{aspect} \times \text{slope}] \times 0.001) + ([\text{aspect} \times \text{foliarN}] \times 0.022) \end{aligned}$$

Equation 2. The full GIS model again selected aspect (calculated as degrees from southwest) and slope (degrees) to predict the rate of decline expected following HWA infestation, with the addition of physiographic position and foliar nitrogen concentration. This model accounted for 79 percent of the variability in the calibration set.

Using this final model based on both landscape variables and foliar N concentrations, we combined coverages of key variables to create a map of relative hemlock

vulnerability to decline following HWA infestation for the Catskills region of New York (Figure 4). The availability of a hemlock distribution coverage from previous work (Pontius and others 2005) allows us to isolate only those areas dominated by hemlock (greater than 40 percent hemlock basal area) for the final risk coverage (Figure 5). The resulting coverage of hemlock and its relative vulnerability to infestation should aid land managers in targeting management activities in the region.

Although these quantitative models were statistically significant, we wanted to ensure that there was a theoretical basis for why these variables might exert influence on hemlock decline rates. The inclusion of landscape characteristics in our risk models has a strong theoretical basis in the literature. Similar to previous HWA research discussed in the introduction, we found that stands with a demonstrated resistance to long-term HWA infestation typically occupy lower physiographic positions, such as streambeds, flats, and toe-slopes ($p = 0.0076$, Figure 6). In addition to physiography, resistant stands were consistently found on

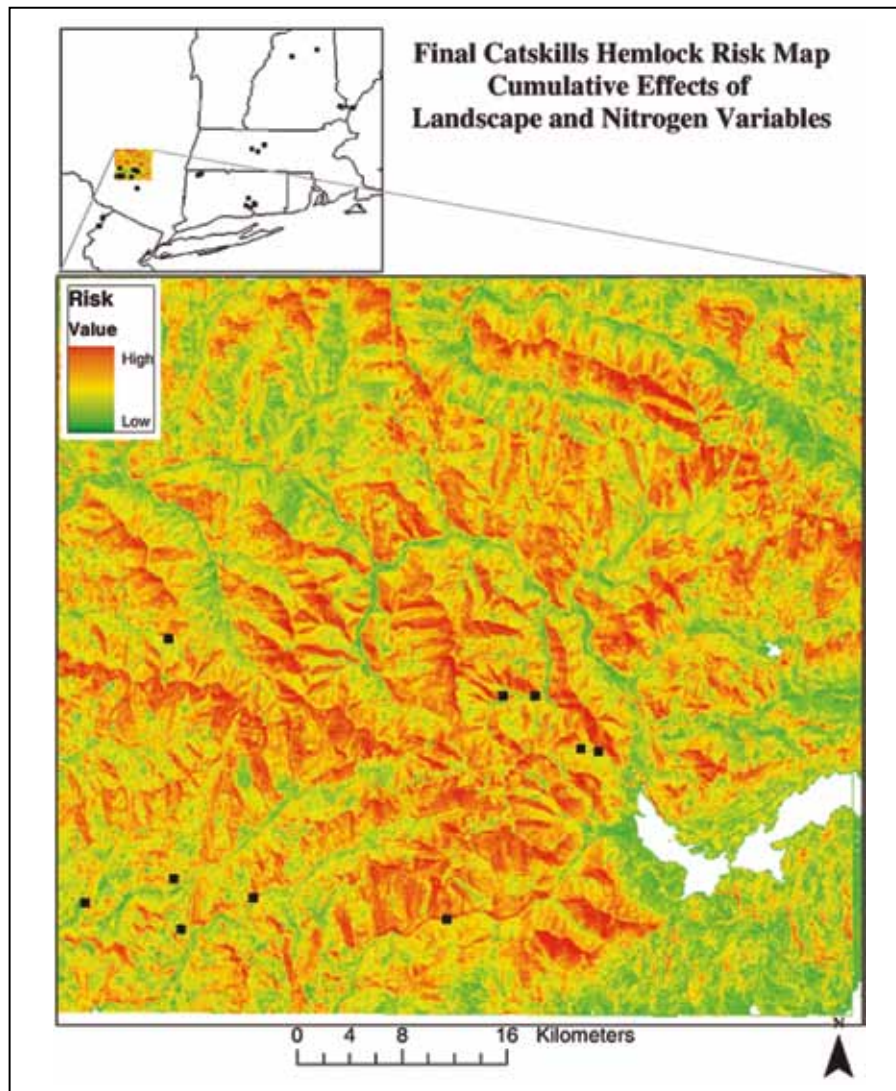


Figure 4—A spatially continuous, comprehensive map of relative hemlock vulnerability to HWA-induced decline in the Catskills, New York, summarizes the cumulative effect of key variables identified in the statistical modeling.

less steep terrain than susceptible stands across our calibration data set ($p = 0.008$, Figure 7). A weak yet significant correlation between aspect (in degrees from southwestern exposure) and the rate of decline ($r = 0.24$, $p = 0.04$) was also seen, with more rapid decline on southern facing exposures ($p = 0.012$, Figure 8). Significant interactions between aspect/slope ($p = 0.06$) and aspect/nitrogen ($p = 0.002$) indicate that aspect may be more significant when other stressors (such as steeper slopes or higher nitrogen concentrations) are involved.

The existing literature suggests that inherently low N concentration may limit HWA success, which, in turn, may impart some measure of resistance for host trees. Under low nitrogen conditions, concentrations may not be sufficient to maintain viable HWA populations. The data presented here support this “palatability-based” relationship between nitrogen and decline rates (Figure 9). The strongest correlate with hemlock decline rates across the region was the percentage infestation ($r = -0.67$, $p = 0.008$), with higher infestation levels associated with more rapid decline rates. In turn, the strongest correlate with HWA infestation

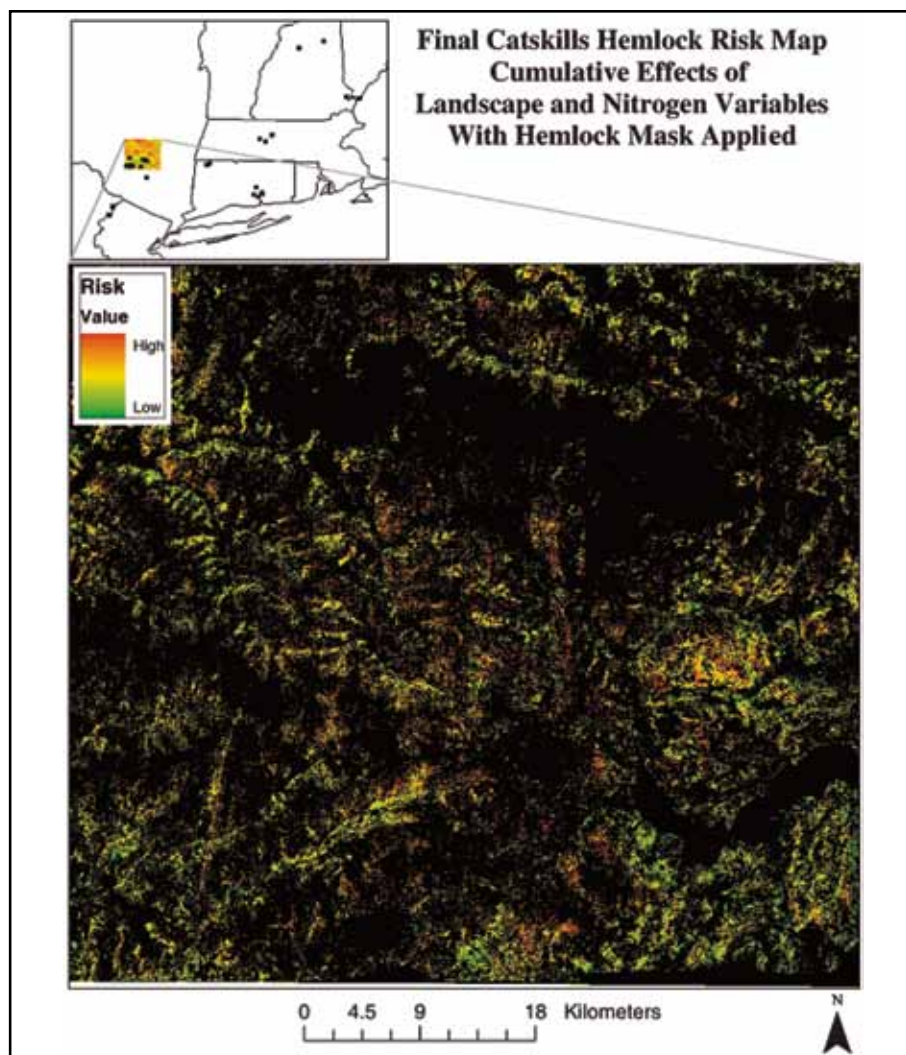


Figure 5—Masking nonhemlock pixels using an AVIRIS-derived coverage of hemlock basal area percentage produces a hemlock only risk map. This allows land managers to identify hemlock-dominated stands and their vulnerability to HWA to direct management efforts.

levels was foliar nitrogen concentration ($r = 0.396$, $p = 0.005$), with higher infestation levels associated with higher nitrogen concentrations. This may explain the significant relationship between hemlock decline rates and foliar nitrogen concentrations ($r = -0.46$, $p = 0.004$). Higher nitrogen levels support a larger, more successful adelgid population, which is able to deplete hemlock of photosynthate more rapidly, leading to more rapid decline.

Whereas these statistics and jack-knifed residuals suggest that this final landscape and foliar nitrogen model is robust enough to apply to new input data, independent validation provides a better assessment of model accuracy.

As a preliminary test, this model was applied to all regional plots, regardless of infestation history, resulting in an $R^2 = 0.51$ and $RMSE = 0.142$ (Figure 10). This reduction in model accuracy when newly infested plots are added is most likely due to a nonlinear decline response over the duration of infestation. Newly infested trees may decline only slightly in the first or second years because photosynthate reserves are available for injury response, defensive reaction, and continued productivity. Once these reserves have been reduced, decline becomes much more rapid. To validate this model, we will continue to track hemlock decline in the remaining plots over the next several years.

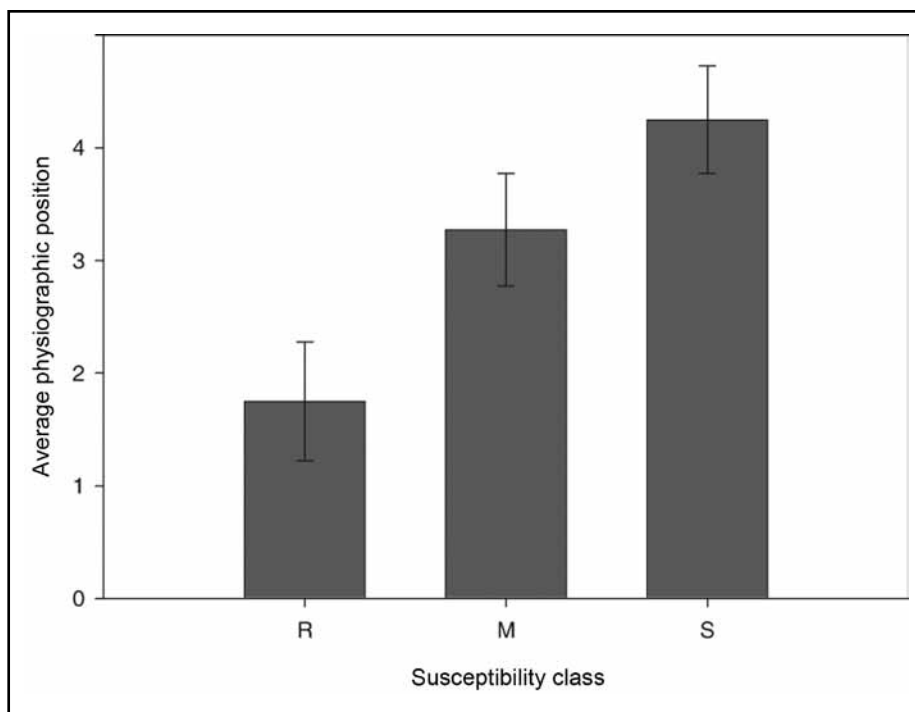


Figure 6—Resistant hemlock stands (R) were consistently found in lower landscape positions such as streambeds, flats, and toe-slopes. Susceptible stands (S) were consistently located on more xeric landscape positions ($p = 0.0076$). Plots with average rates of decline (M) were generally located on midslopes. Physiographic landscape position (class 1 to 6 representing the least to most) is xeric landscape position following Bailey et al. (2004).

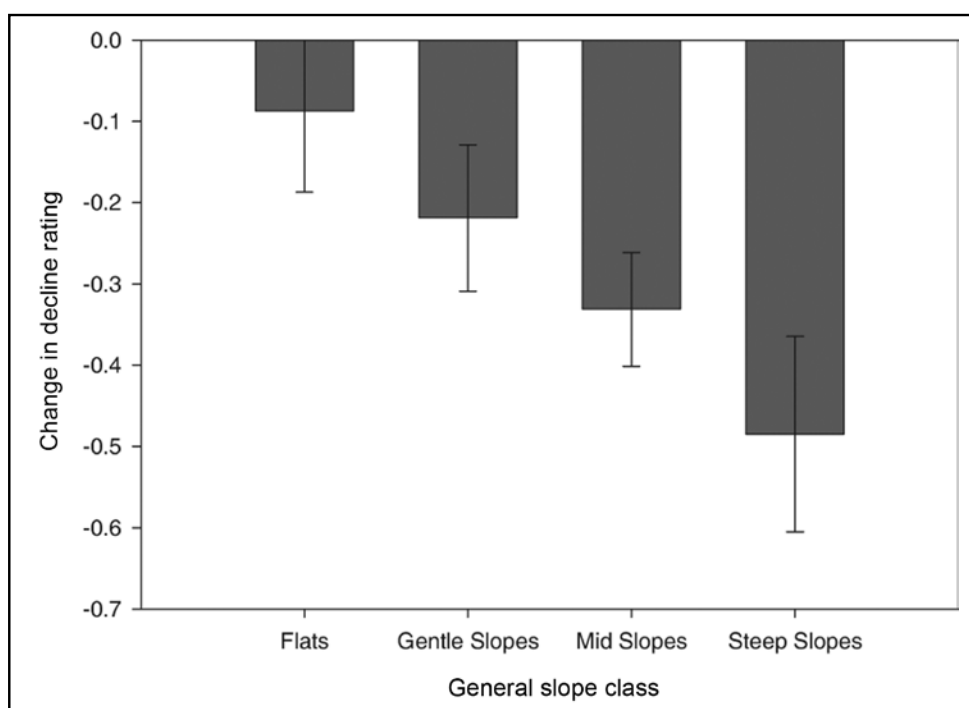


Figure 7—Susceptible stands were consistently found on steeper terrain than resistant stands ($p = 0.008$). Decline rating is a continuous scale from 0 (perfect health) to 10 (dead).

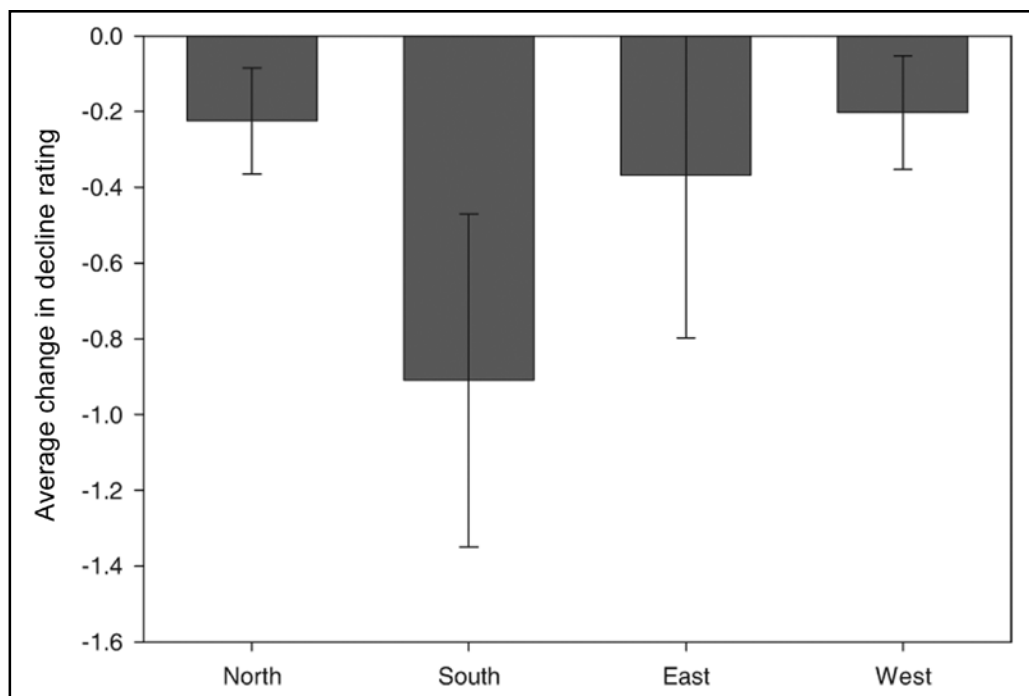


Figure 8—Southern-facing slopes experienced significantly faster decline following HWA infestation ($p = 0.012$). In addition, an interaction between aspect/slope ($p = 0.06$) and aspect/nitrogen ($p = 0.002$) indicates that the significance of aspect may come into play when other stressors (such as steeper slopes or higher nitrogen concentrations) are also a factor. Decline rating is a continuous scale from 0 (perfect health) to 10 (dead).

Conclusions

Creating an acceptable model of a complex, dynamic system is always challenging. Landscape-scale analyses increase the level of complexity by limiting the variables that are available for consideration. Here we applied knowledge- and data-driven approaches to model hemlock decline following HWA infestation. An initial review of existing literature identified potential variables for model inclusion and helped direct field measurement efforts. We used a mixed stepwise linear regression model based on plots that have been infested for at least 4 years to identify the set of landscape and chemical variables that could best predict the average rate of hemlock decline since HWA infestation. Using a continuous output variable instead of a simple tolerant/susceptible classification allows for flexibility in interpretation, depending on the needs of the user. For example, a research scientist with a limited number of HWA predator beetles to release may choose a conservative approach, selecting the stands with the lowest

anticipated rate of decline and, therefore, highest probability of continued health in spite of HWA infestation. Conversely, a forester who wants to preserve a strong genetic pool of hemlock may decide not to cut any hemlock in stands that have even a marginal probability of sustaining long-term infestation with minimal health impacts.

Hemlock vulnerability to HWA is complex and likely results from a combination of landscape and chemical factors. Because the ability to map relative risk on a landscape scale could prove to be a useful tool for managers faced with HWA, we limited ourselves to variables available in digital, raster format for inclusion in a GIS model. A model based only on topographic variables derived from a 10-m DEM was able to account for almost one-third of the variability in hemlock decline rates from infested plots across the Northeast. This is consistent with previous studies that link variables related to soil-moisture availability with hemlock vulnerability. By adding foliar nitrogen concentrations to the model, over two-thirds of the variability in hemlock decline rate following infestation can be accounted

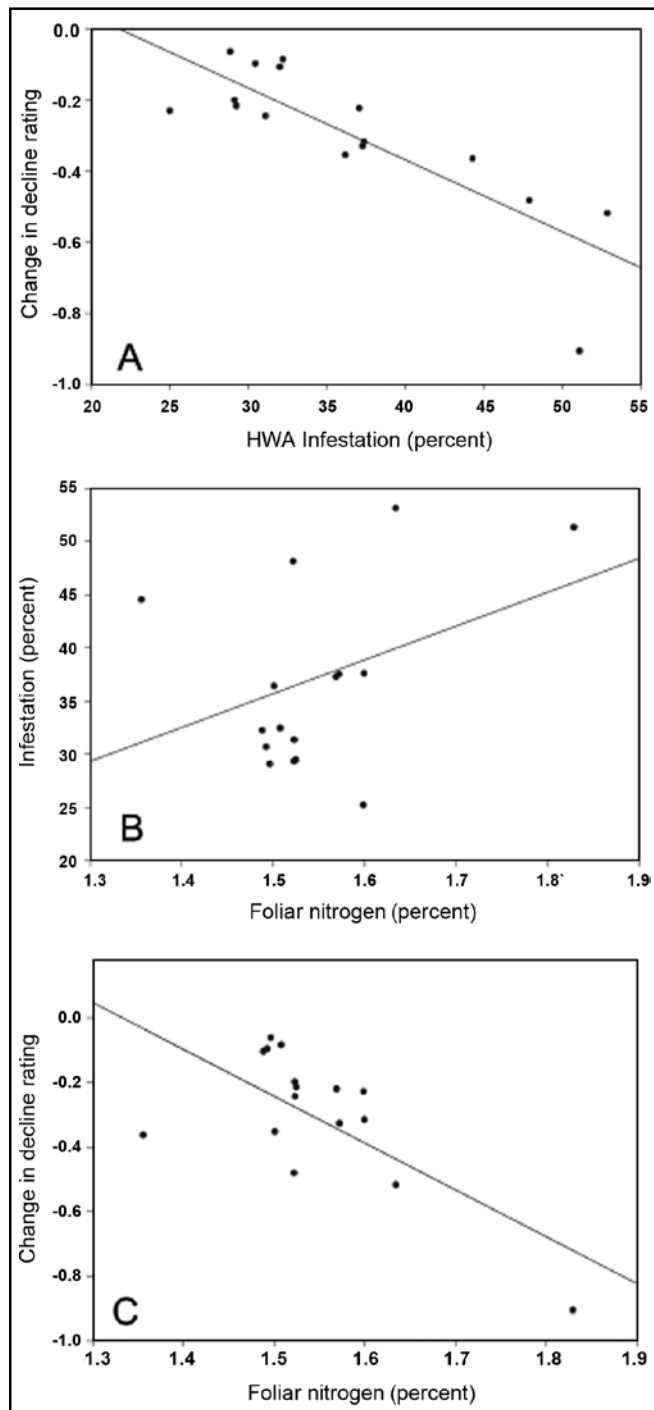


Figure 9—A. Percentage infestation was the strongest correlate with hemlock decline rates across the region ($r = -0.67$, $p = 0.008$). B. In turn, foliar nitrogen was the strongest correlate with percentage infestation ($r = 0.396$, $p = 0.005$), indicating a palatability-based relationship between nitrogen and HWA population success. C. This may be why nitrogen was a significant factor ($r = -0.436$, $p = 0.004$) in determining hemlock decline rates in the final model. Decline rating is based on several normalized variables such as dieback, transparency, live crown ratio, and new growth where 0 = perfect health and 10 = dead.

for. This is also consistent with previous fertilization and foliar chemistry studies, which identify a palatability-based relationship between foliar nitrogen and HWA population levels.

The significant improvement in model accuracy with the inclusion of chemical data highlights the value of hyperspectral data-derived coverages in risk modeling. In addition to improved predictive accuracy, hyperspectral imagery can provide spatially continuous maps of host species abundance and detailed decline assessments for model validation. This will allow land managers to better locate the host resource, identify stands to target management activities, and monitor forest health.

It is likely that the inclusion of other organic compounds, such as phenolics or other defensive chemicals would further improve this model (Bi and others 1997, Zucker and others 1992). However, the ability to use remote sensing platforms to assess secondary compound concentration has not been attempted to date. Other factors such as duration of infestation, climatic variables, and mineral nutrition likely interact, and these factors may exhibit different influence under different situations (Pontius and others 2006). In areas with available hyperspectral imagery, digital soil maps, and climate data, more complex models may soon be available to land managers. The addition of such data layers, which are not typically available for risk modeling, can be incorporated for more detailed and accurate risk maps. This type of spatially continuous information could be used by integrated pest management plans to help target specific areas on the ground where management efforts may be most effective.

This work will continue to be validated and improved in our future research efforts. By tracking infestation as it progresses through these stands and monitoring changes in hemlock health on newly infested plots, we will be better able to test the accuracy of this model. Finalized coverages of foliar nitrogen concentration will also be added to this model so that parameter estimates can be used to predict actual rates of decline instead of relative vulnerability to HWA.

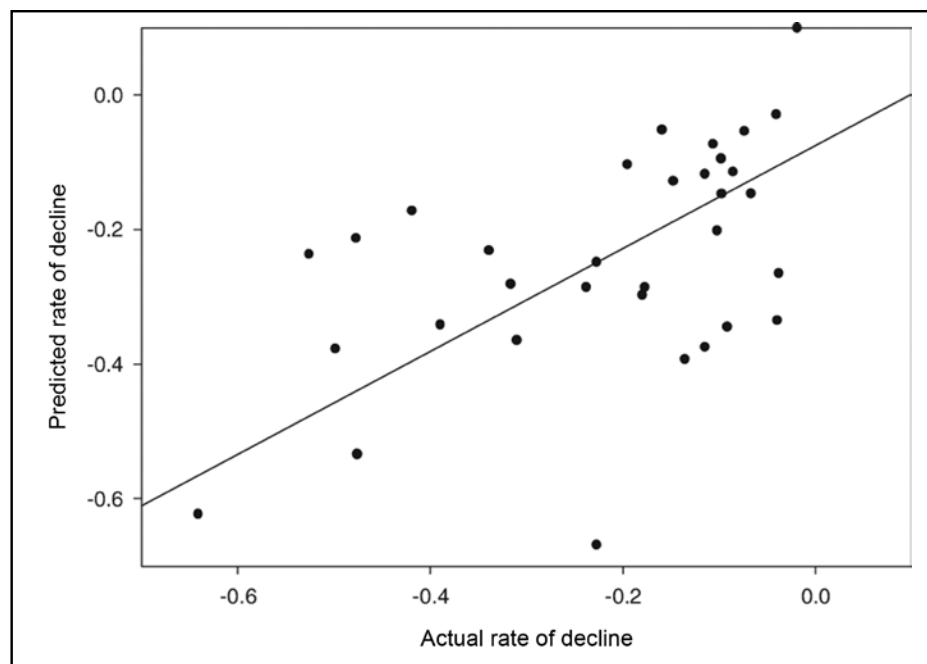


Figure 10—Actual and predicted final model results for all plots, including those excluded from calibration due to insufficient infestation periods to determine decline rates, produced a $p < 0.0001$, $r^2 = 0.51$, and $RMSE = 0.14$.

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Assessment of Habitat Threats to Shrublands in the Great Basin: A Case Study

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Abstract

The sagebrush (*Artemisia* spp.) ecosystem is one of the most imperiled in the United States. In the Great Basin ecoregion and elsewhere, catastrophic wildland fires are often followed by the invasion of cheatgrass (*Bromus tectorum* L.), eliminating or altering millions of hectares of sagebrush and other shrublands. Sagebrush in the Great Basin also is threatened by displacement from encroaching pinyon-juniper woodlands. Despite these changes, the ecoregion retains some of the Nation's largest remaining expanses of sagebrush, most of which are federally managed. Because of these losses and degradation, sagebrush-associated species are declining. To address these issues, we conducted a regional assessment of habitat threats for 40 sagebrush-associated vertebrates of concern in the Great Basin. Our goals were to (1) evaluate habitat conditions for species of concern for conservation planning and management, (2) demonstrate the application of new methods of regional threat assessment in shrubland communities, and (3) describe implications of results for management. Our analyses suggested that more than 55 percent (4.8 million ha) of sagebrush in the Great Basin is at moderate or high risk of being displaced by cheatgrass. Cheatgrass invasion also threatens other shrubland communities, particularly salt desert scrub (96 percent; 7.1 million ha at moderate or high risk). Substantial areas of sagebrush (41 percent; 2.0 million ha) were predicted to be at moderate or high risk of displacement by pinyon-juniper in the eastern Great Basin. Although little sagebrush (less than 1 percent) was at high risk to both threats, more than one-third was at high risk

to cheatgrass and low or moderate risk from woodlands. Habitat loss, including sagebrush and other native plant communities, to cheatgrass could exceed 65 percent (8 million ha) for some of the 40 vertebrate species evaluated. Maintenance and restoration of native shrublands in the Great Basin will require both active and passive management to mitigate the formidable threats posed by cheatgrass and pinyon-juniper woodland expansion.

Keywords: Cheatgrass, Great Basin, models, pinyon-juniper woodlands, sagebrush, threats.

Setting the Stage: Shrublands at Risk

The vast shrublands of Western North America, including the sagebrush (*Artemisia* spp.) ecosystem, provide a wide range of resource values, including recreation, livestock grazing, mining, energy extraction, wildlife habitat, and wilderness. Increasingly, however, these arid and isolated lands have faced a wide range of threats, including wildland fires, invasive plants, roads, oil and gas development, and climate change (Connelly and others 2004, Knick 1999, Wisdom and others 2005a). (In this paper we define a threat as a potentially detrimental human activity or ecological process as it affects native species or their habitats [Wisdom and others 2005a] and risk as “the potential, or probability, of an adverse event” [Burgman and others 1993].)

Despite the recognized values of shrublands and grasslands—collectively known as rangelands—such lands have seldom benefited from the long-term research and monitoring traditionally focused on forested ecosystems in the United States. This lack of a well-established body of research and monitoring presents special challenges in evaluating and predicting effects of disturbances, both natural and anthropogenic, on rangeland habitats and wildlife in the United States. The dearth of attention is especially noteworthy in that rangelands compose about 50 percent (39 million ha) of the 77 million ha managed under the National Forest System. Nationwide, rangelands total 312 million ha, with 43 percent managed by the Federal Government (<http://www.fs.fed.us/rangelands/>).

To address the needs of land managers charged with conservation planning and management of shrublands and associated wildlife, Wisdom and others (2005a) described methods for conducting regional assessments in native shrubland ecosystems of Western North America. They applied those methods in a prototype assessment of habitat threats within the Great Basin ecoregion. Their goals were to (1) evaluate habitat conditions for species of concern for conservation planning and management, (2) demonstrate the application of newly developed methods of regional assessment of threats in shrubland communities, and (3) describe implications of results for management.

The following text draws heavily from their book. A requirement of the Great Basin prototype assessment as funded by the Bureau of Land Management (BLM) was that it be conducted with existing data (spatial and other) as a demonstration of products that could be readily delivered. Such pragmatism is often a necessity in natural resource management.

The reader is referred to the Wisdom and others (2005a) book for additional details of the Great Basin case study, especially background material and methods. Appendices with materials not published within the book can be found at <http://www.fs.fed.us/pnw/pubs/sagebrush-appendices/>.

Status and Threats in the Sagebrush Ecosystem

The sagebrush ecosystem covers more than 43 million ha within the Western United States and Canada and constitutes one of the largest ecosystems in North America (Center for Science, Economics, and Environment 2002; Wisdom and others 2005a). More than two-thirds of the total area covered by sagebrush in the United States is on land publicly owned and managed by State or Federal agencies; the BLM alone manages 52 percent of the sagebrush in the United States (Knick and others 2003). Conservation and restoration of the sagebrush ecosystem are of special concern to State and Federal resource management agencies owing to extensive habitat degradation and loss (Knick 1999, Knick and others 2003, Wisdom and others 2005a). Since European settlement, the area covered by sagebrush has been reduced more than 40 percent (Connelly and

others 2004), and only a small fraction remains unaltered by anthropogenic disturbances (West 1999).

A plethora of threats to sagebrush and other native shrubland communities has been identified (Connelly and others 2004, Wisdom and others 2005a). Among these are invasion of exotic vegetation, altered fire regimes, road development and use, mining, energy development, climate change, encroachment of pinyon (*Pinus* spp.) and juniper (*Juniperus* spp.) woodlands, intensive livestock grazing, and conversion to agriculture or urban areas. The cumulative effects of these stressors have resulted in the sagebrush ecosystem being regarded as one of the most endangered in the Nation (Noss and others 1995), and 20 percent of the plants and animals associated with this ecosystem may be at risk of extirpation (Center for Science, Economics, and Environment 2002).

Several scientific assessments have addressed the effects of multiple threats in the sagebrush ecosystem (e.g., Boyle and Reeder 2005, Connelly and others 2004, West 1999); however, despite the knowledge gained from these studies and others, efforts to abate the degradation and elimination of sagebrush have not been successful at large scales (Hemstrom and others 2002, West 1999).

Ecological Setting and Status of the Great Basin

The Great Basin ecoregion spans more than 29 million ha from the eastern Sierra Nevada Mountains in California to central Utah east of the Great Salt Lake (Figure 1). Federal lands dominate (80 percent) the landscape; thus, the role of Federal agencies is paramount in conservation planning and management in this area. Although classified as arid, with mean annual precipitation of 216 mm, precipitation is highly variable and falls primarily during winter and spring. Climate change has been pronounced during the past 150 years, with warmer temperatures and increased carbon dioxide levels (Tausch and Nowak 2000, West 1999). Riparian systems of the Great Basin are particularly sensitive to effects of climate change (Chambers and Miller 2004).

The Great Basin supports one of the largest extant concentrations of sagebrush in North America at more than

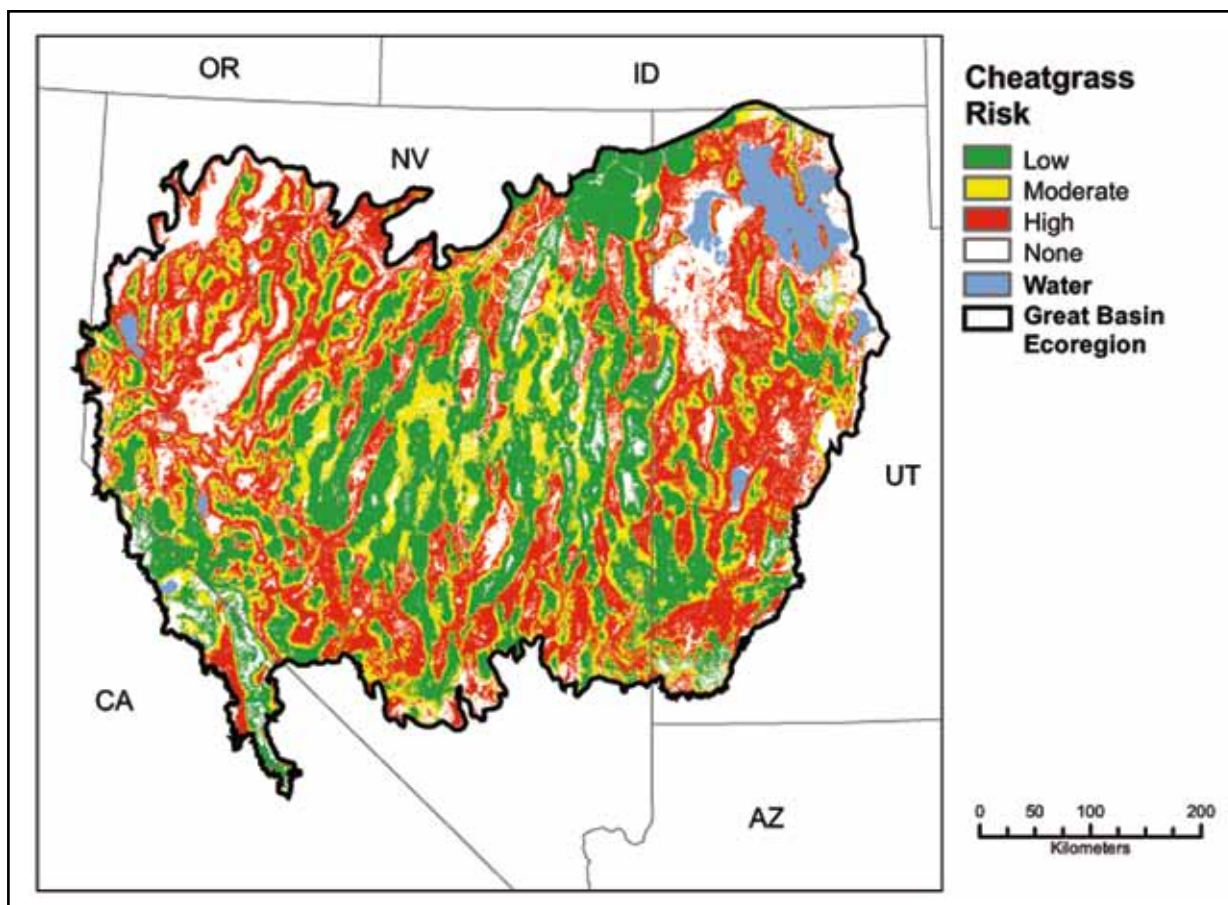


Figure 1—Estimated risk of cheatgrass displacement of sagebrush and other susceptible landcover types in the Great Basin ecoregion (from Wisdom and others 2005a).

8 million ha (29 percent of the landcover in the ecoregion). Indeed, among ecoregions of the sagebrush biome, the Great Basin ranks second in total extent of sagebrush, exceeded only by the Columbia Plateau (Wisdom and others 2005a). Although sagebrush loss in the ecoregion has occurred from a variety of causes, wildland fires have been especially devastating. More than 500 000 ha of sagebrush (6.4 percent of the sagebrush in the ecoregion) burned between 1994 and 2001 (Wisdom and others 2005a). Closely tied to losses from wildfire is the spread of cheatgrass (*Bromus tectorum* L.), introduced to the United States in the 1800s and now a pervasive problem throughout much of the arid West (Billings 1994, Booth and others 2003, Peterson 2006). Cheatgrass is estimated to currently occupy more than 2 million ha (7 percent) of the Great Basin (Bradley and Mustard 2006).

Beyond its reputation as a stronghold for sagebrush, the Great Basin is recognized as an area of rich biodiversity, with its extremes of topography and climate contributing to a diverse assemblage of endemic plants and animals (Nachlinger and others 2001, Ricketts and others 1999). Among terrestrial ecoregions of North America, the Great Basin Shrub Steppe was among the most diverse in both number and endemism of vascular plants (Ricketts and others 1999).

Regional Assessment of Habitats

To address concerns about ongoing degradation of sagebrush habitats and associated species, Wisdom and others (2005a) identified steps for spatial analysis to be used in regional assessment. Those steps were subsequently applied in the Great Basin ecoregion in a case study approach.

The following sections describe the methods and results of the Great Basin assessment, with primary emphasis on results and their interpretation for management. These methods involved selection of species for analysis, describing species-habitat relationships, identification of primary threats, quantification of vegetation and habitat in the study area, and development of predictive models of selected threats.

Selecting and Grouping Species of Concern

Using a variety of screening criteria (e.g., habitat association, estimated risk of extirpation), Wisdom and others (2005a) compiled a comprehensive list of more than 350 sagebrush-associated species of concern. Species of concern were defined as those with populations or habitats that are rare or declining, or both. This list was dominated by vascular plants (68 percent), but also included terrestrial invertebrates (7 percent) and vertebrates (25 percent). For the Great Basin study, this list was narrowed by selecting species suitable for broad-scale assessment in the study area. The screening process yielded 40 species of concern, including 1 amphibian, 9 reptiles, 17 birds, and 13 mammals. Among those selected, greater sage-grouse (*Centrocercus urophasianus*) and pygmy rabbit (*Brachylagus idahoensis*) are of particular concern due to recent petitions to list these species under the U.S. Endangered Species Act (USDI Fish and Wildlife Service 2008, 2010).

The 40 species of concern were assigned to one of five groups according to similarities in habitat associations and amount of habitat in the ecoregion (see Table 1 for a list of species groups). The use of species groups, in combination with individual species, allows managers to:

1. Address either single- or multispecies needs.
2. Identify regional patterns of habitats, especially habitats at risk, that affect multiple species similarly.
3. Address the needs of many species efficiently and holistically.
4. Determine how well regional strategies for groups of species meet the needs of individual species.

Describing Species-Habitat Relationships

Evaluation of the condition (e.g., risk levels) and spatial pattern of habitat for multiple species of concern provides essential context for the development of regionally based management and conservation strategies. A first step in habitat evaluation is defining the relationship between each species and environmental attributes, often vegetation cover types. For the Great Basin assessment, we identified vegetation associated with each species as habitat based on a recently completed landcover map (Comer and others 2002; [<http://sagemap.wr.usgs.gov/>]). This map was developed explicitly for regional assessment of sagebrush habitats across the Western United States and included the delineation of 8 sagebrush and 38 other landcover types in the Great Basin.

To define habitat relationships for Great Basin species of concern, we used the landcover types to construct a species-habitat association matrix, relying on literature sources and existing wildlife-habitat relationships databases. We then asked species experts to review the matrix, and we refined the habitat designations as necessary. This matrix was the basis for subsequent quantification of amount of habitat, and habitat at risk, within each species' range in the Great Basin ("Quantifying Vegetation and Habitat in the Great Basin").

Identifying Regional Threats

Wisdom and others (2005a) outlined a process for identifying threats to consider in regional assessment, using criteria such as spatial extent of the threat, available resources to address the threat, and cost-benefit analysis. The threats confronting sagebrush habitats in the Great Basin ecoregion typify those in most sagebrush-dominated ecoregions of Western North America. Two of these are pervasive and accelerating: the displacement of native vegetation by cheatgrass invasion and by encroaching pinyon-juniper woodlands.

Invasion by exotic species, particularly cheatgrass, is consistently cited as one of the major challenges to maintenance of healthy sagebrush communities (Connelly and others 2004, Knick 1999, Pyke 2000). In addition to its

Table 1—Percentage of watersheds in the Great Basin by combined classes of habitat abundance and risk that habitats will be displaced by cheatgrass, summarized for each of the five groups of species of conservation concern

Species group	Habitat abundance	Risk of habitat displacement by cheatgrass			
		None-low	Low-moderate	Moderate-high	All risk classes combined
Percent					
Sagebrush (<i>n</i> = 519)	Low	14	10	9	33
	Moderate	7	21	18	46
	High	2	14	7	23
	Total ^a	24	44	32	100
Salt desert shrub (<i>n</i> = 507)	Low	6	8	21	35
	Moderate	4	4	23	31
	High	5	3	26	34
	Total	14	15	71	100
Shrubland (<i>n</i> = 521)	Low	2	1	2	5
	Moderate	3	4	10	17
	High	7	21	51	79
	Total	11	26	63	100
Sagebrush-woodland (<i>n</i> = 519)	Low	7	9	9	25
	Moderate	3	18	8	29
	High	5	32	8	45
	Total	15	59	26	100
Generalist (<i>n</i> = 521)	Low	1	<1	1	2
	Moderate	3	1	5	9
	High	14	37	37	88
	Total	18	39	43	100

Total number of watersheds for each species group is given below the group name. See “Selecting and Grouping Species of Concern” for methods used to group the species and “Characterizing Habitat Conditions” for characterization of habitat conditions for species groups.

^a Total refers to all habitat abundance classes (low, moderate, and high) combined.

Source: Wisdom and others 2005a.

displacement of native understory species, the autecology of cheatgrass leads to an increased risk of catastrophic wildfires that eliminate the sagebrush overstory (Billings 1994, Booth and others 2003). The presence of cheatgrass exacerbates fire hazard because of its high flammability. Cheatgrass can drastically shorten the fire recurrence interval in native sagebrush communities from 20 to 100 years to 3 to 5 years in some sites.

Pinyon-juniper woodlands have expanded greatly in the Great Basin when compared to their distribution more than 150 years ago (Miller and Tausch 2001). Tree density also has increased in established woodlands. These changes have been linked to a decrease in area burned by wildfire, a result of increased fire suppression and removal of fine fuels by livestock; climate change; historical patterns of livestock grazing; and increased atmospheric carbon dioxide (Polley and others 2002, Sakai and others 2001).

Detrimental outcomes of pinyon-juniper woodland expansion include increased soil erosion, changes in soil fertility, losses in forage production, reductions in wildlife habitat for some species, and alteration of presettlement native plant communities (Miller and Tausch 2001). In reporting its findings for a petition to list greater sage-grouse under the Endangered Species Act, the U.S. Fish and Wildlife Service explicitly cited habitat degradation from invasive species, such as cheatgrass, and encroachment of pinyon-juniper woodlands as key threats to greater sage-grouse (USDI Fish and Wildlife Service 2010).

Although many other threats affect sagebrush habitats in the Great Basin such as overgrazing by domestic and wild ungulates (Nachlinger and others 2001), empirical data are often insufficient to model or estimate the risks posed by these threats at large scales. We chose these two threats—cheatgrass and pinyon-juniper woodlands—in the Great Basin assessment to exemplify the **process** of evaluating threats to sagebrush ecosystems at regional scales.

Quantifying Vegetation and Habitat in the Great Basin

To estimate vegetation at risk in the Great Basin, we first quantified the total area of each landcover type present in the study area using the landcover map described in “Describing Species-Habitat Relationships” (Comer and others 2002). The area burned in the Great Basin, especially sagebrush, increased dramatically beginning in 1994 (Connelly and others 2004). To accurately assess these landcover changes, areas in the Great Basin burned by large-scale fires since 1994 were reclassified as recently burned. We used fire data from 1994 to 2001 because the cover type map accounted for area burned prior to 1994.

The amount of habitat for each species was calculated by overlaying the landcover map—selecting only those cover types identified as habitat for the species (“Describing Species-Habitat Relationships”)—with the species’ geographic range within the Great Basin. These habitat maps were the basis for subsequent quantification and mapping of habitats at risk for each species (“Species’ Habitats at Risk”) and for species groups (“Characterizing Habitat Conditions”).

Modeling of Displacement by Woodlands and Cheatgrass

We developed two independent, rule-based predictive models of displacement of sagebrush and other native vegetation by encroaching pinyon-juniper woodlands and cheatgrass, employing a variety of topographic and vegetation variables (Suring and others 2005). Simple rule-based models may be effective in displaying broad-scale patterns of risk in susceptible plant communities (Woodbury and Weinstein, this volume). The models were deterministic (i.e., the model assumptions completely determine the outcome of the model, with no element of randomness involved), and were intended to display broad-scale patterns of risk across the ecoregion, summarized at watershed (i.e., 5th hydrologic unit code) or larger scales. The models were applied first to vegetation, and then to habitats for individual species of concern (pinyon-juniper and cheatgrass models) and species groups (cheatgrass model only). The models were intended to display risk of displacement of native vegetation in 30 years by the two stressors; thus, areas mapped as high risk may or may not have been converted to cheatgrass or woodland-dominated sites at present.

We identified environmental variables thought to be most important for estimating the risk that sagebrush will be displaced by pinyon-juniper woodlands or cheatgrass by reviewing the literature, including studies of dispersal mechanisms for pine seeds and juniper berries, and consulting with species experts on the ecological relationships and traits of pinyon and juniper species and cheatgrass. The rule sets for both models used ecological provinces (described by Miller and others 1999b and West and others 1998) as a geographic basis for model development. The characterization of landscape conditions within the provinces provided a useful ecological context for model development.

Model Development

Pinyon-Juniper Woodland Model—

Variables selected for the pinyon-juniper risk model included landcover (i.e., sagebrush taxon), elevation, potential for seed dispersal (based on proximity of pinyon-juniper woodlands to sagebrush), precipitation, and landform. Variables were parameterized differently for groups

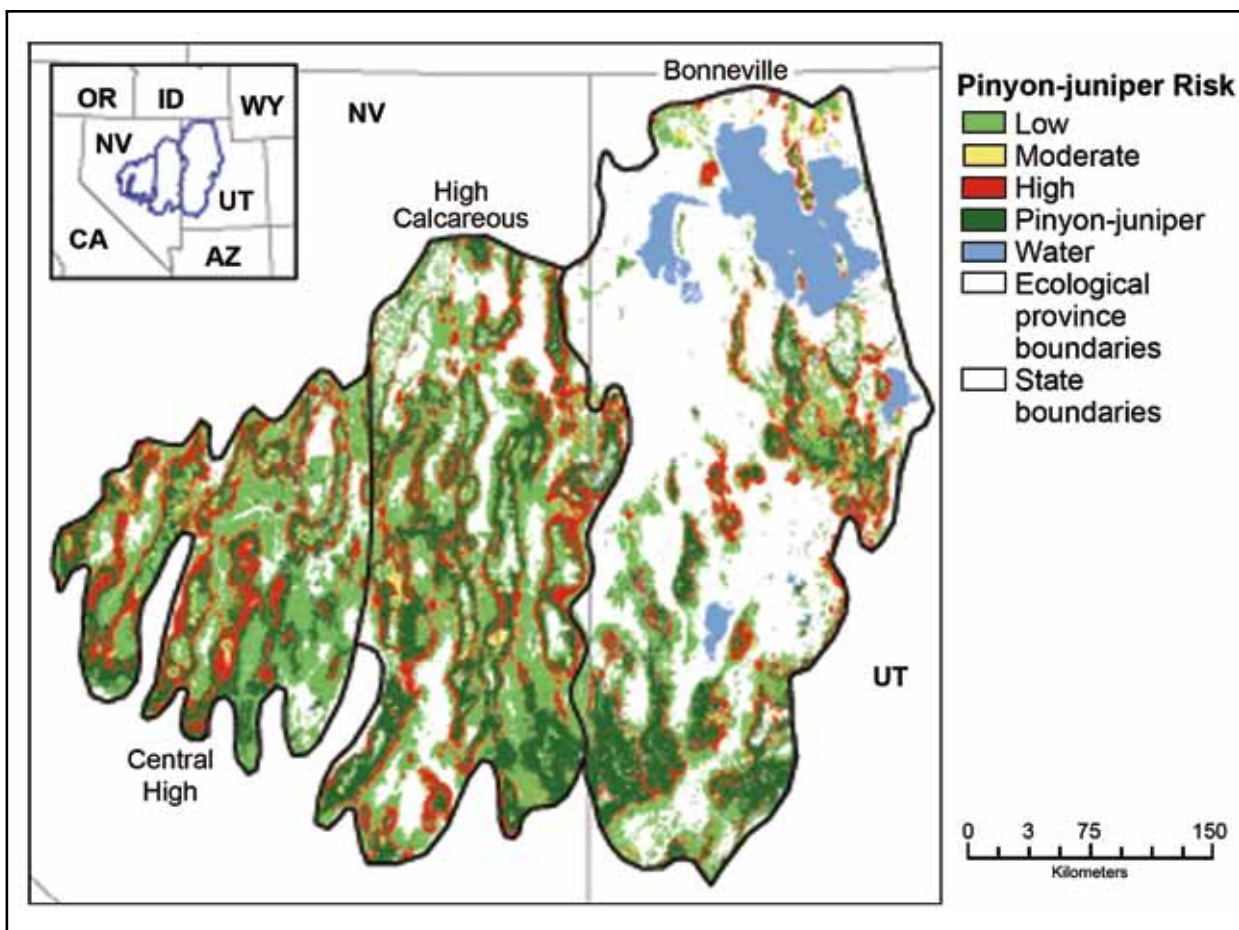


Figure 2—Estimated risk of pinyon-juniper displacement of sagebrush in the Great Basin ecoregion. Levels of risk of sagebrush displacement are mapped in relation to all sagebrush cover types in the three ecological provinces in which the model was applied: Central High, High Calcareous, and Bonneville. Areas not considered to be at risk are landcover types other than sagebrush and are depicted in white (from Wisdom and others 2005a).

of ecological provinces within the Great Basin. Evaluation of the landcover map indicated that, within the study area, the spatial representation of pinyon-juniper woodlands was most accurate in three provinces in the eastern Great Basin. Consequently, our application of the risk model was restricted to those provinces (Figure 2).

Risk classes were defined as follows:

- Low—the probability that pinyon-juniper woodlands will displace existing sagebrush cover types within 30 years is minimal; currently, little or no pinyon-juniper is likely to be present in the overstory of these sage-brush stands.
- Moderate—the probability of displacement within 30 years is likely, but less so than for sagebrush at

high risk; currently, pinyon-juniper is likely to be a minor to common component of the overstory of these stands. This class represents a transition phase in the conversion of sagebrush cover types to pinyon-juniper woodlands (Miller and others 1999a).

- High—the probability of displacement within 30 years is high; currently, pinyon-juniper is likely to be a common to dominant component of the overstory of these stands.

Cheatgrass Model—

Variables selected for the cheatgrass model included aspect, slope, elevation, and landform by ecological province. Recently published research based on remotely sensed data from the Great Basin supports the use of several of these

variables (Bradley and Mustard 2006, Peterson 2006).

The model was applied to all susceptible landcover types within the Great Basin ecoregion (Figure 1). Nonsusceptible landcover types (e.g., marsh/wetland) were assigned to the “none” risk level. The remaining risk classes were defined as follows:

- Low—the probability that cheatgrass will displace existing sagebrush or other susceptible cover types within 30 years is minimal; currently, native plants are likely to dominate the understory of these stands.
- Moderate—the probability of displacement within 30 years is moderate, but lower than for types at high risk; currently, either cheatgrass or native plants can dominate the understory.
- High—the probability of displacement within 30 years is very likely; currently, cheatgrass is likely to dominate the understory.

Results of Model Application in the Great Basin

Outputs from both the pinyon-juniper and cheatgrass models were summarized by landcover type and by species (i.e., habitats at risk). Maps and associated spatial data in tabular form are in Wisdom and others (2005a); we present a brief summary of results below.

Vegetation at Risk of Displacement by Pinyon-Juniper—

Of the 4.8 million ha of sagebrush in the three provinces of the eastern Great Basin, 35 percent (1.7 million ha) was estimated to be at high risk of displacement by pinyon-juniper woodlands, whereas 60 percent (2.9 million ha) was at low risk. Very little sagebrush was in the moderate risk category (6 percent), which represents a transitional phase of encroachment. Mountain big sagebrush (*Artemisia tridentata* Nutt. *vaseyana* (Rydb.) Beetle) appeared to be the most susceptible sagebrush taxon (42 percent at high risk). However, Wyoming-basin big sagebrush (*A. t. wyomingensis* Beetle and Young-*A. t. tridentata*) was the dominant sagebrush cover type in the modeled area and made up 55 percent of all the high-risk sagebrush, exceeding 930 000 ha.

The spatial pattern of areas estimated to be at high risk of displacement by pinyon-juniper closely followed the

distribution of pinyon-juniper woodlands throughout the three provinces (Figure 2). Areas of moderate and high risk tended to occur on the side slopes, with areas of low risk in the valley bottoms.

Vegetation at Risk of Displacement by Cheatgrass—

Nearly 80 percent of the landcover in the Great Basin was estimated to be susceptible to displacement by cheatgrass (i.e., low or greater risk; Figure 1). Of the susceptible area, 26 percent was estimated to be at moderate risk and 40 percent at high risk. Among vegetation types, salt desert scrub was most at risk, with 96 percent (7.1 million ha) estimated to be at moderate or high risk of displacement by cheatgrass. Sagebrush was also at risk from cheatgrass, with 38 percent at moderate risk and 20 percent at high risk, a total of 4.8 million ha. The overwhelming majority (88 percent) of the sagebrush at high risk was Wyoming-basin big sagebrush.

Spatial patterns of risk of displacement by cheatgrass followed the typical north-south alignment of topographic features in the Great Basin. Areas of high risk generally occurred at lower elevations and on valley bottoms, whereas areas of low risk were typically in mountain ranges and higher elevation valleys.

Species' Habitats at Risk—

For 35 of the 40 species of concern considered in our assessment, more than 30 percent of their sagebrush habitat was at high risk of displacement by pinyon-juniper woodlands in the three provinces in which this risk was modeled. Twelve species each had about 1.6 million ha of sagebrush habitats at high risk; these species used all eight sagebrush cover types as habitat and were widely distributed across the three provinces.

Within the Great Basin, the dominant category for species' habitat at risk of displacement by cheatgrass was high risk (mean = 36 percent, $n = 40$). For 33 species (88 percent), more than half of their habitat in the study area was at moderate or greater risk of displacement by cheatgrass. Ten species, including one amphibian, three reptiles, five raptors, and one small mammal, had more than 8 million ha of habitat at high risk, an area equivalent to about

one-fourth of the ecoregion. Relative to other species in our assessment, habitat for greater sage-grouse and pygmy rabbit was at lower risk of displacement by cheatgrass.

Application of Risk Models in Shrubland Management (Application to Management)

Federal agencies need information about habitat requirements and conditions for species and groups of species at spatial extents that are typically used in land management planning. To demonstrate the application of our threats modeling results (“Species’ Habitats at Risk”) to land management in the Great Basin, we used species groups (“Selecting and Grouping Species of Concern”) to generalize the spatial patterns of habitat at risk at the watershed scale in the Great Basin.

Characterizing Habitat Conditions

We used two habitat variables, habitat abundance and habitat at risk of displacement by cheatgrass, to characterize the composite habitat conditions for each species group at the watershed extent within the Great Basin. For each variable, watersheds were assigned to one of three classes, resulting in nine possible combinations of habitat abundance and risk (Table 1, Figure 3).

The percentage of watersheds in each of the nine condition classes was relatively even for the sagebrush and salt desert shrub species groups, but one or two condition classes dominated the other three groups (Table 1, Figure 3). The best condition class, that of high habitat abundance coupled with none to low risk, was consistently uncommon, with the exception of the generalist group of species (Table 1). Notably, the sagebrush group, which included both greater sage-grouse and pygmy rabbit, had the smallest percentage (2 percent) in this class (Table 1, Figure 3).

Prioritizing Restoration Activities

Different combinations of habitat abundance and habitat risk have different implications for conservation and restoration. Those characterizations can inform and guide restoration priorities. For example, watersheds with habitats of relatively higher abundance and lower risk may represent habitat strongholds (i.e., optimal habitat amount,

quality, and spatial arrangement) for a group of species. Management objectives for these watersheds would likely be tailored to maintain current conditions through prudent management of human activities and potential threats that can alter the amount, quality, and spatial pattern of desired native habitats. In general, targeting management of native shrublands at moderate risk may be the most prudent approach, given (1) the relative security of low-risk habitats and (2) the vast resources required to prevent high-risk areas from passing beyond ecological thresholds, after which restoration of native habitats may be difficult or impossible to achieve (Wisdom and others 2005b). Areas at high risk from these stressors may have already passed such thresholds.

By contrast, watersheds with limited habitat area and higher risk may be suboptimal relative to the requirements of a species group. Watersheds in this condition would likely have smaller patch sizes, lower habitat quality, or poor habitat connectivity in relation to the group’s needs. Management objectives for such watersheds would likely focus on retention of existing habitats, combined with substantial emphasis on restoration activities to mitigate past habitat losses and degradation. Identification of broad-scale patterns of habitat risk for species of concern helps determine whether restoration activities need to be initiated (Wisdom and others 2005b).

Integration of Multiple Stressors

The combined risk of displacement by cheatgrass and by pinyon-juniper woodlands was evaluated for the 4.8 million ha of sagebrush in the three ecological provinces in which the pinyon-juniper model was applied. Of the five combined risk classes, the two dominant classes were high cheatgrass (high risk from cheatgrass and low or moderate from pinyon-juniper; 35 percent) and low to moderate (moderate risk from both stressors; 33 percent). Only a trace amount (less than 1 percent) of sagebrush in this region was at high risk to both stressors (see Wisdom and others 2005a for details). Other stressors not considered in our prototype assessment, such as climate change, may be critical in future modeling of risks in the Great Basin from both cheatgrass and pinyon-juniper expansion. Under one modeling approach,

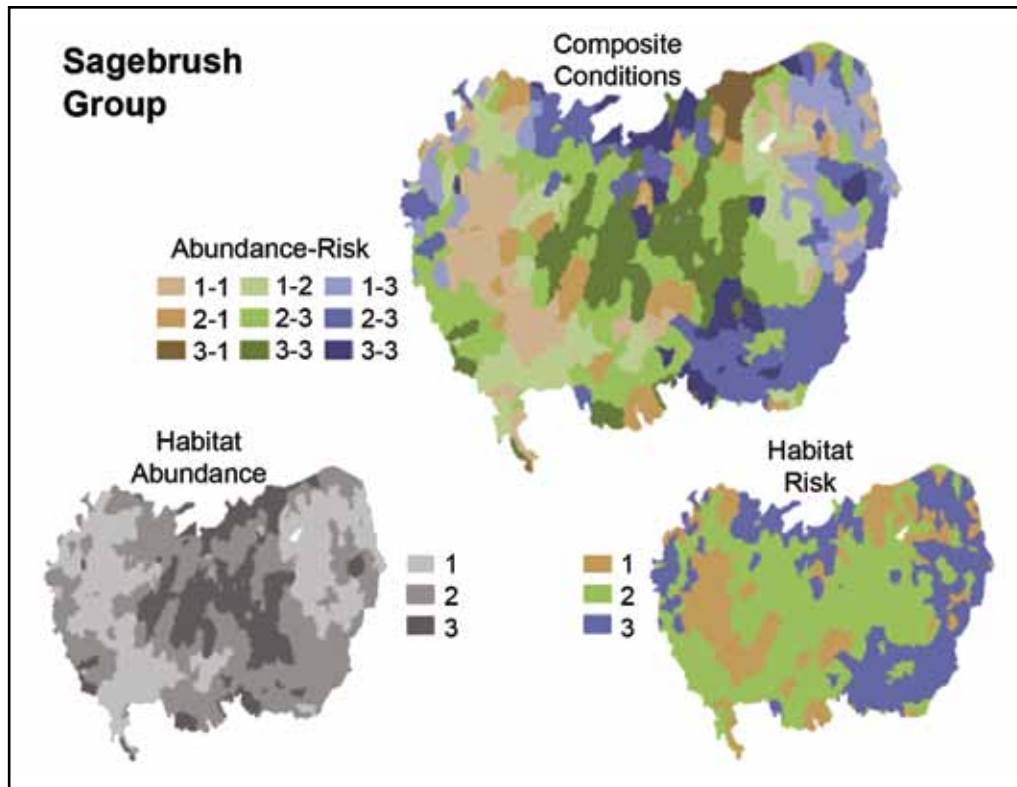


Figure 3—Habitat abundance (lower left), habitat risk (lower right), and composite conditions (top center) for the sagebrush group of species in watersheds of the Great Basin. For habitat abundance, 1 = low (<25 percent of the watershed is habitat), 2 = moderate (25 to 50 percent of the watershed is habitat), and 3 = high (>50 percent of the watershed is habitat). For the map of habitat risk, 1 = watersheds with habitats dominated by none to low risk of displacement by cheatgrass, 2 = watersheds with habitats dominated by low to moderate risk, and 3 = watersheds with habitats dominated by moderate to high risk. For the map of composite habitat conditions, the first number is for abundance, and the second number is for risk, using the definitions above for maps of habitat abundance and risk. For example, 1-1 denotes watersheds with habitats of low abundance and none to low risk, whereas 3-3 denotes watersheds of high habitat abundance and moderate to high risk. Map colors on the composite map follow those for the map of habitat risk (blue = moderate to high risk, green = low-moderate risk, and brown = none-low risk), and intensity of colors follows those for the map of habitat abundance (darkest shade = high habitat abundance, intermediate = moderate, and lightest = low) (from Wisdom and others 2005a). Mean size of the 367 watersheds that occur entirely within the ecoregion boundaries is 66 000 ha.

direct and indirect effects of global warming may result in the elimination of up to 80 percent of existing sagebrush in the United States (Neilson and others 2005).

Management Implications

Our analyses revealed that watersheds differ in spatial patterns of habitat abundance and risk, resulting in different implications for conservation and restoration of habitats. These differences in spatial patterns suggest that regional strategies could be developed for watersheds in each condition class to identify appropriate conservation and

restoration prescriptions needed to meet management goals for each group of species. Targeting areas for management attention through this process complements other strategies to identify threats, such as the conservation by design approach used by The Nature Conservancy (Nachlinger and others 2001).

Results of our regional assessment serve as a working example for analysis of species of concern, but also provide a sound basis for comprehensive land use planning in the ecoregion. Cheatgrass invasion and woodland encroachment are two of the highest priority issues for both research

and management of native shrublands in the Great Basin. Because of the potential mismatch of administrative scales at which management occurs and the ecological scales at which these threats operate, regional assessment can provide key context for local management solutions. Integrated management of both threats is needed to conserve sagebrush habitats affected by both cheatgrass and pinyon-juniper expansion. For example, prescribed burning of some sagebrush habitats may curtail woodland encroachment but also may enhance expansion of cheatgrass on susceptible sites. Alternatively, combinations of mechanical treatments to reduce pinyon-juniper density, followed by chemical treatments to reduce cheatgrass abundance, may effectively reduce the combined risks posed by cheatgrass and woodlands.

With any regional assessment encompassing a diverse set of species and habitats, uncertainty exists at several levels about the assumptions inherent in the analyses. For the Great Basin assessment, example areas of uncertainty include:

- The cover types associated with each species as habitat are assumed to contribute to population persistence, but additional factors beyond habitat abundance and configuration may influence populations. We have limited knowledge of how species respond to landscape-scale changes in native habitats, as brought about by displacement from cheatgrass and pinyon-juniper woodlands. Further studies of this topic are a high priority for research and management.
- Our predictions of risk of displacement of native shrublands are based on a prior knowledge of environmental conditions affecting distribution of cheatgrass and pinyon-juniper and require extensive field evaluation before results are widely applied in management. Such validation is currently underway (Rowland and others 2006) and will reduce the uncertainty associated with model predictions.
- The combined effects of multiple threats, especially the potential synergy among threats, are not well understood. New research is needed to improve

knowledge about cumulative effects on shrubland habitats and species.

Restoration in sagebrush and other arid shrubland communities will require substantial inputs of resources owing to the lack of resiliency in these systems and the recent undesirable trends in vegetation dynamics and fire regimes affecting these communities (Hemstrom and others 2002). Concomitant with restoration, prevention and mitigation of threats that go beyond cheatgrass and woodland expansion, such as infrastructure associated with anthropogenic activities (e.g., powerlines, roads, energy development), are critical for maintaining functioning habitats for shrubland-associated species of concern in the Great Basin and surrounding areas.

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Evaluating the Impact of Invasive Species in Forest Landscapes: The Southern Pine Beetle and the Hemlock Woolly Adelgid

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Abstract

The southern pine beetle, *Dendroctonus frontalis* (Zimmerman) (Coleoptera: Curculionidae: Scolytinae) (SPB), is an indigenous invasive species that infests and causes mortality to pines (*Pinus* spp.) throughout the Southern United States. The hemlock woolly adelgid, *Adelges tsugae* (Annand) (Homoptera: Adelgidae) (HWA), is a nonindigenous invasive species that infests and causes mortality to Eastern hemlock (*Tsuga canadensis* (L.) Carr.) and Carolina hemlock (*T. caroliniana* Engelm.) throughout their range in Eastern North America. Both of these insect species occur in the Southern Appalachians, and both have recently caused tree mortality exceeding historical records. Herbivory by both species is of concern to forest managers, but for different reasons. In the case of the SPB, emphasis centers on forest restoration strategies, and in the case of the HWA, the concern is on predicting the impact of removing hemlock from the forest environment. Both of these issues can be investigated using a landscape simulation modeling approach. LANDIS is a simulation modeling environment developed to predict forest landscape change over time. It is a spatially explicit, landscape-scale ecological simulation

model that incorporates disturbance by fire, wind, biological disturbance (insects and pathogens) and harvesting. Herein, we present a case study using LANDIS to evaluate the impact of herbivory by the SPB and HWA on forest landscapes in the Southern Appalachians.

Keywords: Invasive species, LANDIS, modeling, Southern Appalachians.

Introduction

In 2003, five general areas were identified as concerns to healthy forests in the United States—wildfires, nonnative invasive insects and pathogens, invasive plant species, outbreaks of native insects, and changing ecological processes (USDA-FS 2003). Eastern forests in the United States have been subject to unprecedented threat due to invasion by forest pests (Brockhoff and others 2006, Liebhold and others 1995, Lovett and others 2006) that threaten extinction of host species, engineer fragmented landscapes, and add to fuel loads, which increase risk of wildland fires. Disturbances exert a strong influence on forest structure, composition, and diversity (Connell 1978, Huston 1994, White 1979). However, different types of disturbance have different consequences for vegetation. Surface fires, for example, primarily kill small trees and spare the larger individuals (Abrams 2003, Frelich 2002), often slowing the rate of successional replacement. Canopy disturbances such as insect outbreaks primarily damage larger trees and may accelerate the process of succession (Abrams and Scott 1989, Frelich 2002, Lafon and Kutac 2003, Veblen and others 1989).

Stohlgren and Schnase (2006) suggested that risk analysis techniques, including simulation modeling, that are often used in the assessment of health risks and other hazards, are not only applicable to invasive species, but are needed. Forest managers have been increasingly integrating stand-level forecasting tools, such as the Forest Vegetation Simulator (FVS), in the forest decisionmaking process (Dixon 2002). More recently, landscape models that operate

at a scale of 100s to 1000s of km² have begun to be evaluated for use in forest management (e.g., Shifley and others 2000).

LANDIS (Mladenoff and He 1999) is a simulation modeling environment developed to predict forest landscape change over time. It is a spatially explicit landscape-scale ecological simulation model that incorporates both natural (fire, wind, and biological disturbance) and anthropogenic disturbance (harvesting). LANDIS has been adapted for use in a variety of forest management applications. Examples of applications relevant to this study include He and others (2002b) (forest harvesting and fire disturbance), Akcakaya (2001) (risk assessment and landscape habitat models), Shifley and others (2000), Mehtaa and others (2004) (landscape change and management practices), and Gustafson and others (2000) (forest succession and harvesting).

Landscape models offer the unique ability to assess forest process and pattern over broad spatial and temporal scales. Forest managers increasingly need to implement management strategies that incorporate forest sustainability, ecological restoration, wildlife habitat viability, recreational opportunities, and scenic value. Many of these concerns involve broad spatial and temporal scales. The objective of this study is to demonstrate the effectiveness of using LANDIS for forest threat assessment and restoration. To illustrate this, we examine one nonnative invasive insect, hemlock woolly adelgid (HWA) (*Aldelges tsugae* Annand [Homoptera: Adelgidae]) and one indigenous invasive insect, southern pine beetle (SPB) (*Dendroctonus frontalis* Zimmermann [Coleoptera: Curculionidae]). Both insects currently threaten tree species within the Southern Appalachian Mountains of Eastern North America. In this analysis, we present initial results from our work with LANDIS 4.0 and present a framework for using LANDIS II, which will help evaluate the potential impacts of existing and future multiple interacting forest threats in eastern forests.

Background

The SPB and the HWA are two very different forest-damaging insects that inhabit host tree species that exploit

opposite ends of the moisture gradient found in the Southern Appalachian Mountains (Figure 1), although they occasionally occur together at either end of their natural range. We chose these insects to illustrate the utility of LANDIS in investigating forest insect threats because they represent the extreme cases of an indigenous pest that has the potential to cause great damage (SPB) and an invasive pest that has the potential to remove an entire host plant species from eastern forests (HWA).

Southern Pine Beetle Case

In the Southern Appalachian Mountains, xeric slopes and ridges have historically been dominated by yellow pines (*Pinus* spp.). Because altered disturbance regimes have begun to change the appearance of the landscapes, understanding the dynamics of these systems is important to forest managers in implementing management strategies on public lands. On these landscapes, fire and SPB are the two most influential natural disturbance agents. SPB has caused extensive damage to pine forests throughout the Southeastern United States (Coulson 1980, Coulson and others 2004). On Southern Appalachian xeric ridges, SPB colonizes a variety of pine species including pitch pine (*Pinus rigida* Mill.), Virginia pine (*Pinus virginiana* P. Mill.), Table Mountain pine (*Pinus pungens* Lamb.), and occasionally eastern white pine (*Pinus strobus* L.) (Payne 1980). Interactions between available soil moisture and resin flow, the primary tree defense against SPB (Tisdale and others 2003, among others), have long been noted (Hodges and Lorio 1975, Hodges and others 1979) and are likely affected by such landscape characteristics.

Fire and SPB are thought to drive the regeneration of yellow pine forests on xeric ridges in the Southern Appalachians (Harmon 1980, Harrod and others 1998, Williams 1998). Williams (1998) conjectured that SPB and other non-fire disturbances in xeric pine-oak forests will lead toward hardwood domination in the absence of fire. It has further been hypothesized that these communities are maintained in a drought-beetle-fire cycle (Barden and Woods 1976, Smith 1991, White 1987, Williams 1998). Understanding the relationship between fire, SPB, and mesoscale forest

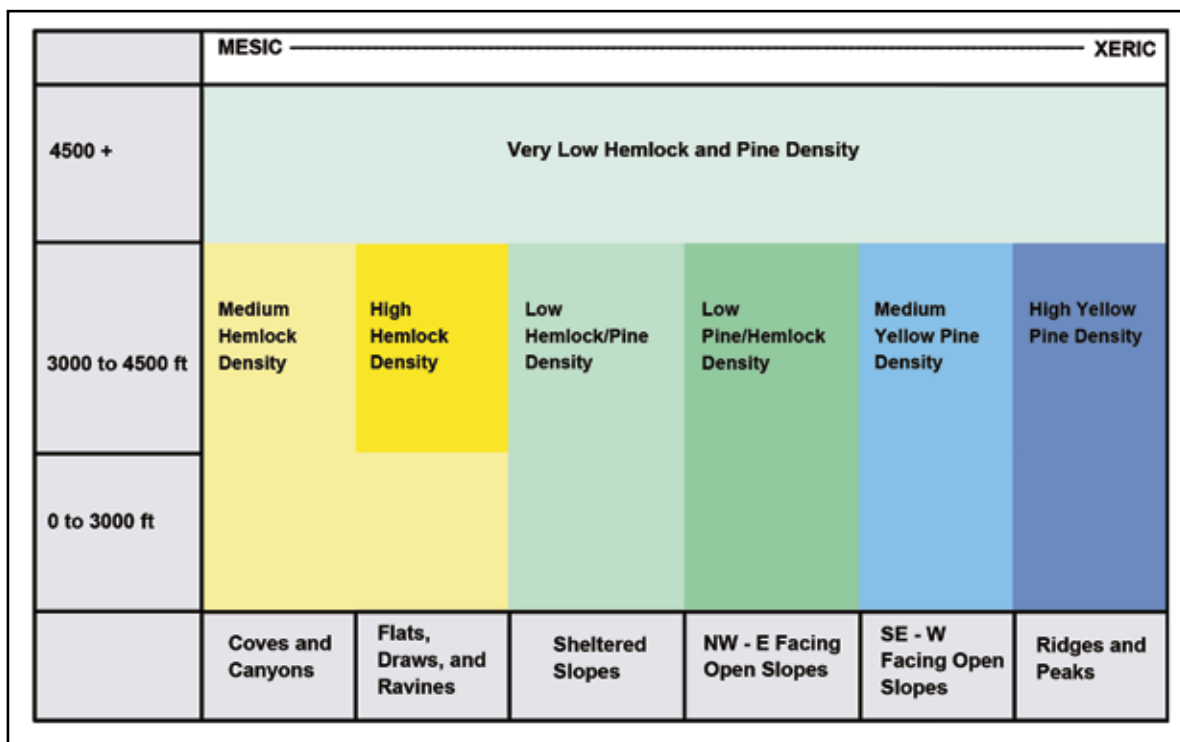


Figure 1—Density of yellow pine and hemlock in the Southern Appalachian Mountains. This illustration depicts the distribution of hemlock and pine density in the southern Appalachian Mountains. The highest densities of hemlocks are found in flats, draws, and ravines at an elevation of 3,000 to 4,500 feet, whereas the highest densities of yellow pines are found on ridges and peaks between elevations of 0 and 4,500 ft (Based on Whittaker 1956).

dynamics can provide direction for forest planners and managers in maintaining and restoring this unique environment.

Hemlock Woolly Adelgid Case

Eastern hemlock (*Tsuga canadensis* (L.) Carr.) and Carolina hemlock (*Tsuga caroliniana* Engelm.) appear in mesic flats, draws, ravines, coves, and canyons of the Southern Appalachian Mountains (Whittaker 1956). Although once more abundant in the forest, hemlock populations declined dramatically approximately 5,500 years ago because of a climatic shift that resulted in summer droughts. These droughts weakened the hemlocks and left them vulnerable to a subsequent widespread insect outbreak (Allison and others 1986, Davis 1981, Haas and McAndrews 2000). In its northern range, canopy gaps were filled by *Acer*, *Betula*, *Fagus*, *Pinus*, *Quercus*, and *Ulmus* (Fuller 1998). Although hemlock did reestablish itself, its recovery may have taken up to 2,000 years and, in many sites, is still not as prominent as it was before the decline (Fuller 1998, Haas

and McAndrews 2000). Now, hemlocks are at risk from the invasive exotic insect pest HWA.

In its native Japan, HWA populations are maintained at low densities on hemlocks (*Tsuga diversifolia* (Maxim.) Mast. and *T. sieboldii* Carr.) by a combination of host resistance and natural enemies (McClure 1992, 1995a, 1995b; McClure and others 2000). The first report of HWA in North America was in the Pacific Northwest in the 1920s; however, western hemlocks were resistant to the adelgid. In the Eastern United States, the first reports of HWA were in 1951 in Richmond, Virginia (Gouger 1971; McClure 1989, 1991). With no natural resistance or natural predators, HWA slowly made its way northeast and has subsequently been moving southwest along the eastern side of the Appalachian Mountains. Little is known about stand-level characteristics that influence HWA susceptibility in the Southeastern United States. However, studies on HWA infestation levels in the northeastern range of this insect noted only latitudinal effects on infestation severity (Orwig and Foster 1998,

Orwig and others 2002). This would seem to suggest that all hemlock stands have the potential of being infested and killed, regardless of site and stand factors.

Methods

Study Area

This study uses a simulated landscape drawn from data approximating the communities and conditions within Great Smoky Mountains National Park. Great Smoky Mountains National Park is a 2110 km² World Heritage Site and International Biosphere Reserve straddling the border between western North Carolina and eastern Tennessee. Great Smoky Mountains National Park serves as an ideal model for this study as most major ecosystems of the Southern Appalachians are represented, and the general topographic distribution of communities and tree species has previously been described (Whittaker 1956).

The Southern Appalachian Mountains, although not representative of all eastern forests, are unique because they represent one of the most biologically diverse regions of the world (SAMAB 1996). A complex system of physiography, environmental site conditions, adaptive life history characteristics, and disturbance history has created a distinctive vegetation structure (Elliott and others 1999). Due to this complexity, Southern Appalachian landscapes contain a variety of community types ranging from mesophytic hemlock-hardwood forests on moist valley floors to yellow pine woodlands on xeric ridges and from low-elevation temperate deciduous forests to high-elevation spruce-fir forests (Whittaker 1956, Stephenson and others 1993). Such high biodiversity areas have been thought by some to act as potential barriers to invasion because of increased competition and by others as at risk of invasion due to the higher potential for suitable habitat niches (Brown 2002, Brown and Peet 2003, Elton 1958, Kennedy and others 2002, Levine and D'Antonio 1999).

Model Description

LANDIS is a spatially explicit computer model designed to simulate forest succession and disturbance across broad spatial and temporal scales (He and Mladenoff 1999a

1999b; He and others 1996, 1999a, 1999b; Mladenoff and He 1999). Whereas LANDIS was originally developed to simulate disturbance and succession on glacial plains in the Upper Midwest (Mladenoff 2004), it has been successfully adapted for use in mountainous areas (He and others 2002a; Shifley and others 1998, 2000; Waldron and others 2007; Xu and others 2004).

LANDIS is raster-based, with tree species (max 30) simulated as the presence or absence of 10-year-age cohorts on each cell. At the site (cell) scale, LANDIS manages species life history data at 10-year time steps. Succession is individualistic and is based on dispersal, shade tolerance, and land type suitability. Disturbances that can be modeled include fire, wind, harvesting, and biological agents (insects, disease) (Sturtevant and others 2004a).

Fire in LANDIS is a hierarchical stochastic processes based on ignition, initiation, and spread (Yang and others 2004). Mortality from fire is a bottom-up process whereby low-intensity fires kill young, fire-intolerant species, whereas fires of higher intensity can kill larger trees and more fire-tolerant species (He and Mladenoff 1999a).

Biological disturbances in LANDIS 4.0 are modeled using the Biological Disturbance Agent (BDA) module. Biological disturbances are probabilistic at the site (cell) level. Each site is assigned a Site Vulnerability (SV) probability value that is checked against a uniform random number to determine if that site has been infected. Site vulnerability can be directly equated with the Site Resource Dominance (SRD) value that ranges from 0 to 1 and is based on species and species age. This value can also be modified by three variables to determine the impact on a given site—Modified Site Resource Dominance (SRDm), Neighborhood Resource Dominance (NRD), and the temporal scale of outbreaks. The functioning of these variables and of the BDA in general is described in detail in Sturtevant and others (2004b).

Simulation Methods

We used LANDIS 4.0 to simulate forest dynamics on a 120-ha idealized landscape. The landscape was a 100 by 120 cell grid with a cell size of 10 m by 10 m, the smallest cell size recommended for use with LANDIS. Using this small cell

size allowed us to operate at approximately the scale of the individual canopy tree, following the logic of gap models. The landscape was divided into 18 individual land types arranged according to the mosaic chart used by Whittaker (1956) to depict the elevation and moisture gradients on the Great Smoky Mountains landscape. The land types are arranged in three rows and six columns. The rows represent (from bottom to top) low (400–915 m)-, middle (916–1370 m)-, and high (1371–2025 m)-elevation zones. The columns represent different topographic moisture classes. Moisture availability decreases from left to right, as follows: (1) coves and canyons; (2) flats, draws, and ravines; (3) sheltered slopes; (4) east- to northwest-facing slopes; (5) southeast- to west-facing slopes; and (6) ridges and peaks. Elevation also influences moisture availability. For example, a low-elevation ridgetop would have drier conditions than a mid-elevation ridgetop. Although the simulated landscape incorporates the full range of environments in the Great Smoky Mountains, our interest in this paper is only on the successional patterns for those land types under the greatest threat by SPB and HWA (Figure 1). We present results for mid-elevation ridges and peaks (SPB) and mid-elevation flats, draws, and ravines (HWA) to illustrate the utility of the model in assessing insect threats.

Results and Discussion

SPB

Our first goal in this study was to investigate the role of fire and SPB in xeric Southern Appalachian landscapes. The modeling projections presented here suggest that the regime of multiple interacting disturbances has important implications for the successional dynamics and vegetation characteristics in yellow pine woodlands of the Southern Appalachian Mountains. When acting alone, fire was projected to create conditions favoring pine presence at levels higher than input, although SPB disturbance acting alone resulted in the removal of yellow pines. Additionally, our model projections suggest that a combination of fire and SPB disturbance creates sustainable yellow pine communities over the long term. This conclusion is consistent with the hypothesis that fire and SPB are part of a disturbance

regime that maintains yellow pine woodlands (Harrod and others 1998, 2000; Lafon and Kutac 2003; White 1987; Williams 1998) (Figure 2).

The results of this study yield several conclusions that are important to forest managers when undertaking restoration efforts. First, our projections suggest that Table Mountain pine (*Pinus pungens*), more than any other species, thrives when in a disturbance regime combining SPB and fire on xeric sites. Because Table Mountain pine is a Southern Appalachian endemic, it is also important for biodiversity conservation (Zobel 1969). These factors suggest that Table Mountain pine could be a species of particular interest for restoration efforts on mid-elevation ridges and peaks in the Southern Appalachians. Our results apply to the restoration of such stands and suggest that periodic burning will be required to maintain the compositional and structural integrity of stands affected by SPB. This conclusion is substantiated by empirical analogue (e.g., Harrod and others 1998, 2000; Lafon and Kutac 2003).

Hemlock Woolly Adelgid

Our second goal was to investigate the impacts of HWA on species composition in the Southern Appalachian Mountains. The results from this study are preliminary, but do show a reduction in hemlock and subsequent replacement by hardwoods (Figure 3). In particular, we see replacement of hemlocks with basswood (*Tilia* spp.), sugar maple (*Acer saccharum* Marsh.), yellow buckeye (*Aesculus octandra* Marsh.), yellow birch (*Betula alleghaniensis* Britton) and northern red oak (*Quercus rubra* L.). These results may not be ecologically correct, as we would anticipate rhododendron (*Rhododendron* spp.) already present in the understory or several potential nonnative invasive species filling many of the gaps created by hemlock removal (Figure 4). Rhododendron and thick shrub cover, in general, have been shown to neutralize tree regeneration in canopy gaps (Beckage and others 2000). Riparian areas in the Southern Appalachians, where we find most hemlocks, have also been shown to contain high exotic species cover and diversity (Brown and Peet 2003). The discrepancies in the landscape approach can be corrected easily by incorporating finer resolution gap models.

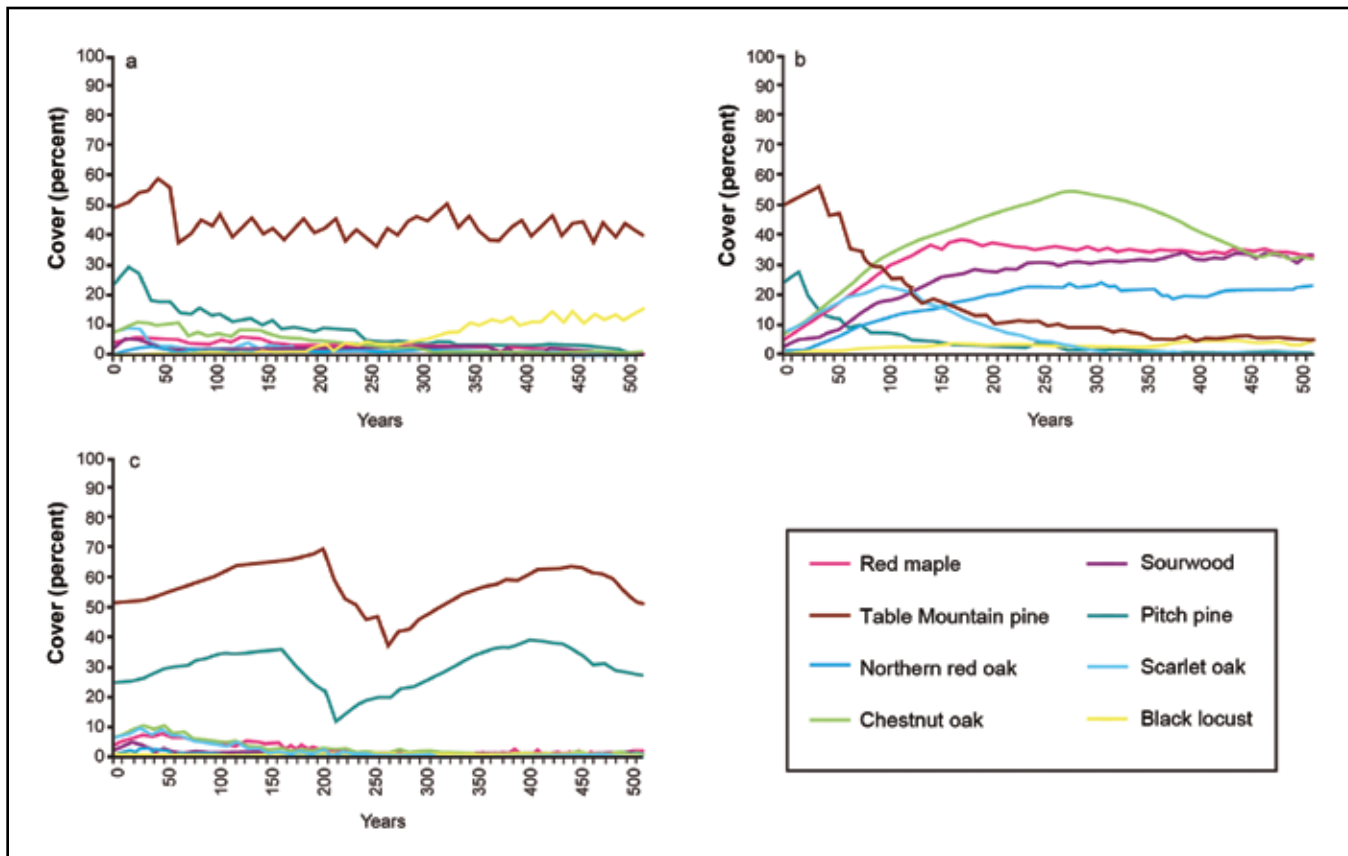


Figure 2—Successional trajectory of mid-elevation Ridges and Peaks. The following three graphs show the successional trajectory of mid-elevation ridges and peaks and represent percentage cover on the y-axis and model run year on the x-axis, which ends at year 500 for mid-elevation ridges and peaks. The fire and SPB scenario (a) show a continued dominance of Table Mountain pine but a reduction in pitch pine. The SPB-only scenario (b) shows a replacement of both pine species with hardwoods. The fire-only scenario (c) demonstrates a continuation of Table Mountain pine and pitch pine. Graphs reflect the percentage of cells occupied on the land type by each species.

Discussion

Through this study, we have demonstrated that different forest pests in different ecological regions, within the same geographic bounds, require different management strategies. If the desired outcome were the maintenance of Table Mountain pine-pitch pine communities, managers would be warranted in using prescribed burning or allowing for natural fires to burn without suppression along with SPB chemical control measures on xeric mid-elevation ridges and peaks. If the maintenance of pitch pine were not considered important, then burning alone would be an acceptable strategy. Although other silvicultural practices could recreate these conditions in the short run, burning is necessary to maintain these conditions as Table Mountain pine needs stand and site disturbance, light, and heat for successful

regeneration (Della-Bianca 1990). On the other hand, there are no immediate controls for HWA. Whereas the imminent destruction of hemlocks by HWA evokes parallels to the chestnut blight fungus (*Cryphonectria parasitica*), which has decimated American chestnuts, there is a major difference. Chestnut blight can survive quite well on the deadwood of a variety of species, and, hence, it will always be present in the environment—therefore constraining the ability of any viable new American chestnut populations. Hemlock woolly adelgid, on the other hand, requires the presence of either eastern or Carolina hemlock to survive in Eastern North America (McClure 1987). One management strategy would simply be to save nursery stocks of hemlocks in a controlled environment for replanting once the HWA has destroyed all the naturally occurring hemlocks and then, itself, perished due to a lack of viable hosts.

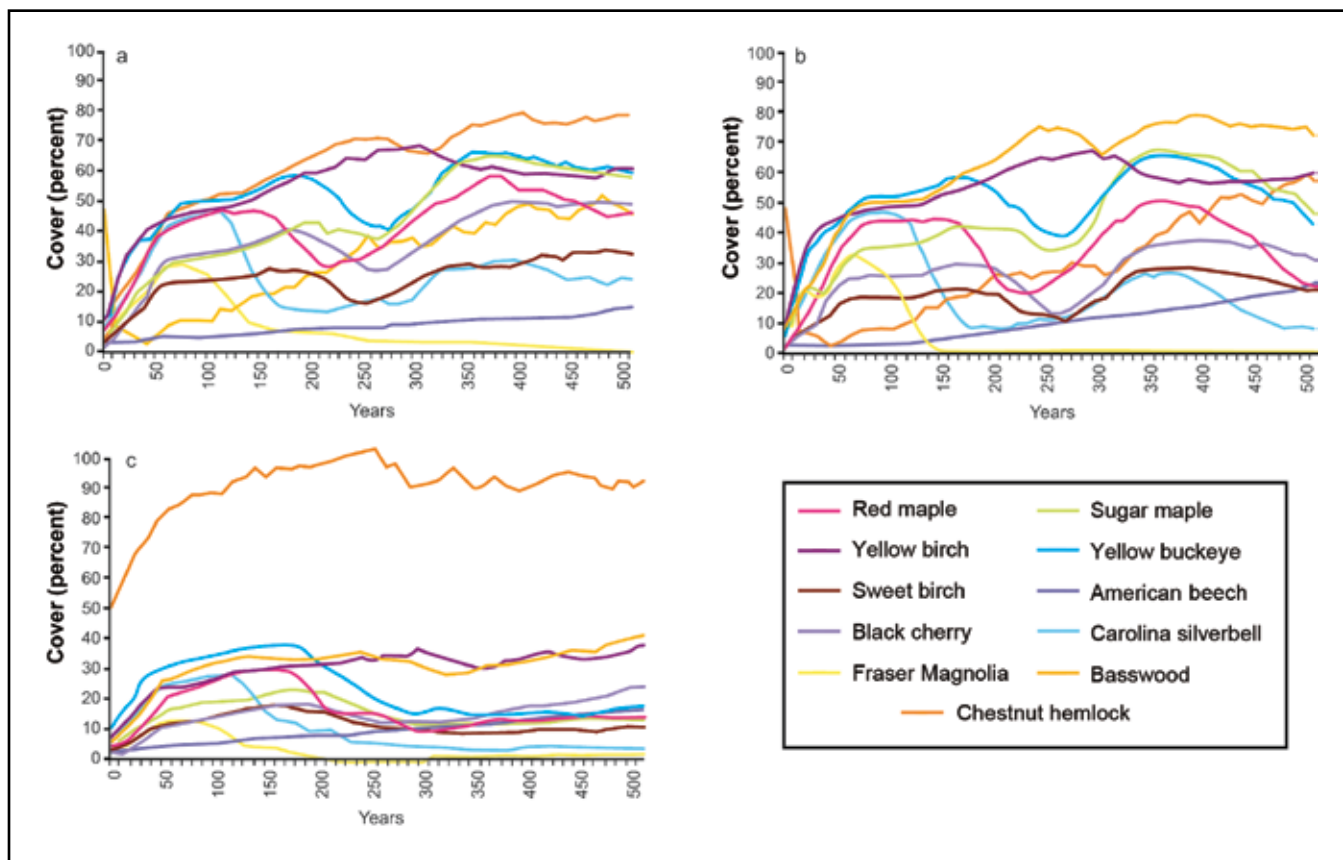


Figure 3—Successional trajectory of mid elevation Flats, Draws, and Ravines. The following three graphs represent percentage cover on the y-axis and model run year on the x-axis, which ends at year 500 for mid-elevation flats, draws, and ravines. In both the fire and HWA scenario (a) and the HWA-only scenario (b), eastern hemlock is replaced by hardwood species, particularly basswood, yellow buckeye, yellow birch, and sugar maple. In the fire-only scenario (c), hemlock maintains its dominance due to the very infrequent fire return on mesic sites. Graphs reflect the percentage of cells occupied on the land type by each species.

We set out to describe a modeling framework for assessing the impacts of SPB and HWA herbivory on forests. The advantage of using LANDIS is that all of these processes and outputs are described and captured in a simple, tractable, and transparent modeling environment. In the context of periodically abundant pests or invasive species, this transparency and simplicity are important because there is often a need for reactive and immediate research into potential impacts. One of the great strengths of LANDIS is that because a well-described and proven model framework already exists, parameterization and hence, model outputs, can be achieved relatively easily by drawing on published literature, expert knowledge, and practical experience. The adoption of a proven model also allows a truly comparative study between the assessments of the impact of different pests. For this initial study, we chose to

illustrate the temporal dynamics of vegetation in response to insect outbreaks within the most vulnerable land types in the Southern Appalachians. The spatial dynamics of vegetation change and insect outbreaks across land types is another key feature of this real-world problem that affects the distribution of pests and the damage they cause and the successional dynamics of impacted vegetation leading to restructuring of the forest. Although the broader spatial patterns and processes were not within the scope of this study, we are currently using LANDIS to explore these issues.

Directions for Future Research

Recently, LANDIS II has been released. In a major change from LANDIS, the life history parameters have been updated to include both minimum and maximum age of resprouting as well as a postfire resprout function, which

allows for serotiny or resprout. LANDIS II also allows for the calculation of aboveground live and dead biomass (as kg/ha) and tracks woody and leaf litter dead biomass. Biomass can also be used as an alternative to the original succession function using species age. Disturbances that can be modeled follow those of LANDIS. In LANDIS II, each ecological process operates on its own individual time step (units: year). For example, fire may operate at a 5-year time step, whereas SPB occurs at a 7-year time step and HWA at a 1-year time step. Also, while LANDIS was limited to 30 species, LANDIS II can have an unlimited number of plant species. Changing to LANDIS II will undoubtedly aid in sorting out some of the problems in the HWA case, as we will be able to incorporate species such as rhododendron and invasive species, as well as have the ability to model HWA annually rather than at 10-year increments.

The Harvesting module in LANDIS II allows for both tree removal and planting within user-defined management areas. There are several functions for species removal. Clearcut removes all species within a stand. Individual species can be removed either as all, a percentage of the species, the oldest cohort within the species, all cohorts except the oldest, the youngest cohort, and all cohorts except the youngest cohort. There are several other differences between the two-model versions. First, climate change scenarios are now possible as land type parameters can be altered according to temporal grouping. Second, the order of disturbances can be randomized so that within a series of runs you might, for example, have fire run either before or after the BDA is run. Finally, LANDIS II is modular. This modularity allows for the relatively easy incorporation of new modules (such as ice storm disturbance) as well as the alteration of existing modules to meet research needs.

Our goal for future research is to test the capacity of LANDIS II using a landscape modeling environment to evaluate changes in composition and structure of Eastern United States forests undergoing multiple interacting environmental threats. Specifically, we are adapting LANDIS II to model the combined effects of key invasive biological disturbance agents and nonnative invasive plant species on the composition and structure of Southern Appalachian

forests. This will allow us to determine the effects of changes in forest structure and composition on fire regimes, biodiversity, and wildlife habitat, to investigate strategies for restoring key ecosystems that may be significantly impacted by multiple-threat interactions, and to test contemporary ecological theory, such as the relationship between biodiversity and invasibility. Also, by incorporating Gap models, we will be able to address additional questions beyond those for which LANDIS II is suitable.

Gap models simulate the establishment, growth, and death of individual trees on small plots (Perry and Enright 2006). Unlike LANDIS, they do not consider the influence of landscape structure on disturbance and succession. Their value lies in their ability to simulate interactions among individual plants in a detailed, mechanistic way. Such local-scale interactions between individual plants that vary in size, growth rate, shade-tolerance, moisture/nutrient requirements, and other attributes are thought to govern successional processes, including exotic species invasions (Huston 2004, Shea and Chesson 2002). Gap models also are capable of representing the interactions between different plant functional types, e.g., trees and shrubs. Although most commonly applied to problems of forest succession, gap models have been used to investigate the dynamics of herbaceous vegetation as well (e.g., Peters 2002). By employing a combined approach of gap and landscape modeling, we will be able to rectify the problems encountered in the HWA study and provide more detailed succession projections.

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Spread of Invasive Plants From Roads to River Systems in Alaska: A Network Model

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Abstract

Alaska has relatively few invasive plants, and most of them are found only along the state's limited road system. One of the most widely distributed invasives in the state, *Melilotus alba* Medik., or sweetclover, has been sown both as a forage crop and as a roadside stabilization species. *Melilotus* has recently been found to have moved from roadsides to the flood plains of at least three glacial rivers. This species has aggressively colonized the lower Stikine River flood plain and occurs there in dense, monospecific stands. It is at an earlier stage of colonization of the Matanuska River and Nenana River flood plains as well. We are developing a network model to examine the spatial relationships among roads, river crossings, and downstream public lands of high conservation significance in interior and south-central Alaska. In 2005 and 2006, we documented the distribution of *Melilotus* on roadsides and near river crossings in this part of the state; we present these data combined with other records of *Melilotus* distribution downloaded from the Alaska Exotic Plant Information Clearinghouse database. When considered together, the distribution data and the network model identify certain road-river interfaces as critical control points for preventing the movement of *Melilotus* toward public land downstream. To illustrate how the model might be used by land managers, results are presented for major crossings upstream of the Kanuti National Wildlife Refuge. When complete, the network model will function as a generally applicable tool to identify the critical control points upstream from land under various ownerships and

for any future invasive species that can disperse via roads and river networks in Alaska.

Keywords: Invasive plants, Alaska, modeling, networks, roads.

Introduction

Alaska has relatively few invasive plants. The state's isolation, lack of development, and cold climate limit the introduction and success of many invasive species. Invasive species often disperse along road networks (Gelbard and Belnap 2003; Koch and Smith, this volume; Lugo and Gucinski 2000; Parendes and Jones 2000), and Alaska has only 0.02 miles of road per square mile of land area, compared to California's 1.08 miles (Jeff Roach. 2002. Personal communication. Alaska Department of Transportation, 2301 Peger Rd., MS-2550, Fairbanks, Alaska 99709).

Melilotus alba Medik., or sweetclover, has recently become a species of concern in Alaska. *Melilotus* has been grown as a crop in some parts of the state and used in a number of mining and industrial area reclamation efforts. *Melilotus* is widely distributed along roadsides around Alaska, a result of both intentional sowing as a roadside stabilization species, and unintentional transport of seed via contaminated soil and gravel. In recent years, *Melilotus* appears to be spreading rapidly along these roadsides. *Melilotus* has been found on roadsides in Denali National Park and Preserve and near several other national parks in Alaska, lands of unquestionably high conservation value (Densmore and others 2001).

Melilotus has been found on several glacial flood plains. It has aggressively colonized the flood plain of the lower Stikine River in southeast Alaska, the lower Matanuska River in south-central Alaska, and portions of the middle and lower Nenana River in interior Alaska (Figure 1). On all three flood plains, *Melilotus* is a major component of the vegetation on recently deposited silt and gravel bars. Although the sources of these populations aren't known for certain, it's likely that *Melilotus* spread onto the flood plains from roads, mines, and agricultural developments upstream. There are vast areas of glacial silt flood plain in interior



Figure 1—The map of Alaska shows the flood plains of three rivers (the Stikine, the Matanuska, and the Nenana) known to have been colonized by *Melilotus alba*. The Nenana is a tributary of the Tanana, which, in turn, is the largest tributary of the Yukon.

Alaska. For example, about 40 percent of the entire Yukon Basin is drained by glacial melt-water dominated rivers (Brabets and others 2000; Paul Schuster. 2006. Personal communication. USGS, National Research Program, 3215 Marine Street, Suite E-127, Boulder, CO 80303). Though many of Alaska's most valuable public lands are located off the road system, they may be vulnerable to invasion by species that gain access to river flood plains from upstream roadside environments.

This project has two objectives. The first is to document the current distribution of *Melilotus alba* on roadsides and on river flood plains near bridges in interior and south-central Alaska. The second is to develop a network model to examine the spatial relationships among roads, rivers, and public lands of high conservation value in the same part of the state. We use the *Melilotus* distribution data as a case

study in the application of the network model, focusing on major crossings upstream of the Kanuti National Wildlife Refuge as an example of how land managers might use model output to prevent the introduction of new invasive plants to the lands they manage.

Methods

Case Study Using *Melilotus alba*

In 2005 and 2006, we surveyed 1,780 miles of eight major highways in interior and south-central Alaska for *Melilotus alba*. These surveys focused on two areas: roadsides and major river crossings. For the roadside surveys, we stopped every 10 miles at State of Alaska Highway milepost markers, and recorded whether or not *Melilotus* was visible on the roadside. We included both sides of the road and as far as we could see in either direction in these rapid

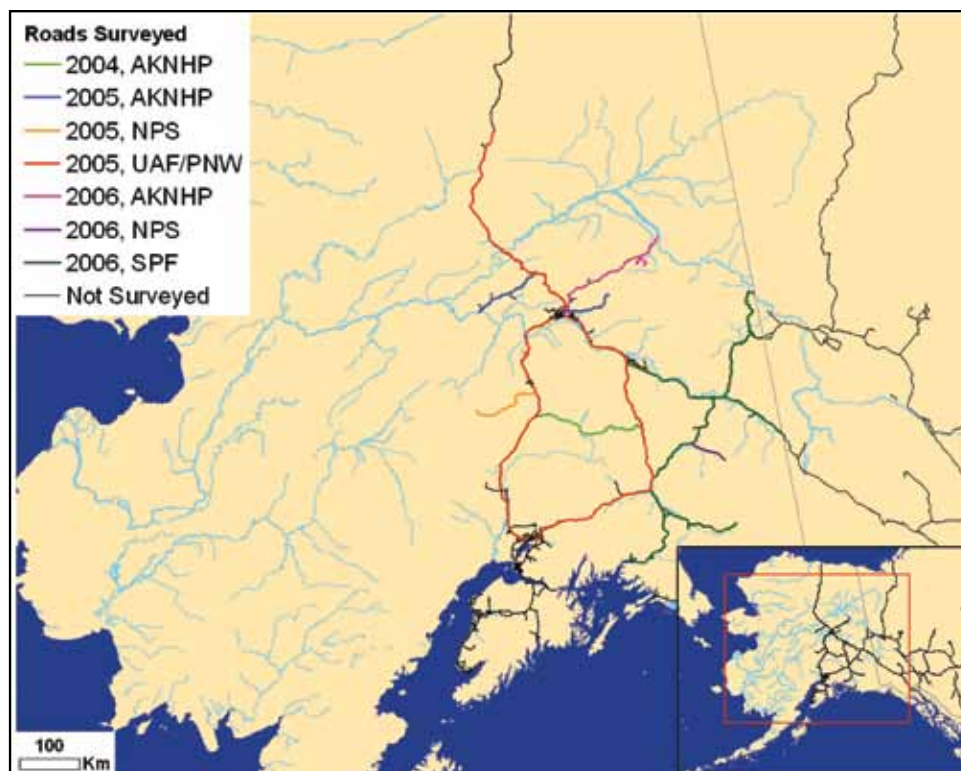


Figure 2—Alaska highways surveyed for *Melilotus alba* between 2004 and 2006. AKNHP = Alaska Natural Heritage Program, NPS = National Park Service, UAF/PNW = University of Alaska Fairbanks/USDA Forest Service, Pacific Northwest Research Station, SPF = USDA Forest Service, State and Private Forestry.

assessments. We acquired *Melilotus* distribution data for several additional highways from the Alaska Exotic Plant Information Clearinghouse database (Alaska Natural Heritage Program, n.d.) (Figure 2).

In addition, we surveyed for the presence or absence of *Melilotus* at 192 major crossings along these highways. We did not survey other types of road-river interfaces (see below). At each crossing, we recorded the presence or absence of *Melilotus* on the roadside immediately adjacent to the bridges and on natural flood-plain surfaces below and in the vicinity of the bridge. We considered both upstream and downstream flood-plain surfaces in making the flood-plain assessments.

We characterized each downstream flood-plain surface in the vicinity of the bridge on its apparent vulnerability to invasion by *Melilotus*, based on our observations of the rivers shown in Figure 1. For example, densely vegetated surfaces were given a score of low, whereas bare silt or

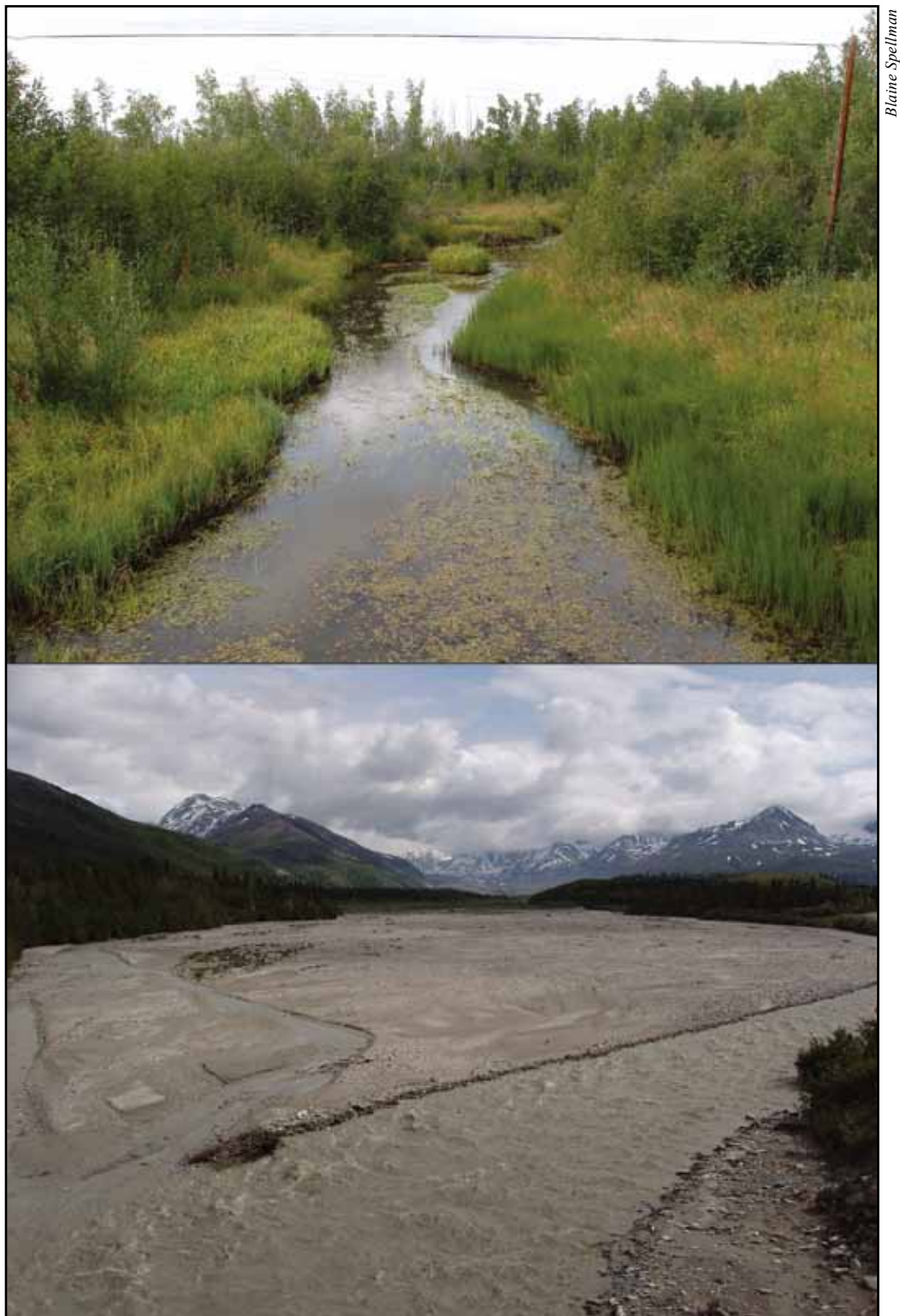
gravel flood plains were considered high in their vulnerability to invasion (Figure 3).

Network Model

The network model examines spatial relationships among roads, river networks, and public land in interior and south-central Alaska. Our analysis is supported by several geographic information system (GIS) data layers. The river network is modeled using the National Hydrography Data set (USGS 1999); the Alaska Department of Transportation global positioning system (GPS) centerline data set provides a detailed road network (Alaska DOT 2006); and the boundaries of federal and state lands were obtained from an Administrative Boundaries data set (Alaska DNR 2001).

National Hydrography Data Set (NHD)

The NHD is managed by the U.S. Geological Survey (USGS) and Environmental Protection Agency (EPA). It



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Figure 3—Flood plain vulnerability to invasion is shown in these examples of flood plains characterized as having low vulnerability to invasion by *Melilotus alba* (top) and as being highly vulnerable to invasion by *Melilotus* (bottom). The top photo shows Fish Creek at the Parks Highway Crossing; the bottom photo shows Lower Miller Creek at the Richardson Highway Crossing.

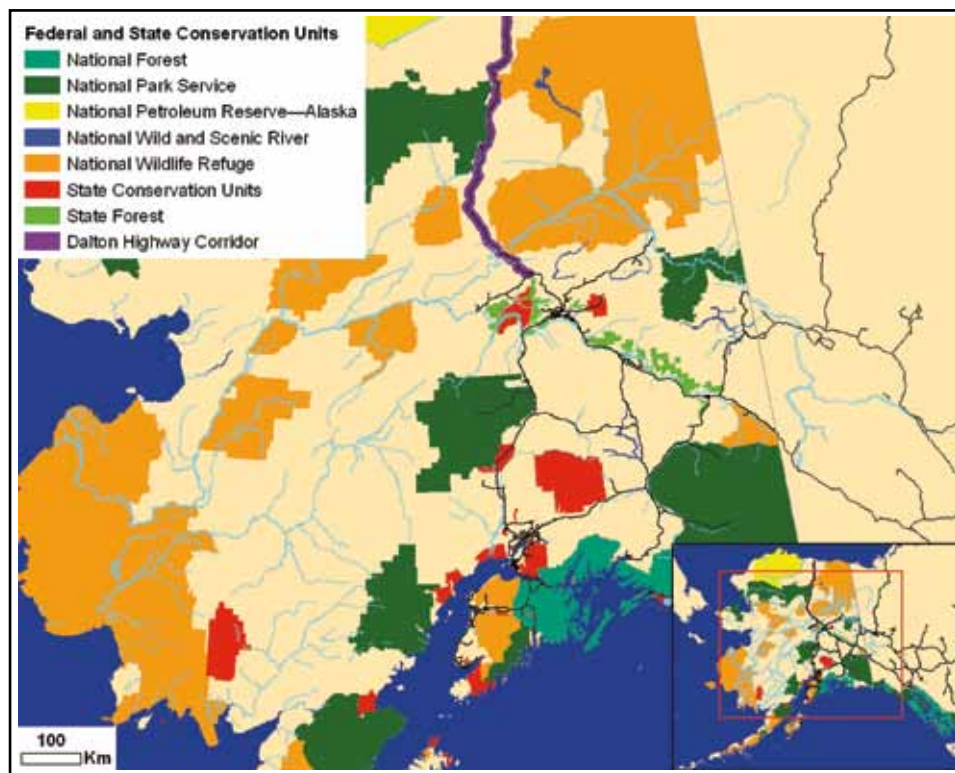


Figure 4—Land classification boundaries identified on the Administrative Large Parcel Boundaries data set from Alaska Department of Natural Resources. We selected parcel types to include in our model by reviewing the agency affiliations of land managers who have been active in Alaska's Committee for Noxious and Invasive Plant Management (CNIPM) over the last few years.

is a comprehensive set of digital spatial data that contains information about surface water features such as lakes, ponds, streams, rivers, springs, and wells. Within the NHD, surface water features are combined to form reaches, which provide the framework for linking water-related data to the NHD surface water drainage network. These linkages make it possible to analyze and display these water-related data in upstream and downstream order (USGS 1999). We started with the assumption that all migration of invasive plants on the river system would be downstream. We then used the NHD as our base layer for river networks because it covers the entire state at a fine scale (1:63360), and also incorporates flow network information allowing us to efficiently identify all river reaches downstream from individual road crossings.

Road Network

There is no comprehensive road network data set available for Alaska, but the Alaska Department of Transportation

(ADOT) provided a draft version of a new GPS road center-line network. It covers the contiguous highway system plus Kodiak and Cordova. Some roads are not included in this data set—for example, the Denali National Park road, State of Alaska-administered logging roads, and private roads—but these road data are the best currently available.

Public Conservation Units

We began with the land classification boundaries identified on the Administrative Large Parcel Boundaries data set from December 20, 2001, produced by the Alaska Department of Natural Resources (Alaska DNR 2001) (Figure 4). We selected parcel types to include in our model by reviewing the agency affiliations of land managers who have been active in Alaska's Committee for Noxious and Invasive Plant Management (CNIPM) over the last few years. Managers who are already aware of and concerned about invasive plants on the public lands they manage may be most likely to benefit from the information provided by

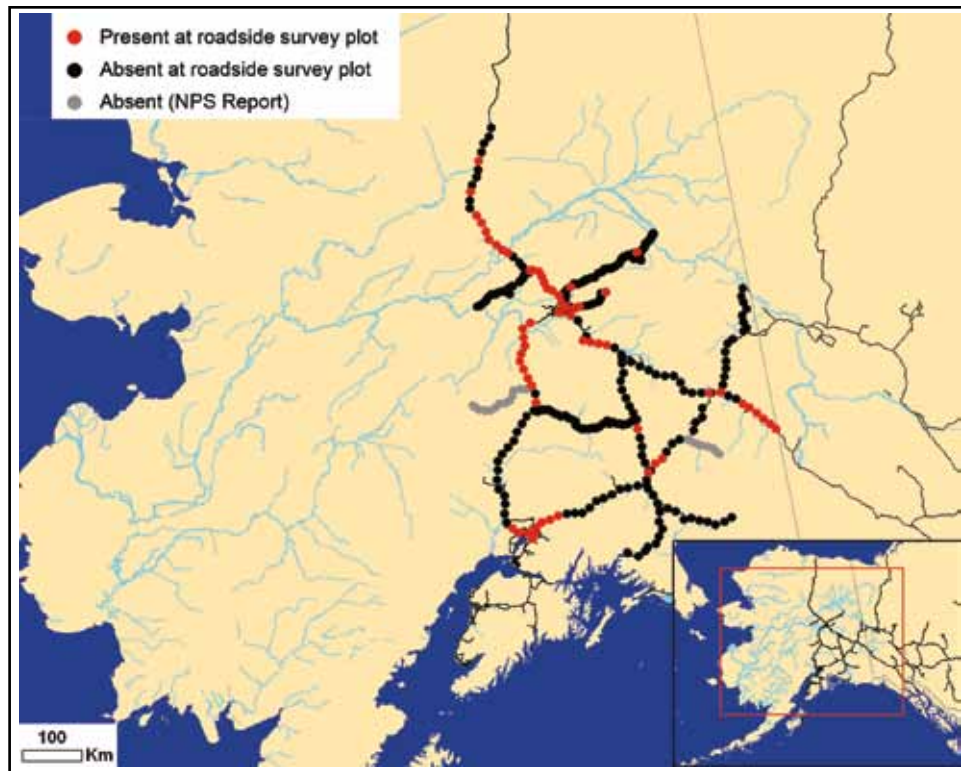


Figure 5—Distribution of *Melilotus* on roadsides, 2004–2006. Most of these data were collected at milepost markers spaced 10 miles apart.

this work. We refer to that list of parcel types in aggregate as conservation units. Land management objectives and agency attitudes and approaches toward invasive species differ widely among these conservation units.

Road-River Interfaces—

The interface between roads and rivers (or streams) can take several forms. Roads can cross, run parallel to, and dead-end at rivers. We defined crossings as intersections between the NHD-derived streams and rivers network and the ADOT road centerline network. At this phase, our analysis does not include the parallel or dead-end road-river interfaces, except for rare occasions where small spatial mismatches cause the river and road linework to intersect. The NHD subregion 1904 (the Alaska portion of the Yukon River drainage) yielded a total of 919 crossings. Where roads cross major rivers or streams, there is typically a bridge; we refer to these points on the landscape as major crossings. Small streams or ephemeral drainages typically pass beneath roads in culverts; we refer to these points as

minor crossings. To associate river reaches downstream from crossings with conservation units, the Administrative Large Parcel Boundary data set was intersected with river reaches downstream from crossings (after the river reaches were buffered by 100 m).

Example

Combining the network model with invasive species distribution data yields information that can help managers determine critical control points for particular conservation units. We provide model output related to the Kanuti National Wildlife Refuge (NWR) as an example of this application. Kanuti NWR is entirely located in NHD subregion 1904, and although it has no direct road access, there are 13 major and 112 minor crossings upstream of the refuge, all on the Dalton Highway. We surveyed only the southernmost 10 of the major crossings, because as we worked north, *Melilotus* had disappeared from the roadsides by the 10th major crossing.

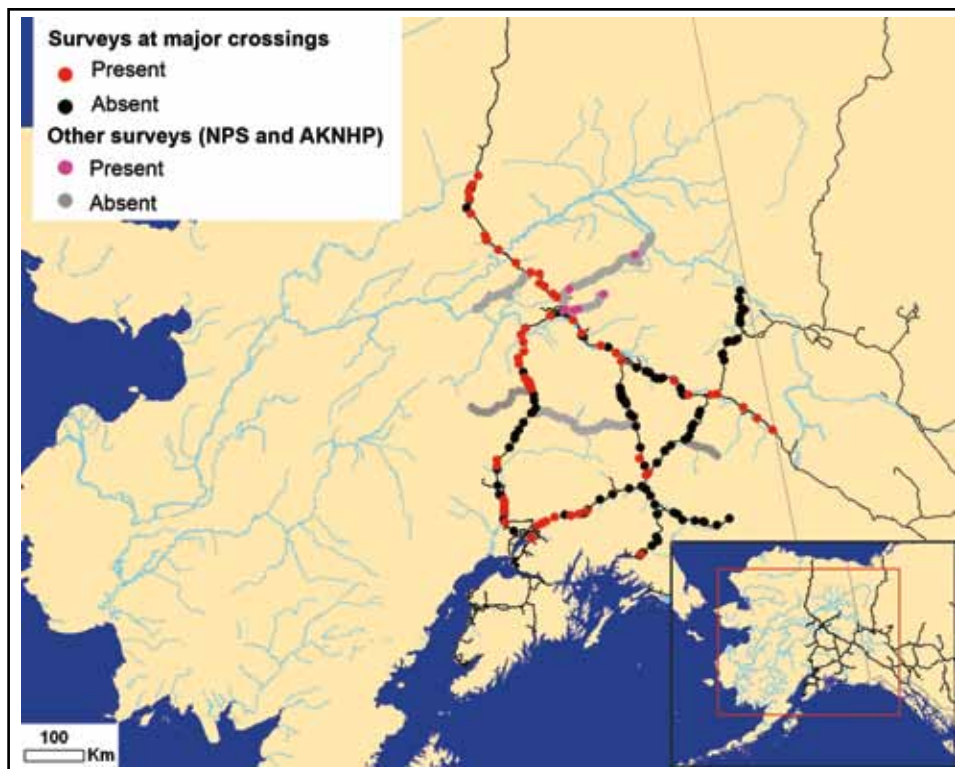


Figure 6—Distribution of *Melilotus* on roadsides immediately adjacent to major crossings, 2004–2006, are displayed. We defined crossings as intersections between the NHD-derived streams and rivers network and the ADOT road centerline network. Crossings with bridges are referred to as major crossings.

Results

Melilotus Distribution

We found *Melilotus* at many points along the highways that we surveyed (Figure 5), with roadsides in the Fairbanks area most consistently colonized. Notably, little or no *Melilotus* was found along several major highways, including the Taylor Highway, the Denali Highway, and the McCarthy Road. In addition, we found *Melilotus* on the roadside immediately adjacent to 64 major crossings (Figure 6).

Of the total of 192 major crossings surveyed, 17 had *Melilotus* growing on a natural flood-plain surface (Figure 7). Most of these were light infestations, but four crossings had moderate to heavy infestations on the flood-plain surface. All 13 of these crossings also had *Melilotus* growing on the roadside immediately adjacent to the bridge.

Though we found *Melilotus* growing on flood-plain surfaces immediately upstream of several crossings, we do not interpret this to mean that this species is spreading

upstream. Instead, all cases where *Melilotus* was found upstream involved heavily colonized individual rivers with multiple road crossings. In these cases, *Melilotus* had spread all along the river from sources upstream of all the surveyed crossings.

Network Model

The network model is a work in progress. At this writing, we have compiled NHD subregion 1904 (the Alaska portion of the Yukon River watershed). Compilation of NHD subregion 1902 (the Matanuska River and Susitna River watersheds) is underway.

Example

Of the 10 major crossings we surveyed leading to Kanuti National Wildlife Refuge, none had *Melilotus* on a natural flood-plain surface either upstream or downstream of the crossing (Figure 8, Table 1). Six had *Melilotus* on the roadside immediately adjacent to the crossing. We characterized

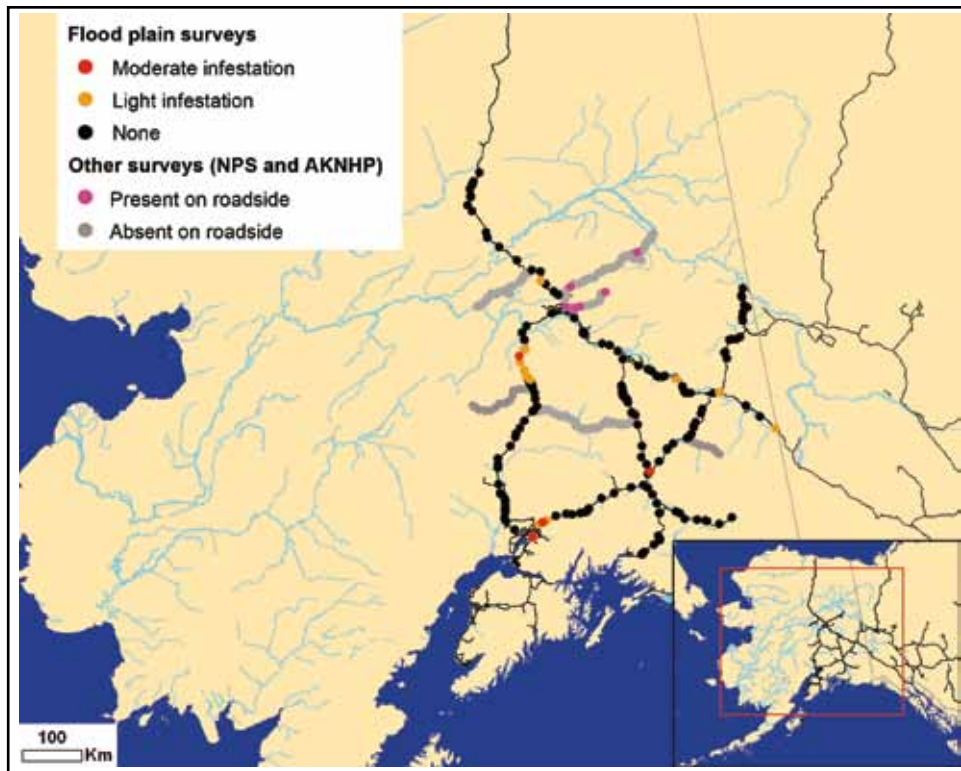


Figure 7—Distribution of *Melilotus* on natural flood-plain surfaces at major crossings, 2004-2006, is shown below. We defined crossings as intersections between the NHD-derived streams and rivers network and the ADOT road centerline network. Crossings with bridges are referred to as major crossings.

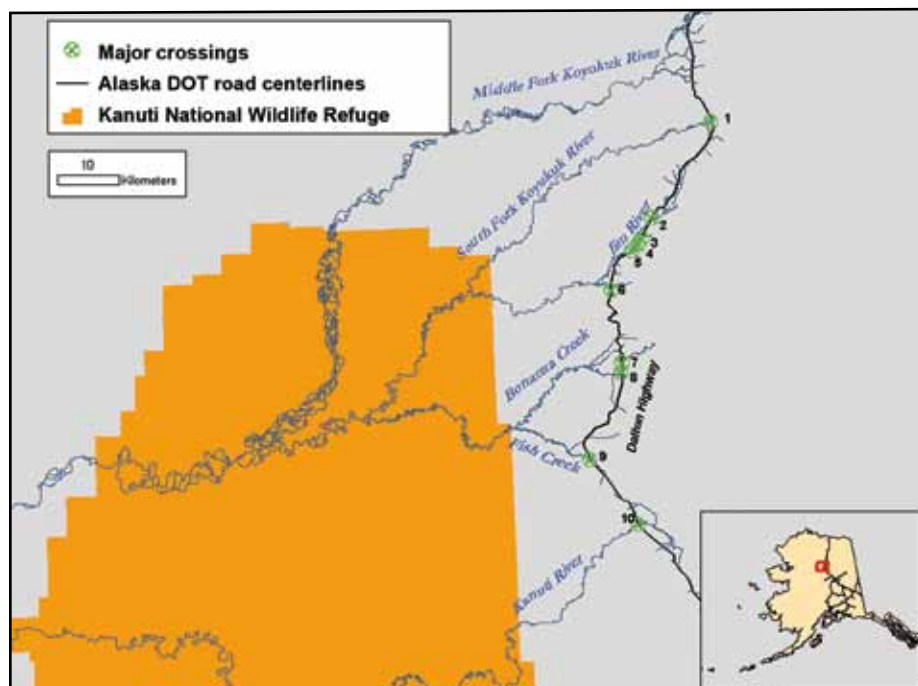


Figure 8—Major crossings upstream of the Kanuti National Wildlife Refuge are outlined below. We defined crossings as intersections between the NHD-derived streams and rivers network and the ADOT road centerline network. Crossings with bridges are referred to as major crossings; identification numbers shown here correspond to numbers in Table 1. There are 13 major and 112 minor crossings upstream of the refuge, all on the Dalton Highway. We surveyed only the southernmost 10 of the major crossings because *Melilotus* had disappeared from the roadsides of the Dalton Highway by the crossing labeled here as 1.

Table 1—*Melilotus* survey results for major crossings upstream of Kanuti National Wildlife Refuge

Number on Figure 8	Highway	Milepost	Crossing name	Distance to Kanuti <i>km</i>	Upstream	Downstream	Roadside	Vulnerability
1	Dalton	156	S_FORK_KOYUKUK_RIVER	80	None	None	Light	High
2	Dalton	144	JIM_RIVER_3	52	None	None	None	High
3	Dalton	142	DOUGLAS_CREEK	46	None	None	None	Low
4	Dalton	142	JIM_RIVER_2	45	None	None	None	Mod
5	Dalton	140	JIM_RIVER_1	42	None	None	None	Low
6	Dalton	135	PROSPECT_CREEK	36	None	None	Light	Low
7	Dalton	125	N_FORK_BONANZA_CREEK	50	None	None	Mod	Low
8	Dalton	124	S_FORK_BONANZA_CREEK	47	None	None	Light	Mod
9	Dalton	113	FISH_CREEK	27	None	None	Light	Mod
10	Dalton	105	KANUTI_RIVER	38	None	None	Light	Low

Milepost refers to state highway milepost marker nearest the crossing. Distance to Kanuti gives the total river distance in kilometers from the Dalton Highway to the eastern boundary of the refuge. The last four columns give data collected in 2005 as part of our *Melilotus* survey.

five of the crossings as moderately or highly vulnerable to invasion by *Melilotus*. Crossings of the Dalton Highway were from 27 river km to 80 km upstream of the eastern boundary of the refuge.

Discussion

The NHD and ADOT road network data sets are both works in progress, and some errors were identified in the NHD data. For example, several small drainages such as the Little Goldstream Creek were isolated from the rest of the hydrography network, and there were several interruptions in the Yukon River flow. These are the best data currently available on a statewide basis, however. We expect that in the future, work in other disciplines will be referenced to the NHD, allowing more detailed data to be incorporated if they become available. For example, additional data may become available regarding water chemistry and flood-plain substrate characteristics.

We are working to link the roads, crossings, reaches, roadsheds, and conservation units so that the data can readily be summarized based on any of these factors and so that an end user can identify a feature (for example, a national park or a national wildlife refuge) and return the crossings, upstream road segments, and river reaches that are associated with it.

In our example with the Kanuti National Wildlife Refuge, our work identified six major crossings where *Melilotus* was found on the roadside immediately adjacent to the bridge. One of these crossings was ranked as highly vulnerable to invasion at the crossing, and two were ranked as moderately vulnerable. Refuge managers could prioritize monitoring and control efforts based on these results. Taken together, the network model and the case study are an effective means of identifying certain river and stream crossings as critical control points for preventing the movement of *Melilotus* toward particular land ownerships downstream.

The general applicability of our network model is demonstrated by examining the structure of data columns in Table 1. The first five columns contain data describing the spatial relationship between roads, crossings, and bridges for the Kanuti National Wildlife Refuge. Other data, such as precise latitude and longitude coordinates and

digital photos, are also available for each crossing. The last four columns contain data from a single survey for a single invasive species. Any future surveys for *Melilotus* or other invasive plant species at these crossings can be directly incorporated using our network model framework and will provide up-to-date information to land managers in Kanuti.

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English Equivalents

When you know:	Multiply by:	To find:
Nanometers (nm)	3.94×10^{-8}	Inches (in)
Millimeters (mm)	0.0394	Inches
Centimeters (cm)	.394	Inches
Meters (m)	3.28	Feet (ft)
Cubic meters (m ³)	35.3	Cubic feet (ft ³)
Meter per second (m/s)	2.24	Miles per hour (mph)
Kilometers (km)	.621	Miles (mi)
Kilometers per hour (kph)	.621	Miles per hour
Hectares (ha)	2.47	Acres (ac)
Microliters (μL)	.0000338	Ounces (fluid)
Milliliters (mL)	.0338	Ounces (fluid)
Grams (g)	.0352	Ounces
Kilograms (kg)	2.205	Pounds
Kilograms per cubic meter (kg/m ³)	.0624	Pounds per cubic foot (lb/ft ³)
Kilograms per hectare (kg/ha)	.893	Pounds per acre (lb/ac)
Megagrams per hectare (Mg/ha)	.446	Tons per acre
Kiloequivalents per hectare (keq/ha)	.405	Kiloequivalents per acre
Trees per hectare	.405	Trees per acre
Micrograms per gram (μg/g)	1	Parts per million
Degrees Celcius (C)	$1.8C + 32$	Fahrenheit (F)

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