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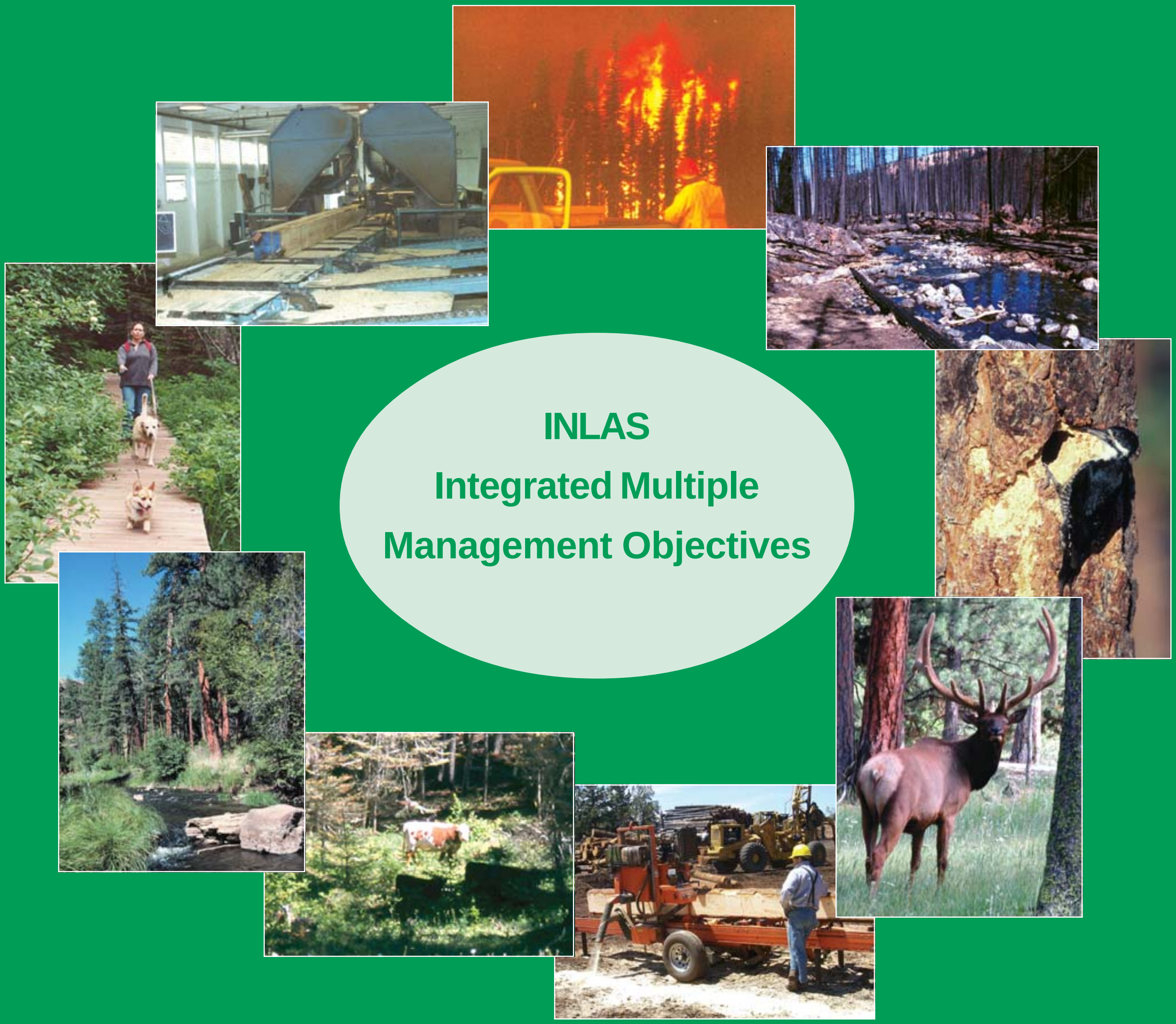
September 2004

Methods for Integrated Modeling of Landscape Change:

Interior Northwest Landscape Analysis System



INLAS Project
USDA FS PNW Research Station

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Abstract

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The Interior Northwest Landscape Analysis System (INLAS) links a number of resource, disturbance, and landscape simulations models to examine the interactions of vegetative succession, management, and disturbance with policy goals. The effects of natural disturbance like wildfire, herbivory, forest insects and diseases, as well as specific management actions are included. The outputs from simulations illustrate potential changes in aquatic conditions and terrestrial habitat, potential for wood utilization, and socioeconomic opportunities. The 14 chapters of this document outline the current state of knowledge in each of the areas covered by the INLAS project and describe the objectives and organization of the project. The project explores ways to integrate the effects of natural disturbances and management into planning and policy analyses; illustrate potential conflicts among current policies, natural distrubances, and management activities; and explore the policy, economics, and ecological constraints associated with the application of effective fuel treatments on midscale landscapes in the interior Northwest.

Keywords: Forest simulation analysis, midscale, vegetation succession, disturbance, management.

Preface

The concept of a process for evaluating policy direction and management options for subbasin-size landscapes in the interior West evolved from the Pacific Northwest Research Station's Research Initiative for Improving Forest Ecosystem Health and Productivity in Eastern Oregon and Washington. The Interior Northwest Landscape Analysis System (INLAS) project was initiated to explore this concept and began with meetings of resource managers and scientists from various disciplines and institutions. This group suggested ways to build an integrated set of tools and methods for addressing resource management questions on large, multiowner landscapes. The papers in this volume are the outcome of these meetings and document our initial approach to developing an integrated landscape analysis framework. Collectively, the papers illustrate the diversity of methods for modeling different resources and reflect the inherent complexity of linking models to create a functional framework for integrated resource analysis. We are still a long way from a perfect tool, the linkages among the chapters are not always apparent, and integration issues have not been consistently addressed. We cannot yet address the interrelationships between many key natural and anthropomorphic processes on large landscapes. We also found that integration forced scientists to generalize relationships and to summarize detailed research findings in order to incorporate their disciplines at the landscape scale of the INLAS framework. With a growing interest in integrated natural resource modeling, we concluded that, despite the fact that we have not solved all the problems associated with integrating information from different scientific disciplines, creating this document will provide a valuable resource for future researchers who want to understand how groups of scientists organize themselves for a project like INLAS. There are few examples of case studies of similar work in other regions, and to our knowledge, none that document such early stages of these projects' organization. The reader can learn from both the continuity and lack thereof among the chapters, and perhaps use this publication to learn new ways to deal with the dilemma of how to hybridize long-term research lineages into coherent ways of thinking about integrated natural resource management.

Acknowledgments

We thank the many disciplinary experts who reviewed the individual chapters. We appreciate the thoughtful reviews of the entire document provided by W. Connelly, T.M. Quigley, and T. Spies. Thanks also to Judy Mikowski for assistance with formatting the entire document. For their continued invaluable input throughout the INLAS project, we thank Kurt Wiedenmann and personnel from the La Grande Ranger District. We are also grateful for the many contributions of Bob Rainville to this project.

Contents

Chapter 1: A Framework for the Development and Application of INLAS: the Interior North-west Landscape Analysis System

R. James Barbour, Alan A. Ager, and Jane L. Hayes[1]

Abstract

The Interior Northwest Landscape Analysis System is a partnership among researchers and natural resource managers from both the public and private sectors. The project is an effort to increase our understanding of the role of vegetative succession, natural disturbance, and management actions at the watershed scale. The effort will advance the development and application of integrated landscape-level planning tools (models, methods, and information) that use consistent assumptions and common data. Focusing on the subbasin (landscape units of about 202 300 ha) and smaller scales, we will demonstrate the use of existing and new landscape simulation tools to project future succession, disturbance, and management under various policy scenarios. These scenarios will compare different approaches to achieving short- and long-term ecosystem goals and the effects of regulatory constraints, ownership patterns, and limited budgets. Of specific interest is the measurement of the long-term cumulative effects of fuels management and other treatments on key resources. The project is using a 178 000 ha watershed in northeast Oregon to prototype modeling tools and methods. The results of landscape simulations will help to inform the debate over sustainability of forest, range, and aquatic ecosystems in the intermountain West.

Keywords: Forest simulation analysis, midscale, vegetative succession, disturbance, management.

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Introduction

Despite a decade of scientific assessments throughout the interior Pacific Northwest (e.g., Caraher et al. 1992, Everett et al. 1994, Gast et al. 1991, Quigley et al. 1996) and elsewhere (e.g., FEMAT 1993, Johnson 1996), the debate continues over management of forested and range lands, and aquatic systems. It is clear from these and other analyses that decades of human activities to reduce the risk of unwanted disturbances like wildfires and to extract goods and services have led to substantial changes in forest conditions and productivity. It is also evident that there is no simple remedy given landownership patterns, ecosystem-level management objectives, existing landscape conditions, and the complex array of state and federal regulatory constraints (e.g., Clean Water Act 1977, Endangered Species Act [ESA] 1973, National Environmental Policy Act 1969, National Forest Management Act 1976). Large areas of forest land in the intermountain West remain in conditions lending themselves to uncharacteristically large and severe wildfires and insect or disease outbreaks (Ottmar and Sandberg 2001). The problems are compounded by finite budgets and changing economic conditions and social concerns that can contribute to constraints on land managers (Quigley et al. 2001).

With the continued dispersal of human populations into areas that were once considered “wild,” the problem of how to manage for natural disturbance over large areas while not impinging on human populations or negatively affecting the conservation of rare or valuable resources becomes increasingly complex. Much of the debate over how to manage federal lands is focused on the tradeoffs among active management to produce goods and services, moderate wildfire and other natural disturbances, and the long-term preservation of federally protected plant and animal species. A relevant policy question is whether short-term goals intended to protect aquatic and terrestrial habitat for species listed under the 1973 ESA might impede forest management activities that are necessary to improve the long-term sustainability of these species. In the Blue Mountains of northeast Oregon, management direction for resource protection and other amenities may prevent treatment of the majority of fuel-laden stands (Wilson et al., n.d.). Market conditions and operational costs further reduce the extent to which management can be applied to reduce risk from natural disturbances (Barbour et al., in press).

One thing that is clear from these debates is that society as a whole does not share a common strategic vision of future forested landscapes. A blueprint for restoring and maintaining these landscapes is needed that considers the combined effects of forest succession, disturbance, and management (Quigley et al. 2001). Many questions remain concerning efficient and cost-effective scheduling and spatial distribution of management activities, such as prescribed fire, thinning, and selective harvesting, on large landscapes to achieve specified goals over the long run (Finney Chapter 9). Further, we do not understand the long-term compatibilities among commodity production, recreational use, fire risks, fuel treatments, cumulative effects of management activities on key resources, and fish and wildlife habitat goals. Some hypothesize that restricting active management will eventually lead to large natural disturbances that will negate the net effect of protective resource policies. Others feel that management itself poses the greatest threat to sensitive resources. Unfortunately, the debate has been fed, in part, from conflicting projections of potential outcomes. Decisionmakers need unbiased and consistent information about the likely outcomes of different policies or management practices as they evaluate options.

Landscape simulation tools can aid in the development of strategic visions for managing forested and range lands by providing a means to project long-term changes from succession, management, and disturbance (e.g., Bettinger et al. Chapter 4, Johnson et al. 1998, Keane et al. 1996, Mladenoff and Baker 1999, Spies et al. 2002). Understanding how landscapes respond over time to perturbations is key to the development of effective forest policy (Turner et al. 2002). Landscape simulation tools also can aid in the growing need to integrate social concerns with tradeoff analyses of natural resource values (Vogt et al. 2002) and provide a framework to build consistent modeling approaches across resource disciplines. Although a number of recent efforts have applied forest landscape simulation modules to analyze policy issues at broad scales (e.g., Johnson et al. 1998, Keane et al. 1996, Spies et al. 2002), there has been little operational use of these tools to examine management issues at the watershed or subbasin scale. In addition, previous work has largely been concerned with modeling of forest vegetation, with relatively little attention to the problem of modeling nonforest conditions and social values.

The overall goal of the Interior Northwest Landscape Analysis System (INLAS) project is to advance the development and application of integrated landscape models and apply these tools to examine the effects of forest management on long-term trajectories of forest, range, and aquatic conditions at the subbasin and smaller scales. A primary focus of this work will be to apply simulation methods to measure the relative effects of forest succession, disturbance, and management on multiple-resource goals (fig. 1).

Decisionmakers can apply the techniques developed in this project to measure the response of large landscapes to different management scenarios ranging from active to passive, while accounting for expected levels of natural disturbance (Quigley et al. 2001). They also can help landowners, managers, and regulatory agencies integrate new scientific information into biological assessments, watershed analyses, subbasin reviews, and forest management plans. The landscape simulation methods that are advanced by this project will also have utility in a wide range of ecological research, especially that pertaining to disturbance processes and their effects on landscape pattern.

The combined development and application of landscape simulation methods will focus on a set of research objectives that will be addressed on a prototype analysis area. These are:

1. Evaluate the combined effects of management, natural disturbance, and succession on current and future resource conditions. The current set of regulations and management directions for individual species and habitats may not allow achievement of long-term, landscape-level ecosystem goals to both manage fuels and protect resources. We will examine possible conflicts created by existing policy and management directions between short-term management for site-specific conditions and the long-term potential for episodic and chronic disturbances to degrade those conditions. We will create a baseline scenario against which we can compare alternative policy and management options. This baseline scenario will follow current guidelines and forest plans. We will then contrast potential outcomes from this scenario with two opposing scenarios: (1) **passive**, i.e., custodial management and (2) **active**, i.e., management actions aimed at accelerated achievement of goals. Each scenario will examine a range of approaches that might be followed by different owners.

2. Develop new knowledge about how to reduce resource impacts by arranging spatial schedules of treatments to manage disturbance. We will use an adaptive approach to apply information developed during early analyses to design spatially explicit schedules of treatments that reduce both immediate adverse effects on desirable site-specific conditions and long-term risks from stochastic disturbances. This work will illustrate ways to make the most of spatial and temporal variation in terrestrial and aquatic conditions to reduce adverse long-term cumulative effects on sensitive resources by taking selective management actions, such as reducing fuel loads.

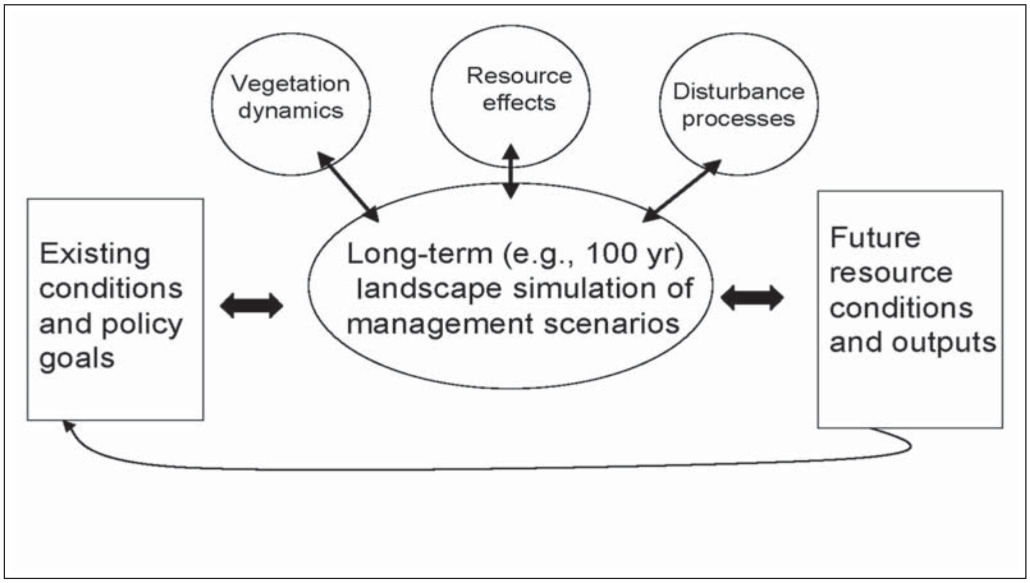

Figure 1—Interior Northwest Landscape Analysis System conceptual framework.

3. Identify the policy, sociocultural, economic, and ecological constraints associated with the application of active or idealized spatially explicit schedules of fuel treatments. Spatial arrangement of fuel treatments on large landscapes can have an important effect on wildfire spread rates (Finney 1998, 2001). Idealized spatial and temporal distributions of fuels management treatments that attempt to minimize wildfire risk may, however, violate constraints imposed by other objectives or policies. Our analysis will identify policy, sociocultural, economic, and resource constraints that might prevent implementation of otherwise theoretically optimal treatment patterns (e.g., Finney 1998).

4. Develop methods to help managers identify problematic watersheds. The difficulty of achieving management objectives in individual watersheds differs considerably owing to particular combinations of physiography, vegetation, social values, economics, and management strategies. Our methods integrate finer resolution variables (e.g., stand density, fire hazard, wildlife habitat, economics, and management restrictions) to help to identify subwatersheds where actions might enhance specific goals or where current policies intended to mitigate risks and effects of unpredicted disturbances might be ineffective over the long term.

5. Examine the long-term consequences and socioeconomic feasibility of density management objectives at the watershed scale. Current assessments by the USDA Forest Service reveal large areas of dense stands that exceed desired stocking levels (Wilson et al., n.d.). What are the most economically efficient ways of altering these conditions, and maintaining stands at desired stocking levels? What are the implications and conflicts associated with achieving these objectives in terms of other resource values? Can we effectively integrate socioeconomic considerations into analyses of subwatershed-scale risk factors that measure deviations from desired stocking? Will the net effects of some management activities result in more long-term stocking problems than they solve? Long-term simulation of forest management will be used to address these and related questions.

Research Approach

The INLAS is building on two alternative methods to create a framework for modeling landscape change. First, a state and transition approach (see Hemstrom et al. Chapter 2) is being used to build a relatively coarse simulation system that integrates conifer succession and disturbance, forest management, fluvial processes, invasive plants, and herbivory. State and transition modeling uses a relatively coarse stratification of landscape conditions into **states** and simulates changes in landscape condition over time by using **transition probabilities** (Hemstrom et al. Chapter 2). These state and transition models evolved from successional studies in ecology and have recently found their way into forest planning and landscape assessment (Hann et al. 1997). Some of the advantages of this system are that software is well developed (Kurz et al. 2000), and data needed to run the model are available. The disadvantage of these methods is that they do not consider tree list type data or other detailed information about vegetative conditions (Hemstrom et al. Chapter 2).

The second approach uses the extension of tree-level growth models (e.g., Forest Vegetation Simulator [FVS], Stage 1973; ORGANON, Hann et al. 1995) to simulate landscapes as an assemblage of individual stands, polygons, or pixels (Ager Chapter 3, Bettinger et al. Chapter 4). This stand-level simulation approach has been the focus of considerable work over the past 10 to 15 years, and many improvements have been made to consider stand contagion, optimization, wildlife, spatial spread of insect epidemics, and consideration of nontimber values, as well as interfaces to the stand simulators to simplify the process of organizing stands for simulation (reviewed in Ager Chapter 3). We are exploring several stand-level simulation approaches, ranging from simple systems that use FVS and FVS postprocessors to model each stand in a landscape (Ager Chapter 3) to systems based on the Simulation and Analysis of Forests with Episodic Disturbance (SafeD) model (Graetz 2000) that can perform spatial optimization and incorporate natural disturbances (Bettinger et al. Chapter 4). Like the state and transition approach, there is a growing interest in this type of modeling for both research and operational applications. Stand-level landscape simulation models are well suited to problems where a relatively high degree of biological and spatial resolution is required. This includes studies where tree-level parameters like size and species are needed, and where management choices are tailored to stand metrics.

By applying both models, we will demonstrate tradeoffs between the two modeling approaches and their relative merits at different biological and spatial scales (fig. 2). For instance, state and transition models are relatively easy to build for large areas because they represent landscapes as a discrete and finite number of states and transitions. They also can be applied to a wide range of problems where process models or data are not available. A state and transition approach may be the most viable approach to building an integrated landscape simulation system. By comparison, stand-level process models require fine-scale quantitative data on stand conditions and can provide detailed data on stand characteristics through time, which are needed for many assessment and planning projects.

By using the two modeling frameworks described above, we will explore how to integrate other important ecosystem components (fig. 1). Each of the ecosystem components represents a model (or models) that is integrated into the framework and takes information from and feeds information to the vegetation simulator. Output from these resource effects models alters vegetative conditions or constrains management or succession and changes the resource outputs available from the landscape.

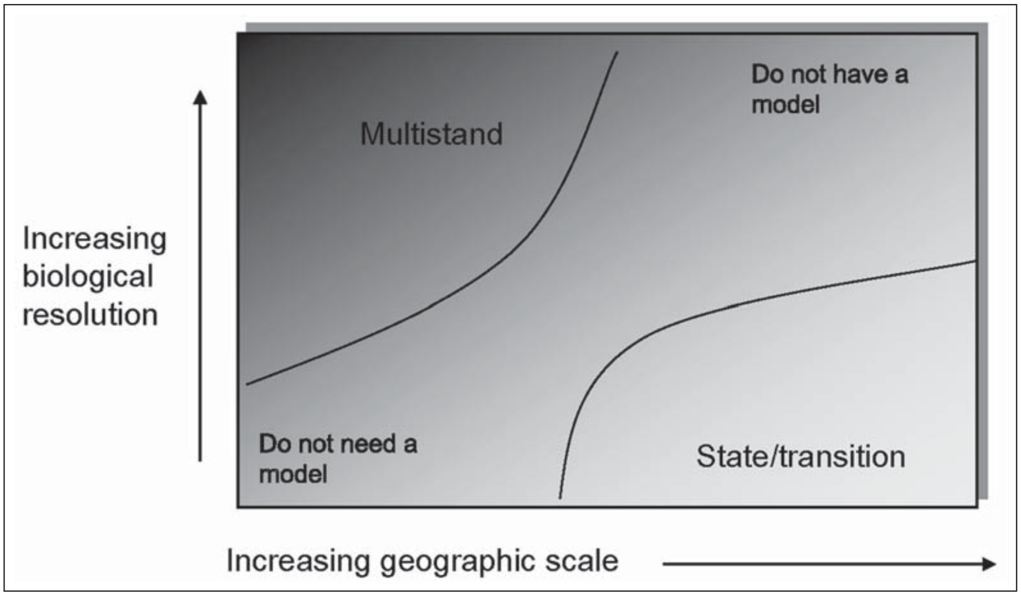

Figure 2—Comparison of a multistand versus state/transition modeling approach for forest landscape simulation in terms of biological and geographic scale.

Descriptions of the status or state of the art of modeling efforts in each of the areas, along with research needs, are provided in the subsequent chapters of this volume. Wales and Suring (Chapter 5) identify methods to describe and evaluate habitat abundance, quality, and distribution across space and time to help managers and policymakers understand how successional processes, natural disturbance, and management actions influence terrestrial habitats. Wondzell and Howell (Chapter 6) review alternative modeling approaches for assessing conditions and prioritizing the restoration of aquatic habitat in the context of biophysical characteristics of streams and watersheds and landscape processes. Chronic disturbance by domestic and wild ungulates is known to significantly affect ecosystem patterns, but as Vavra et al. (Chapter 7) describe, there is a need to develop models that can project the effects of ungulate herbivory at multiple scales. By contrast, episodic disturbances such as insect and disease outbreaks or wildfire have been the subjects of intensive modeling efforts. Ager et al. (Chapter 8) review the quantitative methods for modeling mortality caused by insects and disease and describe the major gaps in this area. Finney (Chapter 9) describes the state of the art and research needs in integrating wildfire into landscape planning models. Lettman and Kline (Chapter 10) examine approaches to evaluate economic impacts of current and alternative management scenarios, and public values and attitudes toward forests. The impacts of human population growth, diversification, movement, and accompanying land use change are important factors in forest management as the wildland/urban interface expands. Clark (Chapter 11) describes approaches for identifying and evaluating the values and places that are important to people. Kline (Chapter 12) describes modeling and analyses of residential and other development scenarios that can contribute to anticipating where land use change is likely to occur. Barbour et al. (Chapter 13) describe techniques for displaying the ecological and economic costs and benefits of timber removal or gathering nontimber forest products.

The specifics of how these individual components might be refined and integrated are part of the major developmental challenges of the INLAS project. Some aspects of integration are covered in the chapters that follow in this volume, whereas others will be developed as the project evolves.

Project Area

Lying within the Upper Grande Ronde watershed, a 4th-hydrologic unit code (HUC4) subbasin, the INLAS project area comprises four HUC5 units occupying about 178 000 ha of mixed forest and rangelands on the eastern flank of the Blue Mountains southwest of La Grande, Oregon (fig. 3). The La Grande Ranger District of the Wallowa-Whitman National Forest administers about 123 000 of these ha (fig. 4). Most of the remaining land is nonindustrial private (about 55 000 ha). Smaller areas are owned by the Confederated Tribes of the Umatilla (about 13 800 ha), Boise Cascade (about 5000 ha), and the state of Oregon (about 810 ha). Numerous residences exist on nonindustrial private lands around the town of Starkey. The topography is highly varied and complex, with deeply dissected drainages feeding into the Grande Ronde River as it runs north through the center of the subbasin (fig. 5). Elevations range from 820 to over 2130 m. Vegetation ranges from xeric, bunchgrass communities at the lower, north end of the project area, to mixed conifer and subalpine fir (*Abies lasiocarpa* (Hook.) Nutt.) on the eastern flanks of the Elkhorn Mountains. Fuel loadings are highly heterogeneous across the project area, and a number of large wildfires have occurred over the last 10 years, burning about 8100 ha. Two additional large wildfires burned as much as 24 300 ha on lands immediately adjacent to the Upper Grande Ronde subbasin and project area. An outbreak of spruce budworm (*Choristoneura occidentalis*) occurred throughout the 1980s causing extensive Douglas-fir (*Pseudotsuga menziesii* (Mirb.) Franco) and grand fir (*Abies grandis* (Dougl. ex D. Don) Lindl.) mortality throughout the Blue Mountains including the Upper Grande Ronde subbasin. Outbreaks of bark beetles (*Dendroctonus* spp.) have also occurred in and adjacent to the project area.

Forest Service lands are managed with emphases ranging from scenic areas to commodity production. The Starkey Experimental Forest and Range (about 8900 ha) is located in the project area on the southwestern portion of the subbasin and includes research facilities of the Starkey Project (Rowland et al. 1997, Vavra et al. 2002). The Upper Grande Ronde subbasin contains habitat for three federally threatened species, the Canada lynx (*Lynx canadensis*), the gray wolf (*Canis lupus*), and the bald eagle (*Haliaeetus leucocephalus*). About 40 additional terrestrial vertebrates of conservation concern identified by Wisdom et al. (2000) are likely to occur in the Upper Grande Ronde subbasin. This area may provide habitat for several of the 15 insect species currently listed as threatened, endangered, or sensitive in east-side forests (LaBonte et al. 2001). The project area includes potential habitat for three federally listed threatened and one candidate plant species (USF&WS 2002). An additional eight plant species, currently designated as sensitive by the USDA, Forest Service, Pacific Northwest Region, have been documented in the INLAS project area. The Grande Ronde River and its tributaries also contain habitat for federally threatened chinook salmon (*Oncorhynchus tshwaytscha* (Walbaum)), bull trout (*Salvelinus fontinalis*), and steelhead (*Salvelinus confluentus)*. For more detailed information about this area, an extensive bibliography of reports and published literature is provided in the final chapter of this volume (Aitken and Ager Chapter 14).

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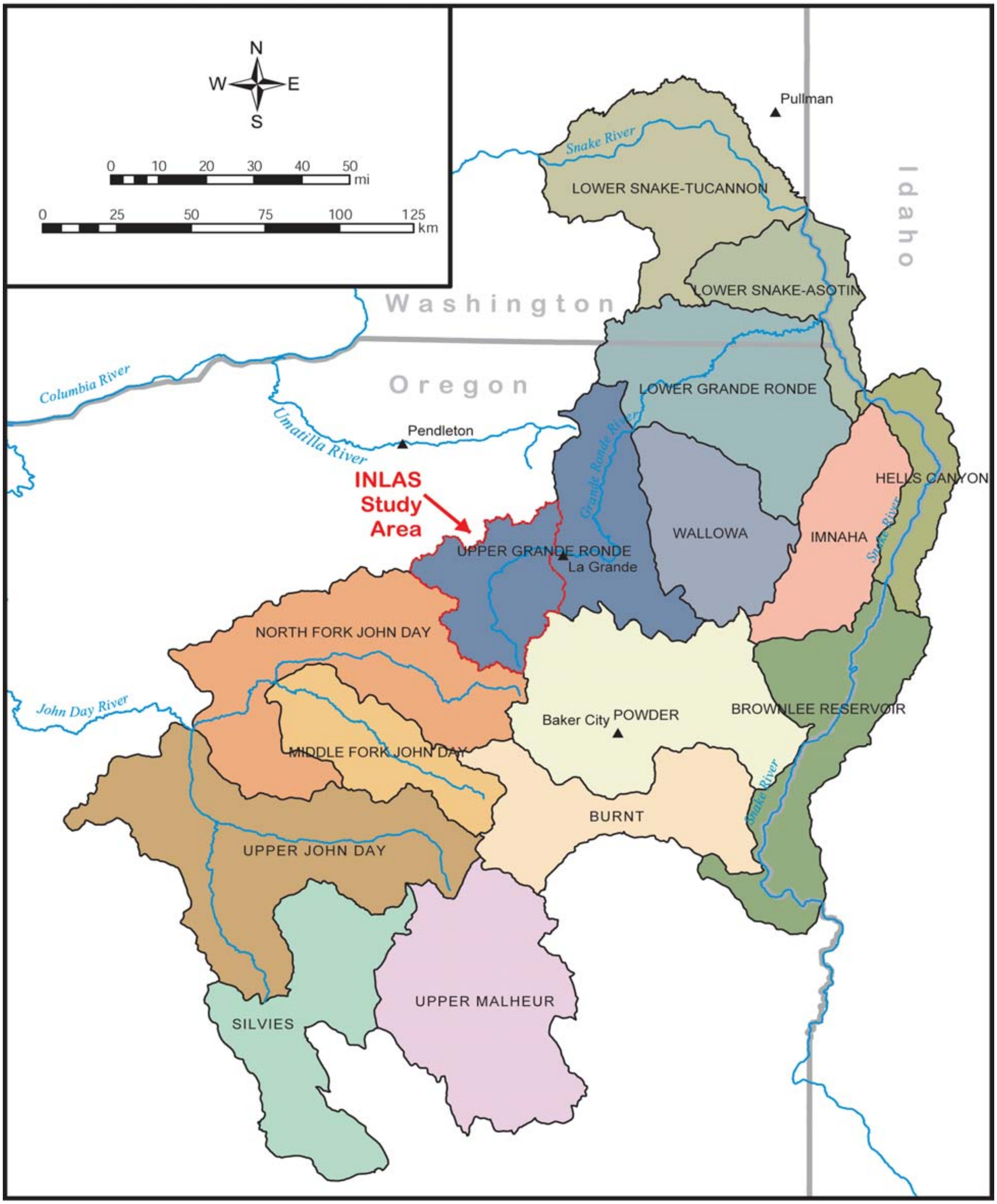

Figure 3—Location of Interior Northwest Landscape Analysis System project area within the Blue Mountains ecoregion (in 4th-hydrologic unit codes).

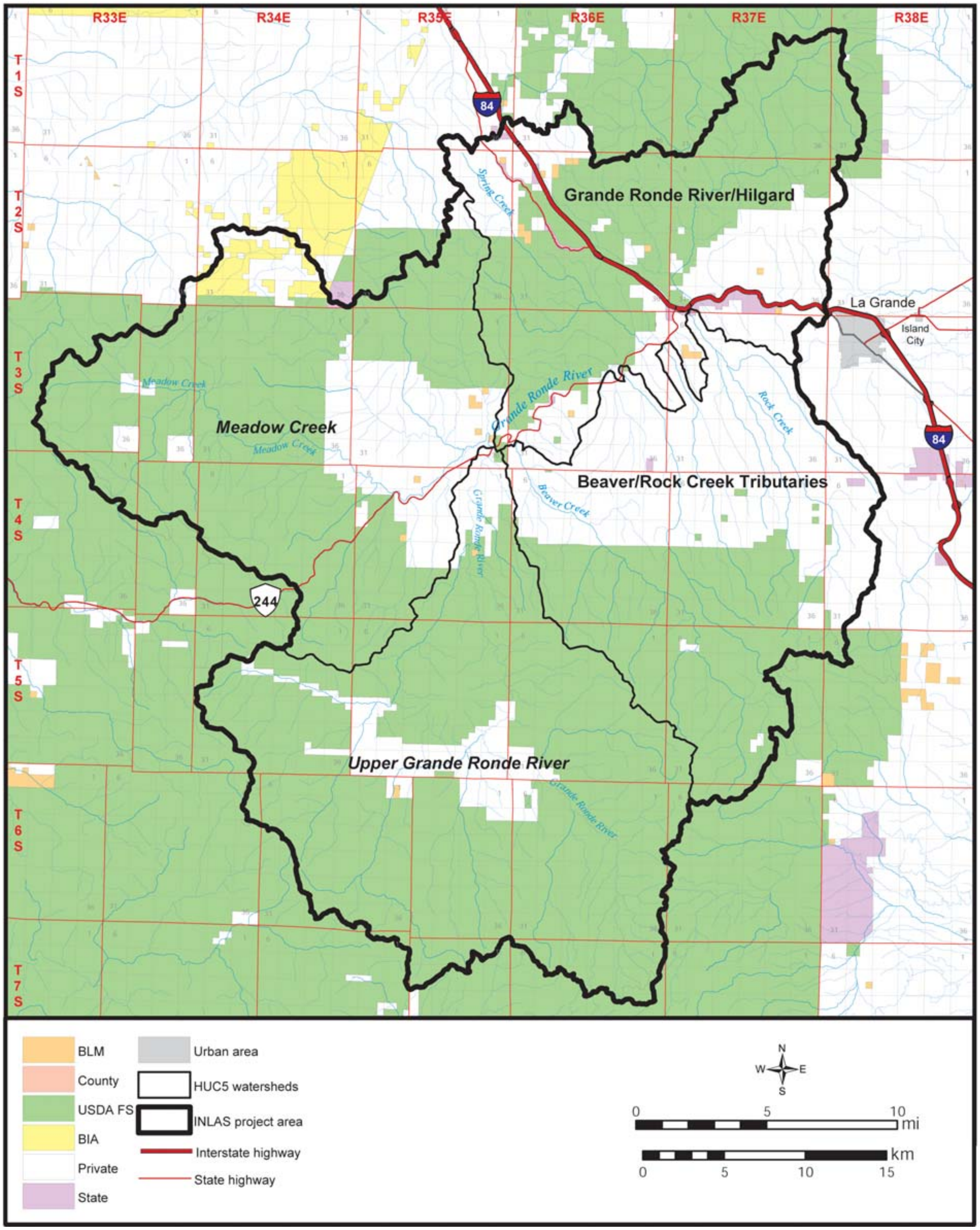

Figure 4—Ownership in Interior Northwest Landscape Analysis System project area.

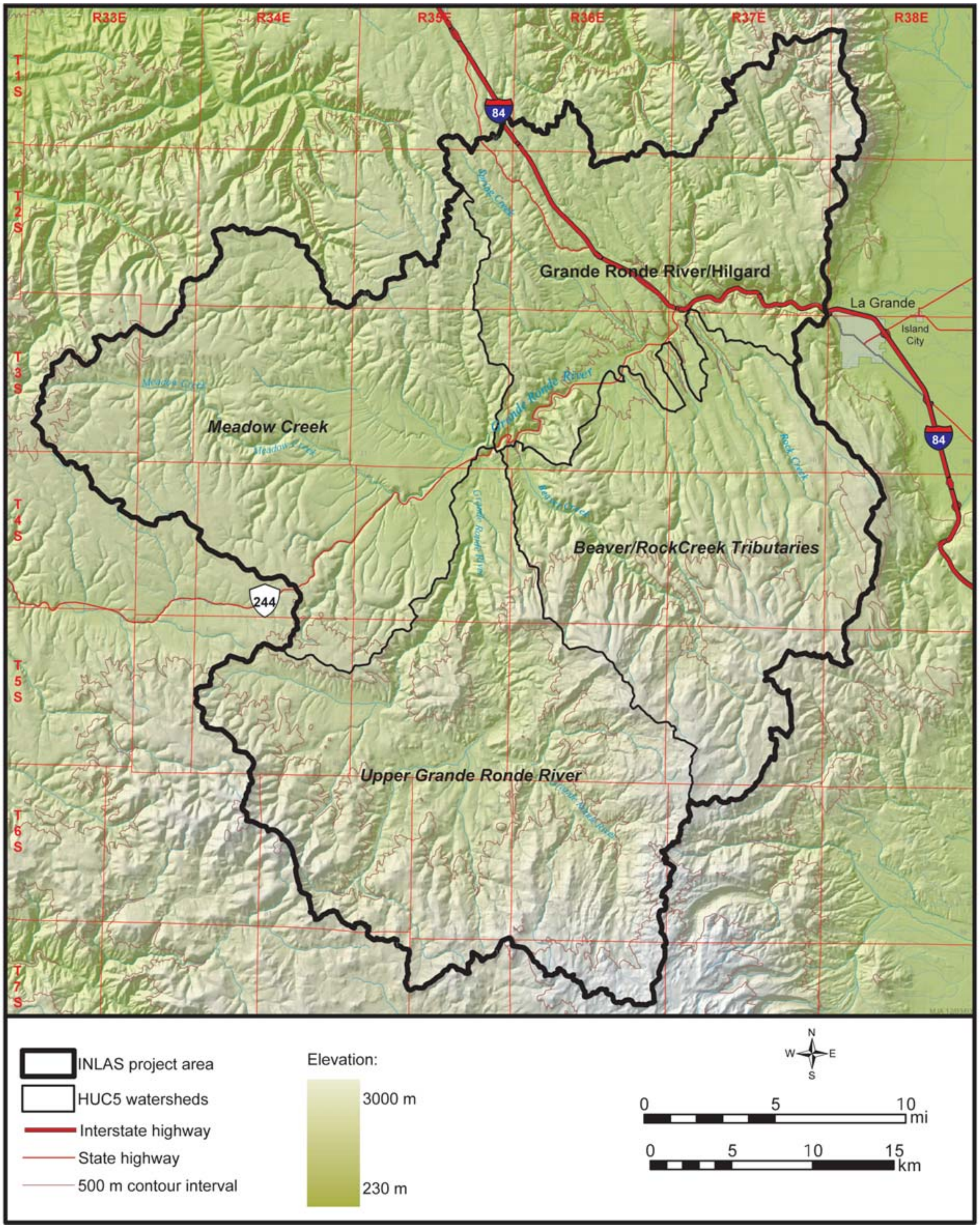

Figure 5—Topography of Interior Northwest Landscape Analysis System project area.

Audience and Products

Although the direct application of analyses from the prototype area is limited in geographic scope, the lessons learned while conducting these analyses will find use in the much broader policy arena. The product mix from the INLAS project will include methodologies, new scientific knowledge, and much information germane to current policy debates over sustainability and conservation of natural resources. Analyses will clarify many socioeconomic and ecological interactions for which we have a poor understanding. This will help scientists identify the most productive areas for future research.

We anticipate that methods we develop for the Upper Grande Ronde prototype area will have applicability to other areas. Some of the methods we develop at the subbasin scale can be “scaled up” to larger areas, e.g., analysis of lynx habitat, or applied on a large number of other subbasins across the Blue Mountains to answer midscale questions. In addition, by developing methods at the midscale, we hope to better understand the larger scale issues and develop ways to use our methods and results at both larger (e.g., forest or regional planning) and smaller scales, (e.g., watershed assessments and project plans). We are working closely with the regional planning staff members to ensure the products produced by INLAS are useful and fit into the planning process. We are also working with the La Grande Ranger District on a relatively small (about 2400 ha) wildland/urban interface fuels-reduction project to prototype some of the analysis tools.

The users of the products developed during this project include those involved in, or interested in the outcome of, watershed assessments, forest planning, and policy analysis. A major drawback of previous landscape modeling efforts is that the data requirements and intricacies of the modeling process rendered existing systems unworkable to most prospective users. Many of our methods and processes are built on existing data, tools, and software to make them more readily adaptable by managers who may already be familiar with the underlying programs. Where new design and development are needed, we plan to work with developers to facilitate the incorporation of our prototype software into preferred systems.

Some of the anticipated outcomes from the INLAS project include:

- Developing methods to perform analyses at the interface of policy, management, and science that rely on a consistent set of assumptions and common data.
- Providing information from landscape analyses to local and state political leaders, government and private resource managers, scientists, and policymakers.
- Demonstrating the breadth of management options to policymakers, resource managers, researchers, and the public.
- Facilitating discussions about realistic balances among goals among managers, policymakers who represent different landowners, and the public.
- Illustrating how actions by nonfederal owners might influence the capability of meeting different policy objectives on federally managed land and vice versa.
- Gaining insights into the influence of scale in determining the importance of management actions within different ownership patterns.
- Identifying specific knowledge gaps in ecological research, management science, and resource planning analysis.

The development and application of different aspects of the work are described in the chapters that follow. Each of these component efforts will produce methods and tools that not only contribute to accomplishing the specific goals of the INLAS project but also can operate outside of the INLAS simulation framework. The integrative products resulting from interactions among sets of two or more components of the framework will help to highlight how different resource values complement or conflict with one another.

English Equivalent

When you know:	Multiply by:	To get:
Hectares (ha)	2.47	Acres (ac)
Meters (m)	3.28	Feet (ft)

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Chapter 2: A State and Transition Approach for Integrating Landscape Models

Miles Hemstrom, Alan A. Ager, Martin Vavra, Barbara C. Wales, and Michael J. Wisdom[1]

Abstract

We will use state and transition modeling (STM) to project landscape dynamics in a portion of the Upper Grande Ronde subbasin, northeastern Oregon. The Interior Northwest Landscape Analysis System effort will develop both process-based models and STM to represent vegetation, disturbance, and management interactions across large landscapes. State and transition models are useful for integrating disturbances, management activities, and vegetation growth and development across large, variable landscapes, but are not currently useful for finding optimal solutions to meet landscape management objectives. Process-based models are useful for detailed modeling of vegetation changes and optimization but can be difficult to develop and parameterize across many disturbances and highly variable vegetation conditions. We discuss advantages and limitations of STM in the context of integrated scientific analysis and land management planning at subbasin and broader scales. We provide an example of how such models might be used to project the integrated effects of vegetation management, fire, invasive plants, ungulate herbivory, and other disturbances on vegetation across a large landscape in northeastern Oregon. We suggest enhancements of existing STMs that will use process-based models to calibrate states and transitions.

Keywords: Landscape simulation, northeastern Oregon, landscape ecology.

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Introduction

Landscape simulation models have been widely applied to address research and land management policy questions in the Western United States and elsewhere (Bettinger et al. 1997, 1998; Graetz 2000; Hann et al. 1997; Mladenoff and He 1999; USDA and USDI 2000). Advances in modeling techniques, computer technology, and geographic information systems (GIS) have made it possible to model large landscapes at increasingly finer scales of spatial and temporal resolution. However, natural resource planning models used in the past focused primarily on conifer succession and management while representing other key ecosystem elements as byproducts (e.g., Alig et al. 2000, Johnson et al. 1986). Although progress has been made in the formulation of multiobjective goals in landscape simulations (e.g., Sessions et al. 1999, Wedin 1999), there remain many challenges to building landscape planning models that include all the important disturbance processes that influence landscape change. Previous efforts have often not included widespread, chronic disturbances (e.g., ungulate herbivory) or have focused on selected environments (e.g., forests) rather than entire landscapes. Of particular interest in the Interior Northwest Landscape Analysis System (INLAS) context are the net, synergistic effects of various disturbances (e.g., fire, invasive plants, large herbivores, and hydrologic processes as they affect geomorphology and associated riparian habitat) across a large landscape that includes a variety of environments. Integrating these kinds of disturbances is exceedingly complex in models that treat vegetation and disturbance on continuous scales.

An alternative approach is to represent the effects of these disturbances in discrete form in state and transition modeling (STM). In parallel with Simulation and analysis of forests with episodic Disturbances (SafeD) developments for INLAS (Bettinger et al. Chapter 4), we will use STM for multiresource integration in a landscape planning model. The broad goal of this work is to develop prototype disturbance models in a STM framework that integrates major environments, vegetation types, ownerships, and disturbances across a large and diverse landscape. This effort will use the Vegetation Development Dynamics Tool (VDDT; Beukema and Kurz 1995) and the associated Tool for Exploratory Landscape Scenario Analysis (TELSA; Kurz et al. 2000), which have many features that make them well suited for developing and testing new approaches to landscape simulation. Ultimately, this work will lead to more refined, integrated approaches to understanding the interplay of disturbances and vegetation across large, variable landscapes. We also will examine the potentially complementary linkage of STM with more detailed, continuous simulations from SafeD. Our expectation is that detailed simulations from SafeD can be used to calibrate states and transitions while STM can examine landscape-wide interactions of vegetation types and disturbances that cannot readily be included in SafeD.

State and Transition Models

State and transition models treat vegetation composition and structure as “states,” connected by transitions that indicate vegetation development over time and disturbance (fig. 6). This STM approach builds from transition matrix models that represent vegetation development as a set of transition probabilities among various vegetation conditions (e.g., Cattelino et al. 1979, Hann et al. 1997, Horn 1975, Laycock 1991, Noble and Slatyer 1980) (figs. 7 and 8). Vegetation states change over time barring management activities or disturbances. For example, grass/forb-closed herblands become shrub/tree regeneration-open midheight shrubs after 15 years. State change along the successional, time-dependent path is deterministic and, without disturbance or management, all the vegetation would ultimately accumulate in one long-term stable state. However, disturbance or management activities can change the course of vegetative development

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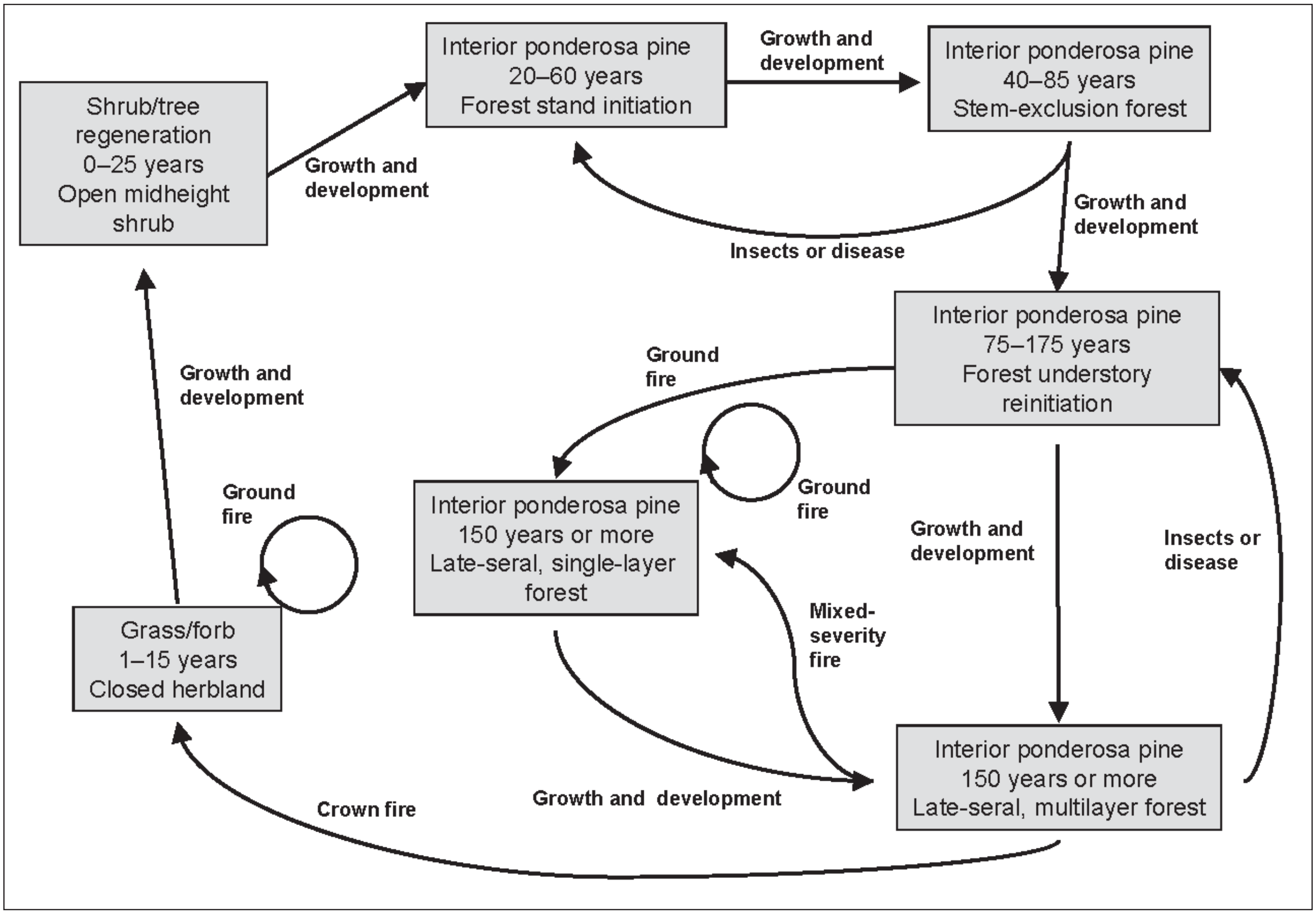

Figure 6—Example of simplified state and transition model for dry forests in the Interior Northwest Landscape Analysis System study area, northeastern Oregon.

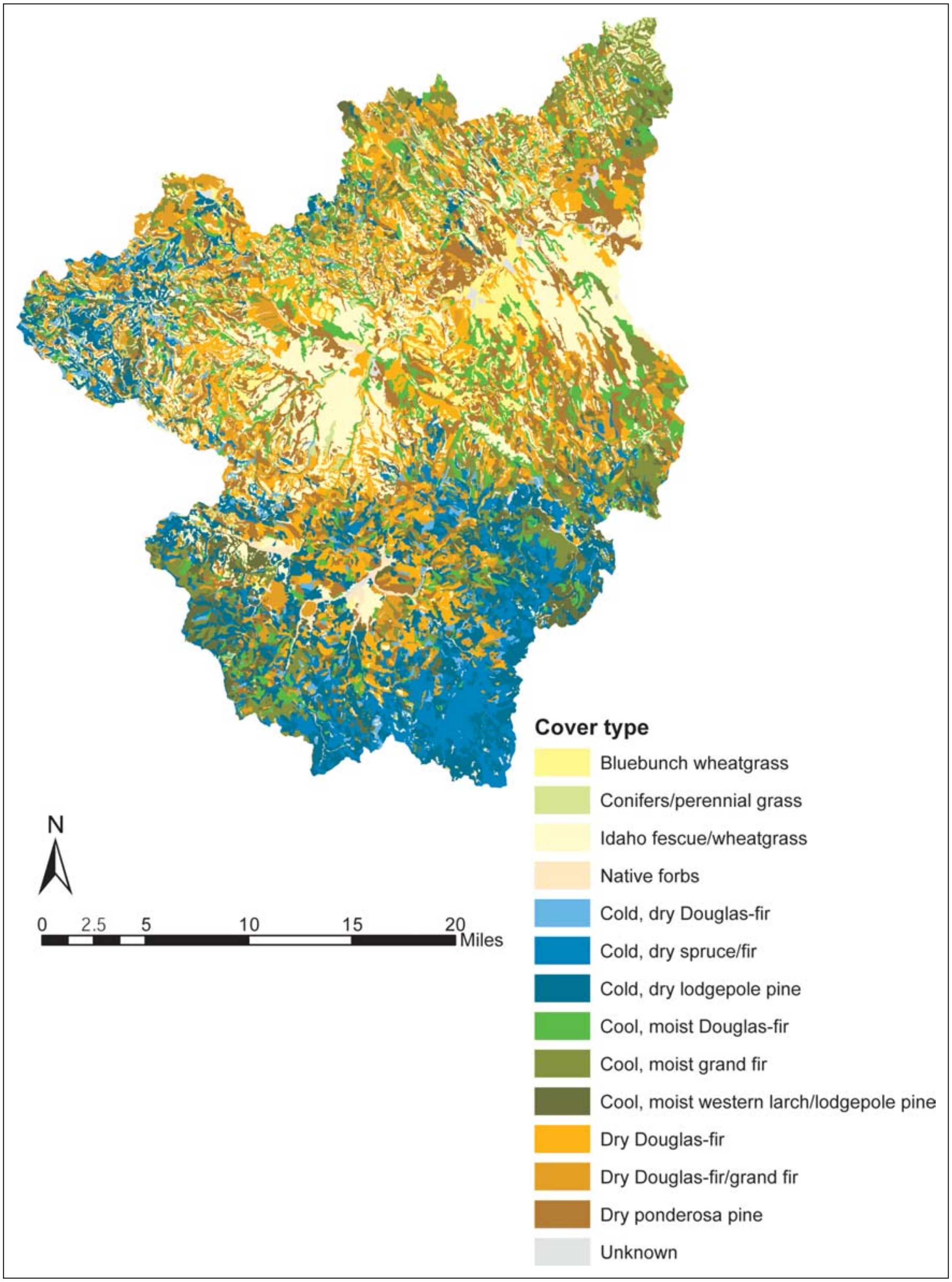

Figure 7—Example of existing vegetation cover type classes in the Interior Northwest Landscape Analysis System study area, Upper Grande Ronde subbasin, Oregon. Classes developed during the study may differ from those shown.

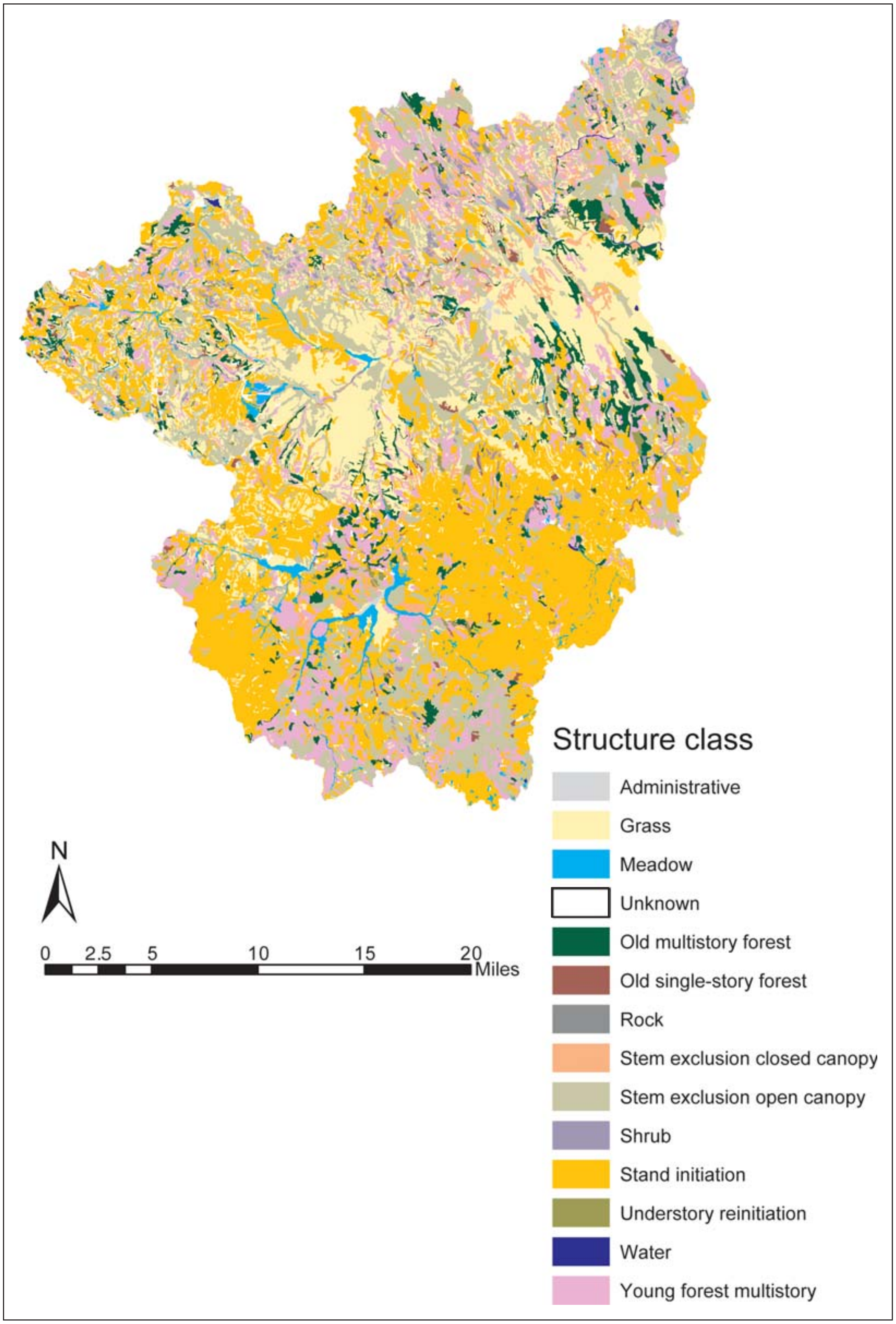

Figure 8—Example of existing vegetation structure classes in the Interior Northwest Landscape Analysis System study area, Upper Grande Ronde subbasin, Oregon. Classes developed during the study may differ from those shown.

at any point. Depending on disturbance probabilities and consequences, very little or no vegetation may actually accumulate in the long-term stable state at the end point of succession. In our example (fig. 6), insect and disease activity may reset interior ponderosa pine (*Pinus ponderosa* Dougl. ex Laws.)/stem-exclusion forest to the stand-initiation condition. In contrast to successional development, disturbances, including management, are probabilistic and possible at each time step, depending on vegetation state. A separate model (states and transitions) is developed for each modeling stratum (groups of potential vegetation types in the study area, see Hall 1998). We anticipate 15 or more modeling strata in the study area (figs. 7, 8, and 9). The example used for illustration (fig. 6) has been substantially simplified. Most models will be considerably more complex. Many of the models we use will contain “transition thresholds” influenced by site degradation or invasive plants, beyond which recovery to previous plant community conditions is difficult or impossible (e.g., Laycock 1991). Hann et al. 1997 and Hemstrom et al. (in press) used several such models to depict vegetation change across the interior Columbia basin.

State and Transition Modeling Systems and Recent Applications

A number of STM systems have been developed in the past 5 to 10 years and applied on Western landscapes either as research or planning tools, including SIMulating vegetative Patterns and Processes at Landscape ScaLEs (Barrett 2001, Chew 1995), LANDscape SUccession Model (Barrett 2001, Keane et al. 1996), and VDDT (Beukema and Kurz 1995). We will use VDDT (Beukema and Kurz 1995) and the associated TELSA (Kurz et al. 2000). The VDDT planning tool is a nonspatial model that allows building and testing STM for a set of environmental strata. The TELSA planning tool is a spatial application of VDDT that includes spatial analyses and spatial contagion of disturbances. Both models contain visual interfaces and other features that make them relatively easy to use. In addition, they have been used in landscape assessments and land management planning in the interior Northwest. The interior Columbia basin landscape assessment (Hann et al. 1997) built VDDT models for a broad cross section of range and forest lands in the interior Northwest. These and similar models are being used by some national forests for revisions to their land management plans (e.g., Merzenich et al., in press). Use of STM is a significant departure for national forest land management planning from past efforts where harvest scheduling models were predominantly used (e.g., Johnson et al. 1986). Harvest scheduling models made extensive use of timber inventories and linear programming to explore resource tradeoffs and marginal costs, as mandated under the planning regulations at the time.

Objectives and Research Approach

We used the following research approach:

1. Build STM by using the VDDT and TELSA modeling systems to simulate future forest, woodland, shrubland, and herbland vegetation conditions across the entire Upper Grande Ronde study area.

2. Link vegetation projections with SafeD (Bettinger et al. Chapter 4) and other resource effects models to examine the use of those models to calibrate STM for forested environments.

3. Explore ways to add states and transitions for large herbivores, invasive plants, and streamside/aquatic systems.

4. Examine stochastic effects and model sensitivity to disturbance probabilities.

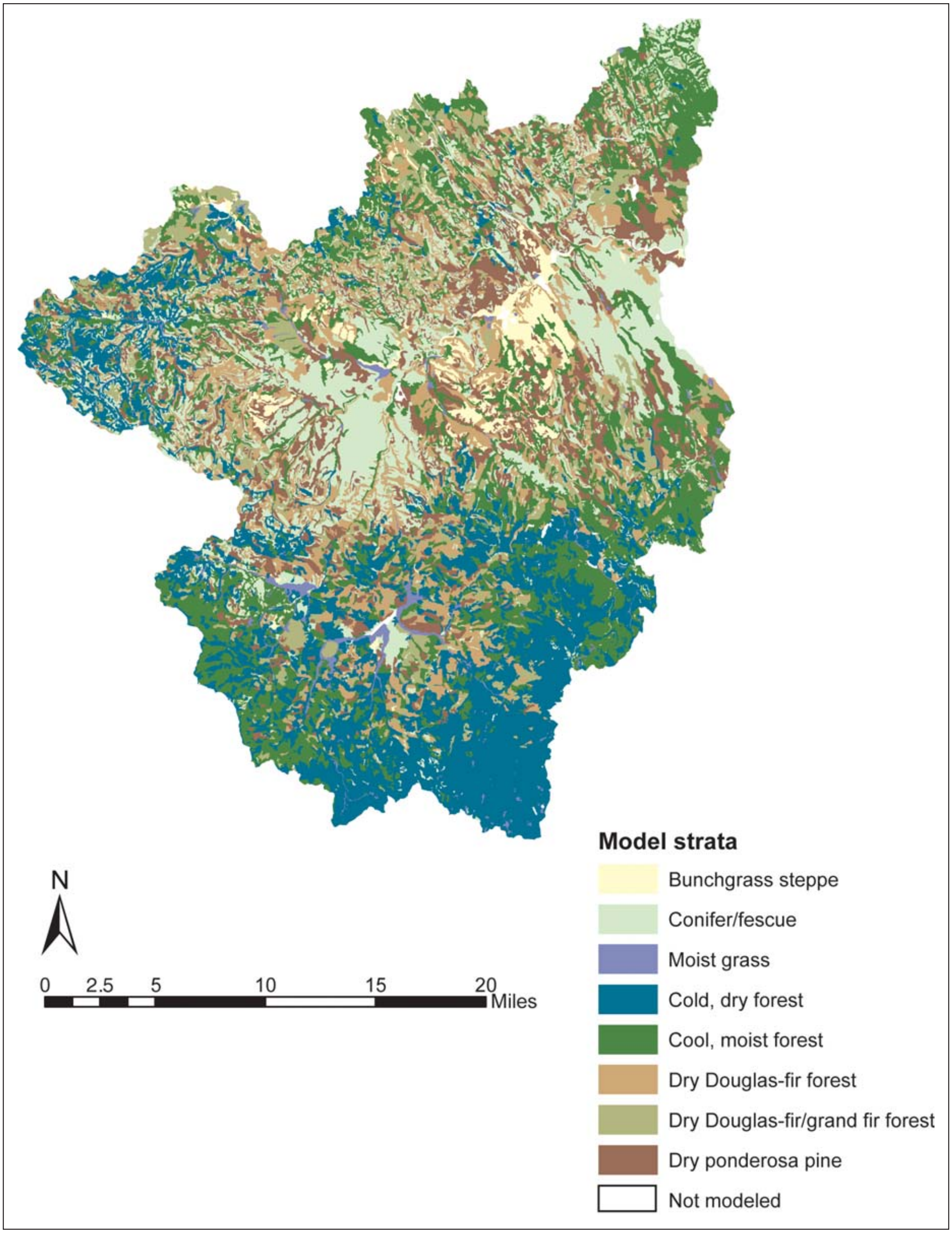

Figure 9—Example of potential vegetation modeling strata in the Interior Northwest Landscape Analysis System study area, Upper Grande Ronde subbasin, Oregon. Classes developed during the study may differ from those shown.

Build State and Transition Models

A number of tasks are required to build VDDT models for use in the INLAS project. Prototype models that might be useful starting points have been built for the interior Columbia basin assessment (Hann et al. 1997) and for the forested lands in the Blue Mountains province.[2] These models will be examined for applicability given the vegetation and environments in the Upper Grande Ronde. A first approximation set of STMs for the Upper Grande Ronde could come from adoption of suitable existing models. We expect that these first-generation models will require considerable refinement, especially those for woodlands, shrublands, and grasslands. A period of model review using the available literature and expert opinion will help refine these initial models to produce a second generation. We will include a variety of management activities by adding them as new pathways and, if necessary, vegetation states. The current version of both VDDT and TELSA can accept more than 400 vegetative states and a number of transitions limited only by computation time—likely more than sufficient for our purposes. We will design vegetation classes based on the need to add detail for wildlife habitat and other models, starting from those in the current Blue Mountains models. We recognize the importance of large dead wood in ecosystem processes and wildlife habitat. We will build structural classes that include abundant large dead wood for one or two decades following some kinds of stand-replacement disturbances.

Link State and Transition Model and Process-Based Models

Professional judgment often has been used to define vegetation states and to derive transition probabilities among vegetation states in existing STM. Although the dominant successional or disturbance transitions might be established in the literature for some vegetation types, many other transitions are not well described. Consequently, expert opinion often is used in model development and calibration. The cumulative effect of many small errors in estimated transition probabilities may undermine the reliability of simulations. Although annual wildfire probabilities often have been developed by using historical fire data, other transitions are more difficult to quantify, and few data are available.

We will examine and adjust states and transition probabilities, including those for various management activities, through the use of process-based models. The Forest Vegetation Simulator (FVS) (Crookston and Stage 1999) provides detailed estimates of tree establishment, growth, and mortality based on forest inventory and other data. Simulations from FVS are forest-based mensurational analyses of tree and stand growth as a function of density effects, disturbance effects, management treatments, and other factors that affect tree and stand growth. Stand-level simulations from FVS or SafeD will be used to refine transition probabilities or state conditions for forested lands to make forest-land projections more accurate. Unfortunately, similar process-based models might not be available for nonforest model strata. In this case, we will continue to rely on expert judgment and will document the sources and assumptions used.

In addition, vegetation classes in existing models are based on classical successional stages (e.g., Hann et al. 1997). We hope to examine this choice more closely given the kinds of stands that develop under human influences, some of which may not have good analogs in natural successional sequences. Detailed stand-level projections from FVS and SafeD models might provide a range of stand structures that should be included in STM for forested areas. We envision development of structural classes that represent stand architecture rather than successional stages that may or may not be representative of current and future east-side forest stands.

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Objective 2 also will require linking model strata to timber inventories. It should be possible to link STM structure and composition classes to plot-level vegetation data from existing forest inventory data. We plan to examine the use of most similar-neighbor analyses (Moeur and Stage 1995, Ohmann and Gregory 2002) to link model strata and tree lists from plot data. We will use multivariate statistical processes to assign tree lists and other information from sampled sites to nonsampled sites based on similarities of environment, photointerpreted attributes, satellite imagery, and other features. This process may both (1) improve the accuracy of current forest composition and structure estimates by using existing plot samples and (2) allow more explicit description of future forest conditions for timber supply and harvest scheduling. True color aerial photographs at a scale of 1:15,840, black and white ortho photography at 1:24,000, and field stand examination data will be used to develop vegetation maps. A subsample of 10 to 20 percent of the photointerpreted polygons will be checked in the field to provide an assessment of photointerpretation accuracy.

Invasive Plants

Native vegetation and associated resources are experiencing significant degradation over wide areas of the interior West from nonnative invasive plants. The cumulative effects go beyond vegetative change because habitat for terrestrial vertebrates and other species is affected (Drake et al. 1989), fire regimes are altered (Billings 1994, Bunting et al. 1987, Pellant 1990), and other ecological processes may be disrupted (Billings 1994, Masters and Sheley 2001). Although the interior Columbia basin project included nonnative invasive plants in some STM, we will examine the potential interaction of invasive plants with other disturbances and management activities. State and transition models are a good choice for initial efforts to model invasive plant interactions across large landscapes because they can be assembled from sparse literature and data and expert opinion.

Ungulate Herbivory

Hobbs (1996) argued that native ungulates are critical agents of change in ecosystems via three processes: regulation of process rates, modification of spatial mosaics, and action as switches controlling transitions between alternative ecosystem states. Huntly (1991) identified the impact of herbivores on plant regeneration as a powerful yet little-studied mechanism of influence on vegetation composition, structure, and diversity. Wild and domestic ungulates should be considered potential agents of chronic disturbance (Riggs et al. 2000).

Cattle grazing often reduces cover of grasses and shrubs as well as total vegetation biomass (Jones 2000). Riggs et al. (2000) reported that in grand fir (*Abies grandis* (Dougl. ex D. Don) Lindl.) forests of northeast Oregon, understory biomass in ungulate exclosures was 2.1 times greater inside than outside, and forest-floor biomass was 1.5 times greater inside than outside. Shrub biomass was influenced more by ungulates than was grass or forb biomass. Augustine and McNaughton (1998) concluded that altered species composition of plant communities in response to selective foraging by ungulates is a general feature of plant-ungulate relations. The authors stated that by ungulates altering the competitive relations among plants, differential tolerance of co-occurring plant species becomes an important determinant of the responses of both woody and herbaceous plant communities to herbivory. Augustine and McNaughton (1998) also summarized ungulate effects on overstory species and listed several species of coniferous and deciduous trees that were herbivory intolerant. Ungulate herbivory is also a driving force shaping vegetation pattern in coastal coniferous forests (Schreiner et al. 1996, Woodward et al. 1994). Research by these authors indicated that ungulates maintained a reduced standing crop, increased forb species richness, and determined the distribution, morphology, and reproductive performance of several shrub species. Woodward et al. (1994) further stated that the extent to which herbivores can change ecosystem processes in forests likely depends on the scales of other disturbances.

Herbivory-induced changes in plant community composition have important habitat ramifications for a number of plant and animal species. Changes in understory structure and litter accumulations may be important to bird and small mammal populations. Individual species of plants and entire plant communities may be at risk under intensive herbivory. Examples of plant species at risk of elimination or severe decline under intensive herbivory include aspen (*Populus tremuloides* Michx.), bitterbrush (*Purshia tridentata* (Pursh) DC.), Pacific yew (*Taxus brevifolia* Nutt.), and mountain mahogany (*Cercocarpus* spp. Kunth) (Parks et al. 1998). Negative effects on vertebrate species that depend on these plants (e.g., cavity nesters in aspen stands, Wisdom et al. 2000) may occur. Inclusion of ungulate herbivory disturbances in STM for the Upper Grande Ronde will allow examination of two important questions:

1. What changes in composition and structure of plant communities occur as a result of herbivory at local and regional scales?
2. How does the grazing regime interact with frequency, intensity, and distribution of episodic disturbances to influence development of plant communities at local and regional scales?

The first question will initially be addressed through a synthesis of existing research data and findings from the Starkey project on diet selection and resource selection functions for ungulates in the Blue Mountains (e.g., from Johnson et al. 1995, Rowland et al. 2000, Wisdom 1998). Data will yield estimates of plant composition with and without herbivory, and the likelihood of herbivory effects occurring in various forest plant communities. The second question will be addressed through development of STM for the Upper Grande Ronde that explicitly includes ungulate herbivory, based on data synthesized for the first question. Plant succession in forests likely operates as a set of states and transitions, much like the models developed and validated for nonforest ecosystems (Laycock 1991, Westoby et al. 1989). Indeed, it now seems possible that the descriptions of many "climax" associations are questionable on this basis (Peek et al. 1978, Riggs et al. 2000, Schreiner et al. 1996). Although our first interest is in building herbivory models for application in the Upper Grande Ronde, we intend to ultimately apply these models at stand, watershed, and basin scales for the entire Blue Mountains province. The models should have some general application throughout the Rocky Mountain west.

Riparian Vegetation and Geomorphology

Riparian and aquatic issues have become critical in the inland Northwest (INFISH 1995, PACFISH 1995), and many upland land management activities have impacts on riparian and aquatic resources. Bettinger et al. (1998) attempted to account for impacts of management and disturbance on stream temperatures across large landscapes but did not project changes in riparian habitat. We will incorporate major physical and biological processes of riparian zones in an STM framework. Many analogies can be formed between existing STM for upland vegetation and the dynamics of valley-floor landforms and riparian plant communities. It may be possible to describe long-term riparian geomorphic and vegetation states, disturbance probabilities, and transitions among states for specific strata of riparian potentials. Drainage networks might be divided into discrete networks with different disturbance regimes similar to the stratification of potential vegetation types. Stream segments might be classified according to both their existing and potential characteristics and their succession described with transition probabilities based on hydrological disturbance regimes. Changes in riparian characteristics could consider both fluvial (e.g., floods) and nonfluvial (e.g., fire) disturbances. Treatment priorities might be based on channel instability and geomorphic and vegetation potentials. In addition, it might be possible to link upland episodic disturbance (e.g., wildfire) and riparian characteristics.

Linkages to Other Modules and Corporate Data

The STM module will generate several spatial and nonspatial data sets that should link well to other INLAS modules. The VDDT and TELSA models project the structural condition and cover type of grassland, shrubland, woodland, and forest vegetation. The VDDT model generates area estimates (hectares) for combinations of structure and cover in several environmental strata (as indicated by potential vegetation) by using an annual time step. It also tracks the area affected by individual disturbance transitions for each simulation year. Outputs are available in text files that can be readily transformed into databases. The TELSA model produces the same kinds of information and GIS coverages (e.g., maps) that can be used to examine spatial patterns of vegetation structure and composition as well as disturbances that drive vegetation change. We will adjust outputs of vegetation conditions and disturbances to fit the needs of wildlife habitat modeling and other modules to the degree that our models can produce appropriate information.

The VDDT and TELSA planning tools use vegetation structure classes that are derived from those suggested by Oliver and Larson (1996) as modified by O’Hara et al. (1996) and used in the interior Columbia basin scientific assessment (Quigley and Arbelbide 1997). Our modification of those structure classes will split some forest structures into classes for wildlife habitat modeling based on diameter of dominant trees. Discussions with USDA Forest Service Pacific Northwest Region planning personnel indicate that our structural classification should fit well with proposed corporate data standards.[3] The Natural Resource Information System (NRIS) proposes the use of the O’Hara et al. (1996) structure classes as one of the acceptable corporate data standards. In addition, a draft structural classification for the Pacific Northwest Region uses tree diameter breaks that are compatible with our structure classes. Our potential vegetation classes also should fit well with corporate data standards because we use aggregates of ecoclasses (Hall 1998).

Our vegetation cover type classes match those currently in use by Blue Mountains national forests (see footnote 2). However, they may not fit well with standard cover types that may be used in the future by the USDA Forest Service Pacific Northwest Region (see footnote 3). The Region’s draft standards match NRIS standards and consist of Society of American Foresters (Eyre 1980) and Society for Range Management (Shiflet 1994) cover types. We found those cover types, which were designed for categorizing vegetation cover across the entire United States, to be insufficiently refined for mapping wildlife habitat and stratifying economic product potential at the scale of our study area.

Validation and Sensitivity Analysis

Model validation is important in evaluating the accuracy and reliability of model projections. Landscape simulation models can be difficult to validate empirically because projections of current conditions into the future may take decades to evaluate, and unforeseen disturbances or management approaches may generate different futures. If we could establish vegetation structure and composition conditions for the Upper Grande Ronde area at some point in the past, we might project those conditions to the present and evaluate differences from current conditions. However, the historical track of disturbances may be only one of many that could have occurred. Actual past disturbances may not have even been those that had a high probability of occurring. Given these difficulties and the relatively short timeframe for our work, we take two approaches to evaluating model projections. First, we will compare the projections from different vegetation

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modeling approaches to look for differences and similarities that may require further examination. Secondly, we will calibrate STM models with stand-scale forest models (e.g., FVS) that have been widely published and evaluated elsewhere.

Our modeling process will be based on stochastic or probabilistic disturbances. Vegetation transitions will be expressed in terms of probabilities. Both VDDT and TELSA have the capability of generating many Monte Carlo simulations by using random number seeds in calculating probabilities. We plan to repeatedly run individual management and disturbance scenarios to examine the effects of stochastic variation on model results. Our intent is to express model results as probabilistic rather than providing only one result for each scenario. In addition, we plan to vary key disturbance probabilities by one or two standard deviations from the calculated or assigned values to gauge model sensitivity.

Products and Audience

Land and wildlife managers in the Blue Mountains province are the targeted users of the research findings and management tools produced from the activities outlined in this paper. Clients include managers of public, private, and tribal lands in the Blue Mountains province, encompassing economic and social interests related to management of timber, livestock, wild ungulates, salmon, vertebrates, and plants of conservation concern. In particular, the Blue Mountains national forests are beginning revision of land management plans. The STM may offer some advantages for land management planning. The modeling framework can be applied to a variety of vegetation types and environments. The models are more easily understood than previous planning models and may provide for better public involvement in the analysis process. The coarse resolution of the internal modeling states in the STM makes them relatively easy to build, edit, and execute. Technical users also may include scientists, public groups, and resource specialists. Application of the concepts and relations developed as part of this research and associated management tools will also extend beyond the Blue Mountains to similar environments in other provinces of the Pacific Northwest and intermountain West.

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Peter Bettinger, Lowell Suring, and Steve Wondzell provided helpful review comments. Funding for this project was provided by the U.S. Department of Agriculture, Forest Service, Pacific Northwest Research Station.

Metric Equivalents

When you know:	Multiply by:	To find:
Miles	1.609	Kilometers

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Chapter 3: Application of the Forest Vegetation Simulator and Related Tools for Integrated Modeling of Forest Landscapes

Alan A. Ager[1]

Abstract

This chapter describes the use of stand-level growth simulators to address landscape planning issues, and outlines work by the Interior Northwest Landscape Analysis System (INLAS) project to enhance the functionality of the Forest Vegetation Simulator and related tools to meet the needs for landscape analysis tools. Stand-level growth models are widely used in the Forest Service and other agencies, and they are logical candidates to use as the core for an integrated framework of the kind envisioned for INLAS. However, a number of modifications are needed to facilitate wider application of these tools to address strategic planning and forest management issues. These proposed modifications include improved data linkages and streamlined methods for building scenarios and summarizing results and are described in this chapter.

Keywords: Landscape simulation, landscape ecology, Forest Vegetation Simulator, forest planning.

Introduction

There is growing interest in applying landscape ecology and simulation methods to forest management problems (Liu et al. 2000, Mladenoff and Baker 1999, Spies et al. 2002). Simulation methods provide the broad and flexible framework needed to model natural disturbances, forest succession, and management on large landscapes. Specific problems of interest include studying the effects of natural disturbance on aquatic and terrestrial habitat reserves (Johnson et al. 1998, Maffei and Tandy 2002) and developing spatially explicit schedules for fuel-reduction treatments (Finney Chapter 9). Analyzing these problems by using traditional methods used in forest operations research (Dykstra 1984) is difficult owing to the stochastic nature of disturbance processes and the need for spatial detail in strategic planning models. Many new simulation modeling lineages have evolved over the past 10 to 15 years, and the application of these models is continuing to grow in scale and complexity, resulting in many sophisticated systems for

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simulating forest succession, management, and disturbance on large landscapes in the Western United States (Bettinger et al. Chapter 4, Graetz 1999, Hof and Bevers 1998, Kurz et al. 2000, Liu et al. 2000, McCarter 1997, Sessions et al. 1999, Spies et al. 2002, Weise et al. 2000). The growing frequency of severe wildfires on interior forests has created a need for strategic planning models that examine the costs and benefits of fuel treatments on large landscapes (Johnson et al. 1998, Sessions et al. 1999) and development of spatially explicit fuel-treatment strategies that best meet multiple resource goals and constraints, including regulatory standards for terrestrial and aquatic species. These problems have complex spatiotemporal dimensions that must consider forest management, natural disturbances, and forest succession over time on large landscapes.

Focusing on recent work in the Western United States, there are two commonly used approaches to simulate changes in forest vegetation. The first, a **state and transition** approach, stratifies forest and other vegetation into states (e.g., forest structure, cover type) that change according to transitions representing disturbance, management, and succession. State and transition models were used for the Columbia River basin assessment (Hann et al. 1997), which led to improved software, and more recently, application for forest plan revisions. State and transition models and their application in the Interior Northwest Landscape Analysis System (INLAS) project are described by Hemstrom et al. (Chapter 2).

A second approach involves the application of tree-level growth simulators (Hann et al. 1995, Stage 1973, Wedin 1999) to model each and every stand on a landscape (Crookston and Havis 2002, Crookston and Stage 1991). Because these methods are relatively well established and readily available, they are logical choices to address specific kinds of resource analyses. However, our review of these tools and methods suggested that simulating management scenarios on forested landscapes remains a complex process. Many specialists on national forest ranger districts who do analyses of alternative management scenarios are largely baffled by the array of existing tools and required data formats, as well as how they can be adapted to project-level work.

This paper briefly summarizes the development and application of stand-level simulation models and their application to management problems in forests in the Western United States. Subsequent sections describe specific improvements to existing methods to help build a coherent modeling framework and facilitate wider application of stand-level models to address strategic planning issues on watersheds with multiple ownerships.

Stand-Level Landscape Simulation and Planning Models

The stand-level simulation approach has been used in a number of applied research projects over the past 10 to 15 years; however, much of this work is either not published or not described in detail in symposia and other documents. In the simplest approach, landscapes were modeled with stand simulators by simply batch processing all the stands in an area and linking the results to geographic information system (GIS) stand maps. Extending Forest Vegetation Simulator (FVS) to landscapes was advanced with the developments of the Landscape Management System (McCarter et al. 1997), the Parallel Processing Extension (PPE) to the FVS (Crookston and Stage 1991), the Prognosis Environmental Indicators model (Greenough et al. 2002), and SUPPOSE, a visual interface to FVS (Crookston 1997). The Parallel Processing Extension added important functionality (Crookston and Stage 1991) in that it provided the means to consider contagion, treatment priorities, and overall landscape condition during the simulation.

Graetz (1999) later demonstrated how stand-level models could be incorporated into landscape optimization systems (Bettinger et al. Chapter 4).

Much of the functionality in current landscape models is derived from the numerous FVS extensions and postprocessors that allow for scaling the simulation system to the problem at hand. An array of simulation capabilities, including the dynamics of fuels and fire effects, (Beukema et al. 1997a), insect and disease mortality (Roberts 2002), economics (Fight and Chmelik 1998, Renner and Martin 2002), etc., enhance the utility of this overall approach for landscape applications. Linkages to visualization systems (McGaughey 2002, 2004) and interfaces like SUPPOSE (Crookston 1997) or the Landscape Management System (McCarter 1997) make it possible to conduct simple landscape simulations with a broad array of capabilities.

One common feature among stand-level modeling projects is that tree lists are usually imputed for stands where data are missing by using a most-similar-neighbor (Crookston and Havis 2002, Moeur and Stage 1995) or K-nearest-neighbor (Ohmann and Gregory 2002) approach. Imputing means that the tree list is obtained from an existing sample of tree lists rather than estimating a new tree list. This process is necessary because tree list data rarely exist for every stand in a project. There have been several recent advances in methods to impute stand data (Crookston et al. 2002, Temesgen and LeMay 2002). More work is needed in this area to determine the effects of imputation errors on different outputs of landscape simulations.

The functionality of the stand-level approach began to more closely match that of traditional forest planning models with the work of Liu et al. (2000), Graetz (1999), Wedin (1999), and others to optimize the scheduling of treatments to meet landscape goals. In this approach, alternative management scenarios are simulated for each stand, and heuristic search algorithms are used to find a combination that best meets the landscape goals. Goal functions are formulated to allow for multiple-weighted goals. Several projects are now using heuristic methods (e.g., Hummel et al. 2002) to sort through simulations a posteriori to find prescriptions that maximize single- or multiple-weighted objectives. In the work of Graetz (1999) and Wedin (1999), the growth and mortality code was extracted from FVS and incorporated into a stand optimizer. This general lineage of landscape simulation/optimization models is reviewed by Bettinger et al. (Chapter 4).

Another significant enhancement to stand-level models was attained when spatially explicit stochastic disturbance was incorporated into landscape planning models (Graetz 2000, Johnson et al. 1998, Sessions et al. 1999). Periodic wildfire was simulated with the Fire Area Simulator (FARSITE, Finney 1999) and fire mortality functions were used to update tree lists after each wildfire. This work represented a significant convergence between landscape ecology models with those used in forest planning and harvest scheduling (Mladenoff and Baker 1999). Spatially explicit models for insect disturbance also have been integrated into landscape simulations (Beukema et al. 1997b, Smith et al. 2002).

Work continues on many aspects of incorporating nonforest products values into landscape simulation models (Greenough et al. 2002). Of particular interest are understory vegetation components, hydrology, wildlife models, and carbon pools. Greenough et al. (2002) provide an example of incorporating an array of environmental indicators into a stand-level simulation system.

Application of stand-level simulation methods to landscapes continues to grow in scale and number. For instance, the Coastal Landscape and Modeling Study (CLAMS) project (Spies et al. 2002) used stand-level modeling (pixels) to simulate 2.6 million ha of the coastal Oregon region.

Improvements to Stand-Level Growth Simulators for Landscape Applications

Somewhere in the various modeling lineages described in this volume lie the needs of ranger district specialists and forest planners who require the stand-level capabilities of FVS and some of the landscape capabilities in experimental models like those described by Bettinger et al. (Chapter 4). Many agency-sponsored development efforts toward this end have not had wide success owing to complex data structures, inflexibility, administrative overhead, accessibility, and other factors. At the same time, experimental systems used for research projects are neither designed for wide deployment nor to address more than a relatively narrow set of questions. Some of the issues that need resolution include appropriate data sources, tree list imputation, mechanics of building spatial scenarios, and linking various resource models. Methods are needed to quickly formulate, execute, and interpret realistic scenarios on large forested watersheds that contain multiple ownerships and complex arrays of management goals and intentions. For instance, a typical watershed in a Western national forest contains numerous management allocations, each having unique long-term management objectives ranging from fiber production, to scenic quality, to protection of habitat for federally listed species. The matrix of forest conditions and management goals is tedious to replicate in a landscape simulation. Further, an efficient simulation system for policy analysis requires a mechanism to rapidly alter the management matrix to test alternative scenarios.

Of prime importance is the ability to model fire and fuel dynamics over time and to visualize treatment response. Landscape planning models need the capability to measure wildfire hazard, as well as simulate prescribed fire and wildfire spread and effects on vegetation and fuels. These capabilities exist with the Fire and Fuels Extension (FFE) to the FVS (Beukema et al. 1997a), FLAMMAP (http://fire.org), and FARSITE (Finney 1999). However, for all these programs, there are significant implementation issues and little published case study in areas like the Blue Mountains. A major obstacle to extending stand-level simulators to landscapes is the problem of organizing spatial simulation units into landscapes and controlling their disposition over time. Simulation units are formed by overlaying GIS layers for stands, management intentions, riparian buffers, treatment alternatives, ownership boundaries, and other layers. The problem is complicated by fine-scale mosaics of federal land management goals and state, federal, tribal, industrial, and nonindustrial private land ownerships within a typical watershed. The resulting matrix of forest conditions and management goals can have several hundred elements for a given HUC4 watershed, making it tedious to formulate a given scenario. Furthermore, analysis of alternative scenarios requires repeatedly changing the array of management intensities to different land strata. Existing interfaces to the FVS like SUPPOSE (Crookston 1997) can simplify the process of organizing stands and management intentions into landscape scenarios by using policy labels (Vandenriesche 2002), although enhancements could significantly simplify the process.

Research Approach and Products

This work focuses on improving operational aspects of the FVS and related software in the context of landscape simulations, as well as adding functionality for resource problems that are of particular concern for the Blue Mountains region. The work will have relevance to efforts that are repackaging other stand-level simulators for landscape applications. Our approach will emphasize, but not be limited to, the improvement of existing software, data linkages, and documentation in terms of a case study. This work will complement the model development work described by Bettinger et al. (Chapter 4) and Hemstrom et al. (Chapter 2) concerned with larger scales and questions that demand features like optimization. The work will be targeted toward specific analyses, like fuels-reduction projects, where existing stand-level tools can be scaled up to address the issues at hand.

Key areas that will be addressed include the following:

1. Framework. A framework will be produced that describes the application of existing simulation tools for rapid development and simulation of management scenarios on landscapes having multiple ownerships. The framework will be a manual of methodology, including a synthesis and compilation of case studies. It also will identify ways to improve linkages among existing data sets and software. The framework will review approaches to address issues such as measuring wildfire risk, simulating prescribed fire, treatment constraints, and landscape visualization.

2. Software development. Modifications to existing software will be explored to build a coherent, functional set of software tools that can be applied to a typical project to capture the differences between various treatment scenarios in terms of potential wildfire risk, fuel loadings, insect mortality, visual impacts, financial outcomes, and other attributes. A significant component of this work will be adding to the capabilities of the FVS PPE. Linkages among FVS-related and other software will be examined to find ways to improve integration of different resource models.

3. Application. The tools will be applied on the Upper Grande Ronde watershed in parallel with other INLAS modeling work to analyze a variety of land management scenarios and their long-term outcomes.

Audience

The primary audience for the products of this work are district specialists charged with National Environmental Policy Act analyses of land management scenarios and forest planners who require detailed projections of forest conditions through time under alternative management scenarios.

English Equivalent

When you know:	Multi by:	To get:
Hectares (ha)	2.47	Acres

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Chapter 4: The SafeD Forest Landscape Planning Model

Pete Bettinger, David Graetz, Alan Ager, and John Sessions[1]

Abstract

We describe quantitative methods in landscape planning and the application of simulation and optimization to analyze alternative policy and management on large landscapes. Landscape planning models can help people see and think in whole-landscape terms and give them a common reference point for discussing conflicting values. Forested landscape conditions are projected through space and time and provide a way to help evaluate the differences among alternative forest policies, and accomplish certain management planning objectives with respect to landscape-level processes and goals. Evaluating alternative forest management policies across the interior West landscape is complicated by the need to recognize the role of stochastic disturbances such as fire, insect, and disease outbreaks. We describe the development of the Simulation and analysis of forests with episodic Disturbances (SafeD) model for the Interior Northwest Landscape Analysis System project. The SafeD model is a multiscale, hybrid simulation/optimization model that addresses both optimization of silvicultural prescriptions at the stand level and the spatial scheduling of these prescriptions on large landscapes to meet multiobjective goals.

Keywords: Forest landscape planning, fire, natural disturbances, forest planning.

Introduction

Resolving the myriad of forest policy problems in the Western United States is hindered by the inability of land managers, policymakers, and planners to analyze tradeoffs of alternative management scenarios on large, heterogeneous landscapes over long timeframes. With the growing emphasis on managing large landscapes, it has become difficult to identify, visualize, and resolve conflicts on landscapes where there is interest in

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multiple, long-term goals. In addition, the current patchwork of regulatory policies, existing landscape conditions, and landownership patterns has created a new matrix of operational constraints that virtually prohibit active restoration of risk-prone habitat (Quigley et al. 2001). In many cases, the federal regulations that protect aquatic and terrestrial habitats are frequently in conflict with management intentions aimed at moderating the threat of severe wildfires or other disturbances. On typical national forest lands in eastern Oregon, resource protection prevents fuel treatments and stocking control on over 75 percent of the nonwilderness lands (Wilson et al., in press), thereby perpetuating the cycle of fuels buildup and catastrophic wildfire. Low-value products from typical restoration activities and finite Forest Service budgets further reduce the areas that can be treated to control density and maintain healthy forest stands.

The past decade has seen the rise of landscape planning models that use simulation and optimization methods to help dissect policy and management goals. Landscape planning models evolved from a fusion of landscape ecology models and forest planning efforts (Mladenoff and Baker 1999, Sessions et al. 1999) and allow for the deduction of results otherwise unattainable owing to the complexity of the planning problem on large landscapes (Mladenoff and He 1999). The goal of these models is to provide a mechanism to simulate landscape change in response to varying levels of management, disturbance, and succession (Mladenoff and Baker 1999, Mladenoff and He 1999, Quigley et al. 1996, Roberts and Betz 1999, Sessions et al. 1999). These models hold promise for solving policy issues, such as those related to the management of disturbance-prone landscapes, while simultaneously meeting the concerns for forest and range sustainability and the viability of terrestrial and aquatic species.

Although these hybrid simulation/planning models are clearly valuable tools to sort out the strategic visions for multiownership watersheds, there remain many gaps, as well as barriers, to more widespread application. One area that deserves attention is the process for allocating an array of stand management goals over space and time to meet landscape-level goals. Management decisions at the stand level, as well as succession and disturbance processes, ultimately drive landscape change, and the linkage between decisions at the stand level and their influence on the attainment of landscape goals is poorly understood. Clearly, decisions at both scales are important components of landscape-scale planning. The integration of stand-level optimization processes and landscape-level optimization processes has yet to be demonstrated.

In this paper, we summarize analytical methods used in landscape planning and describe how this work is being further developed for the Interior Northwest Landscape Analysis System (INLAS) project area. The goal of this work is to create a multiscale (i.e., stand and landscape) model that can be used to sort out management issues on large forest and rangeland areas in the interior West. We discuss the concept of stand-versus landscape-level optimization in meeting multiobjective goals and the integration of these two modeling scales within the INLAS project. Our goal is to apply this modeling method to address the following questions:

1. Can alternative management scenarios designed at the stand level have a significant effect on measures of forest ecosystem health, commodity production, and cumulative effects when portrayed spatially at a landscape level? That is, do stand-level objectives prevent the attainment of landscape-level goals?

2. Do landscape-level objectives prevent the attainment of stand-level goals?

3. When measured at the landscape level, can the threat of fire and cumulative watershed effects be reduced through alternative management policies? Can the spatial distribution of management activities, within and across ownerships, significantly affect measures of forest ecosystem health, commodity production, and cumulative effects?
4. Do landownership patterns and behavior affect forest ecosystem health, commodity production, and cumulative effects, when portrayed spatially at a landscape level?
5. To what extent are commodity production, fire threat, cumulative effects, and fish and wildlife habitat goals compatible?

The answers to these questions will contribute to the ongoing discussion concerning sustainable management of Western landscapes. The flow of this paper proceeds first by discussing the literature associated with the optimization of stand-level goals, then the optimization of landscape-level goals. These two sets of goals are assumed to operate at different spatial scales and thus may not be complementary. We then describe an approach we are developing to integrate the two concepts by using a hybrid landscape simulation/optimization model. Landscape simulation or optimization models may offer some advantages for land management planning. In particular, the modeling framework can be designed to address a variety of management objectives and constraints, incorporate spatial representations of the landscape, and model processes at various scales. The fine resolution that landscape simulation and optimization models can support makes them more complex, yet can provide more detailed analyses of alternative policies. The model we propose developing will support an evaluation of policies in the interior West within the INLAS project.

Stand-Level Optimization

Stand-level optimization methods are used to develop optimal management prescriptions for individual stands, given a set of management goals. Stand-level optimization methods have evolved with the changing demands placed on forests. Initially the goals were to maximize economic or commodity production values but more recently have placed emphasis on noncommodity values. The approaches that can be used to develop optimal stand-level management prescriptions include the Hooke and Jeeves method (Haight et al. 1992, Hooke and Jeeves 1961), dynamic programming (Amidon and Akin 1968; Arthaud and Klemperer 1988; Brodie and Kao 1979; ; Brodie et al. 1978; Brukas and Brodie 1999; Chen et al. 1980a, 1980b; Gong 1992; Haight et al. 1985; Hool 1966; Kao and Brodie 1979; Yoshimoto et al. 1990), nonlinear programming (Kao and Brodie 1980), or specialized heuristics (Bare and Opalach 1987). Many of these approaches key off of whole stand growth-and-yield models or stand age/structure models, which do not tend to provide the tree-level data conditions necessary to facilitate the use of fire behavior models.

Most stand-level optimization methods reported in the literature focus on meeting forest economic or commodity production goals rather than the nontimber goals, such as a reduction in the threat of fire, which is becoming more important in the interior West. In fact, the optimization models that key off of individual tree growth-and-yield models (table 1) were developed with fixed-decision criteria, mainly economic, in mind. There are, however, some exceptions. Haight et al. (1985), for e.g., tracked biological indicators in the development of stand prescriptions although they were not influential in developing the management prescriptions. More recently, Haight et al. (1992) incorporated nontimber outputs into the development of optimal prescriptions by using penalty func-

Table 1—Stand-level optimization research and associated decisions when considering the use of individual tree growth-and-yield simulation models

Decision	Reference
Rotation age, or growing-stock level	Martin and Ek (1981)
	Haight et al. (1985)
	Arthaud and Klemperer (1988)
	Haight and Monserud (1990)
	Yoshimoto et al. (1990)
	Valsta (1992)
Thinning type	Haight et al. (1985)
	Arthaud and Klemperer (1988)
	Haight and Monserud (1990)
	Yoshimoto et al. (1990)
	Valsta (1992)
Planting density	Valsta (1993)
Multispecies management	Haight and Monserud (1990)
	Yoshimoto et al. (1990)
Uneven-age management	Buongiorno and Michie (1980)
	Bare and Opalach (1987)

tions to ensure the attainment of goals. And finally, Gong (1992) developed a multi-objective dynamic programming system to recognize nontimber values. However, Gong (1992) also noted a limitation on the number of dynamic programming state variables that could be used.

The types of forest stand-level goals we should consider for all landowners in the interior West could range from economic (maximize net present value) to biological (maximize mean annual increment) to ecological (minimize fire threat, maximize number of large trees produced). Providing flexibility in the established stand-level optimization techniques requires some level of developmental work, thus access to the computer code associated with the growth models. Also, linking these established techniques to a landscape-level simulation model is problematic. For example, an ideal landscape planning approach may require that the list of trees associated with each forest stand be tracked through time to enable an evaluation of fire hazard and other environmental effects. The number of stand tree lists to simultaneously track could easily exceed 100,000. An optimal prescription for each stand would need to be developed and perhaps adjusted as conflicts with landscape-level goals arise and as natural disturbances are modeled across the landscape. The ability to quickly access a stand's tree list and develop an optimal prescription, while attempting to achieve landscape-level goals, is therefore a priority.

A stand goal within (SafeD) is defined by some set of attributes that are desired of a stand at some future point (or points). An example may be to have a stand that has 60 percent of its basal area in western larch (*Larix occidentalis* Nutt.) with the remaining 40 percent in Douglas-fir (*Pseudotsuga menziesii* (Mirb.) Franco) and grand fir (*Abies grandis* (Dougl. ex D. Don) Lindl.). There is almost an unlimited number of stand goals that can be developed for any one stand—the only restriction is that goals must be based on attributes that are attainable from the data that describe the specific stand of interest (i.e., the "tree list").

Why are stand goals important for INLAS? One objective in INLAS is to create prescriptions for the current landscape that can be evaluated as if they were actually implemented on the ground. This will be done through computer simulation. Stand goals are crucial to the development of specific prescriptions because they give us a target and decision criteria for deciding if certain management actions are needed and in what quantities. In essence, stand goals quantify the desired conditions of a particular stand. When we optimize a stand prescription, we are essentially measuring attainment (or departure) from the stand goal. The closer we are to the goal, the better the prescription.

For any given stand in the INLAS study area, a stand goal may read, "Create a stand that minimizes its departure from a target Stand Density Index (SDI) value and concurrently maximizes the value of the stand–subject to a minimal harvest volume, when harvest is planned."

Mathematically this would be written as:

$$max \quad \sum_{p=1}^{n} \left((\mathrm{w1}) * \mathrm{VALUE} - (\mathrm{w2}) * \mathrm{SDI\_DE} \right)$$

$$\text{Subject to: } \mathrm{H}_p \geq MinHarv_p \qquad \forall\, p,$$

where

p = a single period,

n = the total number of periods,

w1 and *w2* = weights to emphasize importance of each attribute,

VALUE = an attribute that describes the value of the stand,

SDI_DE = an attribute calculated by squaring the SDI deviation (which is the difference between the obtained SDI and the target SDI),

H_p = the harvest level from the stand during period *p*, and

$MinHarv_p$ = a minimum harvest-level threshold during period *p.*

Landscape-Level Optimization

In most landscape planning processes, much consideration is given to the decision variables and the rules for assigning management activities to decision variables, the quantitative rules for selecting new plan configurations, and the length of time the activity selection process (i.e., search process) is allowed to proceed (i.e., how long the computer program is run). Quantitative relationships, or rules, to constrain or guide the assignment of activities across a landscape can be categorized in many ways; one such categorization is whether the relationships require spatial information. The use of spatial information can make goal achievement a complex procedure in forest planning applications, but is important for simulating landscape-level processes.

The spatial arrangement of wildlife habitat and forest management activities is important for a number of reasons, including complying with regulatory restrictions and organizational policies and addressing aesthetic concerns. Forest regulations, for instance, are placing increasingly restrictive limits on the size and spatial relationships of harvest units on both private and public lands (Daust and Nelson 1993). The National Forest Management Act (1976) provides guidance regarding the appropriate harvest unit size on national forest lands, and the Oregon Forest Practices Act (State of Oregon 1999) provides similar guidance for privately owned lands. As a result of a need to manage forest land within regulatory frameworks, forest management planning now often attempts to achieve landscape management goals by placing spatial constraints on the scheduling of management activities (O'Hara et al. 1989).

Landscape-level planning models that allow the optimization of a spatial arrangement of activities to meet a set of management objectives vary from the more traditional optimizations techniques, such as linear or mixed-integer programming (e.g., Hof et al. 1994), to the nontraditional, but increasingly common heuristic programming techniques (e.g., Murray and Church 1995). Classical models such as the Timber Resource Allocation Method (RAM) (Navon 1971) and the Forest Planning Model (FORPLAN) (Johnson et al. 1980) were designed to address the problem of optimal scheduling of harvests with forestwide constraints. These models were used from the 1960s to 1990s and are classical in the sense that they use linear programming to allocate resources and activities to timber stands, and to a limited extent, recognize spatial relationships. However, recognition of spatial features in forest planning generally requires the use of integer-decision variables. Thus as the problem size increases, the potential solution space also increases, but at a disproportionately greater rate (Lockwood and Moore 1993). Mixed-integer programming and integer programming techniques have been used to help solve these problems and produce feasible management plans, but these techniques have substantive limitations (directly related to problem size) when applied to large landscapes (Lockwood and Moore 1993).

To explore the capabilities of traditional techniques, Hof and Joyce (1992) described nonlinear formulations aimed at accounting for the amount of edge, the juxtaposition of different habitat types, the dispersal distance among habitat types, and the minimum size of a patch of habitat. Hof et al. (1994) also described a mixed-integer programming approach that incorporates probabilistic objective functions for wildlife viability concerns. These approaches were theoretical in nature yet expanded the research boundaries and provide valuable insight into a much broader range of capabilities of linear, integer, and nonlinear programming methods. The limitations of these techniques persist, however, and both heuristics and simulation models have since been explored as possible alternatives.

Heuristics

The use of heuristics (solution methods that do not guarantee optimality of objectives has been achieved) in landscape planning is becoming more prevalent, particularly in planning processes where the potential solution space is large, or spatial constraints exist. Many types of complex, nonlinear goals (e.g., spatial and temporal distribution of elk (*Cervus elaphus*) habitat, as described in Bettinger et al. 1997), which have traditionally been considered too complex to solve with traditional optimization techniques, are now being incorporated into heuristics. In recent years, heuristics have been applied to scheduling problems related to forest management (Hoganson and Rose 1984), forest transportation (Murray and Church 1995; Nelson and Brodie 1990; Pulkki 1984; Weintraub et al. 1994, 1995), wildlife conservation and management (Arthaud and Rose 1996, Bettinger et al. 1997, Haight and Travis 1997), aquatic system management (Bettinger et al. 1998b), and the achievement of biological diversity goals (Kangas and Pukkala 1996). Monte Carlo simulation, tabu search (TS), and simulated annealing (SA) are three of the more popular heuristics. Three other more recently developed heuristics, the great deluge algorithm (GDA), threshold accepting, and genetic algorithms, also seem to operate as well as the others. Some effort also is being made to integrate the aspects of each into hybrid heuristic techniques, although this research is in its preliminary stages in natural resource management. Although the use of heuristics does not guarantee that a global optimum solution can be located for a particular landscape planning problem, heuristics can produce feasible (and often very good) solutions to complex problems, in a reasonable amount of time.

Simulated annealing is a search technique that began to be widely used during the early 1980s in operations research fields (Dowsland 1993). The foundation for SA was first published by Metropolis et al. (1953) in a scheduling algorithm that simulated the cooling of materials in a heat bath–a process known as annealing. The SA technique is a Monte Carlo method that uses a localized search process, where a subset of solutions is explored by moving from one solution to a neighboring solution with a simple change of a characteristic of a single-decision variable (1-opt moves), such as the timing of harvest of a management unit.

Threshold accepting (TA) is similar to SA, and was introduced by Dueck and Scheuer (1990). The TA technique also uses a localized search process but uses a slightly different, and somewhat simpler, set of acceptance rules for a new solution than does SA. Threshold accepting accepts every new (proposed) solution that is **not much worse** than the previous solution (within a preset limit of the value of the current solution), whereas in SA, the probability that a lower quality proposed solution would replace the current solution is a function of the quality of the solution and a stochastic element.

The great deluge algorithm is similar to SA in that it uses a localized search process. The GDA was introduced by Dueck (1993) and derives its name from the conceptual framework on which the algorithm works. Consider a problem where the objective is to find the highest elevation in a fictitious landscape by simply walking around and measuring elevations. Logically you would want to continuously measure higher and higher ground rather than lower ground. The GDA starts at some unknown location in the landscape, and subsequently weather conditions would be modeled as though it is “raining without end,” flooding the landscape and making it easier to locate the higher elevations. As the water rises, the GDA moves around the landscape (the solution space) trying to “keep its feet dry” (by only walking on higher and higher ground), and eventually finding what it considers the highest spot on the landscape, or an estimate of the global optimum solution to a planning problem.

Tabu search has been successfully applied to a number of scheduling problems outside of forestry and wildlife management, such as those in telecommunications, transportation, shop sequencing, machine scheduling, and layout and circuit design problems (Glover 1990, Glover and Laguna 1993). Within forestry it has been applied to timber harvest scheduling problems with adjacency (green-up) requirements (Murray and Church 1995), as well as for developing forest plans that have landscape goals for elk (Bettinger et al. 1997) and aquatic habitat (Bettinger et al. 1998b). Tabu search with 1-opt moves such as the harvest timing of a management unit, short-term memory, and aspiration criteria is a good scheduling technique, but generally not as good as SA, TA, or GDA (Bettinger et al. 2002). Using 2-opt (the swapping of choices among two decision variables) (and greater) moves has allowed TS to produce results as good as SA, TA, or GDA (Bettinger et al. 2002), but at a fairly large computing cost (Bettinger et al. 1999). One advantage of TS is that it is well suited to parallel processing.

Genetic algorithms (GA) were developed initially by Holland (1975) in the 1970s. Diverse fields such as music generation, genetic synthesis, strategic planning, and machine learning have benefited from the application of GAs to the scheduling of resources (Srinivas and Patnaik 1994). The GAs have been applied to a limited extent in forestry (Falcão and Borges 2000, Lu and Eriksson 2000, Mullen and Butler 1999). Although GAs have proven to be fairly good in developing moderately complex forest plans (Bettinger et al. 2002), it is more difficult to implement GAs than SA, TA, or GDA. A hybrid GA/TS heuristic technique that utilizes 1-opt and 2-opt TS processes as well as a GA crossover process (Boston and Bettinger 2002) also has shown promise for developing moderately complex forest plans.

Simulation Models

Simulation models that schedule forest management activities similar to heuristics and traditional mathematical programming techniques can be developed to provide the spatial and temporal context to help guide policymakers who are given the task of evaluating strategic alternatives. These models might be considered favorable to use in situations where stochastic elements are modeled, making optimization difficult. Simulation models generally are developed to capture relevant features of the dynamic nature of some "target system" under study (Birta and Özmizrak 1996), and their reliability is highly dependent on the degree to which the models reflect reality (Li et al. 1993). Gaining reliability in a simulation model is not a trivial task. For example, ecological consequences can differ dramatically depending on the pattern of land use activities imposed on a landscape (Franklin and Forman 1987); thus one measure of reliability is in modeling realistic land use activities.

Many simulation models have been developed in the last two decades to model events or behaviors across landscapes. Franklin and Forman's (1987) was one of the first to simulate the ecological consequences of forest management activities on a landscape, and indicated that the pattern of management applied to landscapes can result in varying ecological consequences. Others (Flamm and Turner 1994; Gustafson and Crow 1994, 1996; Gustafson et al. 2000; Johnson et al. 1998; Li et al. 1993; Turner 1987; Wallin et al. 1994) have since developed models for forested landscapes that simulate a variety of activities or disturbances at various spatial and temporal scales. Simulation models have been widely used in other natural resource areas as well. For example, they have been developed to focus on other types of disturbances and landscapes, such as gypsy moth (*Lymantria dispar*) outbreaks (Zhou and Liebhold 1995) and grasslands (Gao et al. 1996). As with heuristics, the use of simulation models does not guarantee that a global optimum solution can be located for a particular landscape planning problem; in fact, most simulation models do not claim to be attempting to locate optimal solutions. Simulation models can, however, produce feasible (and often very good) solutions to complex problems, in a reasonable amount of time.

Some common drawbacks, however, of forest landscape simulation models include:

- Resolution of the landscape scale is low.

- Integration of activities within a hierarchical spatial structure is low. For example, small basic simulation units might be aggregated into larger management units, which might be aggregated into larger management units, which might be aggregated into even larger harvest blocks.
- Only a few variables are used to track and allocate activities, such as transition probabilities or stand age.
- Use of other socioeconomic or ecological information to track and allocate activities is low.
- Landownership is not explicitly recognized.
- Spatial allocation of harvests is stochastic.
- Key landscape variables, such as topography and stream networks, are not recognized.
- Regeneration harvest sizes are determined by using a normal distribution of harvest sizes.
- Broad management strategies are stochastically implemented.
- Initial conditions of the landscape are randomly assigned.

Two projects have been undertaken in the past 5 years to develop simulation models to overcome most of these limitations. The Coastal Landscape Analysis and Modeling Study (CLAMS) (http://www.fsl.orst.edu/clams/), centered in the Coast Range of Oregon, is developing the LAndscape Management Policy Simulator (LAMPS) model to evaluate alternative forest management policies across all landownerships, long timeframes (100 years), and large areas (2 million ha). The LAMPS model does not, however, incorporate stochastic fire events in the simulation of management policies. The Applegate Project (http://www.cof.orst.edu/research/safefor/) developed a hybrid landscape optimization/simulation modeling system called "Simulation and analysis of forests with episodic Disturbances," or SafeD (Graetz 2000), that incorporated stochastic fire events. Table 2 presents a comparison of a few of the more important aspects of four forest landscape simulation models: "Safe Forests" (Johnson et al. 1998), LANDIS (Gustafson et al. 2000), LAMPS (Bettinger and Lennette 2002), and SafeD.

Approach and Design of a Landscape Planning Model

The approach we are suggesting would be useful in evaluating the aggregate effects of policies across a forested landscape and centers on the ability to use spatial simulation or optimization techniques. This type of approach can provide managers, policymakers, and planners with the ability to think about forests and their management in ways unimagined only a few decades ago. Often called "landscape assessment and planning," these approaches help people see and think in whole-landscape terms (not simply single ownerships) and give them a common reference.

In support of the INLAS project, we are proposing the development of a spatial landscape simulation model that will use spatial analysis techniques to model forest change across all ownerships and over long timeframes. Although the model will use both strategic (long-range, coarse-scale) and tactical (short-range, fine-scale) planning methods, it is more appropriate to call it a midscale, or regional, simulation model than a fine-scale tactical planning model. Successful implementation requires effective interdisciplinary collaboration that addresses the economic, ecological, and social dimensions of proposed management policies. Bettinger (1999) proposed that four elements were required at appropriate levels for a system to be implemented effectively: people, databases,

Table 2—A comparison of recently developed landscape simulation models

	Simulation model			
Comparison criteria	**Safe forest**	**LANDIS**	**LAMPS[a]**	**SafeD[b]**
Spatial data components:				
Analysis area (ha)	400 000	600 000	600 000	200 000
Data structure	Vector	Raster	Vector	Raster
Minimum mapping unit	Varies	200 x 200 m	25 x 25 m[c]	25 x 25 m
Model characteristics:				
Recognize ecological and economic goals	Both	Ecological	Both	Both
Optimize multiple goals	Yes	No	No	Yes
Represent forest management activities	Yes	Yes	Yes	Yes
Represent landowner behavior	No	No	Yes	No
Represent stochastic events	Yes	Yes	Yes	Yes
Represent fire disturbances (spatially)	Partially	Yes	No	Yes
Represent insect disturbances (spatially)	No	No	No	Yes

[a] LAMPS = LAndscape Management Policy Simulator.
[b] SafeD = Simulation and analysis of forests with episodic Disturbances.
[c] Raster databases are converted to vector databases for use in the LAMPS model.

technology, and an organizational commitment to the project. This paper mainly addresses the development of appropriate technology for modeling management and stochastic disturbances at the midscale in the interior West. Although the four elements are interdependent, our assumption is that data development, hiring and management of highly trained personnel, and a commitment by the main supporters of the INLAS project (USDA Forest Service, Oregon Department of Forestry, and College of Forestry at Oregon State University) will be supplied at the appropriate levels and appropriate times. No system is perfect, as Bettinger and Boston (2001) point out, but how setbacks are addressed is important in maintaining a level of progress consistent with project time lines.

On completion of the model, managers, policymakers, and planners will have the capability to (1) evaluate the effects of fuel treatments on wildfire behavior; (2) identify economic, ecological, and social constraints associated with the application of various policies; and (3) locate areas (perhaps watersheds) that are particularly difficult to manage under various constraints. With this in mind, we now concentrate on the technical development of a spatial landscape simulation model, its components, and the types of activities we envision modeling. Obviously a recognition of economic, ecological, and social goals is important. However, given that a project of this scope involves multiple collaborators, a linkage from one model to the other is more likely; facilitating the linkage between models is important. In addition, landowner objectives may range from relatively simple (maximize net present value) to more complex (maximize timber volume produced with acceptable fire threat, or minimize fire threat with high volumes produced), and the ability to develop an analysis that recognizes the need to optimize multiple goals. The representation of a range of forest management activities is also important because a wider set of potential management activities may facilitate the achievement of certain goals. And finally, the ability to recognize or model stochastic events is becoming more important as these events shape the condition of forests much more than activities by humans. Spatially representing fire spread and insect outbreaks as a function of forest conditions and landscape characteristics is important. Obviously the previous forest management practices can affect the risk of a stochastic event occurring; thus when projecting future conditions, the planned activities will also likely affect these risk levels.

The Simulation and Analysis of Forests with Episodic Disturbances Model

As noted earlier, the main objective of this paper is to develop a spatial forest landscape simulation model that allows the portrayal of processes (management activities, stochastic events, etc.) and subsequent analysis of silvicultural treatments at both the stand and landscape levels. The approach and design of this modeling effort build on the efforts of Graetz (2000), who developed a preliminary model (SafeD) to incorporate fire and insect disturbances in a landscape planning system. The SafeD model evolved from the efforts of the Sierra Nevada Ecosystem Project (Sessions et al. 1999) and the Applegate Project (Graetz 2000). The SafeD model is a spatially explicit, hybrid simulation/optimization model that allows the achievement of multiple resource goals at both the stand and landscape levels, while recognizing stochastic disturbances, and management behavior. It uses a distance-independent individual tree growth model (similar to the Forest Vegetation Simulator) to facilitate the development of optimal stand prescriptions, a heuristic scheduling model to allocate prescriptions across the landscape, and a raster-based fire-spread model called Fire Area Simulator (FARSITE, Finney 1998) to model fire on the landscape. This modeling framework is attractive because it can schedule management activities that attempt to meet long-term landscape goals under an uncertain future of stochastic disturbances.

To recognize the achievement of optimal stand-level prescriptions, optimal landscape-level objectives, and to recognize stochastic events, the operation of the SafeD model is segmented into four processing stages (fig. 10). To recognize the importance of both stand- and landscape-level goals, SafeD first develops a set of optimal prescriptions for each stand. It then allocates the prescriptions to the landscape to achieve landscape-level goals. Stochastic events are then applied to the landscape in a spatial manner. Finally, the stand- and landscape-level goals are reevaluated and adjusted, if necessary, to reflect the changes that have occurred on the landscape and in affected stands.

Optimizing Stand-Level Goals

Within SafeD, prescriptions for timber stands are dynamically generated by a stand optimization model that uses a combination of the region-limited strategy and path (RLS-PATH) algorithm (Yoshimoto 1990). A number of potential stand-level objectives can be recognized, and an optimal prescription for each can be developed. One challenge for the INLAS science team and collaborators will be in defining the types of stand-level objectives that should be modeled. The types of objectives modeled in the Applegate Project (Graetz 2000) included limiting fire hazard, limiting insect and wind-throw hazard, enhancing wildlife habitat, improving fish habitat, and maximizing net present value. To achieve these objectives, tree harvesting and snag creation rates were varied, and the resulting residual tree growth monitored. Goal achievement was then measured by using both live- and dead-tree characteristics.

Because an optimal stand-level management prescription is developed for all stand-level objectives, a second challenge becomes deciding which prescription to actually apply to each stand. For example, if we had three potential objectives (maximize net present value [a single goal for a stand], minimize fire hazard [a single goal for a stand], or maximize net present value with an acceptable fire hazard [multiple goals for a stand]), three

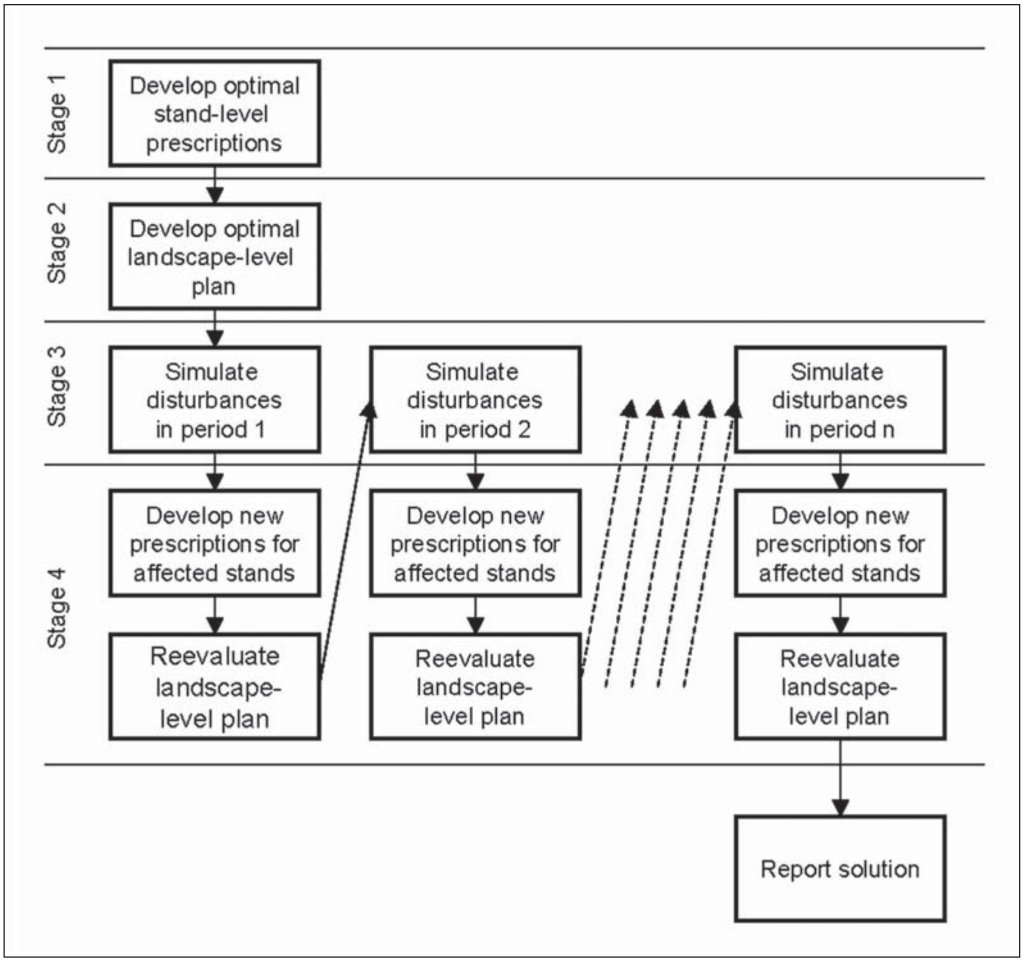

Figure 10—Four-stage process of the Simulation and analysis of forest with episodic Disturbance model.

optimal prescriptions would be developed for each stand on the landscape. Only one of these prescriptions can be applied to each stand, however. The prescription choice will be based on its contribution to overall landscape goals when applied to a stand, which may be a function of the spatial location of each stand.

Optimizing Landscape-Levels Goals

The second stage of SafeD consists of a landscape simulation model that distributes the optimal stand-level prescriptions through time and space given landscape-level goals and constraints. It is often confusing to those not closely familiar with forest planning efforts that stand- and landscape-level objectives are not necessarily compatible. A brief example may help clarify this notion. Let's say we have four hypothetical stands, each containing a different set of stand conditions. Applying stand-level optimization techniques to each to maximize net present value, e.g., may lead to a schedule that indicates each should be clearcut immediately. Although this may seem extreme, these prescriptions are optimal for each stand and represent decisions that are independent of the other stands. If the overall landscape objective of the landowner is to spread the harvests out evenly over time to avoid surges and dips in timber production, one or more of the optimal stand-level prescriptions could not be used, and some other prescription needs to be developed to represent the management of these stands. Therefore although optimal stand prescriptions can be generated, it is highly unlikely that they will lead directly to an optimal landscape-level scheduling solution (unless both stand- and landscape-level objectives are exactly the same).

The SafeD model designs planning problems as Model I nonlinear integer problems, where individual stands are tracked through time as they are regenerated or disturbed. The spatial location of each stand, as well as certain stand structural conditions, is important in adequately modeling management behavior and natural disturbance events.

It is clear from the previously provided summary of the literature that traditional techniques, such as linear or integer programming, are not appropriate for management planning at the landscape scale when integer variables are required to represent spatial landscape features. Therefore, a heuristic scheduling technique, the GDA, was chosen for use as the landscape-level optimization technique in the SafeD model. In the Applegate Project implementation of SafeD (Graetz 2000), the following objective function was used:

maximize:

$$\sum_{k=1}^{m}\sum_{j=1}^{q}\sum_{t=1}^{n} r_{k,j,t}\, x_{k,j,t} \quad , \tag{3}$$

where

k = a stand;

m = total number of stands in a landscape;

j = a prescription;

q = total number of possible prescriptions;

t = a time period;

n = total number of time periods in a planning horizon;

$r_{k,j,t}$ = the value of some stand attribute residing in stand k, when managed under prescription j, during time period t; and

$x_{k,j,t}$ = a binary (0-1) variable indicating whether prescription j was assigned to stand k during time period t.

A variety of constraints can be included in the SafeD model; however, the current version of the SafeD model uses only two. The first is a constraint limiting the number of prescriptions applied to a stand in each time period,

$$\sum_{j=1}^{q} x_{k,j,t} = 1 \qquad \forall\, k,t \quad , \tag{4}$$

and the second is a constraint on the level of equivalent roaded acres (ERA) that resulted from management activities. The ERA (McGurk and Fong 1995) is a measure used by the national forests to estimate cumulative impacts to a landscape, and to some extent, explains the hydrologic recovery of watersheds. There has been some debate, however, about the ability of ERA to be correlated with changes in measures of aquatic habitat (sediment and temperature) (Bettinger et al. 1998a). Within SafeD, an ERA constraint was applied in each time period and to each subwatershed:

$$\left[\frac{\sum_{k=1}^{p} ERA_{k,t}\ A_k}{\sum_{k=1}^{p} A_k}\right] \le ERA\_Threshold_{t,w}\ , \qquad \forall\, t, w \tag{5}$$

where

k = a stand,

p = the total number of stands in a subwatershed,

t = a time period,

w = a watershed,

$ERA_{k,t}$ = the contribution to equivalent roaded acres by stand k during time period t,

A_k = the area of stand k, and

$ERA\_Threshold_{t,w}$ = the upper limit on equivalent roaded acres allowed in subwatershed w during time period t.

Other constraints could be added to the SafeD model to guide the scheduling of management activities during the landscape-level optimization process. These may include timber harvest volume flow constraints, harvest adjacency (green-up) constraints, or the maintenance of a distribution of habitat patch sizes. Constraints also can be applied to individual landowner groups, or land allocations within landowner groups. Collaborators of the INLAS project will be called on to provide guidance in the development of appropriate landscape-level processes that must be recognized in future versions of SafeD; the landscape-level objectives and constraints will arise from these discussions.

Recognizing Stochastic Events

The third stage of SafeD distributes stochastic events across time and space. The brief description of stochastic events that follows is not meant to minimize their importance in a landscape planning effort. Within the SafeD model, fire events are applied in a spatial manner across the landscape in response to climatic variables and the management activities prescribed for each stand. Insect disturbance models were based on expert advice and are designed to simulate the expected growth-and-yield losses from forests over the long run. Episodic mortality of trees is embedded in the SafeD model to occur during drought periods (which are determined in a stochastic manner). Mild and severe drought periods will trigger the application of insect disturbances to the landscape. During these insect events, the structural condition of each stand in the landscape is examined, and a decision is made regarding the application of accelerated mortality rules.

Wildfires are applied to the landscape in the SafeD model by using the FARSITE model developed by Finney (1998). The FARSITE model is a fire growth-and-spread model that requires a spatial database describing the landscape. It includes methodology that allows the modeling of surface fire spread, crown fire spread, fire spotting, and fuel moisture content. Enabling the use of FARSITE requires knowing how many fires will occur during a specific period, how long they will burn, and where the initiation points are on the landscape. Probability distributions were used in the Applegate Project (Graetz 2000) to determine these parameters.

A number of sources of information are brought to bear on the modeling of stochastic events, including expert knowledge and functional relationship models. The literature on the effects of fire and insect events on landscapes is broad, yet little exists when one considers including these events in a forest landscape planning model. Some examples include Armstrong et al. (1999) who modeled the effects of natural disturbances (fires) on boreal landscapes nonspatially by assuming a distribution of forest types would be regenerated each year, and Reed and Errico (1986) who modeled the effects of fire in a linear programming model (again, nonspatial), but found that although fire losses may be stochastic, a close approximation to an optimal solution for a forest plan can be developed by using deterministic fire distributions that closely resemble the stochastic disturbance levels.

Reoptimize Stand-Level Prescriptions

The fourth stage of SafeD provides for a reoptimization of stand-level objectives in those stands affected by the distribution of stochastic events across the landscape. Landscape-level objectives are then reexamined, and prescriptions reassigned to reflect attainment of these goals.

Products and Audience

The approach we describe represents a refined forest landscape simulation model that is able to prescribe, schedule, and locate treatments dynamically in response to stochastic disturbances (fire, insects, etc.). This type of planning or policy analysis model will be useful in efforts aimed at evaluating the aggregate effects of policies across a forested landscape, and can provide managers, policymakers, and planners with the ability to think about forests and their management in ways unimagined only a few decades ago. Often called “landscape assessment and planning,” this type of approach helps people see and think in whole-landscape terms (not simply single ownerships) and promotes a common understanding of the basic processes that underlie landscape change.

We will apply SafeD to evaluate several alternative forest management policies and practices of each landowner in the pilot test area. The economic, ecological, and social effects will be measured for management scenarios that achieve specific goals related to fuels reduction, riparian management, threatened and endangered species habitat, and other values. At the initiation of the INLAS project, the intent (from the Oregon State University modeling perspective) was to support the Oregon Department of Forestry’s effort at evaluating landscape management alternatives for eastern Oregon, thus supporting the Forestry Program for Oregon and providing spatial projections of how the landscape might look under different management scenarios. It is hoped that simulations from SafeD also could provide national forest managers direction for choosing forest landscape management systems that address the tradeoffs associated with timber production, fire risk, and ecosystem health.

Analysis of alternative policies is the primary product of this modeling effort and will likely be a learning process for all involved. Outputs from the modeling effort will include a set of GIS databases that provide an indication of the effects of alternative management policies on the forest resources of eastern Oregon. Associated with these GIS databases are forest structural conditions (as represented by tree lists) that can facilitate further analyses of the effects of policies on wildlife and aquatic habitat resources on forested lands. Evaluating the impact of policies in a spatial context will require thinking about forests and forest resources in a manner heretofore difficult to perform. Although sets of data describing economics and commodity production levels will allow a relative comparison of alternative policy scenarios, examining alternative policies at a landscape scale (with maps) will likely require both quantitative and qualitative approaches.

A secondary product of the modeling effort involves a separate analysis of silvicultural treatments at the stand level. Here our goal is to understand which management practices are most beneficial (from a variety of perspectives: reducing fire hazard, maximizing net present value, etc.) to implement across broad classes of forests. Examining the resulting stand-level decisions, in light of forest-level goals and landscape disturbances, may provide management direction for both federal and private landowners, where multiple-resource goals influence the management of interior West forests.

The tertiary products developed by the modeling effort will be knowledge, algorithms, and software for modeling the effects of stand management and development on fine litterfall tree mortality, and snag longevity. The decay of large dead wood certainly is important for modeling wildlife, insects, and disease response to management and disturbance processes. Snags are tracked through time in the stand-level prescription model. Down wood, however, is not tracked through time, nor is the decay of either resource. The decay of wood is important for various biological effects models. The type of “bottom-up” analysis that would be provided (from trees to landscapes) and the growth projections that will ensue after natural disturbances may be useful in calibrating the INLAS state and transition modeling effort (Hemstrom et al. Chapter 2). Estimates of decay rates for fine litter, coarse woody debris, and snags would also then logically follow and provide a mechanism for summarizing these conditions over space and time, then facilitate an evaluation of the effects of management on wildlife species that utilize these resources. In addition, there has been only a limited amount of work aimed at incorporating fuel dynamics into the prediction of fire occurrence and behavior. In fact, usually only the mean rates of litter inputs and decomposition are used in modeling efforts, with no provision for variation based on stand structure and density levels (e.g., Keane et al. 1996). Yet, stand density strongly influences fuel accumulation (Maguire 1994) and litter decomposition (Piene 1978). Thus the development of models that estimate the effects of stand management on the production and decay of these resources is important.

The development of landscape simulation or optimization models requires a major collaboration between scientists, planners, managers, and policymakers to ensure that the kind of model developed will have widespread application and acceptance at the spatiotemporal scales at which it is used. As with most large-scale landscape modeling efforts, collaborators of the INLAS project will be called on to provide guidance in their areas of expertise. In large projects, with 10 to 20 internal collaborators and numerous outside interest groups shaping the look and feel of an analysis system, the expected goals of the project will likely change. For example, a fire specialist will be asked to assist in fine-tuning parameters related to the fire spread model. As refinements are made to these and other important components of an overall landscape modeling system, previously developed model components may need to be adjusted.

Although the modeling system we describe is well suited to address a wide spectrum of issues relating to the dynamics of change in coniferous forests, there remain a number of gaps in our knowledge about important disturbance factors that affect other significant resources. Of most interest are invasive plants, large herbivores, and hydrologic processes that regulate stream geomorphology and associated riparian conditions. Data and models are lacking to incorporate the effects of these factors into detailed simulation models like SafeD that model processes like stand growth in a continuous scale. In the absence of refined data and models, an alternative approach to building a landscape model that considers these factors is described by Hemstrom et al. (Chapter 2). Finally, for demonstration purposes, stochastic processes are incorporated into the results only once. For extensive analysis purposes, multiple runs will be required to assess the capacity of the landscape to not only produce the goals suggested by the policymakers but also to evaluate the projected dynamics of change that affect natural resources.

Two rather difficult issues to address are those related to sensitivity analysis and validation of the SafeD process. Sensitivity analyses give the customer of the simulation system products a sense of the importance of variables in the model. At present, no sensitivity analysis has been planned. Given the number of variables included in the fire model, the stand-level prescription generator, and the SafeD landscape model, the number of potential scenarios that can be modeled is infinite. The difficulty for a sensitivity analysis effort will be in determining which parameters to keep constant, and which to vary. It may be more difficult to determine which to keep constant in a sensitivity analysis, because it assumes that these variables are reflective of the human or natural system. Validation of large forest landscape models has generally been limited to an assessment of how well the model simulates what is typically known (e.g., recent harvest levels, recent areas treated with various management prescriptions). It has been suggested that one should use simulation models to evaluate paths from past conditions to the present. This would allow one to evaluate how well the models can explain past behavior of landscapes and may provide a good clue as to how they can help explain future behavior. However, projecting a historical landscape to the present is problematic. One would need databases that describe the landscape 20, 30, or more years ago to do this, an effort not planned within the INLAS project. Thus validation of large-scale forest landscape planning models is elusive. A number of verification processes are used to determine whether submodels within the larger SafeD modeling framework are working as intended, by comparing various output products to the models (e.g., the Forest Vegetation Simulator) from which the processes were derived. In addition, the cumulative results from a landscape simulation (harvest volumes, areas treated, fire risk, etc.) will be evaluated for reasonableness, which while not a validation, may suggest how well the simulation model is performing.

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English Equivalents

When you know:	Multiply by:	To get:
Hectares (ha)	2.47	Acres
Meters (m)	3.28	Feet

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Chapter 5: Assessment Techniques for Terrestrial Vertebrates of Conservation Concern

Barbara C. Wales and Lowell H. Suring[1]

Abstract

The quantity and quality of habitat for many wildlife species have changed throughout the interior Western United States over the last 150 years owing to a variety of natural and human-caused disturbances. Results from regional landscape models indicate that many species in this region are currently at risk of extirpation. Little is known, however, about how landscape mosaics and patterns of vegetation contribute to the viability of wildlife populations at finer scales. The increased ability to model vegetation and disturbances, including insects and fire, allows the opportunity to explore how potential changes in vegetation structure and composition may affect wildlife populations at finer scales. We identify methods to describe and evaluate habitat abundance, quality, and distribution across area and time, considering alternative management goals and assumptions at a landscape scale. Landscape simulation modeling results associated with a prototype subbasin in northeastern Oregon will be used to develop a decision-support tool to help managers and scientists design and schedule management activities that provide for conservation and recovery of terrestrial vertebrates.

Keywords: Decision support, habitat modeling, species of concern, wildlife.

Introduction

In recent work associated with the Interior Columbia Basin Ecosystem Management Project (ICBEMP), an approach was developed to evaluate how wildlife habitat for species of conservation concern is distributed across the interior Columbia basin (Raphael et al. 2001, Wisdom et al. 2000) (tables 3 and 4). These analyses provided insight into the abundance, quality, and distribution of habitats and to the status of associated terrestrial species across the basin. Findings demonstrated large declines in old forests,

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Table 3—Species of conservation concern occurring in the Upper Grande Ronde assessment area and considered for use in the development of INLAS

Common name	Habitat association	Federal status	Oregon Department of Fish and Wildlife status	Oregon Natural Heritage rank
Amphibians;				
Columbia spotted frog	Riparian	SoC	SU	S2?
Birds;				
Bald eagle	Riparian	T	T	S3B,S4N
Black-backed woodpecker	Broad-elevation, old forest		SC	S3
Brown creeper	Broad-elevation, old forest			S4
Brown-headed cowbird	All habitats			S5
Flammulated owl	Broad-elevation, old forest		SC	S4B
Great gray owl	Broad-elevation, old forest		SV	S3
Northern goshawk	Broad-elevation, old forest	SoC	SC	S3
Olive-sided flycatcher	Broad-elevation, old forest	SoC	SV	S4
Pileated woodpecker	Broad-elevation, old forest		SV	S4?
Pine grosbeak	Broad-elevation, old forest			S2?
Pygmy nuthatch	Low-elevation, old forest		SC	S4?
Three-toed woodpecker	Broad-elevation, old forest		SC	S3
White-headed woodpecker	Low-elevation, old forest	SoC	SC	S3
Williamson's sapsucker	Broad-elevation, old forest		SU	S4B,S3N
Willow flycatcher	Riparian	SoC	SV	S4
Mammals:				
American marten	Broad-elevation, old forest		SV	S3
Canada lynx	High-elevation forest	T	T	S1
Fringed myotis	Forest, woodland, and sagebrush	SoC	SV	S2?
Long-eared myotis	Forest, woodland, and sagebrush	SoC	SU	S3
Long-legged myotis	Forest, woodland, and sagebrush	SoC	SU	S3
Silver-haired bat	Broad-elevation, old forest	SoC	SU	S4?
Western small-footed myotis	Forest, woodland, and sagebrush	SoC	SU	S3
Yuma myotis	Riparian	SoC		S3

State Natural Heritage ranks
S1= critically imperiled
S2 = imperiled
S3 = vulnerable
S4 = apparently secure
S5 = secure
? = inexact rank
B = breeding range
N = nonbreeding range

Federal status
SoC = listed as species of concern by the U.S. Fish and Wildlife Service
T = listed as threatened by the U.S. Fish and Wildlife Service

Oregon status
SC = sensitive species, critical category
SV = sensitive species, vulnerable category
SU = sensitive species, undetermined status
T = listed as threatened by Oregon Department of Fish and Wildlife

Table 4—Scientific names of species of conservation concern

Common name	Scientific name
Amphibians:	
Columbia spotted frog	*Rana luteiventris*
Birds:	
Bald eagle	*Haliaeetus leucocephalus*
Black-backed woodpecker	*Picoides arcticus*
Brown creeper	*Certhia americana*
Brown-headed cowbird	*Molothrus ater*
Flammulated owl	*Otus flammeolus*
Great gray owl	*Strix nebulosa*
Northern goshawk	*Accipiter gentilis*
Olive-sided flycatcher	*Contopus borealis*
Pileated woodpecker	*Dryocopus pileatus*
Pine grosbeak	*Pinicola enucleator*
Pygmy nuthatch	*Sitta pygmaea*
Three-toed woodpecker	*P. tridactylus*
White-headed woodpecker	*P. albolarvatus*
Williamson's sapsucker	*Sphyrapicus thyroideus*
Willow flycatcher	*Empidonax traillii*
Mammals:	
American marten	*Martes americana*
Canada lynx	*Lynx canadensis*
Fringed myotis	*Myotis thysanodes*
Long-eared myotis	*M. evotis*
Long-legged myotis	*M. volans*
Silver-haired bat	*Lasionycteris noctivagans*
Western small-footed myotis	*M. subulatus*
Yuma myotis	*M. yumanensis*

native grasslands, and native shrub lands at 1-km resolution. This information has provided a basis for potential additional analysis and development of management direction at smaller scales and greater resolution.

The Interior Northwest Landscape Analysis System (INLAS) provides an opportunity to develop and implement a prototype approach for applying and focusing the results of the ICBEMP to regional and local natural resource planning efforts, in particular for updating land and resource management plans on national forests throughout the Northwest. Land managers working at these finer scales (e.g., province or national forest) need tools to help them evaluate habitat for terrestrial vertebrates at midscales. To provide for the conservation of all species across their ranges, as per the National Forest Management Act (NFMA 1976) regulations, national forest land managers require analyses that will incorporate areas large enough to encompass several home ranges of all species of concern. Such analyses also will provide insight into the potential contribution other public and private lands may make to the conservation of species and their habitats.

Prototype Study Area

The Upper Grande Ronde subbasin has been selected as the study area for initial development and application of INLAS. There are approximately 40 terrestrial vertebrate species of concern within the Upper Grande Ronde subbasin (see Wisdom et al. 2000). This initial list received additional screening against the State of Oregon Heritage Status Rank (Association for Biodiversity Information 2001) for species ranked S1–S3 (e.g., vulnerable or below) and against the state of Oregon sensitive species list (Oregon Department of Fish and Wildlife 1997) for species ranked vulnerable or critical. Occurrence of each of the resulting species within the study area was verified with local species checklists (e.g., Bull and Wisdom 1992) and the results of the Oregon Gap Analysis program (Kagan et al. 1999). Probability of occurrence also was evaluated based on habitats available in the study area. These screens resulted in a list of 24 potential species for analysis (table 3). Most of the species of conservation concern within the Upper Grande Ronde also occur throughout large areas within the interior West, and many of these species have home ranges that span multiple subwatersheds or larger scales. The tools developed through this project will be used to facilitate planning and evaluation of various management activities and should be useful at multiple scales. Such planning tools will be useful to help restore and conserve natural landscape patterns and functions over the long term.

Research Objectives

We propose to develop methods to describe and evaluate habitat abundance, quality, and distribution through time considering different management objectives and activities. To accomplish this, we will address the following:

- How will the current quantity, quality, and distribution of habitats that contribute to the long-term persistence of species of concern change in the future under different management regimes in the Upper Grande Ronde subbasin?
- How do the effects of roads, recreation, fire, insects, disease, timber harvest, grazing, and other disturbances (and their interactions) influence the viability and vulnerability of terrestrial vertebrates of concern in the Upper Grande Ronde subbasin?
- Develop analytical tools that are user-friendly and flexible to accommodate available data in other locations, thereby facilitating widespread application.
- Describe how effective broad-scale habitat models are in providing a useful context for mid- and fine-scale analyses and land management planning.

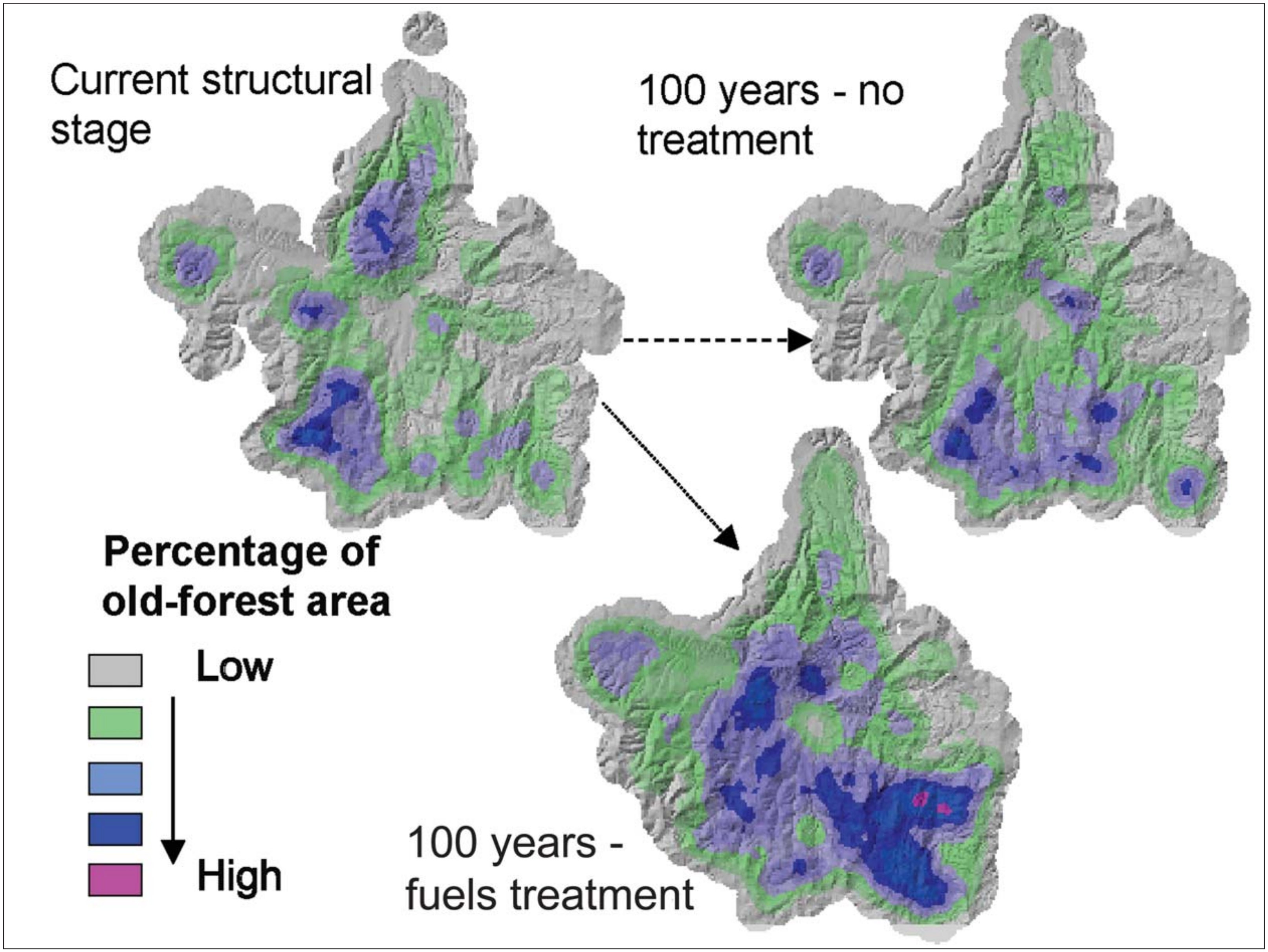

Figure 11—One potential modeling output comparing suitable habitat for an old-forest-dependent species for the current period and as estimated under two different management scenarios in 100 years.

Methods

Evaluating landscape change may be more important than current landscape structure in developing an understanding of long-term population dynamics of terrestrial vertebrates (Dunn et al. 1991, Knick and Rotenberry 2000). By using a combination of geographic information system spatial modeling and decision-support models (DSMs), we will evaluate changes in wildlife habitat under different management regimes through time as well as develop assessment processes for wildlife species at a subbasin scale. Figure 11 displays how one potential output might look comparing two different management scenarios through time. We will explore the use of Bayesian belief networks (BBNs), a type of DSM, as well as other more traditional modeling techniques such as habitat suitability index models. See Wondzell and Howell (Chapter 6) for more discussion regarding the use of DSMs.

Bayesian modeling is just one of numerous types of wildlife habitat modeling that can calculate an index of population response.[2] It can provide a modeling approach that (1) displays major influences on the persistence of wildlife populations and their values and interactions, (2) combines categorical and continuous variables, (3) combines empirical data with expert judgment, often from multiple experts; and (4) expresses predicted outcomes as likelihoods as a basis for risk analysis and risk management (Marcot et al. 2001). The models can rely on outputs from other models, such as projected vegetation, to estimate the amount of habitat available, and other environmental factors, to estimate the quality of habitat (Raphael et al. 2001). It is likely models will be developed at two scales, site-specific and subbasin, which will be hierarchically nested. The site-specific model will estimate habitat quantity and quality at the scale of a pixel (or stand), whereas the subbasin model will summarize those results to assess the overall conditions within a subbasin. Figure 12 shows an example of a site-specific belief network modified from the work of Raphael et al. (2001). Within the subbasin model, it is possible to assess the connectivity of hiqh-quality habitats, another important aspect for some wildlife species.

The wildlife models will rely heavily on the outcomes of the vegetation modeling described by Hemstrom et al. (Chapter 2) and Bettinger et al. (Chapter 4). Many of the species of concern in our study area are dependent on snags and coarse woody debris (CWD). Because insects, disease, and fire are imbedded in the vegetation modeling efforts, snags will be addressed. We will develop methods to quantify snag and CWD development within the vegetation models. In addition, a companion project in the same study area will be developing landscape models to predict snag and CWD densities in relation to vegetation type and landscape characteristics, such as distance to nearest roads and towns, elevation, and slope, which we will build into our habitat models (Bate and Wisdom 2001). We also will be working to develop close links with other resource modules such as recreation, social, and riparian. Although little empirical data exist on species distribution across the subbasin, we will use any available data to help build the models and use existing models such as those developed by McGrath et al. (2003), Sallabanks et al. (2002), and Roloff and Haufler (1997) (also footnote 2). Our knowledge on species environmental requirements and population dynamics differs widely per species, so some models will be better developed than others.

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[2] Roloff, G.J. 2001. Breeding habitat potential model for northern goshawks in the Idaho Southern Batholith. [Pages unknown]. Unpublished document. On file with: Timberland Resources, Boise Cascade Corporation, 1564 Tomlinson Road, Mason, MI 48854.

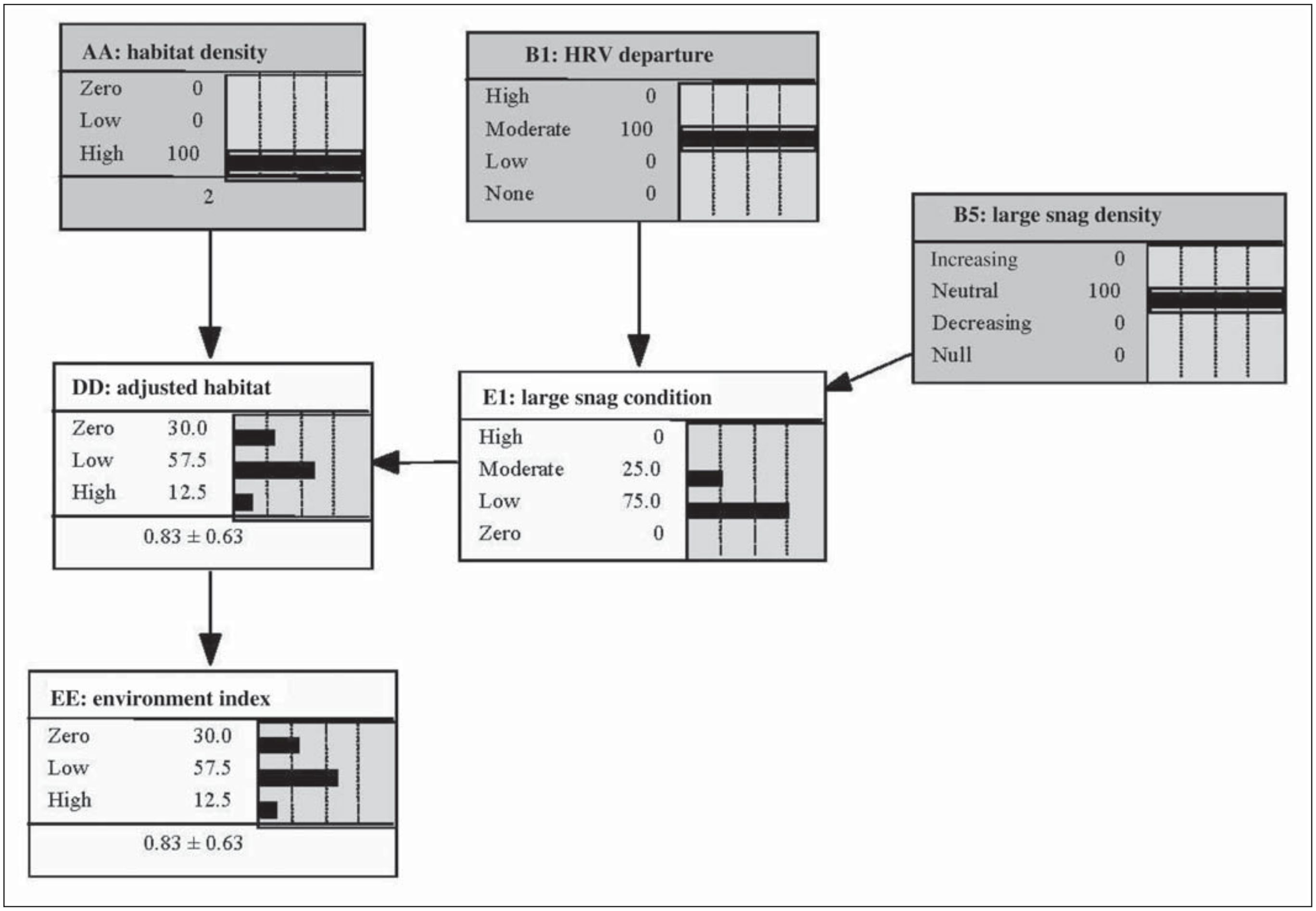

Figure 12—Example of a Bayesian belief network model adapted from Raphael et al. (2001). This example was used to quantify habitat quality and quantity within a subwatershed for pygmy nuthatch.

The performance of BBN models, as well as alternative approaches to BBN models, also may be evaluated where alternative models exist that are compatible with vegetation data generated by the INLAS base models (e.g., Hemstrom et al. Chapter 2 and Bettinger et al. Chapter 4). Performance of BBN and other models may be evaluated in various ways, including the use of Bayesian statistics (Lee 2000), or through other analyses of model predictions versus empirical observations (Rowland et al. 2003, Wisdom et al. 2002). Tests of model performance will provide an opportunity to explore how different procedures for modeling wildlife habitat compare in terms of their results, veracity, and compatibility with INLAS models for other resources. In addition, the models developed during this analysis will be evaluated through a companion project to be conducted in the study area by scientists from the University of Idaho, which will provide information in developing a final set of user-friendly models.

Products and Audience

As a result of this work, we will provide prototype decision-support models that can be used to describe the amount, distribution, and quality of habitat for terrestrial vertebrates throughout the interior West. These analytical tools will help managers and scientists design and schedule management activities that will provide for the conservation of terrestrial species at a landscape scale through time. We will apply this prototype to the Upper Grande Ronde subbasin. The models we build will give a relative index to habitat quality or species persistence, depending on the scale.

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English Equivalent

When you know:	Multiply by:	To get:
Kilometers (km)	0.06125	Miles

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Chapter 6: Developing a Decision-Support Model for Assessing Condition and Prioritizing the Restoration of Aquatic Habitat in the Interior Columbia Basin

Steven M. Wondzell and Philip J. Howell[1]

Abstract

The INLAS Aquatic Module is part of the larger Interior Northwest Landscape Analysis System (INLAS)—a multidisciplinary effort to develop midscale analytical tools to project succession and disturbance dynamics across landscapes in the interior Northwest. These tools are intended to be used to examine change in ecological and socioeconomic systems under various policy or management options (Barbour et al. Chapter 1). For the Aquatics Module, we are developing tools to assess midscale aquatic habitat in the context of the biophysical characteristics of streams and watersheds and landscape-scale processes, including natural disturbances such as fire, and alternative management scenarios. We will apply these analytical tools to a demonstration area (the Upper Grande Ronde River subbasin), where we will assess factors influencing conditions of aquatic habitat and water quality and evaluate the potential cumulative effects of alternative management scenarios on aquatic habitat, hydrology, and erosion. The tools we are developing are intended to help natural resource specialists and managers define the types of management most likely to be compatible with guidelines for aquatic species and their habitat and management objectives for other resources.

Keywords: Decision-support models, aquatic habitat, water quality, salmon, steelhead, bull trout, alternative management scenarios.

Introduction

Chinook salmon (*Oncorhynchus tshawytscha*) (Walbaum), steelhead (*O. mykiss* [formerly *Salmo gairdneri* Richardson), and bull trout (*Salvelinus confluentus* (Suckley)) have been eliminated from much of their historical range and are now listed as threat-

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ened or endangered within most of the interior Columbia River basin (USDA and USDI 2000). Other native fishes also have declined (Lee et al. 1997). Many factors have contributed to declines, including (1) overharvest; (2) blocked access and increased mortality of migrating fish from dams; (3) interactions between wild fish and hatchery stocks, which appear to impair fitness of wild stocks; and (4) degradation of spawning and rearing habitat (Federal Caucus 2000). Degraded water quality is closely linked to issues surrounding degraded spawning and rearing habitat. Thousands of miles of streams throughout the Columbia River basin, including the Upper Grande Ronde subbasin (Grande Ronde Water Quality Committee 2000), have been listed as impaired by the states under section 303d of the Clean Water Act for failing to meet water quality standards (Lee et al. 1997). Streams in USDA Forest Service (USDA FS) ownership are most commonly listed for failure to meet standards for sediment/siltation/turbidity, water temperature, and flow (Lee et al. 1997).

The USDA FS and other federal agencies, including National Marine and Fisheries Service (NMFS), U.S. Fish and Wildlife Service (USF&WS), and USDI Bureau of Land Management (BLM) have been developing broad-scale approaches to address aquatic and other land management issues within the region (FCRPS Biological Opinion 2000, Federal Caucus 2000, USDA and USDI 2000). These broad-scale plans recognize the importance of maintaining existing high-quality habitat in tributaries of the Columbia basin and restoring habitat that is currently degraded.

The success of broad-scale management depends on the ability of natural resource specialists to convert broad-scale management direction into mid- and fine-scale management practices. To do this, natural resource specialists, managers, and planners must be informed as to the nature and extent of potential impacts resulting from current management practices and proposed changes in those practices (Rieman et al. 2001). Specifically, natural resource specialists need to be able to assess (1) the ability of a stream (or watershed) to support species of interest and other desired resource values, (2) the current condition, and (3) the potential impacts of management decisions on future conditions. Managers and planners must be able to use this information to determine the type and location of management activities most likely to meet desired objectives and to prioritize these activities on the basis of multiple and sometimes conflicting objectives.

Management actions occur in systems with high natural variability and that have been altered by a number of historical and current land and water management practices. Thus, predictions of the potential effects of management actions are fraught with uncertainty associated with the ecological responses and the complexity of multiple management objectives and strategies under consideration (Rieman et al. 2001). To aid evaluations, land managers in the inland Northwest need tools that formalize these complex relationships into a common framework that describes aquatic habitat in the context of landscape processes and conditions, potential effects of management actions, and sources of uncertainty. There are currently no analytical tools available that provide managers the ability to assess conditions of aquatic habitats at mid to fine scales (i.e., 4th to 6th hydrologic unit codes or HUCs) in a landscape context and to analyze potential cumulative effects of management decisions, including forest harvest, fuels reduction, herbivory, and riparian management, on aquatic species and their habitats.

The goal of the proposed research is to develop a decision-support tool to help inform management decisions at midscales. The proposed research is guided by four primary questions:

- How have changes in landscape processes, such as fire, over the last 100 to 150 years affected aquatic habitat and populations of aquatic species?
- What and where are the principal opportunities to maintain and restore aquatic species and water quality?
- What are the cumulative effects of alternative management approaches on aquatic habitat and water quality?
- How can stream restoration opportunities be better integrated with management for other resources?

Review of Alternative Modeling Approaches

A variety of modeling approaches are available to address the questions we pose above. Below, we briefly review these modeling approaches and evaluate their suitability for this project.

Mechanistic Models

Existing tools are unable to adequately address the questions listed above for various reasons. First, many models are narrowly focused and thus do not include other factors that are likely to influence aquatic and riparian habitat. For example, the Stream Segment Temperature Model (SSTemp) (USGS 1999) is typical of reach-scale temperature models that calculate shading/sun exposure to the stream surface and use temperature and volume of water flowing into a reach to estimate a new temperature at the bottom of a reach. These models reliably predict the effect of site-scale modifications on stream temperatures within relatively short stream reaches. However, they are not designed to analyze temperature changes within entire stream networks. Secondly, most existing models have been designed to answer questions at different scales. For example, the aquatic-effects analysis model developed for the interior Columbia basin (Rieman et al. 2001) operates at too coarse a scale, whereas models such as SSTemp work at too fine a scale for subbasin planning. Thirdly, most mechanistic models are too complex, requiring extensive data and a high degree of expertise to run and analyze, both of which are frequently not available. Examples of these models include network-scale stream-temperature models such as SNTemp (USGS 2000) or distributed hydrology models, such as the Distributed Hydrology Soil Vegetation Model (DHSVM) (Wigmosta et al. 1994). The DHSVM, e.g., is designed to predict event-based stream discharges and annual water yield at watershed scales but requires detailed inputs of soil and topographic characteristics and is driven by spatially distributed energy and precipitation budgets. The DHSVM would need to be calibrated to match observed hydrographs and then validated by predicting hydrographs for a different series of storms or a different watershed. However, it would usually be difficult to obtain local calibration data, and the calibrated model will not be readily transferable to other watersheds. Fourthly, most existing models lack followup support for technology transfer to agency management units to help natural resource specialists parameterize the models to local conditions and then run the models. Finally, only a few empirical models have been developed for the interior West that relate landscape variables and processes to aquatic habitat or species because the empirical basis for these relationships is limited. All these factors limit the use of complex, mechanistic models as planning tools that can be applied to subbasins across the entire Columbia River basin.

Each of the models described above offers some utility toward analyzing a specific problem related to land management practices and their effect on aquatic habitat. None of these models, however, attempts to link landscape processes and the range of land management practices to cumulative effects on either habitat capacity or water quality. We do not know of a linked series of models that would enable a user to simultaneously examine multiple, midscale land management issues and their effect on aquatic habitat capacity and water quality.

Expert System, Expert Evaluation, and Statistical Models

Recently, several models (for example, the Ecosystem Diagnosis and Treatment [EDT] Method, the Plan for Analyzing and Testing Hypotheses [PATH], and the Cumulative Risk Initiative [CRI]) have been developed to help inform decisions related to salmon management in the Columbia River basin. The EDT model (Mobrand Biometrics 1999) was designed to compare effects of alternative strategies for managing hatcheries, hydropower, and harvest. The EDT model was designed to be a comprehensive model, accounting for spatial and temporal interactions between habitat conditions, competition, and predation, and projecting cumulative effects (ISAB 2001). Consequently, the model is relatively complex, requiring qualitative and quantitative habitat information about species, which are represented as a set of rules relating survival to habitat conditions. The model is fine scale, utilizing habitat information at the 6^{th} HUC (HUC6 level) and some 40 habitat parameters (ISAB 2001). The EDT model will be required in future subbasin assessments in the Columbia River basin, and work is currently underway to integrate EDT into a broader assessment framework to evaluate fish and wildlife species across aquatic, riparian, and terrestrial environments (Marcot et al. 2002). Although EDT is a habitat-based model, it was not designed to link instream features to processes occurring in upland areas—processes such as fire and other natural disturbances or land management activities such as harvest or grazing. Also, EDT does not directly assess uncertainty in predicted outcomes, and because the model is complex, it is difficult to ground-truth all input data and to review or edit rules linking habitat to the survival of fish species (ISAB 2001). These factors would make EDT difficult to use in INLAS.

The PATH and CRI models are statistical modeling approaches focused on population dynamics of anadromous salmonids. The PATH model (Marmorek et al. 1998) was designed to examine Snake River listed salmon and steelhead and to evaluate management options for these species as affected by survival in specific life stages. The model's main focus is the survival of fish migrating through the mainstem river corridor and the influence of variations in the management and operation of the hydropower system on fish survival. The CRI model statistically examines the survival of fish in freshwater habitats as one generalized component of the overall extinction risk for all listed anadromous salmonids in the Columbia River basin (CRI 2000). However, CRI does not link survival to specific habitat attributes nor does it consider how habitat might change under different management scenarios. These factors make PATH and CRI unsuitable for use in INLAS.

Decision-Support Models

Decision-support models (DSMs) are based on decision analysis and provide possible alternatives to the more traditional modeling approaches described above. Decision analysis can be broadly divided into two components: (1) risk analysis and (2) risk management. Risk analysis is the process of identifying the results of alternative decisions. Thus, risk analysis can help natural resource specialists examine the expected effects of different management strategies (Varis and Kuikka 1999). Further, because risk analysis uses explicit, quantitative methods to examine uncertainty (Clemen 1996), risk analysis can be used to assess the influence of various sources of uncertainty (e.g., variability) on the probability of achieving specific outcomes given a particular decision. Additionally, risk analysis can be used to estimate the value of additional information (e.g., monitoring, watershed analysis). Risk analysis, however, cannot choose the "best" management strategy. Risk management is the process of assessing the value of possible outcomes. A formal risk management plan requires that decisionmakers (i.e., managers) define their attitudes about risks and assign quantifiable values (e.g., an economic cost or a societal benefit) to each possible outcome identified in the risk analysis.

The use of DSMs to conduct risk analysis for the INLAS aquatic module offers several specific advantages that meet our modeling needs. The DSMs can:

- Provide a quantitative framework to describe the current understanding of the complex interrelationships between landscape properties and aquatic habitat, to explicitly define these relationships within the model structure, and then to test the influence of each variable on expected outcomes.
- Use outputs from other models (e.g., the projected changes in vegetation, fire severity and extent, management activities, and other variables from other INLAS modules) to project changes in aquatic habitat units at selected points in time.
- Use expert opinion to parameterize input variables when empirical data are lacking. Additionally, the influence of those opinions and the underlying assumptions are explicit and consistent within the model. The model is transparent in that key assumptions and the values of all variables, including those based on expert opinion, are displayed.
- Incorporate empirical data, mechanistic models, meta-analyses, and subjective probabilities from experts into a single model, integrate information from several disciplines, and use that information to analyze alternative management scenarios.
- Be used to test effects of alternative assumptions on outcomes.
- Determine the relative contribution of each variable to model outcomes through sensitivity analysis of model variables.

At least two DSMs have been developed and are currently in use in the Pacific Northwest and interior Columbia basin. The Ecosystem Management Decision Support System (EMDS) (Reynolds 1999, Reynolds et al. 2000), developed by the Pacific Northwest Research Station, is a fuzzy logic rule-based model providing decision-support tools for landscape analysis and restoration priority setting. However, the aquatic applications to date have primarily focused on disturbance from landslides and debris flows, rather than fire, in basins west of the Cascade Range. Further, current applications of EMDS are driven primarily by inchannel variables, such as large wood and pools, rather than upland characteristics and management activities. Aquatic applications also have not been integrated with other resource areas (e.g., vegetation management, terrestrial species).

A Bayesian belief network (BBN) model was developed for the aquatic effects analysis of management alternatives proposed in the environmental impact statement for the interior Columbia basin (Rieman et al. 2001). This model has been used to evaluate broad-scale effects of federal land management alternatives on aquatic habitat and species for the interior Columbia basin. However, the model is designed for broad-scale analyses of Interior Columbia Basin Ecosystem Management Project (ICBEMP) management alternatives. Also, the model does not directly examine the effects of specific management practices. Rather it uses measures of management activity, such as road density, to project habitat condition over large spatial scales. Although neither the existing versions of EMDS nor the ICBEMP BBN model are sufficient to meet our objectives, they are examples of the types of DSMs most likely to meet the modeling needs identified above.

Modeling Approach

We will develop a DSM to evaluate the effects of alternative land-management scenarios on salmonid habitat at the subbasin scale within the interior Columbia basin. The work described here is focused on risk analysis. Objectives include:

Objectives

- Develop midscale analytic tools to:
    - Assess aquatic habitat condition in the context of the biophysical characteristics of streams and watersheds and landscape-scale processes.
    - Compare potential cumulative effects of alternative management scenarios on aquatic habitat.
    - Help define where and what types of land and water management treatments may be most compatible with aquatic habitat considerations (e.g., key habitats and limitations of species, sensitive soils, existing roads).
- Develop analytic tools that can incorporate new information to resolve key uncertainties in an adaptive management framework.
- Develop analytic tools that are spatially explicit (i.e., can analyze and report information at various fine and mid scales),
- Develop analytic tools that are sufficiently flexible to accommodate a variety of available data and that facilitate widespread application,
- Complement other existing midscale aquatic analytic tools (EDT and EMDS).

Methods

The initial phase of decision-model development will be to identify the decision context(s), responses to be modeled and management alternatives. Decision models will then be structured specifically to address each decision situation and to link with other INLAS modules. Conditional dependencies will be parameterized by using the existing data from the region and data gathered from published studies via meta-analysis (Gelman et al. 1995). Where empirical data or other model output are lacking, expert opinion will be solicited from a panel of species and habitat experts and used to parameterize variables included in the models (Morgan and Henrion 1990). To explicitly incorporate uncertainty, relationships between environmental variables and habitat capacity will be modeled as conditional dependencies (probabilities), combined in a BBN (influence diagram) (Haas 2001), dynamic optimization model (Williams 1996), or similar decision-model form. Sensitivity analysis will then be performed on these models.

Links to Other INLAS Modules

Although the streams make up only a tiny percentage of the total land base of the Upper Grande Ronde watershed, they can be impacted by land use activities occurring anywhere within the watershed. Thus, the decision-support tool developed for the INLAS Aquatic Module needs to be linked directly to many other INLAS modules. Potential direct linkages between the vegetation, disturbance, riparian, wood utilization, herbivory, recreation, and economic modules are illustrated (fig. 13). We will use inputs from these INLAS modules to characterize watershed attributes that directly or indirectly influence the aquatic system and then analyze those projected landscapes to evaluate likely habitat capacity and water quality effects for short-term (e.g., 5- to 10-year) and long-term (e.g., 100-year) timeframes that would result from specific management scenarios. Aquatic habitat capacity potential also is affected by physical attributes of the subbasin, attributes such as slope steepness, soil types, and valley floor widths, which are fixed

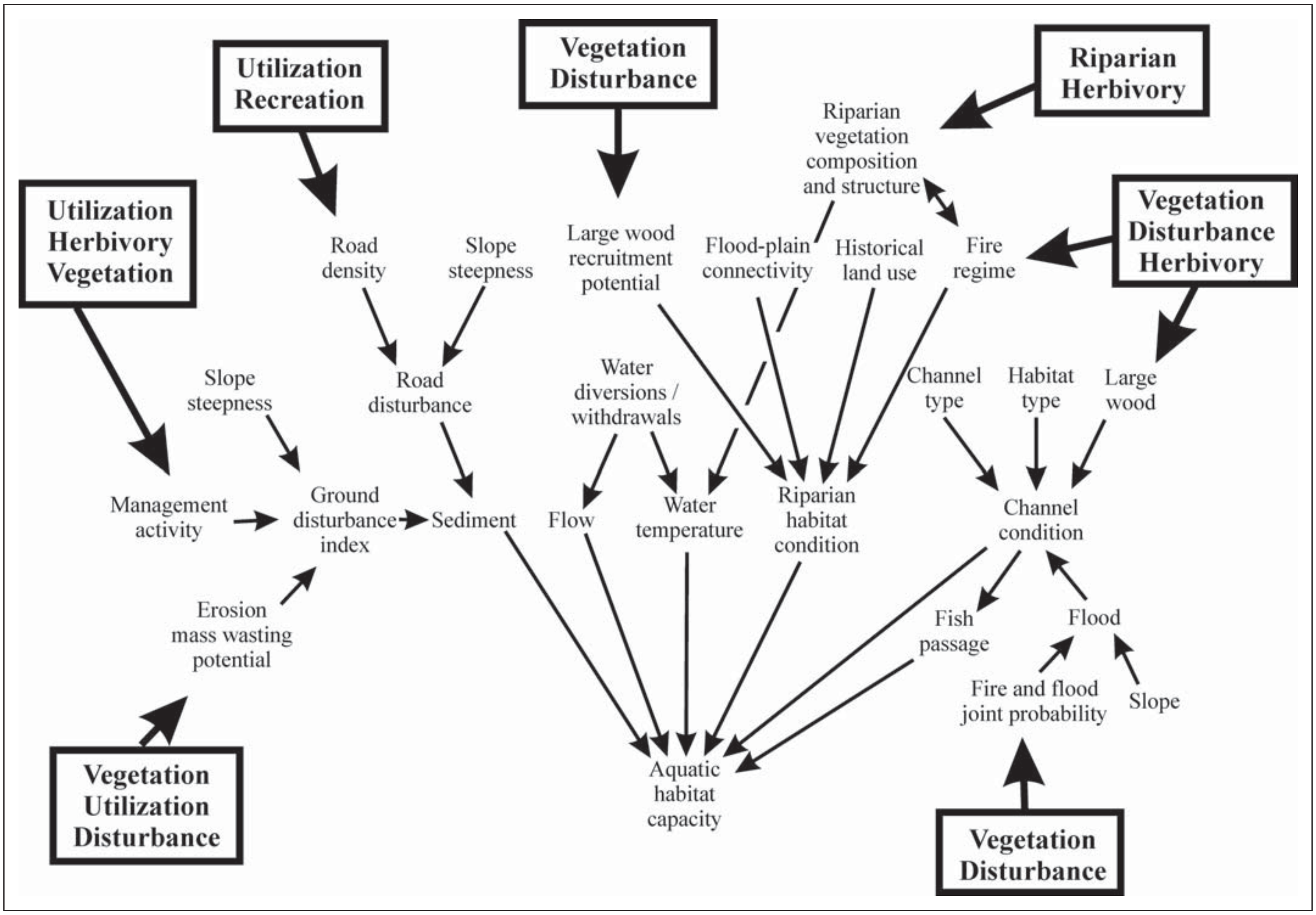

Figure 13—Example of possible linkages between physical conditions, land management practices, and aquatic habitat capacity to be used for decision analysis (The actual decision analysis framework will be developed with the use of expert panels during the project). Potential links to other Interior Northwest Landscape Analysis System modules are illustrated (bold text in boxes).

physical attributes and insensitive to management-caused changes. Many of the other INLAS modules require similar descriptive information. Spatially explicit databases will be compiled for the INLAS project and available to all INLAS modules so that effects of specific management scenarios will be based on identical watersheds.

Expected Outputs

We will develop DSMs and provide detailed documentation of those models including methods used to incorporate data into the decision models, the sensitivity analysis, and evaluation of the relative value (cost benefit) of collecting additional data to better parameterize model variables. The latter also will be used to make recommendations regarding future studies or monitoring efforts.

We also will develop a user-friendly electronic version of the DSMs for use by Forest Service biologists.

The DSMs will be applied to the Upper Grande Ronde subbasin to evaluate the influence of alternative management scenarios developed to address aquatic and other resource issues.

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Chapter 7: Modeling the Effects of Large Herbivores

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Abstract

Knowledge about the effects of ungulate herbivores on forest and range vegetation in the Blue Mountains in northeastern Oregon is reviewed, and future research needs to improve our understanding of herbivory on ecosystem processes are identified. Herbivores have had a major influence on the development of current vegetation conditions, yet their effects are largely ignored in most planning analyses, especially the wild ungulates. We discuss alternative modeling approaches to help understand herbivory as a disturbance process and identify gaps in knowledge and data that need attention before models can be fully integrated with landscape planning systems. For the Interior Northwest Landscape Analysis System we plan to develop the framework for a conceptual model of herbivory effects on succession. This model should run at multiple scales but ultimately function to deliver landscape-level products. The model ultimately will consider herbivore density and distribution as inputs.

Keywords: Herbivory, succession, disturbance, modeling, ungulates.

Introduction

Herbivory by wild and domestic ungulates has profound effects on ecosystem patterns and processes and direct economic implications for production of nearly every commodity and amenity associated with forests and rangelands in the Pacific Northwest. Many

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factors determine the level of herbivory, and in turn, the magnitude of herbivory effects on ecosystems. Impacts associated with the management of ungulate herbivory in relation to ecosystem properties potentially involve millions of dollars. Moreover, the effects of ungulate herbivory on the dynamics of plant succession have strong legal and policy implications related to federal requirements to maintain viable populations of native species. Mandates by the Endangered Species Act (1973) and National Forest Management Act (1976) make the issue of ungulate herbivory of interest to nearly every user and manager of forests and rangelands.

Enough data are available to develop a conceptual framework for linking proposed herbivory research with potential management products and address three major parts of ungulate-ecosystem relationships: (1) direct effects of ungulate herbivory on ecosystems, (2) factors affecting ungulate herbivory, and (3) integration of relevant, unpublished data and existing publications to augment parts 1 and 2. Our paper focuses on herbivory by three ungulates that dominate landscapes of the Blue Mountains and Pacific Northwest: elk (*Cervus elaphus*), mule deer (*Odocoileus hemionus*), and cattle.

Positive results of appropriate management of these three ungulate species need to be fully recognized and articulated. A wealth of existing but unpublished data needs to be integrated with existing publications and findings from future research. Specifically, models need to be built and validated that project effects of ungulate herbivory at multiple scales, particularly stand, watershed, and basin scales. Such products can be used as endpoints for management application in wildfire rehabilitation and prescribed fire and fuels-reduction programs. They also can be developed as large-scale hypotheses for further testing and validation through adaptive management. In this manner, our proposed research and potential management products have a strong foundation in research but are of direct utility to managers of ungulates, ungulate herbivory, and the forest ecosystems in which ungulates occur.

Model development will occur in a progressive manner. First, broad-scale models of ungulate resource selection that predict spatially explicit distributions of ungulates on landscapes will be constructed from information available from the Starkey Project (Johnson et al. 2000). Ongoing research and published information (Riggs et al. 2000) will be used to develop a model of herbivore forage preference and resulting forage depletion. These models form the underpinning of models that predict the effects of herbivory on flora and fauna at landscape scales such as found in range allotments. An ungulate keystone effects model will then be targeted as a primary end product. Such a model could be used to understand the effects of herbivory on other resources of interest (e.g., timber production, avian species richness, or nutrient recycling) and to assess the degree to which successional trajectories and vegetation states can be maintained or altered in desired ways.

Herbivory Effects on Forest and Range

Succession in forests has been traditionally assumed to progress predictably to climax plant associations (Clements 1936). Evidence is growing that succession can be controlled or altered dramatically by chronic herbivory (Augustine and McNaughton 1998, Hobbs 1996, Jenkins and Starkey 1996, Peek et al. 1978, Riggs et al. 2000, Schreiner et al. 1996). Variation in the herbivory regime (i.e., variation in the herbivore species, and timing, duration, or intensity of grazing) can vary the pattern and rate of successional change, and even vary the apparent endpoint (i.e., trajectory) of succession. Thus, to predict landscape vegetation dynamics with confidence, one must understand the herbivory regime and its influence on succession in the form of vegetation states, transitions, and potential thresholds. Knowledge of the role of chronic herbivory in altering succession is critical to managers dealing with the results of wildfires, prescribed fires, and fuels-reduction projects as well as understanding current steady states.

Current conditions in forests of the Western United States are associated with a high risk of catastrophic events that could dramatically change ecosystem patterns and processes (Hann et al. 1997, Hemstrom et al. 2001). Years of fire suppression and resulting forest ingrowth, combined with tree mortality caused by insect and disease outbreaks, have contributed to widespread development of forest conditions that deviate dramatically from background or historical range of variability (Quigley and Arbelbide 1997). These current conditions are associated with a high risk of lethal fire events (Hann et al. 1997). The role of herbivory in developing current conditions is not well understood but is implicated (Belsky and Blumenthal 1997).

Management actions may be taken to reduce tree density and fuels and to increase prescribed burning as means of reducing fire risk during the next several years and decades. Concomitant with such management activities, however, will be the continuing risk of conflagrations in areas yet to be treated, given the substantial portion of forest landscapes that may not receive management attention because of limitations of time, money, and practicality of application. Consequently, vast acreages have been and may continue to be altered by wildfire (Hemstrom et al. 2001). For example, 17 percent of the Wallowa-Whitman National Forest has burned in the last 10 years. These disturbances will set in motion secondary plant succession that can result in trajectories influenced by herbivory.

Hobbs (1996) made the case that native ungulates are critical agents of change in ecosystems via three processes: regulation of process rates, modification of spatial mosaics, and action as switches controlling transitions between alternative ecosystem states. Huntly (1991) identified the impact of herbivores on plant regeneration as a powerful yet little-studied mechanism of influence on vegetation composition, structure, and diversity. Wild and domestic ungulates should be considered agents of chronic disturbance, capable of influencing succession, nutrient cycles, and habitat characteristics to extents equal to episodic fire or timber harvest (Riggs et al. 2000).

An extensive review by Jones (2000) revealed that grass and shrub cover as well as total vegetation biomass are often reduced by cattle grazing. Riggs et al. (2000) reported that understory biomass at seven grand fir (*Abies grandis* (Dougl. ex D. Don) Lindl.) and Douglas-fir (*Pseudotsuga menziesii* (Mirb.) Franco.) exclosure sites averaged 2.1 times greater inside than outside, and forest-floor biomass was 1.5 times greater inside than outside the exclosure sites. Shrub biomass was influenced more by ungulates than was grass or forb biomass. Photos from exclosures illustrate the effect of large herbivores on forest understory vegetation (fig. 14). Augustine and McNaughton (1998) concluded that altered species composition of plant communities in response to selective foraging by ungulates is a general feature of plant-ungulate relations. They stated that by ungulates altering the competitive relations among plants, differential tolerance of co-occurring plant species becomes an important determinant of the responses of both woody and herbaceous plant communities to herbivory. They also summarized ungulate effects on overstory species and listed several species of coniferous and deciduous trees that were herbivory-intolerant. Ungulate herbivory is also a driving force shaping vegetation pattern in coastal coniferous forests (Schreiner et al. 1996, Woodward et al. 1994). Research by

Figure 14—Visual comparison of vegetative structure inside (top) and outside (bottom) ungulate-proof exclosures following 30 years of protection from ungulate herbivory at the Hoodoo site, Walla Walla Ranger District, Umatilla National Forest, Oregon. Photographed by Robert A. Riggs.

these authors indicated that ungulates maintain a reduced standing crop, increase forb species richness, and determine the distribution, morphology, and reproductive performance of several shrub species. Woodward et al. (1994) further stated that the extent to which herbivores can change ecosystem processes in forests likely depends on the scales of other disturbances. However, we hope to demonstrate that it is the balance between the scale of episodic disturbance and the density of ungulates that is the primary driver of change.

Of particular interest in areas like the Blue Mountains are the interactions between grazing and conifer stand density (Belsky and Blumenthal 1997, Karl and Doescher 1993, Krusi et al. 1996, Madany and West 1983, Rummell 1951, Savage and Swetnam 1990, Zimmerman and Neueschwander 1984). There is increasing evidence that under certain conditions, the net effect of long-term cattle grazing is higher conifer density (Belsky and Blumenthal 1997) and concomitant increase in the risk of large-scale crown fires and insect epidemics. The role of other ungulates (deer, elk) has not been demonstrated but can be implicated (Augustine and McNaughton 1998, Hobbs 1996, Riggs et al. 2000). Higher rates of conifer seedling survival associated with some large herbivore grazing regimes probably result from the combined impacts of selective avoidance of conifer foliage by herbivores and decreased ground-fire frequency as a consequence of reduction in understory fine fuels (i.e., grass, see Zimmerman and Neuenschwander 1984) and less seedling competition from preferred forage species. It is interesting to note that if grazing does indeed promote short-term overstocking of conifer stands that, without intervention management such as precommercial or commercial thinning, such stands are likely to be ultimately predisposed to disease and insect epidemics and crown fires. As a result, ecosystems subjected to intensely chronic herbivory may be predisposed to more marked oscillations in the amount and distribution of transitory range, although this is arguably influenced by the fuels mosaic and ignition rate as well as the herbivory regime. Along with potential for periodic instability of the plant-animal equilibrium are instabilities in forest structure.

With the potential for herbivory-induced changes in plant composition of forest understories and overstories, important habitat ramifications for a number of plant and animal species occur. Changes in understory structure and litter accumulations may be important to bird and small mammal populations (DeCalesta 1994, Fagerstone and Ramey 1996). Individual species of plants and entire plant communities may be at risk under intensive herbivory. Native steppe species in the interior Northwest are not adapted to frequent and close grazing (Mack and Thompson 1982). Examples of plant species in the Blue Mountains that are at risk of elimination or severe decline under intensive herbivory include aspen (*Populus tremuloides* Michx.), Pacific yew (*Taxus brevifolia* Nutt.), bitterbrush (*Purshia tridentata* (Pursh) DC.), and mountain mahogany (*Cercocarpus* spp. Kunth) (Parks et al. 1998). Negative effects on vertebrate species that depend on these plants (e.g., cavity nesters in aspen stands) are implied (Wisdom et al. 2000).

Herbivory Models

We identified two modeling approaches that could be useful to address research and management questions related to herbivory in the Blue Mountains province. The first uses a state and transition approach (Laycock 1991, Westoby et al. 1989) and builds on an existing model of succession and disturbance in the Blue Mountains (Hemstrom et al. Chapter 2). The second is a fine-scale individual animal foraging model that brings together previous work on foraging behavior of ungulates with data from Starkey on forage production and animal distributions (Johnson et al. 1996, 2000). These two approaches are described in more detail below.

State and Transition Models

Plant succession in forests likely operates as a set of states and transitions, much like the models developed and validated for rangeland ecosystems (Laycock 1991, Westoby et al. 1989). Indeed, it now seems possible that the veracity of many “climax” associations is questionable on this basis (Peek et al. 1978, Riggs et al. 2000, Schreiner et al. 1996).

State and transition models (Laycock 1991, Westoby et al. 1989) for specific forest plant communities can be built from the succession-disturbance regime models that were developed and applied to forest landscapes of the interior Columbia basin (Hann et al. 1997; Hemstrom et al. 2001, in press). These models were designed as state and transition models. The models projected successional change for each potential vegetation type and management prescription that was associated with each unique combination of disturbance regimes of herbivory, fire, disease, insects, and human activities. The models were built and parameterized with the use of the Vegetation Dynamics Development Tool (VDDT) (Beukema and Kurz 1995, as cited and used by Hann et al. 1997; Hemstrom et al. 2001, in press) and projected through time in a spatially explicit manner by using the Columbia River Basin Succession Model (CRBSUM) (Keane et al. 1996) and corollary rule sets (Hann et al. 1997).

An example state and transition model of ponderosa pine (*Pinus ponderosa* Dougl. ex Laws.) forests in the Blue Mountains is shown in figure 15. This model was built by using the VDDT program (Beukema and Kurz 1995). This is the type of model that we will modify to construct herbivory-disturbance-regime models that integrate the effects of herbivory on succession after episodic disturbances of fire, insects, disease, and human activities. This type of model provides the greatest utility for multiscale management inferences. This modeling approach has several advantages: (1) effects of all disturbance regimes and management prescriptions for all vegetation types can be accounted for at any spatial and temporal scale desired, provided sufficient empirical data exist for their substantiation; (2) the role of herbivory can be explicitly modeled in relation to all potential interactions with other disturbance regimes and management; (3) sensitivity and validation of herbivory effects relative to the interactive effects with other disturbance regimes can be tested; and (4) spatial and temporal scales of herbivory/disturbance effects can be modeled. Ultimately, these models could be applied at the stand, watershed, and basin scales for the entire Blue Mountains province provided that their predictions can be substantiated empirically. The models should have some general application throughout the Rocky Mountain West.

Starkey Foraging Model

A second approach to modeling the effects of herbivory involves building on earlier work at Starkey to simulate forage consumption and performance of ungulates on summer range conditions (Johnson et al. 1996). In contrast to the state and transition approach, this work is built on basic processes of herbivores moving across landscapes and foraging for preferred plants in preferred habitats. This modeling approach uses empirical models of animal distributions, forage production, and animal energetics, coupled with process-based models of foraging behavior to simulate foraging by cattle, elk (*C.elepnas*), and deer (*Odocoileus* spp.) at the landscape scale (5000 to 50 000 ha). The original formulation of the Starkey Foraging Model (SFM) was completed within a linear programming framework (Johnson et al. 1996) and later refined within a simulation framework that modeled individual animals, their movements, and foraging behavior at the bite level (fig. 16). Much of this work is based on previous models of movement and foraging behavior (Cooperrider and Bailey 1984; Kueffer 2000; Seagle and McNaughton 1992; Spalinger and Hobbs 1992; Van Dyne et al. 1984a, 1984b). The model considers foraging site selection, forage consumption, energy balance, and forage regrowth on a daily time

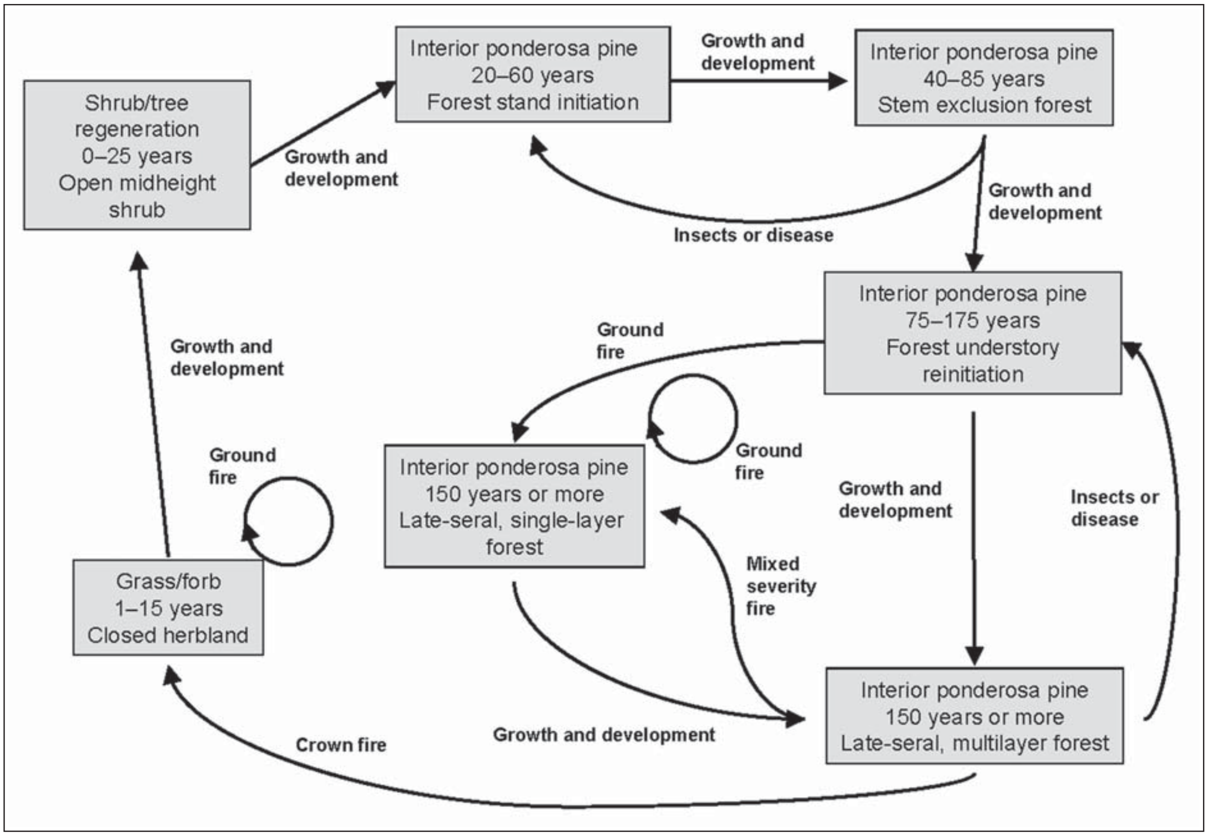

Figure 15—A hypothetical state and transition model for the interior ponderosa pine forest (from Hemstrom et al. Chapter 2).

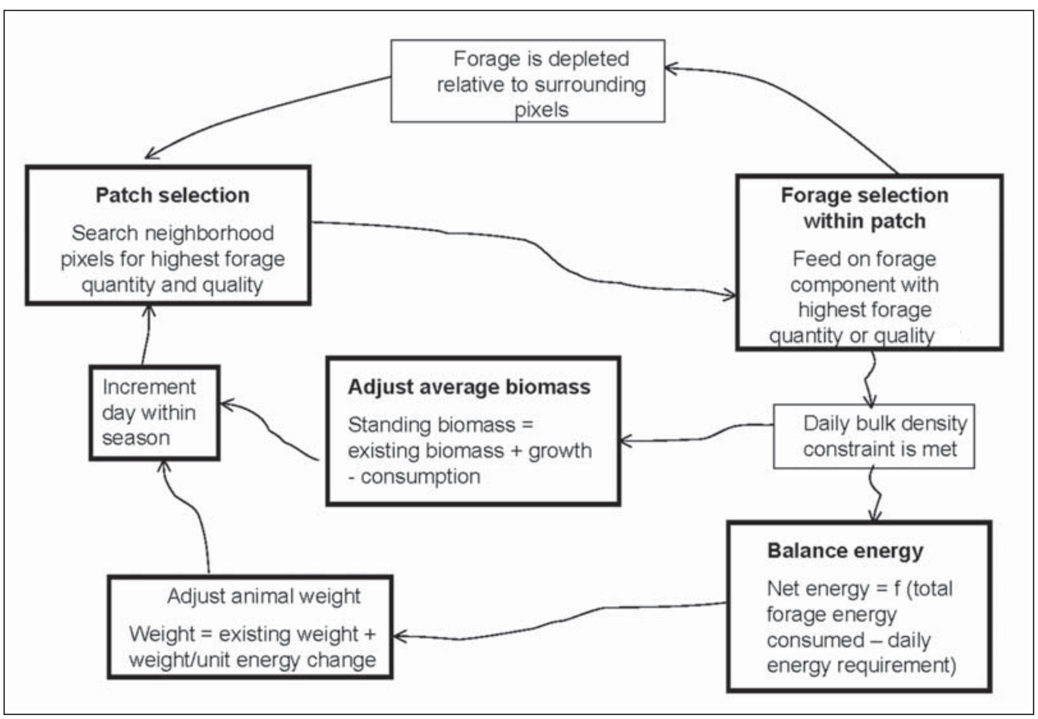

Figure 16—Simulation sequence for selecting forage in the Starkey Foraging Model and the resulting energy balance and resulting gain or loss of body mass for grazing ungulate on a daily time step.

step by using an array of empirical and conceptual information. Habitat preferences for elk, deer, and cattle were modeled by using resource selection functions developed at the Starkey Project (Coe et al. 2001, Johnson et al. 2000). Resource selection functions were developed on a monthly time step for elk, deer, and cattle (fig. 17) and represent the probability of an animal visiting a particular pixel.

Forage production was estimated by using several empirical models built from Starkey data (1993-2000) and the literature. Data from Starkey were used to build functions that predicted herbage production as a function of Julian day (figs. 18 to 20). These data came from clipped plots at Starkey and were constructed for grasslands, ponderosa pine, and riparian ecotypes as represented in the area sampled. The equations for grasslands, ponderosa pine, and riparian ecotypes were extrapolated to the seven plant association groups in the model: moist meadows (MM), dry meadows (MD), bunchgrass and shrub lands (GB), warm dry forests with grass (WDG), warm dry forests with shrub understory (WDS), cool moist forest with grass understory (CMG), cool moist forest with shrub understory (CMS), and subsequently partitioned into forbs, grass, and shrubs by using scaling factors developed from Hall (1973) and Johnson and Hall (1990). The growth functions also were adjusted for canopy closure on a pixel basis by using relationships developed at four grazing exclosures at Starkey and the data of Pyke and Zamora (1982) (fig. 21). Forage quality, as measured by in vitro digestible energy (IVDE) of forage, was obtained from the literature (Holechek et al. 1981, Sheehy 1987, Svejcar and Vavra 1985, Westenskow 1991) and data at Starkey (fig. 22). Digestible energy was calculated from IVDE by using the methods of McInnis et al. (1990).

The dynamics of animal foraging are modeled as a two-step process that involves the selection of feeding patches and subsequent selection of forage within the feeding patch. The form of this two-stage model was motivated by literature and concepts in optimal foraging theory and ecology of ungulates (Gross et al. 1993, 1995; Shipley and Spalinger 1995; Spalinger and Hobbs 1992). Feeding patches were defined as 30- by 30-m pixels, a size chosen to be compatible with geographical information system (GIS) data on vegetation strata. Movement to foraging patches was modeled by using a neighborhood search algorithm that searched a 300-m radius for pixels that maximized the expression:

$$\frac{(RSF_{spm})^{a_s} \text{ x } (DE_{pm})^{\,b_s} \text{ x } (F_{pm})^{c_s}}{(Distance_{i,j})^{1/d_s}} \quad ,$$

where

$(RSF_{spm})^{\mathbf{a}}$ = resource selection function score ($0 \geq \text{RSF}^{\mathbf{a}} \leq 1$) for pixel *p*, species *s*, month *m;*

DE_{pm} = digestible energy in mcal/kg forage for pixel *p* and month *m;*

$(F_{pm})^{c}$ = forage (kg/ha) present on pixel *p* and month *m;* and

$(Distance_{i,j})^{\mathbf{1/d}}$ = distance (m) required to move from the current pixel (*i*) to the pixel (*j*) being evaluated.

The *a*, *b*, and *c* are species-specific, real valued weighting coefficients that are used to control the relative importance of habitat, forage quality, and forage energy content and movement distances in the foraging process. All these factors influence the selection of feeding sites by elk, deer, and cattle (Johnson et al. 1996). Initial simulations with the

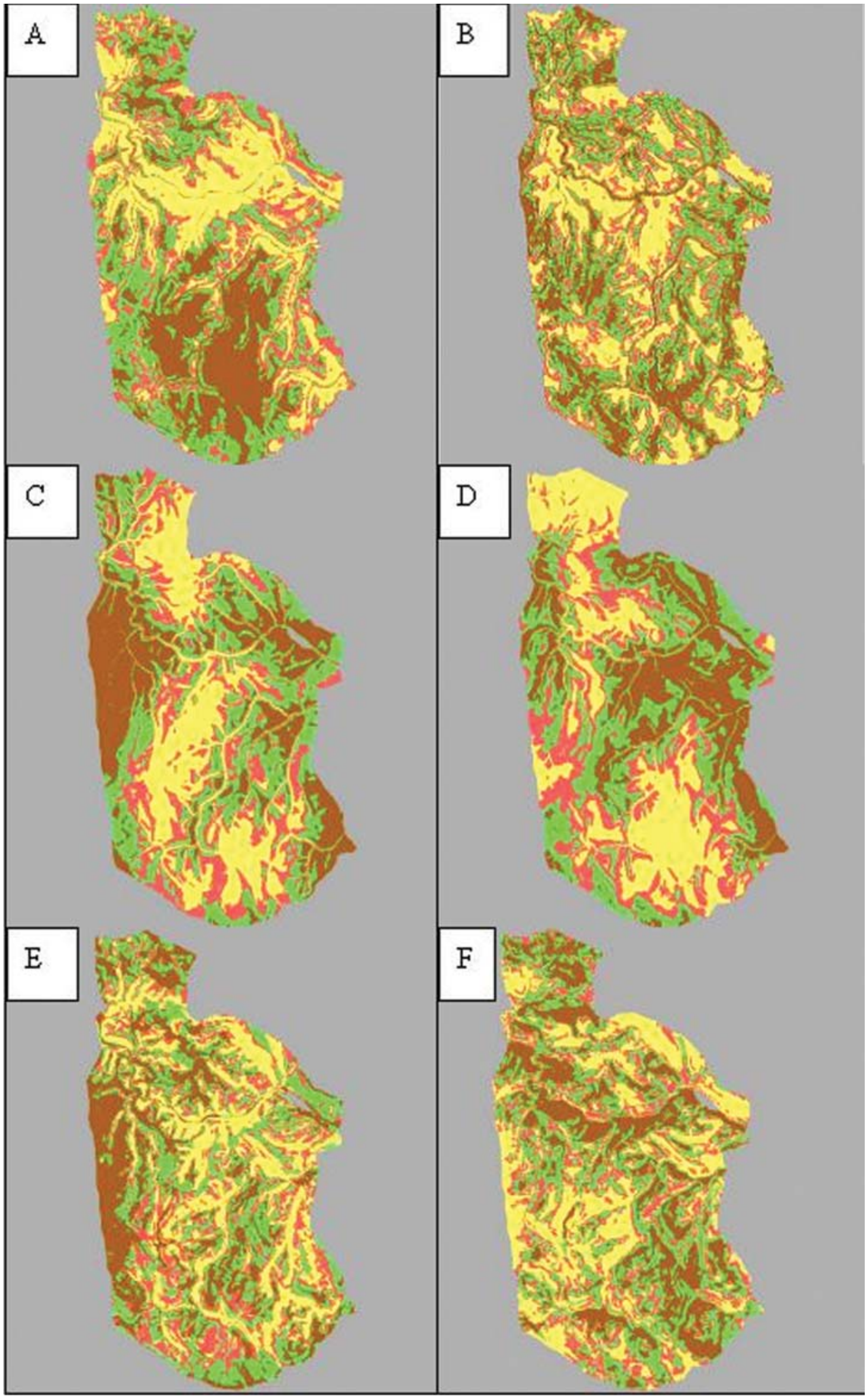

Figure 17—Predicted distributions of cattle in June (A) and August (B), mule deer in May (C) and August (D), and elk in May (E) and August (F) from resource selection functions. Colors depict probability of use from high (brown), moderately high (green), moderately low (red), and low (yellow).

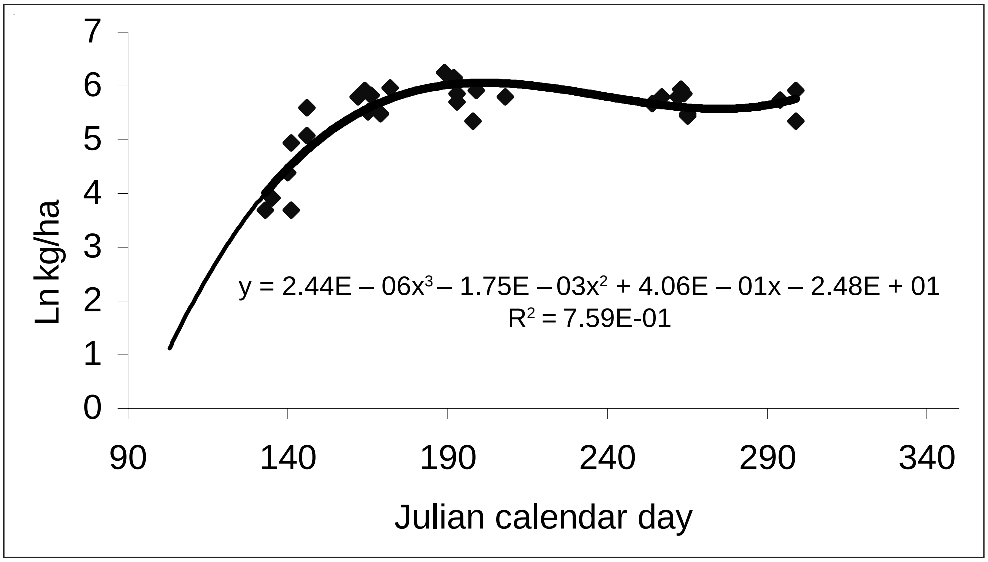

Figure 18—Herbage production in ponderosa pine habitat collected from clipped plots at the Starkey Experimental Forest and Range during 1993–99.

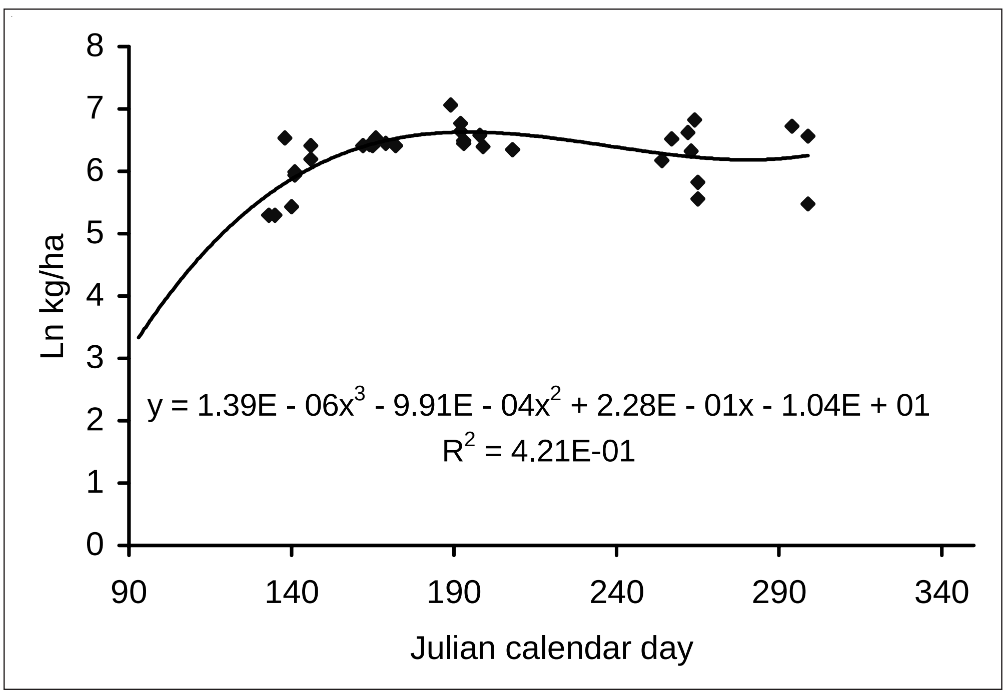

Figure 19—Herbage production in grasslands at the Starkey Experimental Forest and Range estimated from data collected from clipped plots during 1993–99.

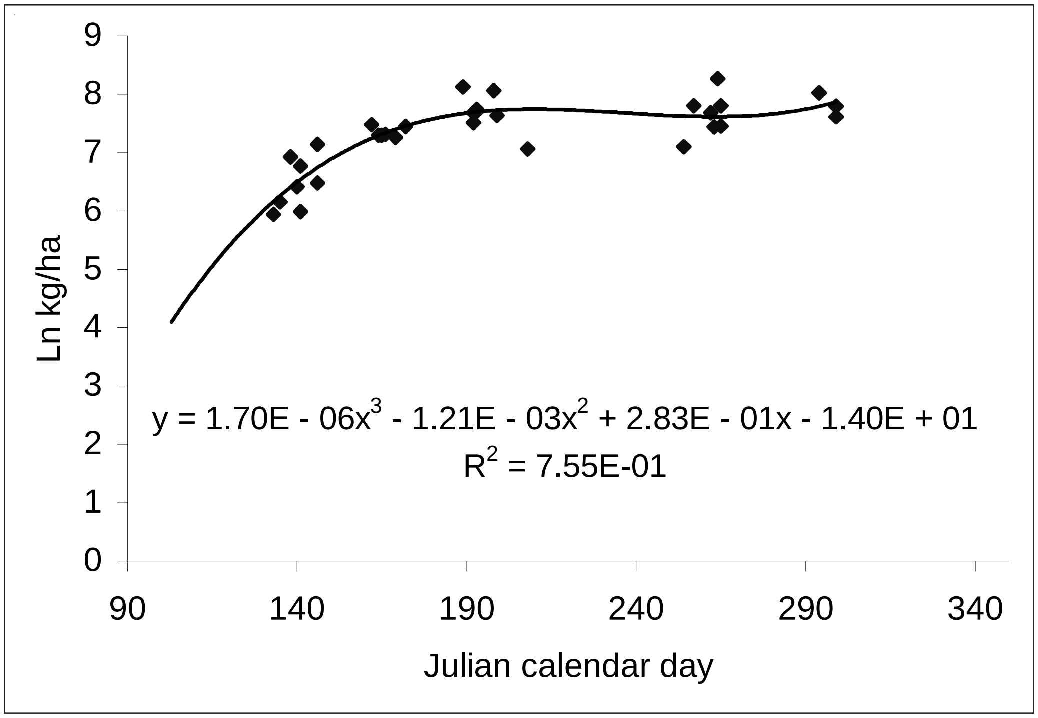

Figure 20—Estimated forage production in riparian areas collected from clipped plots determined for Starkey Experimental Forest and Range during 1993–99.

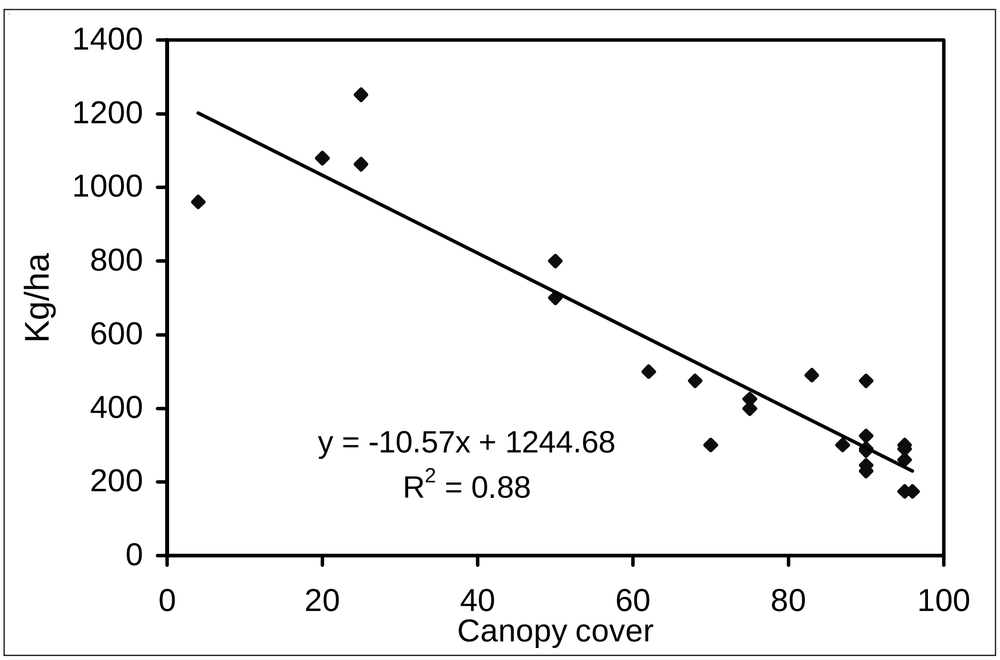

Figure 21—Predicted herbage production in grand fir habitat as a function of canopy cover. Data for canopy cover >50 percent were adapted from Pyke and Zamora (1982), and the four data points in the upper left of the graph are unpublished data from Starkey Experimental Forest and Range.

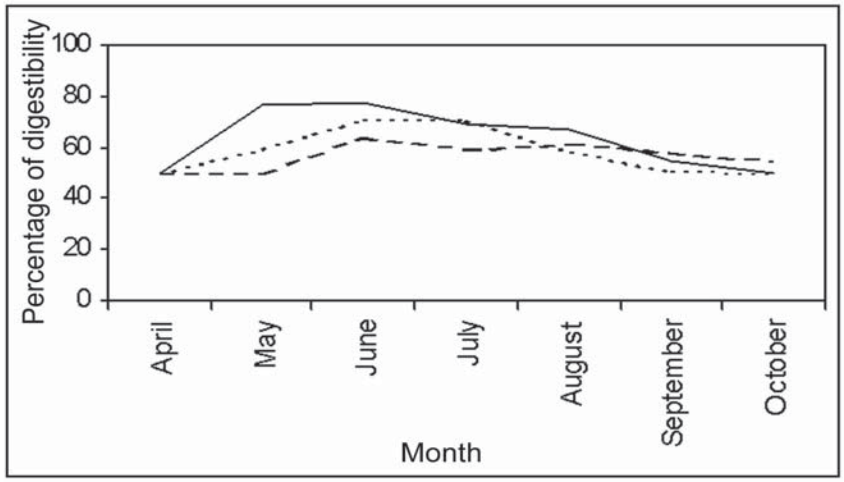

Figure 22—Hypothetical example of in vitro digestibility of grasses (dashed line), forbs (solid line), and shrubs (dotted line) in cool moist forests at Starkey Experimental Forest and Range. Forage digestibility estimates were from unpublished data collected at Starkey Experimental Forest and Range (Holechek et al. 1981, Krueger and Bedunah 1988, Skovlin 1967, Svejcar and Vavra 1985, Urness 1984, and Walton (1962).

SFM used coefficients of 1.0 for all species. Once a foraging pixel was selected, consumption of forage (grass, forbs, and shrubs) was modeled as a Monte Carlo process that simulated successive bites that removed forage types in proportion to the product of total forage available times forage digestible energy. Specifically,

$$P_{ts} = \frac{(F_{pm})^{w_s}\ (DE_{ptm}),}{\sum_{1}^{t}[(F_{\mathrm{pm}})^{w_s}\ (DE_{ptm})]} \qquad (2)$$

where

P_{ts} = probability of removing forage type t for species s $(0 \geq P_{ts} \leq 1)$;

F = forage on pixel p at month m;

DE_{ptm} = digestible energy in mcal/kg for forage type t, pixel p, and month m; and

W_s = species-specific weighting factor.

The initial simulations used W_s = (body weight)$^{0.75}$. This resulted in deer emphasizing forage quality versus cattle emphasizing forage bulk. Elk, with their intermediate body weight, were simulated as having a foraging behavior in between that of deer and cattle. Bite size was held constant for each species (elk = 0.22 g, deer = 0.06 g, cattle = 0.53 g), although in future work, the type II functional response between bite size and plant size could be coupled with a maximum number of bites constraint to reduce foraging efficiency for smaller plant size (Gross et al. 1993). Foraging on a particular pixel ceased when either 90 percent of the forage was consumed or the total energy was 20-percent below average for all pixels in the 300-m-radius neighborhood. Using the foraging rules described above, animals foraged until they reached 3 percent of body weight per day. When average forage quality for a particular animal and day was below 55 percent, the bulk forage constraint was reduced to 2.5 percent. Animal energy balances were calculated daily and used monthly energy requirements prorated to a daily basis (table 5). Daily energy balance was calculated by using the energy conversion equation as:

$$Me = 1000^{x} \text{ kg forage x } (0.038 \text{ x } \%DE + 0.18)/1.22, \qquad (3)$$

where

DE = digestible energy (mcal/kg forage).

Negative energy balances were translated into a weight loss by using a conversion of 6000 mcal/kg. Positive daily energy balances were translated into a weight gain by using the conversion 12 000 mcal/kg.

Test simulations with this model (figs. 23 and 24) for a summer grazing system (April 15 to November 15, 210 days) were performed by using a herd of 500 cows, 450 elk, and 250 deer. Initial weights were set at 450, 230, and 60 kg per animal for cows, elk, and deer. The initial simulations showed good correspondence with known levels of forage consumption and animal weight gains (losses) on the Starkey area (7800 ha). The simulations also show the effect of lower forage production on foraging patterns by elk and deer (figs. 23 and 24). Although much work remains on this modeling approach, the initial simulations indicated that it is feasible to build a fine-scale foraging model for cattle, elk, and deer that can simulate consumption of individual plants on large landscapes through time.

Research Needs

Managers need information on herbivory to understand its impacts on succession, forest productivity, and biodiversity. Research should focus on, among other things, providing tools to better understand the role of herbivory in shaping plant communities in interior Northwest forests. Primary questions of interest are:

1. What are the patterns of resource selection by deer, elk, and cattle that influence composition and structure of plant communities at multiple scales?
2. What changes in composition and structure of plant communities occur as a result of herbivory at local and regional scales?
3. How does the herbivory regime interact with frequency, intensity, and distribution of episodic disturbances to influence development of plant communities at local and regional scales?

Questions 1 and 2 can be addressed through a synthesis of existing research data and findings from the Starkey Project on resource selection functions for ungulates in the Blue Mountains (Johnson et al. 2000, Rowland et al. 2000, Wisdom 1998). These data can provide estimates of plant composition with and without herbivory and the likelihood of herbivory effects occurring in various forest plant communities. We will develop products for questions 1 and 2. Question 3 requires the development of a multiscale model of ungulate herbivory, based on data synthesized for questions 1 and 2 and by using the modeling frameworks discussed above. This question will be addressed through the development of a conceptual herbivory model.

Table 5—Daily energy demands of adult female deer, cow, and elk by month

Species	April	May	June	July	August	September	October
				mcal per day			
Cattle	23	23	23	22	21	19	18
Elk	6.7	7.3	10.2	10.3	9.3	7.5	7.0
Deer	3.2	3.3	4.8	4.9	4.3	3.6	3.1

Sources: Hudson and White 1985a, 1985b; Nelson and Leege 1982; Sheehy 1987; Wallmo et al. 1977.

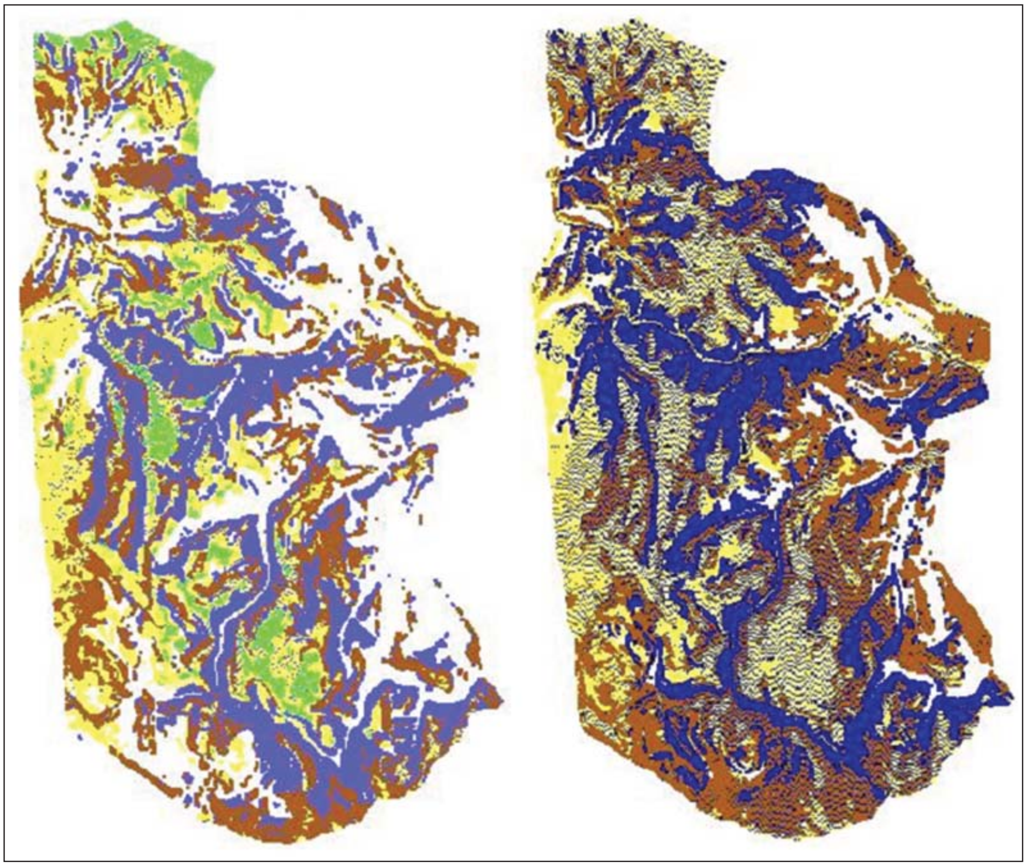

Figure 23—Results of simulating foraging by 450 elk at Starkey for April 15–July 15 by using the Starkey Foraging Model. Images show the areas foraged over time. Colors depict the sequence in which forage was removed (green 0 to 20 days, yellow 21 to 40 days, brown 41 to 60 days, and blue >60 days) for normal forage production (left) and drought conditions (10 percent of normal forage production, right panel).

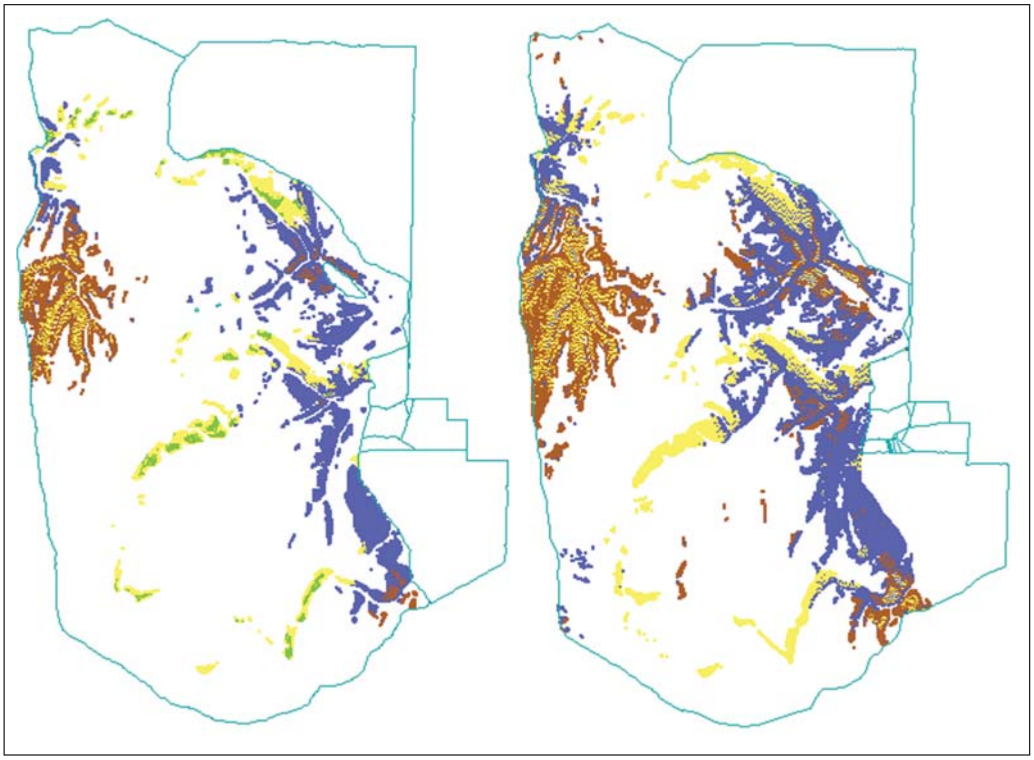

Figure 24—Results of simulating foraging by 250 mule deer at Starkey for April 15–July 15 using the Starkey Foraging Model. Images show the areas foraged over time. Colors depict the sequence in which forage was removed (green 0 to 20 days, yellow 21 to 40 days, brown 41 to 60 days, and blue >60 days) for normal forage production (left) and drought conditions (10 percent of normal forage production, right panel).

For the state and transition approach, appending new states and transitions specific to herbivory would provide a prototype framework and identify the major gaps in terms of unknown transitions and states. The transitions will be modeled within the context of a disturbance, where assumptions about the frequency and magnitude can be changed to simulate specific management scenarios analogous to prescribed fire or wildfire. In contrast to other types of disturbances, the herbivory transitions will not be periodic or involve epidemics and will be associated with relatively low transition probabilities. There will exist states that can only be achieved after long periods of chronic herbivory.

Further work on the SFM needs to focus on refining the coarse stratification of vegetation types (forbs, grass, and shrubs) for both production and consumption by herbivores. Information to fill this gap can come from literature and ongoing studies at Starkey and industrial forest land. For instance, diet selection data are available from the study of Riggs et al. 2000, although these data are for a limited set of plant associations in the Blue Mountains. These data are in the form of species depletion curves for individual taxa (fig. 25). Incorporation into the SFM would require extending the array of forage types for individual species. However, although this might accomplish species-specific consideration of plant biomass for a given season, the long-term multiseasonal effects on particular species would require additional modeling of the plant response to grazing. If virtually all of a species is consumed over successive years, we need to know how long it will take before the species is extirpated from a particular foraging patch, and how the extirpation progress relates to the abundance of a species in neighboring patches. The life

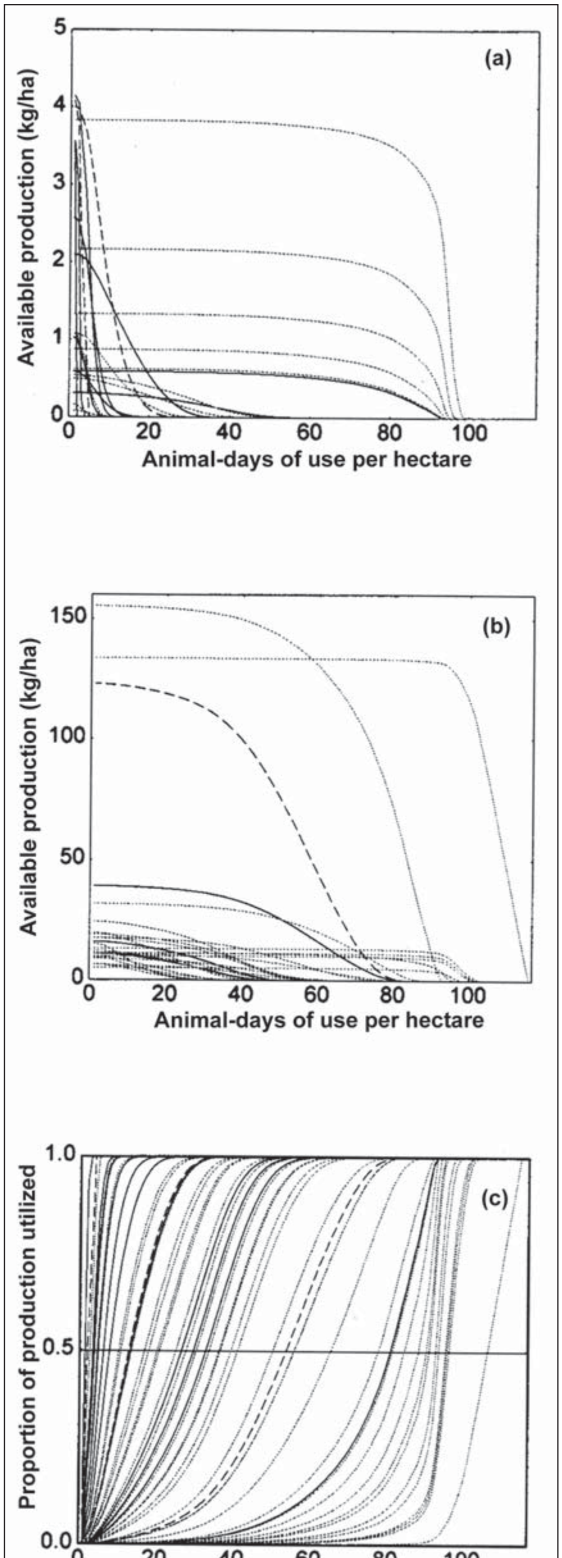

Figure 25—Model profiles for taxon-specific depletion of (a) rare and (b) common plant taxa, and (c) corresponding whole-community utilization, within 2 years of clearcut timber harvest of an *Abies grandis* forest, Mottet study site, Umatilla National Forest. Depletion and utilization of shrubs (solid lines), graminoids (dashed lines), and forbs (stippled lines) were modeled on empirical estimates of elk forage preferences, animal-days of elk grazing, and total production partitioned among plant taxa in the postdisturbance community. Rare, highly preferred plant taxa are depleted at relatively high rates. Preference-abundance relationships among species, particularly during the first few years following episodic disturbance, may determine to a large extent the potential for alternate successional pathways in disturbance-adapted mixed-conifer forests (Riggs et al. 2000). Such definitive modeling at the community level requires reliable estimation of grazing preferences and reliable knowledge of animal density and distribution. Current databases are inadequate and further field work is required. Graphs excerpted from Riggs et al. (2000).

history of each shrub needs to be considered to model response to grazing. Qualitative information on species response to grazing is included in plant association guides for the Blue Mountains in terms of whether species increase or decrease when subjected to grazing pressure. Forage production data also need refinements in terms of modeling the growth of the major plant species. These data also are limited but available in plant association guides (e.g., Hall 1973). In addition, studies are underway at Boise Cascade to build empirical models of nonconifer plant production and composition for a subset of the plant communities in the Blue Mountains.

Linking a spatial foraging model like that developed at Starkey with tree growth simulators (e.g., the Forest Vegetation Simulator [FVS]) is a complex problem. The simulation of herbivory as a spatial disturbance within a stand-level simulation model will require the kind of formulation described by Bettinger et al. (Chapter 4) to simulate wildfire within a vegetation growth model like FVS. Integration of herbivory models into vegetation growth models presents a significant challenge for future work.

Products and Audience

Forest, rangeland, and wildlife managers in the Blue Mountains province are the targeted users of the research findings and management tools produced from the activities outlined in this paper. Clients include managers of public, private, and tribal lands in the Blue Mountains province, encompassing economic and social interests related to management of timber, livestock, wild ungulates, salmon, and vertebrates and plants of conservation concern. Technical users of the research findings and products outlined here include spatial analysts, planners, and resource specialists of public, private, and tribal lands in the Blue Mountains. Application of the concepts and relations developed as part of this research and associated management tools also will extend beyond the Blue Mountains to similar environments in other provinces of the Pacific Northwest and intermountain West. These extensions will target the above-named clients in these similar environments.

English Equivalents

When you know:	Multiply by:	To get:
Hectares (ha)	2.47	Acres
Meters (m)	3.28	Feet
Grams (g)	.0352	Ounces
Grams (g)	.0022	Pounds
Kilogram (kg)	2.205	Pounds

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