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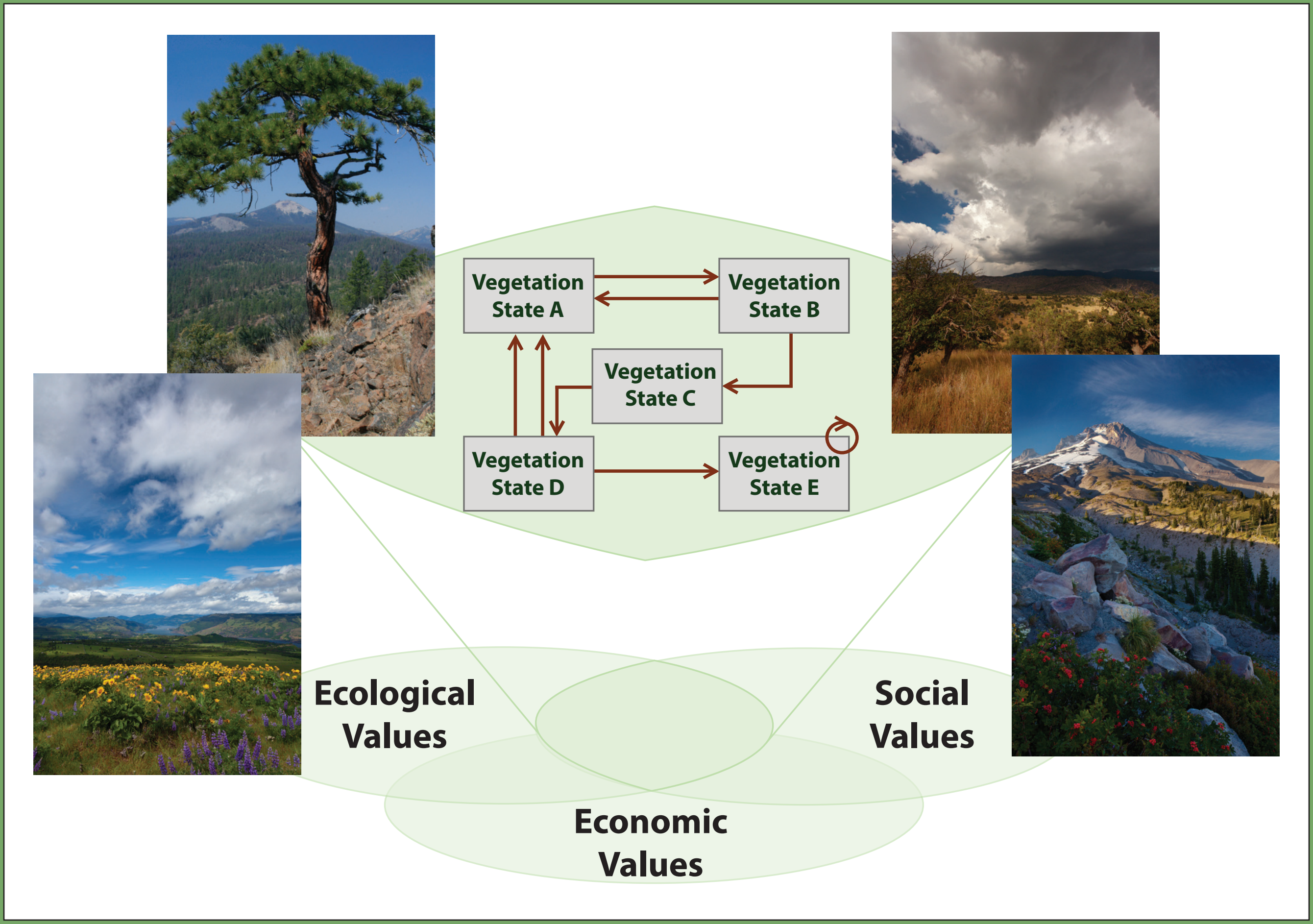
Integrating Social, Economic, and Ecological Values Across Large Landscapes



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Editors

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Abstract

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The Integrated Landscape Assessment Project (ILAP) was a multiyear effort to produce information, maps, and models to help land managers, policymakers, and others conduct mid- to broad-scale (e.g., watersheds to states and larger areas) prioritization of land management actions, perform landscape assessments, and estimate cumulative effects of management actions for planning and other purposes. The ILAP provided complete cross-ownership geospatial data and maps on current vegetation, potential vegetation, land ownership and management allocation classes, and other landscape attributes across Arizona, New Mexico, Oregon, and Washington. State-and-transition models, developed to cover all major upland vegetation types in the four states, integrated vegetation development, management actions, and natural disturbances to allow users to examine the mid- and long-term effects of alternative management and disturbance scenarios. New model linkages to wildlife habitat, economics, aboveground carbon pools, biomass, and wildfire hazard were developed and integrated through decision-support systems. Models incorporating potential effects of climate change were also developed for focus areas in Oregon and Arizona. This report includes an overview of the structure and components of ILAP along with descriptions of methods and example results for state-and-transition modeling, fuel characterization, treatment economics, wildlife habitat, community economics, and climate change. This report serves as a guide to ILAP. Complete collections of the project's models, maps, data, and tools will be archived and available online through the Western Landscapes Explorer portal (www.westernlandscapesexplorer.info) so that scientists and managers will be able to use and build upon ILAP's products.

Keywords: Landscapes, state-and-transition modeling, vegetation mapping, fuels, wildfire hazard, treatment costs, biomass, wildlife habitat, decision support, climate change, watersheds, community economics, all ownerships, geographic information systems, landscape assessment.

Contents

Chapter 1: Overview of the Integrated Landscape Assessment Project

Miles A. Hemstrom, Jessica E. Halofsky, F. Jack Triepke, R. James Barbour, and Janine Salwasser[1]

Introduction

Fire suppression, vegetation management activities, wildfires, grazing, climate change, and other factors result in constantly changing vegetation and habitat conditions across millions of hectares in the Western United States. In recent years, the size and number of large wildfires has grown, threatening lives, property, and ecosystem integrity. At the same time, habitat for species of concern is often becoming less suitable, the economic vitality of many natural resource-dependent human communities is declining, and resources available for land management are limited. Techniques are needed to prioritize where natural resource management activities are likely to be most effective and result in desirable conditions. Solutions driven by single-resource concerns have proven problematic in most cases, as multiple ecological resources and human systems are necessarily intertwined.

To help resource managers prioritize management actions across large landscapes, the Integrated Landscape Assessment Project (ILAP) produced databases, reports, maps, analyses, and other information showing mid- to broad-scale (thousands to hundreds of thousands of hectares and larger areas) vegetation conditions and potential future trends, key wildlife habitat conditions and trends, wildfire hazard, potential economic value of products that might be generated during vegetation management, and other critical information for all lands and all major upland vegetation types in Arizona, New Mexico, Oregon, and Washington. The ILAP work involved gathering and consolidating existing information, developing new information to fill data holes, and merging vegetation model information with fuel classifications, wildlife habitat models, community and economic information, and potential climate change effects. Information resulting from ILAP highlighted priority areas for management, considering a combination of landscape characteristics.

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ILAP was designed to allow resource managers, planners, analysts, and other potential users to answer many questions about the integrated effects of vegetation change, management activities, natural disturbances, and climate change on important natural resources across all major upland ecological systems in the four-state study area.

The ILAP was designed to allow resource managers, planners, analysts, and other potential users to answer many questions about the integrated effects of vegetation change, management activities, natural disturbances, and climate change on important natural resources across all major upland ecological systems in the four-state study area. Questions addressed by ILAP included, but were not limited to, the following:

1. What are the conditions and trends of vegetation and natural disturbances in forests, woodlands, shrublands, grasslands, deserts, and other ecological systems in Arizona, New Mexico, Oregon, and Washington?
2. What are the implications of vegetation change, management activities, and natural disturbance trends on key wildlife habitats, wildland fuel conditions, nonnative invasive plant species, and other landscape characteristics?
3. How might those trends play out in the future under alternative land management approaches or scenarios?
4. How will alternative vegetation management scenarios meet land management objectives and generate economic products that might offset treatment costs and benefit local communities?
5. What areas and management regimes might be most likely to produce high combined potential to reduce critical fuels, sustain or improve key wildlife habitat, and generate positive economic value?

To ensure that relevant and useful information was produced, ILAP worked with local collaborative groups in focus areas (fig. 1.1) to forecast the potential effects of alternative land management scenarios on important landscape characteristics. Questions addressed in these landscapes were developed in collaboration with local users, in particular the Tapash Sustainable Forestry Collaborative in central Washington and the Firescape-Sky Islands group in southern Arizona. Alternative landscape management scenarios were simulated for each area. Examples of the questions addressed in focus areas include:

1. Central Washington landscape area—How might the Tapash Sustainable Forestry Collaborative partners simultaneously achieve individual landscape objectives while sustaining or improving critical wildlife habitat, reducing wildfire hazards, and generating economic benefits for local communities?
2. Sky Islands landscape area—How could fuel treatments be used to move vegetation toward desired or reference conditions in the Sky Islands landscape and how much would that cost? What effects might climate change have on the effectiveness of fuel treatment programs and associated wildfire hazards?

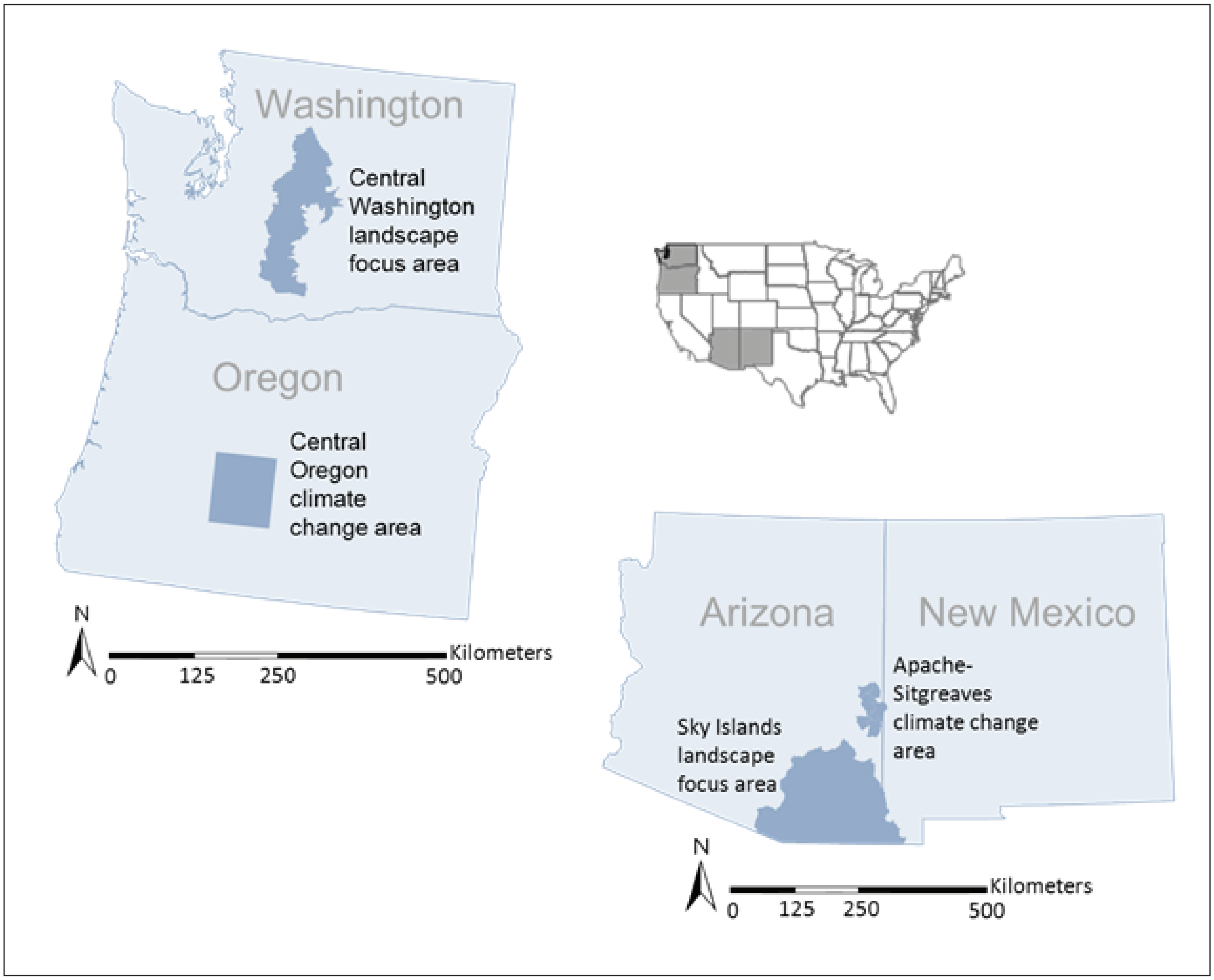

Figure 1.1—The focus areas and climate change areas within the Integrated Landscape Assessment Project.

In addition to working with collaborative groups at the local level, ILAP also worked to address issues that were important over larger regional landscapes. The Interagency Mapping and Assessment Project (IMAP) User Group served as the advisory group to ILAP. This group included regional and state land managers and planners from Oregon and Washington. They identified the east-side dry forests in Oregon and Washington as highest priority for ILAP to address, with these specific management questions:

1. What are likely trends in dry forest conditions in eastern Oregon and Washington under a fire suppression only and hypothetical resilience/forest health scenario?
2. Where are dry forests most at risk for loss of large, old trees?

3. What are the land ownership and management allocation circumstances (ownership-management hereafter) in these areas?
4. Which landowners might need to collaborate to achieve mutual forest management objectives?

Each of these landscape-level questions was addressed by ILAP through vegetation mapping and modeling and integration with newly developed information, as well as subsequent discussions with each of the collaborative groups in each focus area. The structure of ILAP is described below and represented in figure 1.2, and methods used for ILAP are outlined below and described in detail throughout this report.

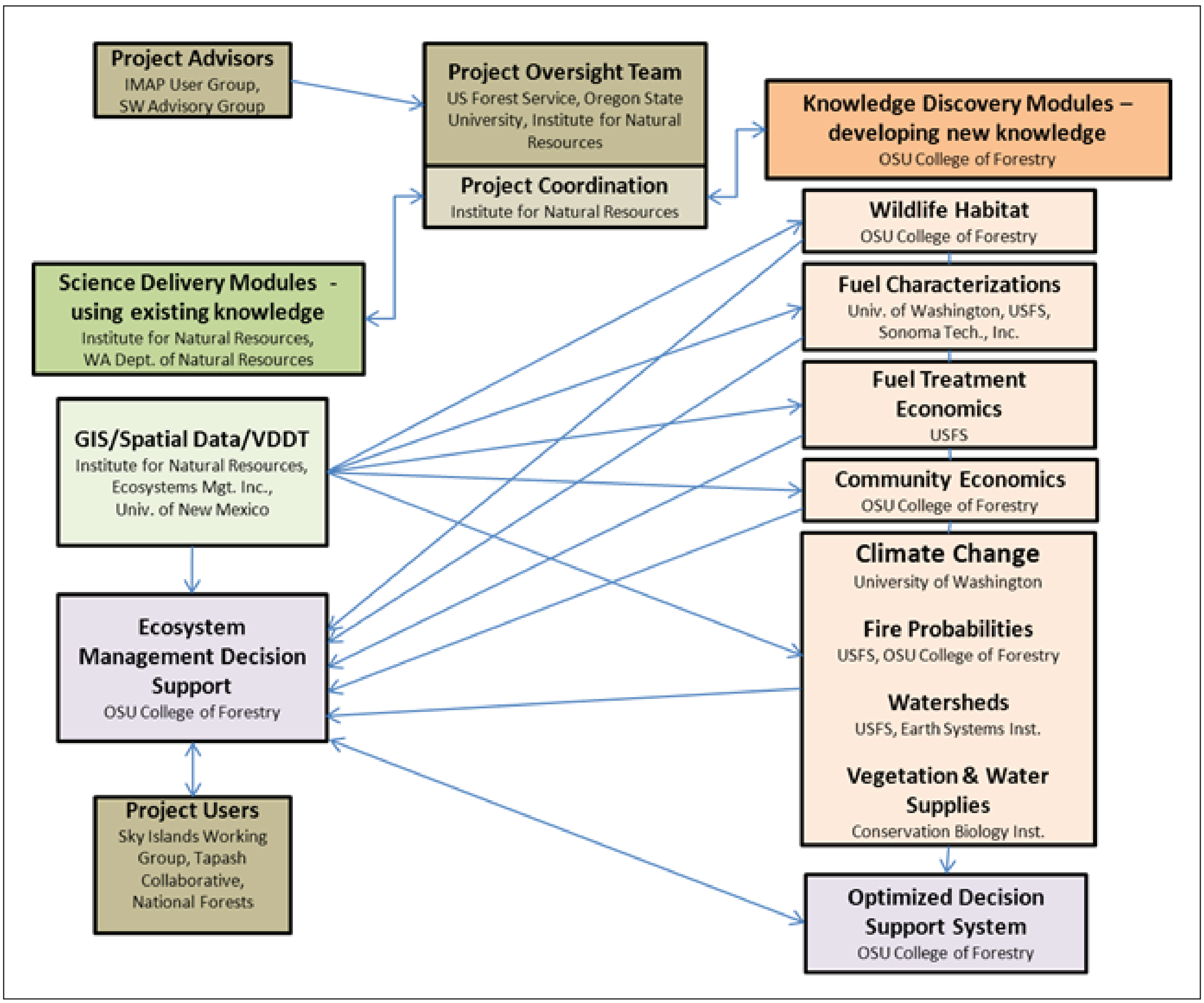

Figure 1.2—Organization of science delivery and knowledge discovery partners in the Integrated Landscape Assessment Project. IMAP = Interagency Mapping Assessment Project, OSU = Oregon State University, WA = Washington, USFS = U.S. Forest Service, GIS = geographic information system, VDDT = Vegetation Dynamics Development Tool.

Structure of ILAP

Partners and Oversight

The ILAP was a collaborative effort and incorporated expertise from several institutions and disciplines (fig. 1.2). An oversight team, composed of representatives from the major collaborators (the Institute for Natural Resources, Oregon State University College of Forestry, the U.S. Department of Agriculture Forest Service [USDA FS], Pacific Northwest Research Station, and the USDA FS Southwest Region) provided overall direction at monthly meetings. Other clients and partners include the USDA FS Pacific Northwest Region, Washington Department of Natural Resources, Oregon Department of Forestry, University of Washington, University of New Mexico, University of Arizona, The Nature Conservancy, and others. Two groups of project advisors, one from Oregon and Washington and one from Arizona and New Mexico, connected the project goals, objectives, and products to state and federal agencies, nonprofit organizations, private contractors, universities, and the interested public by providing comment, feedback, and review throughout the project. The project lead scientist and project coordinator oversaw the technical and outreach aspects of project work. Science delivery, as a whole, was jointly led by scientists from the Institute for Natural Resources and the Washington Department of Natural Resources. Each science delivery module (described below) had a lead investigator and production team, as necessary. Knowledge discovery modules (described below) were led by several universities and nonprofit organizations, and each module had a lead scientist, and as appropriate, a production team. User involvement was critical throughout the project, particularly in the development of management scenarios and review of draft products.

Science Delivery and Knowledge Discovery

The ILAP was organized into two broad themes that roughly followed each other sequentially—science delivery and knowledge discovery. Science delivery teams generally worked with existing methodologies to develop landscape-level information (primarily related to vegetation conditions), while the knowledge discovery teams developed and applied new methodologies to link the science delivery vegetation outputs with associated landscape-level information on wildlife habitat, fuel conditions, treatment economics, community impacts, and climate change impacts.

Science delivery modules—

The science delivery portion of ILAP consisted of two main modular components: geographic information system (GIS) data and state-and-transition models (STMs). These two modules worked together to create the following across the four-state area:

- Maps of potential vegetation that depicted the kinds and spatial extents of all major upland ecosystem types in the four-state area (riparian, aquatic, agricultural, urban, and barren areas were not modeled for this project).
- Maps of current vegetation characterizing current vegetation cover and structure conditions. These maps were summarized into state classes that matched STMs.
- Parameterized STMs used for simulation forecasting of future vegetation condition.

The potential and current vegetation mapping work, STM development, vegetation classification, and simulation forecasting are described briefly below and in detail in chapter 2.

The GIS data sets compiled by ILAP include more than 40 statewide spatial data sets detailing current and potential vegetation conditions, watershed boundaries, ownership-management categories, and others.

Geographic information system module—The GIS module processed the best available spatial data for use by the other ILAP modules. The GIS data sets compiled by ILAP include more than 40 statewide spatial data sets detailing current and potential vegetation conditions, watershed boundaries, ownership-management categories, and others. The team gathered data from various sources, merged and appended it, combined attribute data into consistent formats, and created detailed documentation. The GIS module delivered standardized data in raster/grid and vector formats from various hardware and software platforms. A data management process and protocol were put in place to facilitate the incorporation of any data updates or improvements and maintain and house original data sets over the long term.

Where existing maps were not available or contained insufficient detail, current and potential vegetation were mapped using imputation methods and geo-referenced plot data from various sources. Plot data for mapping came from the USDA FS Forest Inventory and Analysis (FIA) program (USDA FS 2012), the LANDFIRE plot database (www.landfire.gov; Rollins 2009), U.S. Department of the Interior Bureau of Land Management data, and various other sources. Current and potential vegetation were mapped using a combination of gradient nearest neighbor imputation (Ohmann and Gregory 2002) and random forest nearest neighbor imputation (Crookston and Finley 2008), which rely on a combination of remotely sensed information and other geographic data. The resulting spatial data are 30-m grids that contain information on key attributes of current vegetation and an assignment of potential vegetation across the four-state region.

The STM module collected, integrated, and as necessary, built new vegetation models for forest, woodland, shrubland, grassland, and desert vegetation types across the four-state study area.

State-and-transition modeling module—The STM module collected, integrated, and as necessary, built new vegetation models for forest, woodland, shrubland, grassland, and desert vegetation types across the four-state study area. Using a summarization of layers from the GIS data to establish initial (current) conditions, STMs were used to project future landscape conditions according to alternative vegetation management scenarios.

The STM approach represents vegetation types by state classes (boxes), each characterizing combinations of cover type (i.e., dominant species or functional group composition) and structural stage (i.e., vegetation cover, size class, and canopy layers) within a particular biophysical environment. Boxes are linked by arrows (transitions) that represent natural disturbances, management actions, or vegetation growth and development (fig. 1.3). The ILAP team sometimes augmented existing STMs used by various organizations for land management planning, restoration planning, and ecoregional assessments in Arizona, New Mexico, Oregon, Washington, and elsewhere (e.g., Evers 2010, Forbis et al. 2006, Hann et al. 1997, Hemstrom et al. 2007, Holsinger et al. 2006, Merzenich and Frid 2005, Weisz et al. 2009). In some cases, STMs consistent with the project framework did not exist and new models were constructed using similar existing models as templates. State-and-transition models were developed for units of potential vegetation within 18 modeling zones (see chapter 2, fig. 2.4), resulting in 275 STMs across the four-state study area. Transition probabilities in the STMs come from a combination of expert opinion, available literature, and empirical data analysis (see chapter 2). However, the structure of ILAP STMs allows relatively easy updating of transitions and probabilities from empirical data (e.g., annual insect and disease surveys) and

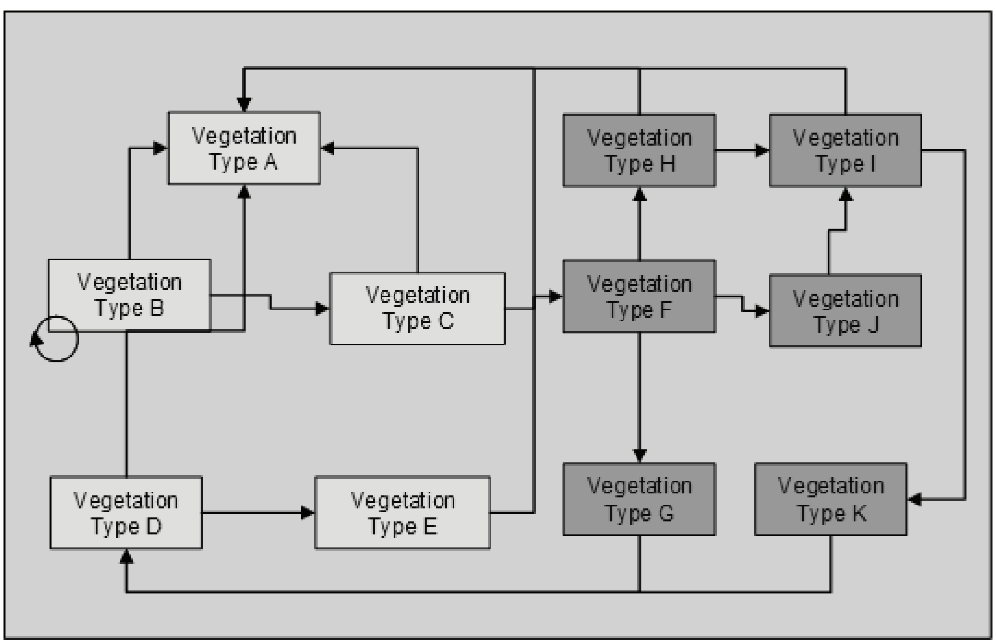

Figure 1.3—Generalized state-and-transition model (STM) diagram used in the Integrated Landscape Assessment Project. Boxes represent state classes, comprised of cover type and structural stage. Arrows represent transitions that simulate processes that can cause an area to move from one state class to another. One STM is built for each potential vegetation type.

ancillary models, such as the Forest Vegetation Simulator (Crookston and Dixon 2005, Stage 1997) and new inventory data.

Models were stratified and run on combinations of land ownership-management classes, potential vegetation, and watershed (fifth-code hydrologic unit [HUC5]; USGS and USDA NRCS 2011). Results forecast potential future amounts and distributions of important landscape characteristics, but vegetation data accuracy limit appropriate use to landscape scales rather than vegetation stands. The ILAP analyses were often summarized by watershed as an appropriate spatial scale. Alternative land management scenarios can easily be generated by changing assumptions about vegetation management treatments and rates by land ownership-management allocation. The resulting forecasts of vegetation conditions, management activities, and natural disturbances were linked to wildlife habitat characteristics, economic values, and other important attributes (see below and chapters 3 through 7).

The ILAP was organized into nine knowledge discovery modules, each generating new information on a particular natural resource issue.

Knowledge discovery modules—

The ILAP was organized into nine knowledge discovery modules, each generating new information on a particular natural resource issue. Knowledge discovery modules are described briefly below and selected modules are described in detail in chapters 3 through 7.

***Fire and fuel characterization module*—**The fire and fuel characterization module evaluated current and potential future fuel characteristics and fire hazard for forests and woodlands across Arizona, New Mexico, Oregon, and Washington. The module team built fuel beds (descriptions of burnable biomass extending from the forest floor to the canopy) in the Fuel Characteristic Classification System (Ottmar et al. 2007) from inventory plots for each vegetation state class in the STMs. The resulting fuel beds (over 14,000) were analyzed for fire potential and linked to STM output, allow clients and users to assess current conditions and trends in fuels and potential fire behavior over time under different management scenarios. See chapter 3 for further detail.

***Fuel treatment economics module*—**The fuel treatment economics module estimated potential tree-based biomass (by diameter classes and tree species groups), timber volume, and aboveground, tree-based carbon pools by STM state class and potential vegetation type for all forests and woodlands in Oregon and Washington. The study used STM simulation outputs of the removed timber products from proposed treatments over the simulation period to perform cost-benefit analyses. The analyses considered harvesting costs associated with each treatment using the Fuel Reduction Cost Simulator (Fight et al. 2006), transportation cost to mill locations, products prices, and other economic factors. It provided data and methods to allow managers and others to assess the financial feasibility of proposed forest vegetation management treatments. See chapter 4 for further details.

Wildlife habitat module—The wildlife habitat module developed species-habitat relationships for 24 species in Oregon and Washington and 13 species in Arizona and New Mexico, linking habitat and nonhabitat classifications to STM state classes in forest, woodland, shrubland, grassland, and desert models. The team generated tools that estimate the amount of current and potential future habitat area for selected species across the four-state area. These species-habitat relationships can be used for mid- to broad-scale assessments of management and on potential wildlife habitat. See chapter 5 for further detail.

Community economics module—The community economics module addressed the question of whether large-scale forest vegetation treatment programs can stimulate economic activity and contribute to well-being in communities that have been negatively impacted by recent federal forest policy changes. The team produced indicators for each HUC5 watershed (and ownership-management allocation within watershed) that describe the potential for fuel treatment in those watersheds to produce benefits to communities for the forested landscapes in Arizona, New Mexico, Oregon, and Washington. See chapter 6 for further detail.

Climate change and vegetation module—The climate change and vegetation module used the MC1 dynamic global vegetation model (Bachelet et al. 2001) to inform vegetation change and wildfire trends in STMs for two focus areas: central Oregon and the Apache-Sitgreaves area in eastern Arizona (fig. 1.1). The result is a set of "climate-informed" STMs that can be used to determine likely shifts in vegetation structure and species composition and abundance with climate change, and can be used by land managers to weigh potential benefits or tradeoffs associated with alternative management approaches under a changing climate. Analyses were conducted for three climate change scenarios that bracket the range of projected climatic changes for the study areas. See chapter 7 for further detail.

In addition to the coupled model approach for two focus areas, the climate change and vegetation module team ran coarser scale (4-km grid) simulations for Arizona, New Mexico, Oregon and Washington. These simulation data cover the historical (1895–2009) and future (2010–2100) time periods for the entire four-state area, and an Envision software-based tool (Bolte 2007) was developed and constructed as a GIS plug-in that allows users to extract climatic, hydrologic, vegetation, and other data from these MC1 outputs.

Climate change and watersheds module—The climate change and watersheds module developed information on the current condition of all HUC5 watersheds that contain national forest lands in Oregon and Washington. The module used NetMap (http://www.netmaptools.org/; Benda et al. 2007) to generate estimates of erosion hazards, in-channel habitat conditions, and other important watershed characteristics. It also allows estimates of potential future changes in watershed condition under climate change by analyzing the likely effects of changes in precipitation amount and seasonality, along with changes to wildfire and vegetation conditions that might result under different future climates.

Climate change and fire probabilities module—The objective of the climate change and fire probabilities module was to provide refined insight into the potential variation of wildfire probabilities and vegetation dynamics that may occur with climate change. Wildfire probabilities and vegetation transitions are key to STM parameterization, but we know little about how variations in wildfire probability or shifts in vegetation composition may vary with changes in climate. This module used spatially explicit landscape simulation models to examine variation in wildfire probabilities and vegetation dynamics with climate change in a prototype area of the upper Deschutes subbasin in Oregon. The methods and results of these analyses will be described in a forthcoming report.

Ecosystem management decision support module—This module used the Ecosystem Management Decision Support system (Reynolds 1999, Reynolds and Hessburg 2005) to integrate information on current and potential future vegetation, fuels, wildlife habitat, and economic conditions into a combined, flexible assessment and prioritization tool. This tool will help managers and others explore and set priorities using color-coded maps, tables, and reports based on various combinations of characteristics that best reflect local values. The methods and results of this work will be described in a forthcoming report.

Optimized decision support module—The optimized decision support module developed methods that integrate fuels, wildlife habitat, and economic conditions into a spatial analytical decision support process. Weighting and prioritization are driven by direct user input or through an evaluative technique, such as the Analytical Hierarchy Process (e.g., Kangas 1992, Thomas 1990). Methods and results of this work will be described in a forthcoming report.

Access to ILAP Data and Information

The ILAP created a number of methods and outputs that were developed to help land managers and planners integrate and prioritize management activities. Through the Western Landscapes Explorer (www.westernlandscapesexplorer.info), the project's publications, models, maps, data, and tools will be archived and available online so that scientists and managers will be able to use and build upon the project's products. Land managers, planners, analysts, scientists, policymakers, and large-area landowners can use the project's tools and information for many applications, including:

- Watershed restoration strategies
- Land management planning across all lands
- Statewide assessments and bioregional plans

Through the Western Landscapes Explorer, the project's publications, models, maps, data, and tools will be archived and available online so that scientists and managers will be able to use and build upon the project's products.

The ILAP data and information are being used to support USDA FS Forest Plan Revisions, Collaborative Forest Restoration Program projects, and can be used to inform upcoming statewide forest assessments, statewide wildlife action plans, and ecoregional assessments. As this first phase of ILAP concludes, a strong foundation of landscape-level data, STMs, tools and expertise has been built for future landscape planning and assessments across a broad range of western landscapes.

Acknowledgments

The ILAP was funded by the American Recovery and Reinvestment Act, and the USDA FS Pacific Northwest Research Station, Pacific Northwest Region, and Southwestern Region. The USDA FS Washington office and Intermountain Research Station provided project advice and review. The U.S. Department of the Interior Bureau of Land Management in Arizona, New Mexico, and Oregon/Washington contributed plot data and other valuable information. Several state agencies, including the Arizona Department of Forestry, Arizona Department of Fish and Wildlife, Oregon Department of Forestry, and Washington Department of Natural Resources provided data, input on project priorities, and review of products. The Nature Conservancy in Arizona, New Mexico, Oregon, and Washington provided data and review of products. Oregon State University, the University of Arizona, University of New Mexico, and University of Washington provided data, product review, contracted work on science delivery and knowledge discovery, and other valuable input. Ecosystem Management Inc. was contracted to compile plot data, wildlife habitat information, and other supporting data in the Southwest.

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Chapter 2: Dynamic Vegetation Modeling of Forest, Woodland, Shrubland, and Grassland Vegetation Communities in the Pacific Northwest and Southwest Regions of the United States

Theresa K. Burcsu, Joshua S. Halofsky, Simon A. Bisrat, Treg A. Christopher, Megan K. Creutzburg, Emilie B. Henderson, Miles A. Hemstrom, F. Jack Triepke , and Melissa Whitman[1]

Chapter Summary

Land management planning at broad scales requires integrative techniques to understand and synthesize the effects of different land management activities and address socioeconomic and conservation concerns. The Integrated Landscape Assessment Project was developed to support the vital but complex task of broad-scale integration of information to assess ecological sustainability at multiple scales. The project supports ecosystem management planning at a regional scale across all lands of Arizona, New Mexico, Oregon, and Washington by simulating landscape dynamics using state-and-transition models (STMs) and linking model output to management planning considerations such as fuel conditions, wildlife habitat, community economics, and climate change. The stakeholders and target users for the

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project products include natural resource planners, decisionmakers, and modelers who can provide additional analyses in support of planning and policy. This chapter reports on the STM component of the project. The STMs were designed for all major potential vegetation types in the study area with a focus on watershed-level prioritization of land management actions. One baseline scenario, depicting vegetation dynamics with no management activity, was applied across the full study area. Other management scenarios were developed for focus areas within the four-state project area. The modeling framework was sometimes linked with other modeling systems (e.g., Forest Vegetation Simulator) and other data sets for validation or calibration, and incorporated expert opinion where data were lacking. The process was flexible and modular to allow alternative data sources for vegetation and other base data to be incorporated. Products resulting from this work include STMs of vegetation dynamics, tools for preparing and initializing models, data summarization and visualization tools, and model output data sets across the four-state study area. Most data, tools, and products will be available online via the Western Landscapes Explorer (www.westernlandscapesexplorer.info).

Introduction

Understanding shifts in ecosystem dynamics in the face of continued human influence is vital for sustaining ecosystems (Christensen et al. 1996). Twentieth-century management practices, including timber harvest and fire suppression, have altered the ecosystem dynamics of many native communities throughout the Western United States. In forested communities, management has resulted in changes to forest structure, composition, and ecological function (Franklin et al. 2002), as well as shifts in natural disturbance regimes such as wildfire and insect outbreaks, particularly in drier environments (Ottmar and Sandberg 2001). In the Southwestern United States, management has contributed to changes in forest and woodland fire regimes, as witnessed by the massive Whitewater-Baldy Fire of 2012, as well as community structure, composition, and function (Allen et al. 2002). In warm, dry shrub steppe and desert environments across the Western United States, invasion by exotic grasses and forbs has promoted frequent, stand-replacing fire (Knapp 1996, Whisenant 1990). In many cooler, upland shrub steppe sites, juniper expansion beyond its historical range has converted historical grassland and shrub steppe communities to woodlands, resulting in loss of sagebrush and other habitat (Burkhardt and Tisdale 1976, Miller et al. 2005).

Management plans are being revised for many of these altered landscapes, and there is an opportunity to provide increased understanding of coupled human-natural systems through methods that incorporate current models for landscape dynamics, agency management treatments, and common conventions for vegetation description. Integrating management methods and accepted concepts, socioeconomic and conservation concerns, and other needs into land management and planning is challenging but provides a means to weigh the influence of a wide range of system drivers, especially those related to vegetation dynamics at regional scales (Christensen et al. 1996, Dale et al. 1998, Hann and Bunnell 2001).

The Integrated Landscape Assessment Project (ILAP) was designed to support ecosystem management planning at a regional scale with a focus on watershed-level prioritization of land management actions across all forests, woodlands, and arid lands of Arizona, New Mexico, Oregon, and Washington. The project explored the dynamics of broad-scale, multiownership landscapes by integrating information about current and future vegetation and fuel conditions, climate change, wildlife habitat, fuel treatment economics, and community economics. These factors were chosen because they are being managed for or considered as part of the management planning process. The stakeholders and target users for the project products included natural resource planners, decisionmakers, and modelers who can provide additional analyses in support of planning and policy. At the heart of the project were state-and-transition models (STMs) depicting vegetation state classes and processes that cause vegetation change over time. The models were built to represent the range of vegetation types from forested to arid lands in all four states, and project changes from vegetation community development, natural disturbances, and management events. One baseline scenario depicting no management except for fire suppression (fire suppression only [FSO]) was modeled for the entire project study area, and several other scenarios were customized for geographic focus areas. Products from the work included vegetation models, spatial data sets describing current vegetation conditions, and projections of vegetation conditions. A number of tools were also developed to facilitate project work and applications beyond the life of the project, such as visualization tools and two decision support frameworks for land management. In this chapter, we describe the data and methods used to construct, parameterize, and run the STMs for ILAP.

State-and-Transition Modeling Framework

In landscape ecology and resource management, STMs have been applied to simulate vegetation change in rangelands, woodlands, and forests. The STMs can be thought of as box and arrow diagrams (fig. 2.1), where boxes represent state classes describing structural and functional attributes, and may include one or more successional phases (Bestelmeyer et al. 2009). Arrows represent the drivers causing state class change, such as succession, disturbance, and management (Westoby et al. 1989). The STMs can be designed to address both broad- and fine-scale research

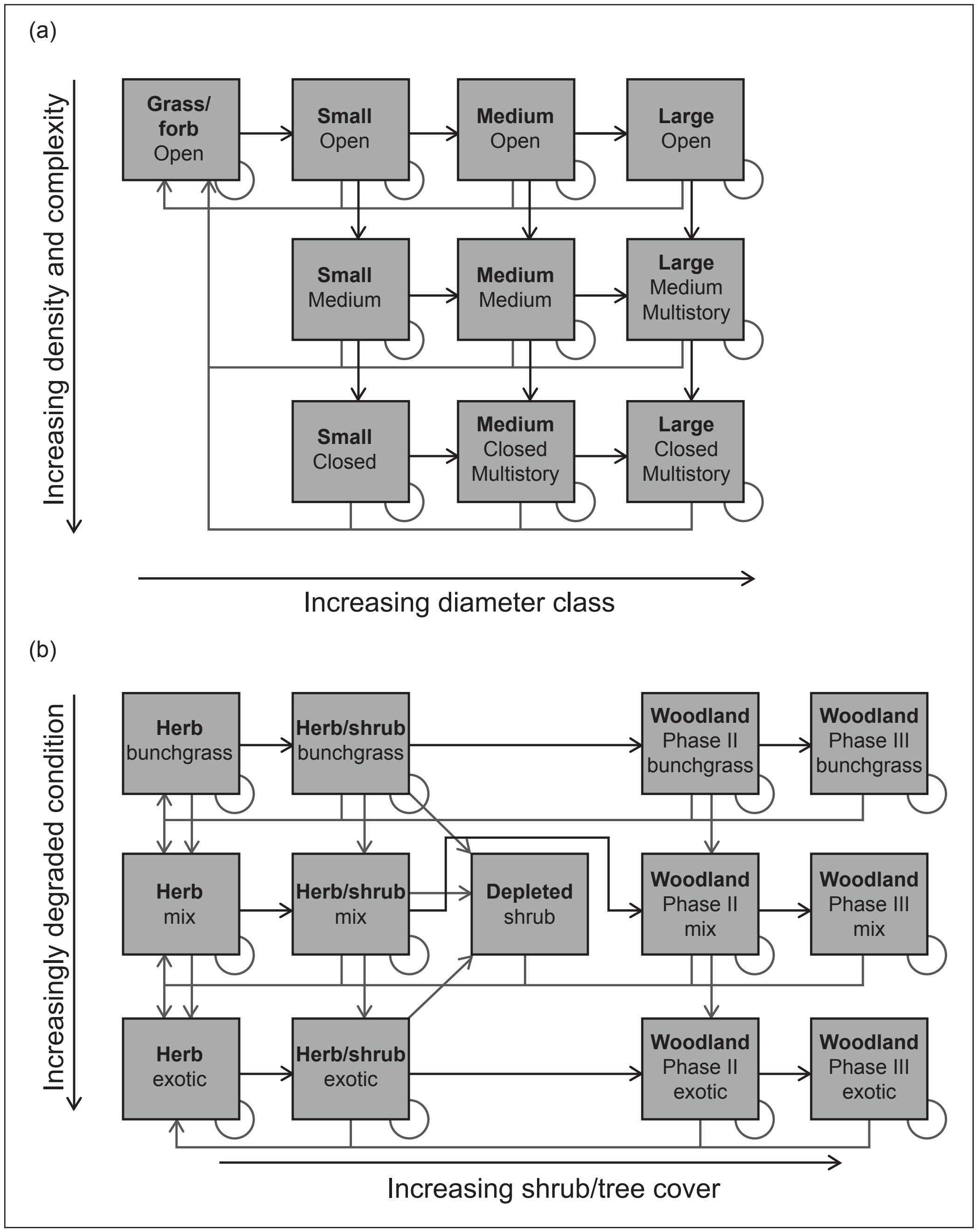


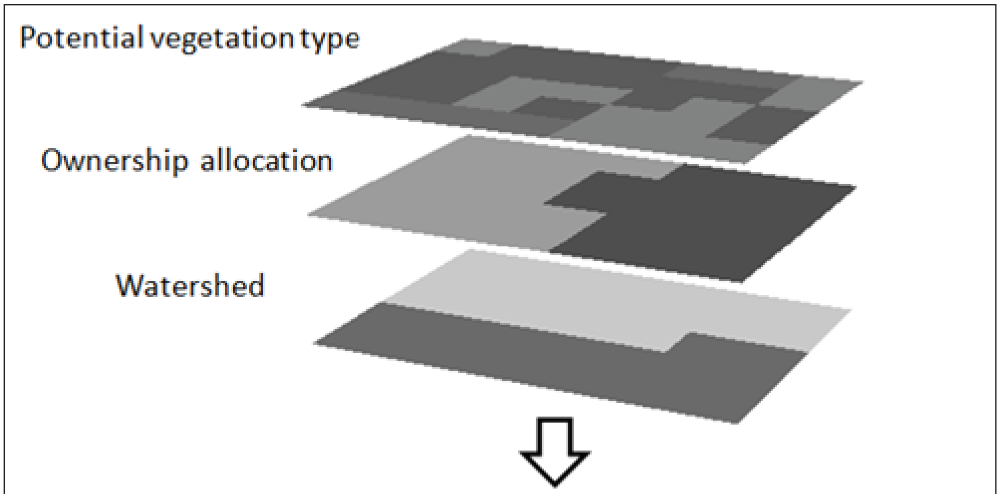
Figure 2.1—Simplified state-and-transition models (STMs), with state classes (boxes) depicting combinations of vegetation cover and structure, and transitions (arrows) showing process that causes vegetation change. Black arrows indicate growth and succession, and grey arrows represent disturbances and management activities. The figure includes (a) a schematic of the basic structure of a forest or woodland STM. Columns in a model represent the progression (left to right) from younger to older trees (and larger size classes). Rows typically represent the progression (top to bottom) from open to closed density classes, based on percentage of canopy cover. (b) An example of an arid land STM of sagebrush-steppe vegetation dynamics. Succession proceeds from left to right as shrubs and trees become dominant (woodland phases from Miller et al. 2005). Fires and other major disturbances return cells to early successional state classes. Rangeland condition is characterized by the composition of the herbaceous layer, ranging from good-condition native bunchgrass (top) to a semidegraded mix of native and exotic species (middle) to an exotic-dominated herbaceous layer (bottom).

questions. They have been used extensively in rangeland management to represent highly dynamic and perturbation-sensitive rangeland ecosystems (Briske et al. 2005, Petersen et al. 2009, Westoby et al. 1989), examine ecosystem resilience and the effects of restoration (Forbis et al. 2006), and project the distribution of state classes on the landscape through time as part of an integrative modeling framework for planning (Baker 1989, Bestelmeyer et al. 2009, Hemstrom et al. 2007, Vavra et al. 2007, Wales et al. 2007). As a decision support tool, STMs allow synthesis of assumptions about vegetation growth, natural disturbance regimes, and management regimes in a single modeling environment (Bestelmeyer et al. 2003, Hemstrom et al. 2007). Models developed under this framework are relatively simple to parameterize and can integrate expert opinion and information derived from data (Provencher et al. 2009, Westoby et al. 1989).

For this project, individual STMs represented vegetation dynamics (alternate state classes, successional processes, disturbance, etc.) within broad units of potential vegetation. Potential vegetation provided a useful basis for building ecological units because common species assemblages, site productivity, and disturbance patterns are represented, and finer classifications (plant association groups and plant associations) commonly used by federal land management agencies for planning and project implementation (e.g., Hall 1998) could also be included. The resolution of potential vegetation is also compatible with the strategic and landscape-scale perspectives of contemporary resource managers. More importantly, potential vegetation units provide concise descriptions of biophysical conditions and disturbance regimes. Within our project area, potential vegetation maps provided the spatial extent in which each STM operated, similar to the use of biophysical setting by the LANDFIRE project (Rollins 2009), or the ecological site descriptions by the Natural Resources Conservation Service (www.nrcs.usda.gov).

Because spatial modeling would be impractical at the scale of the project area, we used nonspatial STMs. However, our modeling process combined spatial layers of potential vegetation, watersheds, and land ownership-management allocation types (hereafter ownership-management) to form discrete spatial units called modeling strata (see “Spatial Data”; fig. 2.2). The features in each layer worked much like a set of interconnected cookie cutters that divided the landscape into modeling strata. Each modeling stratum was linked to an STM model run, and STM results were summarized for each modeling stratum. This allowed STM output to be summarized and mapped onto meaningful landscape units to illustrate changes in modeled state classes through space and time and link to other characteristics of interest (fig. 2.3; chapters 3 through 7). We also used spatial data from the current vegetation layer to initialize STMs with current vegetation conditions in the first time step of the simulations (see “Spatial Data”).

This allowed STM output to be summarized and mapped onto meaningful landscape units to illustrate changes in modeled state classes through space and time and link to other characteristics of interest.



Watershed	PVT	Owner	Modeling Strata	Area
101	OSW_fmh	f5	101_OSW_fmh_f5	10
101	OSW_fmh	o1	101_OSW_fmh_o1	6
102	OSW_fal	f1	102_OSW_fal_f1	15

Figure 2.2—Modeling strata combined land ownership-management, potential vegetation type (PVT), and watershed and provided the basic spatial unit for state-and-transition modeling. Modeling strata organized modeling activities so that divisions of the landscape that share model assumptions were assigned parameters that were specific to the conditions found in sites with those characteristics (e.g., a wildfire rate specific to a PVT and a management rate specific to the ownership-management type).

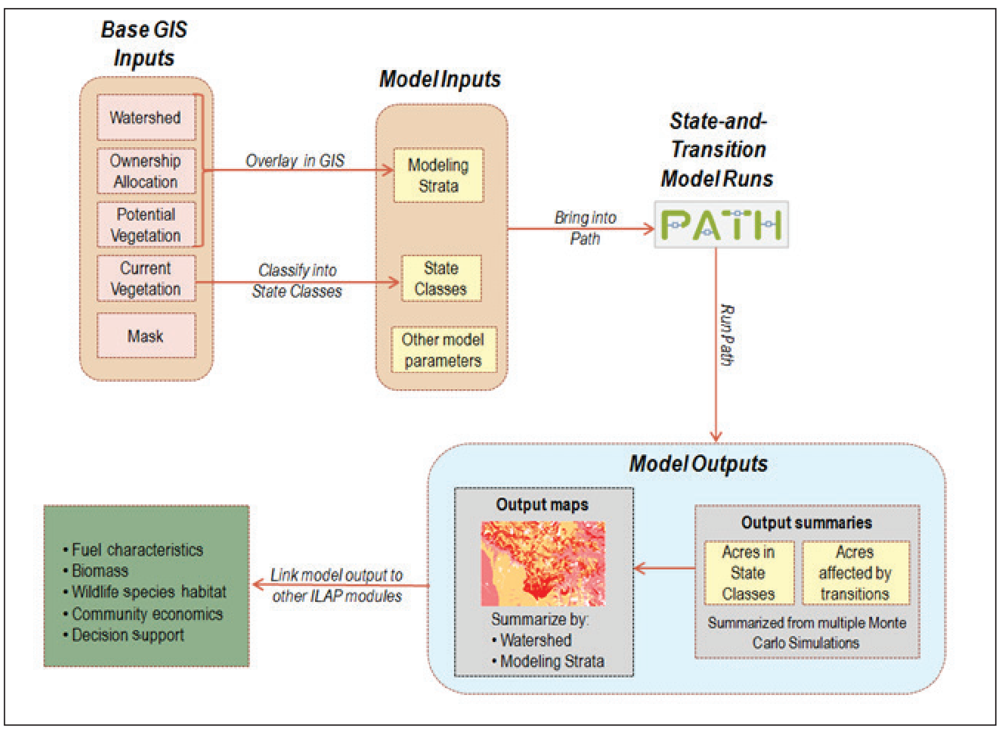


Figure 2.3—Flow diagram illustrating the data inputs and outputs for the state-and-transition modeling process. Base geographic information system (GIS) layers were processed to comprise model inputs, which were brought into the Path modeling platform. Path model output was used to summarize vegetation dynamics in graphical and map form, and output was linked to other characteristics of interest by other project modules. ILAP = Integrated Landscape Assessment Project.

The individual state classes (boxes) within ILAP STMs represented cover types, usually the dominant species or vegetation assemblage, and structural stages, based on physical attributes such as vegetation height, percentage cover, and canopy layers. For example, in a forest STM (fig. 2.1a), a state class may represent ponderosa pine of the 25- to 38-cm diameter class in multiple structural stages, depending on whether it had open-, mid-, or closed-canopy cover. In a shrub steppe model (fig. 2.1b), a state class may represent shrub steppe with native bunchgrasses and sagebrush, where grass cover was open and shrub cover was sparse. To link these conceptual STM state classes to current landscape conditions, we utilized spatial data of current vegetation. The spatial data in the current vegetation map (cover type and structure) was allocated to each of the state classes within a STM, forming the modeling initial conditions (see "Data Processing"). These initial conditions provided the starting point (initial time step) from which a given STM simulated vegetation dynamics (fig. 2.2).

The transitions (arrows) in the STMs simulated successional processes such as growth and development, natural disturbances such as wildfire and insect outbreaks, and management actions such as prescribed fire, seeding of native bunchgrasses, and tree harvesting. Transitions were classified as either deterministic or probabilistic. Deterministic transitions occurred at a specific vegetation age, whereas probabilistic transitions were defined by an annual transition probability. Transitions could be linked to other transition types using a "time-since-disturbance" clock, allowing the modeler to simulate interaction effects, such as a the delayed onset of an insect outbreak with tree size, or the undesirable effect of livestock grazing following a major disturbance. Some transitions moved vegetation from one state class to another; for instance, a stand-replacing disturbance may move vegetation from a dense forest to a grass/forb state class. Other transitions resulted in vegetation remaining in the same state class, such as surface fires, mild insect activity, drought, and some types of grazing.

The STMs were developed in the Vegetation Dynamics Development Tool (VDDT), version 6.0.25 (ESSA Technologies Ltd. 2007), and simulations were run using the Path landscape model (Path), version 3.0.4 (Apex Resource Management Solutions 2012; Daniel and Frid 2012). The software simulated vegetation dynamics by dividing the landscape into cells and simulating movement among state classes according to user-defined transition probabilities at each time step (ESSA Technologies Ltd. 2007, Kurz et al. 1999). VDDT is a nonspatial STM framework that has been used to support broad landscape modeling efforts such as the Interior Northwest Landscape Analysis System (Barbour et al. 2007) and the LANDFIRE project (www.landfire.gov; Rollins 2009). More recently, it has been folded into the Path landscape model interface (Apex Resource Management Solutions 2012; Daniel and Frid 2012), which facilitates running multiple STMs concurrently over large and diverse landscapes.

State-and-Transition Model Collection and Development

Using and adding to the collection of STMs for major vegetation types in the study area was part of the original vision for the project and its timeline.

A collection of STMs representing the major vegetation types already existed for much of the study area; using and adding to this collection was part of the original vision for producing project results rapidly. The STMs used for this project were catalogued and run in three overarching groups: forests, woodlands, and arid lands (nonforested types), which were further divided by geographic regions. In the U.S. Department of Agriculture Forest Service (USDA FS) Southwestern Region 3 (Arizona and New Mexico; AZ/NM hereafter), forest and woodland areas were generally defined as areas with the potential to support mature trees and a canopy cover >10 percent, and dominant species composition was used to separate forests from woodlands. In USDA FS Pacific Northwest Region 6 (Oregon and Washington; OR/WA hereafter), forested areas were generally defined using a ≥10 percent tree cover cutoff, with the exception of juniper communities and locations identified as burned or cut. In both regions, arid lands included areas that contained <10 percent tree cover historically and have the potential to support shrub and herb vegetation. Model sources included the USDA FS Region 6, USDA FS Region 3, the LANDFIRE project, the USDI Bureau of Land Management (USDI BLM), and The Nature Conservancy (TNC). Many existing STMs did not require modifications, but for those that did, modifications included calibration of transitions and vegetation state classes using the Forest Vegetation Simulator (FVS; see Forest Vegetation Simulator Calibration), modifying fire probabilities based on the Monitoring Trends in Burn Severity (MTBS) data (see "Wildfire Probabilities"), reconciling multiple model versions using expert opinion and literature, and altering transition probabilities and model state classes based on expert review and scientific literature. New models were developed for many of the AZ/NM arid lands and for some OR/WA arid land vegetation types. In all, 275 STMs were adapted or constructed for the project.

Oregon and Washington Forest Models

Modeled Vegetation Types

The ILAP effort compiled or constructed 53 STMs that covered all major forested vegetation types in OR/WA (table 2.1). The FVS was used for model calibration in the Washington Coast Range, Washington West Cascades, and Washington North Cascades areas of OR/WA (table 2.1; see Forest Vegetation Simulator Calibration).

Table 2.1—Descriptions of forested vegetation types modeled in Oregon and Washington[a]

Potential vegetation type	Description
Douglas-fir-dry	Dry Douglas-fir with ponderosa pine as a long-lived, early-seral species.
Douglas-fir-moist	Moist Douglas-fir with ponderosa pine and incense cedar as long-lived early-seral species. Pacific madrone, big leaf maple, and Oregon white oak are also common.
Douglas-fir-white oak	Douglas-fir on sites too dry for western hemlock and white fir. Oregon white oak and Pacific madrone are also common.
Douglas-fir-xeric	Douglas-fir on sites too dry to support western hemlock or white fir but moister than Douglas-fir/white oak.
Grand fir-cool/moist	Grand fir and Douglas-fir on mid- to high-elevation sites in eastern Oregon and Washington.
Grand fir-valley	Grand fir and Douglas-fir on low-elevation, warm, dry sites in western Washington.
Grand fir-valley (Oregon)	Grand fir and Douglas-fir on low-elevation, warm, dry sites in western Oregon. Big leaf maple, Oregon white oak, and Pacific madrone may also be common.
Grand fir-warm/dry	Grand fir and Douglas-fir on low- to mid-elevation, warm, dry sites in eastern Oregon and Washington.
Lodgepole pine-dry	Lodgepole pine found on pumice parent material but without the presence of ponderosa pine.
Lodgepole pine-wet	Lodgepole pine found in cold-air pockets and riparian zones with transitions to meadows, willow, and quaking aspen.
Mixed conifer-cold/dry	Mixed conifer at mid to high elevations in Oregon's eastern Cascades.
Mixed conifer-dry	Mixed conifer at mid elevations in the eastern Cascades.
Mixed conifer-dry (pumice soils)	Mixed conifer at mid elevations in Oregon's eastern Cascades and defined by the pumice parent material.
Mixed conifer-moist	Mixed conifer at mid to high elevations in the eastern Cascades.
Mountain hemlock-cold/dry	Mountain hemlock at high elevations in the eastern Cascades. Lodgepole pine is a frequent seral species.
Mountain hemlock-cold/dry (coastal/west Cascades)	Mountain hemlock at high elevations west of the Cascades or in the coastal ranges. Lodgepole pine is a frequent seral species.
Mountain hemlock-intermediate	Mountain hemlock with temperature and moisture intermediate to the cold/dry vegetation type. Alaska cedar and Pacific silver fir may also be present.
Mountain hemlock-wet	Mountain hemlock at high elevations on wetter locations. Lodgepole pine is infrequent.
Oregon white oak	Oregon white oak and ponderosa pine in southwestern Oregon. Sites are too warm and dry to support Douglas-fir.

Table 2.1—Descriptions of forested vegetation types modeled in Oregon and Washington[a] (continued)

Potential vegetation type	Description
Oregon white oak-ponderosa pine**	Oregon white oak and ponderosa pine in the eastern Cascades. Sites are too warm and dry to support Douglas-fir.
Pacific silver fir-intermediate**	Pacific silver fir and Douglas-fir at mid to high elevations of the coastal and Cascade mountain ranges.
Pacific silver fir-warm	Pacific silver fir and Douglas-fir at lower elevations of the coastal and Cascade mountain ranges.
Pacific silver fir-wet	Pacific silver fir and Douglas-fir on very moist sites at mid to high elevations in western Washington.
Ponderosa pine-dry**	Ponderosa pine and Douglas-fir found at lower elevations in Washington's east Cascades or limited areas of southwestern Oregon.
Ponderosa pine-dry, with juniper**	Ponderosa pine and Douglas-fir found at lower elevations in Oregon's east Cascades where western juniper may encroach.
Ponderosa pine-lodgepole pine	Ponderosa pine and lodgepole pine found on pumice parent material.
Ponderosa pine-xeric	Ponderosa pine on sites that are transitional to dry woodland species such as curl-leaf mountain mahogany and western juniper.
Shasta red fir-dry	Mixed conifer at mid to high elevations in the eastern Cascades. Shasta red fir is the dominant late-seral species.
Shasta red fir-moist	Douglas-fir at moist, high-elevation locations in southwestern Oregon. Shasta red fir is the dominant late-seral species.
Sitka spruce	Near-coastal, low-elevation forest dominated by Douglas-fir and with Sitka spruce and western hemlock as late-seral species.
Subalpine fir*	Subalpine fir at high elevations in western Washington. A variety of species may be codominant in this type.
Subalpine fir-cold/dry	Subalpine fir at high elevations in eastern Oregon and Washington. A variety of species may be codominant in this vegetation type.
Subalpine parkland**	Subalpine fir at the highest elevations still capable of supporting trees. Tree distribution is patchy and surrounded by shrubby or herbaceous communities.
Subalpine woodland	Subalpine forests in eastern Oregon, historically dominated by white bark pine or limber pine at the highest elevations still capable of supporting trees. Tree distribution is patchy and surrounded by shrubby or herbaceous communities.
Tan oak-Douglas-fir-dry	Douglas-fir and tan oak on moderately dry sites in southwestern Oregon.
Tan oak-Douglas-fir-moist	Douglas-fir and tan oak on moist sites in southwestern Oregon.
Tan oak-moist	Tan oak on moist sites in coastal Oregon.

Table 2.1—Descriptions of forested vegetation types modeled in Oregon and Washington[a] (continued)

Potential vegetation type	Description
Ultramafic	Vegetation type defined by serpentine and peridotite soil parent materials. Jeffrey pine, Douglas-fir, and incense cedar are common species.
Western hemlock-coastal	Western hemlock and Douglas-fir found in a relatively narrow strip along the coast in southwestern Oregon.
Western hemlock-cold	Western hemlock and Douglas-fir on very moist sites in the higher, colder portions of the western hemlock zone.
Western hemlock-hyperdry	Western hemlock and Douglas-fir on the very driest sites still capable of supporting western hemlock. Often transitional to Douglas-fir and grand fir vegetation types.
Western hemlock-intermediate	Western hemlock and Douglas-fir at lower to middle elevations on well-drained but not dry sites.
Western hemlock-moist	Western hemlock, Douglas-fir, and western redcedar on moist and productive sites west of the Cascade crest.
Western hemlock-moist (coastal)	Western hemlock, Douglas-fir, and western redcedar on moist and productive sites in the coastal mountain ranges.
Western hemlock-wet	Western hemlock, Douglas-fir, and western redcedar on wet sites from poor soil drainage or concave topography.
Western redcedar/western hemlock-moist	Western hemlock, Douglas-fir, and western redcedar in northeastern Washington.
White fir-cool	White fir and Douglas-fir in southwestern Oregon. In this area, white fir has replaced Pacific silver fir in mid- to high-elevation forests.
White fir-intermediate	White fir and Douglas-fir in southwestern Oregon with intermediate temperature and moisture.
White fir-moist	White fir and Douglas-fir in southwestern Oregon.

[a] Model source is indicated as a superscript where: * = new model built for the project, and ** = model built by the U.S. Department of Agriculture Forest Service (USDA FS) and Washington State Department of Natural Resources. Vegetation types with no superscript represent models that were developed by the USDA FS.

Modifications to these STMs included calibration of existing growth and succession transition rates and the omission or addition of state classes, depending on the STM and potential vegetation. In addition, wildfire probabilities under current climate, vegetation, and fire suppression conditions were computed from MTBS data for most models of potential vegetation on the east side of the Cascade Mountains (table 2.1; see “Wildfire Probabilities”).

Model Structure

The ILAP models necessarily abstracted the broad range of structural variety found in forests into four density classes and six diameter size classes (table 2.2). Models typically included growth and development transitions, natural disturbance transitions, and management transitions (fig. 2.1a, table 2.3). Growth and development transitions increased forest age and complexity. Natural disturbance transitions differed across modeling zones (fig. 2.4), but most models included disturbances related to wind, wildfire, insects, or fungal pathogens. Depending on disturbance severity, the effects of a transition ranged from simply delaying deterministic growth to returning vegetation to an open, seedling-sapling state class. Management transitions also varied across models and included regeneration harvest, salvage harvest following a natural disturbance, precommercial thinning, commercial thinning, thinning from below, and others. Transitions can be customized to better fit the management plans of landownership types or other conditions by altering rates and the removal or addition of transition types as necessary.

Table 2.2—Structural attributes modeled in forest state-and-transition models for Oregon and Washington

Density classes		Size classes	
Density (percent canopy cover)	**Model density classes**	**Diameter at breast height range**	**Model size classes**
		Centimeters	
<10	Grass/shrub	Nonstocked	Grass/shrub
10 to 40	Open	<13	Seedling/sapling
40 to 60 >60	Medium	13 to 25	Pole
	Closed	25 to 38	Small
		38 to 51	Med
		51 to 76	Large
		>76	Giant

Table 2.3—Probabilistic transition types used in Oregon and Washington forest state-and-transition models

Transition type	Transition description
Natural transitions:	
Insects and disease	All insect and disease damage to trees, usually delaying growth or transitioning to postdisturbance state classes
Succession	All succession types, representing both the growth of trees and changes to tree species composition within a stand
Wildfire	All wildfire transitions
Alternate succession	Alternative successional pathways, such as when snags fall over and result in a change in state class
Insects and disease-low-severity	Low-severity effects of insects and diseases (background levels) that delay deterministic growth from one state class to another
Forest Vegetation Simulator (FVS) growth	Probabilistic growth transitions derived from the FVS to replace deterministic growth transitions
Canopy growth	Canopy growth that increases stand density after a set number of years without any disturbance or management
Hemlock looper	Damage related to hemlock looper insect outbreaks
Mistletoe	Damage related to dwarf mistletoe
Mountain pine beetle	Damage related to mountain pine beetle insect outbreaks
Natural regeneration	Natural conifer regeneration
Other bark beetles	Damage related to bark beetles other than the western pine beetle, such as Jeffrey and mountain pine beetles
Root disease	Damage related to root diseases
Spruce budworm and tussock moth	Damage related to spruce budworm and tussock moth insect outbreaks
Understory development	Understory development that moves stands from single to multistory state classes after a certain number of years following a fire or management treatment
Mixed-severity wildfire	Wildfire that kills 25 to 75 percent of the forest canopy
Nonlethal wildfire	Wildfire that kills less than 25 percent of the forest canopy
Stand-replacing wildfire	Wildfire that kills more than 75 percent of the overstory
Windthrow	Damage related to severe windthrow
Wooly adelgid	Wooly adelgid insect outbreaks causing subalpine fir mortality
Western pine beetle	Damage related to western pine beetle insect outbreaks
Management transitions:	
Timber harvest	All harvest types including partial, salvage, selection, and regeneration
Partial harvest	All partial harvest types including selection

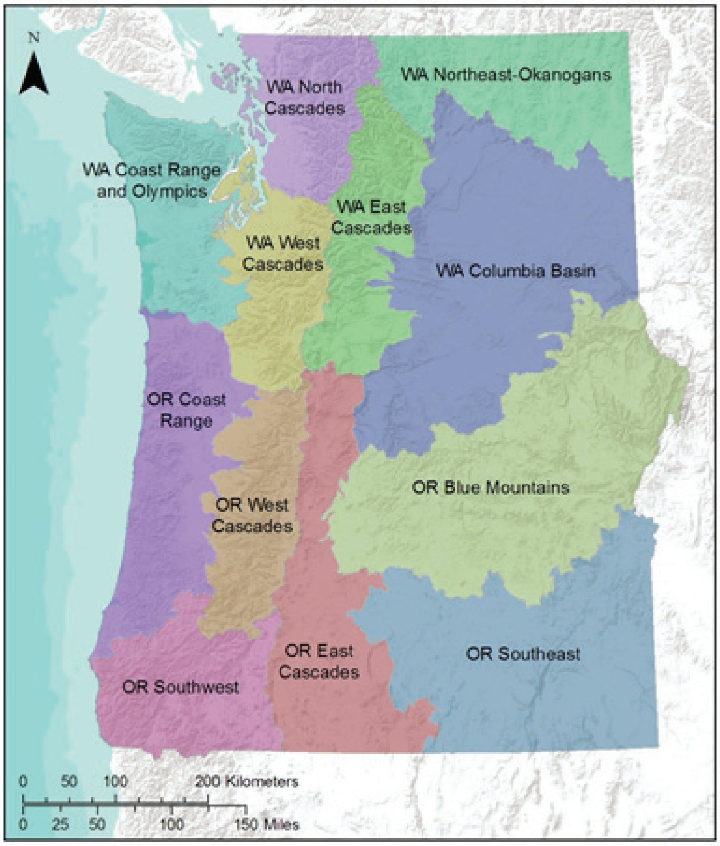
Table 2.3—Probabilistic transition types used in Oregon and Washington forest state-and-transition models (continued)

Transition type	Transition description
Grazing and miscellaneous	The effects of grazing on seedling/sapling development, generally accelerating the transition from a postdisturbance state class to seedling/sapling and pole-sized state classes in dry forests
Precommercial thinning	Precommercial thinning in which vegetation and trees are removed to increase available resources for the most vital and valuable trees
Prescribed fire	Prescribed fire transitions of all severity levels
Partial harvest-medium and closed stand	Partial harvest (commercial thinning from below) in medium and closed stands
Partial harvest-open stand	Partial harvest (light maintenance thinning) to maintain an open canopy
Partial harvest-pole stand	Partial harvest in pole-sized stands, representing fuel treatments that do not generally produce a commercial product and are applied after the normal age of a precommercial thin
Replant-postdisturbance	Replanting with seedlings after harvesting or disturbance
Regeneration harvest	Regeneration harvest, including clearcut or shelterwood
Salvage harvest	Salvage harvest following stand-replacing disturbance, often followed by replanting
Selection harvest	Selection harvest that represents a periodic individual tree selection of mature stands

Model Sources

Most OR/WA forest STMs were adapted from existing models, built through the Interagency Mapping and Assessment Project (IMAP; http://ecoshare.info/projects/interagency-mapping-and-assessment-project/), with state classes and transitions informed primarily through local, expert opinion. Forest Service ecologists worked with silviculturists and other specialists to develop and parameterize most of the models (table 2.1). Preexisting models did not exist for several modeling zones (Washington Coast Range, Washington West Cascades, Washington North Cascades, and Washington Northeast) and, in those cases, new models were derived from existing models in neighboring regions. Model adaptation involved maintaining general model state classes but altering disturbance regimes and vegetation growth and development. Changes to growth and development transition probabilities were informed by simulations in FVS (see "Forest Vegetation Simulator Calibration"), and in some cases, vegetation state classes were omitted or added. Disturbance regimes were derived from both expert opinion and existing literature.

(a)



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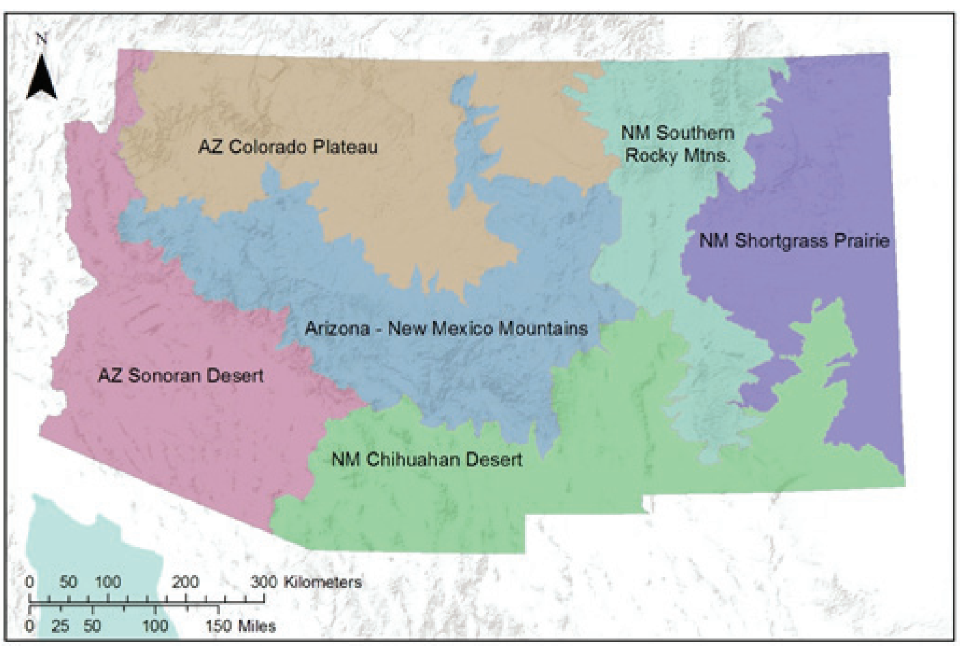

Figure 2.4—State-and-transition modeling zones in (a) Oregon and Washington (OR/WA), and (b) Arizona and New Mexico (AZ/NM). ILAP = Integrated Landscape Assessment Project.

Oregon and Washington (OR/WA) Arid Land Models

Modeled Vegetation Types

We adapted or constructed 19 STMs characterizing contemporary vegetation dynamics for all major nonforested systems (called arid lands) in OR/WA (table 2.4). The model set primarily covered rangeland vegetation types, but also included other nonforested systems such as subalpine meadows and montane/canyon shrublands. Detailed documentation of OR/WA arid land models is available on the Western Landscapes Explorer website (www.westernlandscapesexplorer.info)

Model Structure

Arid land state classes for OR/WA STMs were defined by combinations of cover types and structural stages (fig. 2.1b). Arid land cover types generally specified the dominant shrub and the dominant grass functional group(s). Alternative state classes within STMs were distinguished primarily by dominant life form (grass/shrub/tree) and composition of the herbaceous layer, which often indicated the level of degradation compared to historical plant communities. Grass functional groups included native grass (i.e., bunchgrass), robust grass (grass species that are fairly tolerant of grazing pressure, and therefore often indicate semidegraded conditions), and exotic grass, including cheatgrass and medusahead. Structural stages indicated the cover of the dominant life form, and included herbaceous stages, open, mid and closed shrub steppe, woodland phases I, II, and III (Miller et al. 2005) that characterized the level of site dominance by western juniper, and others (table 2.5).

Growth and succession were represented as deterministic transitions, and other processes such as wildfire, insects, and management activities were represented as probabilistic transitions (table 2.6). Wildfire probabilities were derived from the MTBS data for potential vegetation groups and three levels of exotic grass invasion (native, semidegraded, and degraded), reflecting changes in fire return interval with the introduction of exotic species (see "Wildfire Probabilities" and www.westernlandscapesexplorer.info). Management transitions were built into the models but were deactivated for the baseline ILAP FSO scenario.

Model Sources

The ILAP arid lands STMs were obtained from a variety of sources and were adapted for use across all OR/WA arid lands (table 2.4). Final models often incorporated information from multiple sources, including expert opinion, literature, and data. The STMs developed by USDA FS ecologists for the Blue Mountains of northeastern Oregon were the primary source for many ILAP arid land models, because the model set covered most of the major potential vegetation in eastern Washington and Oregon, and because the models were thoroughly documented by the original

Table 2.4—Descriptions of arid land vegetation types modeled in Oregon and Washington[a]

Potential vegetation type	Vegetation description
Bitterbrush-no juniper	Upland sites dominated by bitterbrush shrub steppe, often with some sagebrush
Bitterbrush-sand	Dry, sandy deposits of the Columbia River basin with bitterbrush, other shrubs, and needle-and-thread
Bitterbrush-with juniper	Upland sites dominated by bitterbrush shrub steppe, often with some sagebrush, and with potential for juniper encroachment
Bluebunch wheatgrass-Sandberg bluegrass	Low-elevation grasslands dominated by bluebunch wheatgrass and Sandberg bluegrass
Idaho fescue-prairie junegrass	Mid-elevation grasslands growing on productive plateau, canyon, and ridge sites, dominated by Idaho fescue
Low sagebrush-mesic, no juniper*	Upland sites with soils that are shallow or have a restrictive layer supporting low sagebrush steppe
Low sagebrush-mesic, with juniper*	Upland sites with soils that are shallow or have a restrictive layer supporting low sagebrush steppe and with potential for western juniper encroachment
Montane and canyon shrubland	Low- to mid-elevation montane shrub community with a diversity of species, including mountain snowberry, mallow ninebark, and many others
Mountain big sagebrush-no juniper*	Cool, moist sites with mountain big sagebrush that are outside of the range of juniper (most of Washington)
Mountain big sagebrush-with juniper*	Cool, moist sites with mountain big sagebrush steppe and high potential for juniper encroachment
Mountain mahogany	Rocky sites dominated by curl-leaf mountain mahogany
Rigid sagebrush and low sagebrush-xeric	Shallow soil sites supporting rigid sagebrush, low sagebrush and/or buckwheat
Salt desert shrub-lowland**	Lowland salt desert shrub flats, dominated by greasewood and salt grass and intermittently flooded, usually on Pleistocene lake beds
Salt desert shrub-upland**	Salt desert shrub sites above playas, dominated by saltbush and sparse grasses
Subalpine meadows-green fescue	Montane to subalpine grasslands occurring on mountain slopes and ridges, dominated by green fescue
Threetip sage	Highly productive three-tip sagebrush steppe sites
Western juniper woodland	Sites historically and currently considered western juniper woodlands
Wyoming big sagebrush-no juniper*	Warm, dry low- to mid-elevation sites supporting Wyoming big sagebrush steppe
Wyoming big sagebrush-with juniper*	Warm, dry mid-elevation sites supporting Wyoming big sagebrush steppe with enough moisture for potential juniper encroachment

[a] Model source is indicated with a superscript where: * = U.S. Department of the Interior, Bureau of Land Management, and ** = The Nature Conservancy of Nevada. Vegetation types without a superscript were developed by the U.S. Department of Agriculture, Forest Service.

Table 2.5—Major structural stages used in Oregon and Washington arid land state-and-transition models

Name	Description
Herbland	Grassland with few shrubs and trees (shrub cover less than 5 percent, tree cover less than 2 percent)
Open shrub	Open shrub steppe (shrub cover greater than 5 to 15 percent, tree cover less than 2 percent)
Mid shrub	Mid shrub steppe (shrub cover greater than 15 to 25 percent, tree cover less than 2 percent)
Closed shrub	Closed shrub steppe (shrub cover greater than 25 percent, tree cover less than 2 percent)
Depleted shrub	Depleted shrubland with high shrub cover and overgrazed herbaceous layer (shrub cover greater than 25 percent, herb cover less than 5 percent, tree cover less than 2 percent)
Woodland-phase I	Phase I juniper invasion, consisting of shrub steppe with shrubs or grasses dominant and scattered juniper (juniper cover greater than 2 to 10 percent)
Woodland-phase II	Phase II juniper invasion, with shrubs/grasses codominant with juniper (juniper cover greater than 10 to 20 percent)
Woodland-phase III	Phase III juniper invasion—juniper woodland with herbaceous understory and few shrubs (juniper cover greater than 20 percent)

Source: Shrub cover cutoffs are from Karl and Sadowski (2005), and woodland phases are from Miller et al. (2005).

model creators.[2] Models developed by Evers (2010) for the Malheur High Plateau region of southeastern Oregon were our primary source for sagebrush steppe STMs. Lastly, we adapted models from TNC of Nevada (Provencher and Anderson 2011) for salt desert shrub communities and from TNC of Idaho (Landscape Toolbox Project; www.landscapetoolbox.org) for three-tip sagebrush communities.

Arizona and New Mexico Forest and Woodland Models

Modeled Vegetation Types

We included five forested and seven woodland potential vegetation types in AZ/NM (table 2.7). The AZ/NM forest and woodland models were developed to examine ecosystem sustainability (Weisz et al. 2009) and for forest planning.

Model Structure

The AZ/NM forested and woodland STMs were similar in structure to the OR/WA models (fig. 2.1a), but contained fewer, more generalized state classes. Structural stages characterized canopy cover and tree size class, and sometimes included dominance and the number of canopy layers. Canopy cover classes included grass/forb/shrub (0 to 10 percent tree cover), open (10 to 30 percent tree cover), and closed

[2] Swanson, D. unpublished documentation.

Table 2.6—Probabilistic transitions used in Oregon and Washington arid land state-and-transition models

Transition type	Transition description
Natural transitions:	
Alternative succession	Recovery of herbaceous species (from exotics to natives) following rest from grazing
Browse	Browsing of palatable shrubs (e.g., bitterbrush and mountain mahogany)
Drought	A multiyear drought that, when combined with grazing, may result in degradation of rangeland condition
Insects and disease	General transition for insects or disease that can kill shrubs or trees
Insect-cyclical outbreak	Cyclical transition that simulates outbreaks of sagebrush-defoliating insects, occurring at an interval of 20 to 48 years
Natural recovery	Natural reseeding of native species. Species differ depending on the model
Severe drought	Severe drought that can kill sagebrush, occurring only once every 100 to 200 years
Wildfire-mixed-severity	Mosaic fire that results in a heterogeneous burn pattern
Wildfire-nonlethal	Surface fire (only modeled in phase III juniper woodlands with exotic grass)
Wildfire-stand-replacing	Stand-replacing wildfire that leads to early successional (herbaceous) state classes
Livestock grazing transitions:	
Graze-degrade	Grazing-related degradation that is unrelated to disturbance
Postdisturbance graze-degrade	Heavy grazing following major disturbance (within 2 years of fire or drought) that can lead to grazing-related degradation. The transition probability is 10-fold higher than graze-degrade
Maintenance graze	Maintenance grazing that does not affect vegetation composition or structure
Moderate graze	Moderate grazing that decreases herbaceous cover and alters succession but does not cause degradation. This can either accelerate or delay succession, depending on the model
Juniper cutting	Mechanical treatment (i.e., chain saw removal) of juniper
Shrub treatment	Mechanical treatment of depleted sagebrush, leading an open sagebrush stand
Prescribed fire	Prescribed fire treatment
Seed-exotic	Seeding of a degraded site with nonnative grass (i.e., crested wheatgrass)
Seed-native	Seeding of a degraded site with native species
Agriculture	Conversion of native vegetation to agricultural land

Table 2.7—Descriptions of forest and woodland vegetation types modeled in Arizona and New Mexico[a]

Potential vegetation type	Description
Forest models:	
Ponderosa pine forest*	Warm, dry forests with ponderosa pine, border pinyon, juniper species, and evergreen oak species
Ponderosa pine-evergreen oak*	Ponderosa pine, Apache pine, Chihuahuan pine, and evergreen oak forests
Mixed conifer with aspen*	Mixed-conifer forests occurring in small patches at a broad elevation range, with Douglas-fir, Engelmann spruce, white fir, and big tooth maple as the dominant species, and quaking aspen as an important seral component
Mixed conifer dry**	Dry mixed-conifer forest containing Douglas-fir and white fir associations and with Arizona fescue in the understory
Spruce-fir forest*	Engelmann spruce, subalpine fir, and corkbark fir forests, containing quaking aspen as an important seral species and twinflower as an important understory component
Woodland models:	
Pinyon-juniper woodland**	Pinyon-juniper woodlands occurring in either mild or cold climates, with Arizona orange, rubber rabbitbrush, Apache plume, and Toumey oak as the dominant shrub species
Pinyon-juniper sagebrush**	Pinyon-juniper woodlands occurring as mix of trees and shrubs in cold climates, typically above the Mogollon Rim in northern Arizona and New Mexico, with sagebrush or blackbrush in the shrub layer
Pinyon-juniper evergreen shrubland**	Pinyon-juniper woodlands occurring below the Mogollon Rim in southern Arizona and New Mexico with crucifixion thorn, Sonoran scrub oak, pointleaf manzanita, alderleaf mountain mahogany, and gray oak as the dominant evergreen species
Pinyon-juniper grassland**	Pinyon-juniper woodlands occurring in either cold or mild climates, with understory typically dominated by grass species, and less abundant shrubs and trees occurring as individuals or occasionally in small groups
Juniper grassland**	Woodlands comprised of one or several juniper species (e.g., oneseed, alligator, and Pinchot's juniper), with an understory dominated by grass species
Madrean pine-oak woodland*	Woodland bounded by semidesert grasslands at the lowest elevations and montane forests at higher elevations. Emory oak, Arizona white oak, Mexican blue oak, and gray oak are the common oak species, and Douglas-fir, ponderosa pine, and Chihuahuan pine are the common conifer species
Madrean encinal woodland**	Oak woodlands occurring in isolated mountain ranges grading into semidesert grasslands at lower elevations and pine-oak woodlands at higher elevations, with Emory oak, Mexican blue oak, Arizona white oak, and gray oak as the common species

[a] Models described below were adapted and calibrated by the U.S. Department of Agriculture Forest Service Southwest Region. Original model source is indicated by a superscript where: * = The Nature Conservancy, and ** = LANDFIRE.

(>30 percent tree cover). Tree size classes included seedling/sapling (<13 cm diameter at breast height [DBH]), small (13 to 25 cm DBH), medium (25 to 50 cm DBH), large or very large (>50 cm DBH). Where canopy layers were included, the number of canopy layers was either single storied or multistoried. Transitions represented growth and development, natural disturbance, and management transitions (table 2.8). The AZ/NM forest and woodland models did not contain any deterministic transitions; all succession transitions were calibrated using FVS runs based on Forest Inventory and Analysis (FIA) plot data (Weisz et al. 2010) and treated as probabilistic transitions. Wildfire probabilities for some models came from field plot data across the national forests in the region and expert opinion, and for others were derived using the Fire and Fuels Extension (FFE) of FVS (Reinhardt and Crookston 2003). Wildfire transition multipliers describing interannual variation in fire weather were derived from an analysis of the MTBS data (see "Interannual Variation Using Monte Carlo Multipliers").

Model Sources

Individual national forests in AZ/NM developed their own versions of forest and woodland models based largely on LANDFIRE and TNC efforts to describe potential natural vegetation and vegetation dynamics in AZ/NM (table 2.9). The models used for ILAP were adapted from those model variants developed at each national forest in AZ/NM.

Arizona and New Mexico Arid Land Models

Modeled Vegetation Types

The Integrated Landscape Assessment Project compiled and developed models to simulate contemporary vegetation dynamics for all arid land (nonforested) potential vegetation in AZ/NM (table 2.9). The AZ/NM arid lands encompassed a broad range of temperate life zones, from very dry lowland desert scrub to upland chaparral and montane meadow systems. Detailed documentation of AZ/NM arid land models is available on the Western Landscapes Explorer website (www.westernlandscapesexplorer.info).

Model Structure

Model structure varied considerably, depending on model source (table 2.9). In general, cover types indicated the dominant life form(s), such as perennial grass, forb, sagebrush, or juniper. Structural stages defined the cover range in each of the three major vertical structure layers: grass/forb, low to mid shrub, and tall shrub/tree. Cover ranges of each vertical layer were specified as absent (zero percent), sparse (1 to 10 percent), open (>10 to 25 percent), mid (>25 to 60 percent), and

Table 2.8—Probabilistic transition types used in Arizona and New Mexico forest and woodland state-and-transition models

Transition type	Transition description
Natural transitions:	
Growth and succession	Forest succession and growth transition where models are calibrated by the Forest Vegetation Simulator
Nonlethal fire	Wildfire resulting in less than 25 percent top kill
Mixed-severity fire	Wildfire resulting in 25 to 75 percent top kill
Stand-replacing fire	Wildfire resulting in 75 to 100 percent top kill
Wildfire—low conditions	Wildfire under low burning conditions, as modeled by the Fire and Fuels Extension of the Forest Vegetation Simulator (FFE-FVS)
Wildfire—moderate conditions	Wildfire under moderate burning conditions, as modeled by the FFE-FVS
Wildfire—high conditions	Wildfire under high burning conditions, as modeled by the FFE-FVS
Insects and disease	Transition caused by forest insect and disease agents
Wind/weather/stress	Transition caused by wind, weather, and stress
Alternative succession	Transition resulting from a lack of disturbance
Elk herbivory	Elk browsing, according to summer range mapping
Natural regeneration	Natural regeneration and recruitment by species
Management transitions:	
All initiate regeneration	All management activities that initiate tree regeneration
All prescribed burning	All management activities related to prescribed burning, typically low severity
All fuel treatments	All fuel treatment management activities
Intermediate thin—low basal area	Precommercial thinning of intermediate intensity to a low basal area that takes place as part of the maintenance of a stand in preparation for a commercial harvest
Intermediate thin—moderate basal area	Precommercial thinning of intermediate intensity to a moderate basal area that takes place as part of the maintenance of a stand in preparation for a commercial harvest
Intermediate thin—41 cm limit	Precommercial thinning of intermediate intensity of trees less than 41 cm (diameter cap) that takes place as part of the maintenance of a stand in preparation for a commercial harvest
Shelterwood	Harvest in which some large trees are retained to provide some shade and other protection to remaining trees. Trees may be removed in several harvests over a 10- to 15-year period to encourage regeneration of medium to low shade-tolerant species
Free thin to target basal area	Free thin of all sizes to a target basal area to reduce risk and allow growth in the residual, by favoring the removal of suppressed, defect, and excess trees beyond the target basal area of each size class
Thin from below to target basal area	Thin from below to a target basal area. Trees of sizes below the target size class are thinned from the stand. Typically results in a state class change to a more open larger size class

Table 2.8—Probabilistic transition types used in Arizona and New Mexico forest and woodland state-and-transition models (continued)

Transition type	Transition description
Thin under 41 cm to target basal area	Thinning trees under 41 cm in diameter to achieve a target basal area. Typically results in a state class change to a more open larger size class, where repeat entries result in a closed, single-storied stand
Group selection/ matrix thin	Group selection with a matrix thin in which groups of trees are removed from an area over a period of several entries and trees in the surrounding matrix are thinned to a target basal area
Shelterwood seed cut to target basal area	Shelterwood harvest to achieve a target basal area in which retained trees also provide seed for regeneration
Clearcut-legacy trees	Clearcut in which some trees are retained (also called clearcut with reserves)
Clearcut-coppice	Clearcut to produce even-aged stand of species that regenerate by sprouting from the roots or stem base (coppice)
Plant seedlings	Replant with seedlings following harvest or disturbance
Prescribed fire	Application of controlled fire
Thin under 23 cm to target basal area	Thinning trees under 23 cm in diameter to achieve a target basal area

closed (>60 percent). Processes modeled in AZ/NM arid land models included a range of natural processes, livestock grazing practices, and management activities (table 2.10).

Model Sources

Southwestern arid land models came from a variety of sources (table 2.9). Eleven new models for the major vegetation types in the Chihuahuan Desert and Short-grass Prairie modeling zones (fig. 2.4b) were constructed by Natural Heritage New Mexico. Models for many Colorado Plateau, Sonora-Mojave, and interior chaparral types were adapted from The Nature Conservancy in Nevada and Arizona. LAND-FIRE reference condition models (www.landfire.gov; Rollins 2009) were the basis for many STMs in the Sonoran desert and other regions. In most of these cases, substantial additions were made to the models to include contemporary state classes and transitions not represented in the reference condition models, such as tree and exotic grass invasion (see documentation for details).

Wildfire is an important natural disturbance in nearly all ILAP models and differs regionally. We developed methods to use wildfire records from the Monitoring Trends in Burn Severity program (MTBS).

Model Parameters

Wildfire Probabilities

The STM framework provided a means to customize transitions to region-specific disturbance probabilities. Wildfire is an important natural disturbance in nearly all ILAP models and differs regionally. We developed methods to use wildfire records

Table 2.9—Descriptions of arid land vegetation types modeled in Arizona and New Mexico

Potential vegetation type	Vegetation subclasses	Description
Intermountain salt scrub*		Warm, dry desert scrub environments dominated by saltbush, greasewood, and other shrub species, often occurring in alkaline or calcareous soils above playas
Colorado Plateau/ Great Basin grassland*	No juniper	Xeric Colorado Plateau grasslands characterized by drought-tolerant bunchgrasses such as blue grama and scattered shrubs
	Juniper potential	Xeric Colorado Plateau grasslands characterized by drought-tolerant bunchgrasses and scattered shrubs, occurring adjacent to juniper woodlands
Sagebrush shrubland*		Dry, open sagebrush shrublands with scattered semiarid grasses such as blue grama and Indian ricegrass, sometimes with rabbitbrush
Sand sheet shrubland**		Sand sheets of the Colorado Plateau characterized by joint fir species at low cover, with sparse grasses
Interior chaparral***		Mid-elevation chaparral with a diverse community of shrubs, including oak, manzanita, ceanothus species, and others
Mountain mahogany mixed shrubland**		Mid-elevation rocky outcrops dominated by curlleaf mountain mahogany, sometimes with sagebrush, oak, manzanita, and ceanothus species
Gambel oak shrubland**		Mid- to upper-elevation shrublands with Gambel oak as the dominant overstory species and serviceberry or sagebrush forming the understory
Montane/subalpine grassland **		Montane and subalpine grasslands characterized by Arizona fescue, mountain muhly, Kentucky bluegrass, and other grasses
Sonora-Mojave creosote-bursage desert scrub*	Sand	Sonoran dunes and sand flats dominated by white bursage, big galleta, and creosote bush
	Valleys	Creosote bush—white bursage desert scrub found in valleys of the Sonoran desert
	Mountains	Creosote bush and white bursage desert scrub found on low-elevation mountain slopes in the Sonoran desert
Sonoran paloverde-mixed cactus desert scrub* **	Mountains	Mountain slopes with paloverde, desert ironwood, saguaro and other cacti
	Bajada/foothills	Alluvial fans and pediments with paloverde, desert ironwood, saguaro and other cacti
Sonoran mid-elevation desert scrub**		Mixed desert scrub consisting of varying assemblages across its range, occurring at elevations between paloverde-mixed cactus and chaparral vegetation types above the Sonoran desert

Table 2.9—Descriptions of arid land vegetation types modeled in Arizona and New Mexico (continued)

Potential vegetation type	Vegetation subclasses	Description
Sonoran mixed salt Desert scrub*		Sonoran desert valleys dominated by saltbush, with sagebrush, creosote bush, or white bursage as codominants
Mojave mid-elevation desert scrub*		Mixed desert scrub dominated by blackbrush but including other codominants, occurring between the creosote-bursage desert scrub and montane woodlands above the Mojave desert
Chihuahuan desert scrub		Chihuahuan desert scrub occupying a variety of land forms, dominated by creosote bush, tarbush, and acacia
Chihuahuan salt desert scrub		Chihuahuan desert scrub found in basin bottoms where salts and other minerals have accumulated in the soil, characterized by saltbush and other species
Semidesert grassland	Piedmont	Semidesert grasslands on coalesced alluvial fan piedmonts, characterized by black grama, bush muhly, and fluffgrass
	Foothill	Semidesert grasslands on colluvial foothill slopes, dominated by sideoats grama, curlyleaf muhly, New Mexico feathergrass, and bullgrass
	Lowland	Semidesert grasslands found in lowland basins and playas, with tobosagrass, burrograss, alkali sacaton, big sacaton, or vine mesquite
	Sandy plains	Semidesert grasslands typical of sand sheets, with black grama, dropseed, and muhly species
Great Plains grassland	Mixed-grass prairie	Southern Great Plains grasslands dominated by warm-season grasses, including sideoats grama, blue grama, switchgrass, and bluestem
Shortgrass prairie	No juniper	Western and southern Great Plains grasslands characterized by blue grama, buffalograss, New Mexico feathergrass, James' galleta, and other grama species
	Juniper potential	Western and southern Great Plains grasslands found on the edges of the prairie adjacent to juniper woodlands
Sandsage		Scrubland vegetation typical of sandy, wind-deposited soils and characterized by sand sagebrush
Shinnery oak		Scrubland vegetation typical of sandy, wind-deposited soils and dunes, with shinnery oak as the dominant shrub

[a] Model source is indicated as a superscript, based on the primary source of the model, including: *The Nature Conservancy (TNC) – Nevada, ** LANDFIRE , and *** TNC – Arizona. Vegetation types with no superscript indicate models that were built specifically for the project.

Table 2.10—Probabilistic transition types used in Arizona and New Mexico arid land state-and-transition models

Transition type	Transition description
Mosaic fire	Mosaic fire that results in a patchy burn pattern or partial mortality
Stand-replacing fire	Wildfire that results in high mortality of shrubs or trees
Surface fire	Wildfire that burns the herbaceous or shrub layer under tall shrubs or trees
Drought	Decrease in precipitation for 1 or more years
Exotic invasion	Invasion by exotic species, including cheatgrass, red brome, buffelgrass, and others
Freeze	Freezing temperatures that kill frost-intolerant species, such as some cacti
Increased winter precipitation	Increased winter precipitation that causes increased shrub growth compared to grasses
Natural recovery	Natural recovery of the herbaceous layer from exotic to native species
Oak expansion	Expansion of shinnery oak into grasslands
Perennial establishment	Invasion by and growth of perennial grass species
Prairie dog establishment	Prairie dog colony establishment
Prairie dog extinction	Prairie dog colony extinction
Recreation—off-highway vehicles (OHV)	Recreation by OHV that results in vegetation and soil disturbance
Shrub encroachment	Invasion by shrubs, particularly mesquite, into grasslands
Tree encroachment	Encroachment by juniper into shrublands or grasslands
Very wet year	Periodic flooding of low-lying basins
Weather and stress	Abiotic stress
General livestock grazing	A general transition for livestock grazing
Excessive grazing	Excessive livestock grazing that can degrade range condition
Managed grazing	Managed livestock grazing that alters succession
Graze-mesquite invasion	Invasion of grasslands or shrublands by mesquite owing to livestock grazing
Native grazing	Grazing by native wildlife
Abandoned agriculture	Abandonment of agricultural planting
Herbicide treatment	Herbicide applied singly or in combination with mowing or seeding
Prescribed fire	Application of controlled fire
Root plowing	Conversion of native species to agriculture
Tree lopping	Chainsaw removal of trees
Tree thinning	Thinning of trees, occurring singly or in combination with herbicide and seeding

from the Monitoring Trends in Burn Severity program (MTBS) (Eidenshink et al. 2007, www.mtbs.gov) as a basis for many wildfire probabilities. We used the MTBS data to calculate wildfire probability in southwestern Oregon, all eastern OR/WA forests, and all OR/WA arid lands. The MTBS data set included burn perimeters and burn severity classes for all fires over 405 ha in the Western United States over a 25-year timespan (1984 to 2008). To obtain wildfire probabilities, MTBS burn perimeter polygons were overlaid in a geographic information system (GIS) with our potential vegetation map. We grouped potential vegetation into groups representing similar fire regimes (e.g., dry forests, moist forests, semidesert shrub steppe, mesic shrub steppe) to increase our wildfire sample size in each category. We then calculated the number of hectares burned each year for each potential vegetation group and calculated annual fire probabilities by dividing the area burned in each potential vegetation group by the area in the group and the number of years in the fire record. These annual values became the wildfire probabilities in our models. For OR/WA arid lands, we included an additional spatial data layer containing exotic annual grass to account for variation in fire return interval from invasion by exotic species. This exotic grass layer was developed from the current vegetation layer (see "Spatial Data") and was categorized into low (0 to 10 percent), moderate (10 to 25 percent), and high (>25 percent) exotic grass cover. We combined the exotic grass data layer with potential vegetation groups and MTBS fire perimeters to obtain fire probabilities for each combination of potential vegetation group and exotic grass level.

The MTBS data set provided a tractable improvement to reliance on expert-opinion derived wildfire transition rates. However, there were limitations to the MTBS data and our derivatives. Fire-severity information was available as part of the MTBS data but was not used for this study owing to inconsistencies with data reporting and image classification. Fire probabilities derived from MTBS data likely underestimated fire activity in an area, as some fires may not have been reported throughout the 25-year record, and no fires under 405 ha were included in the data set. We did not have a sufficient sample size in western OR/WA to conduct similar analyses. However, we did not consider this to be critical, because fire intervals are quite long in these moist forest types. MTBS data were not used to obtain fire probabilities for forests and woodlands in AZ/NM because local analysts were already able to provide inventory plot data, local severity mapping, and FFE-FVS to estimate wildfire effects and probabilities. In addition, the MTBS data set for arid lands of AZ/NM appeared to be incomplete and therefore were not used to derive fire probabilities. Note that the MTBS data produced wildfire probabilities that assumed a continuation of fire suppression at current rates, and thus fire suppression was incorporated into the models inherently through the probabilities rather than explicitly as a separate disturbance or probability multiplier.

Forest Vegetation Simulator Calibration

We used FVS to calibrate forest succession transitions for all forest and woodland STMs in AZ/NM and some forest models in OR/WA. The FVS is an individual tree growth and yield model system that predicts distance-independent tree mortality and tree growth in terms of tree diameter, height, crown ratio, and crown width over time (Dixon 2006, Hoover and Rebain 2011, Moeur and Vandendriesche 2010). The FVS model system uses empirically derived life-history equations to simulate stand dynamics over time and has been used to support management decisionmaking at scales from the local project-level to regional assessments (Crookston and Dixon 2005). Forest Inventory and Analysis (USDA FS 2012) plot data provide long-term data collected from private and public forest land in the United States and served as reference data for input to FVS.

Transition Multipliers

Transition multipliers are values that increase or decrease probability values simultaneously for all transitions of one type. Transition multipliers operated at modeling strata and regional levels to simulate variation owing to vegetation type and land ownership or management. They were used to (1) deactivate a transition completely (multiplier equals zero), (2) decrease a transition rate (multiplier between zero and one), or (3) increase a transition rate (multiplier greater than 1). For the ILAP default FSO scenario, all management transitions were deactivated using a multiplier of zero, but for other scenarios, such as the resilience scenario developed for focus area analyses, multipliers were used to adjust the average probability of a management event, such as a thinning from below, to levels corresponding to management prescriptions in the scenario. Transition multipliers were linked to our modeling strata via the ownership-management layer, so management transitions could be adjusted differently for each ownership-management combination. In addition, transition multipliers could be used to generate trends for selected transitions.

Interannual Variation Using Monte Carlo Multipliers

The modeling framework provided a means to simulate year-to-year variability in STM simulations using Monte Carlo multipliers (MCMs). We incorporated interannual model variation for fire weather, insect outbreaks, and severe drought (in a small number of OR/WA arid land models) to simulate stochastic or cyclic variation in fire weather, insect outbreaks, and other disturbances by creating a stream of randomly generated multiplier sequences. The interannual multipliers were applied to individual model simulations and selected transition types, and augmented the software's built-in variability. The MCM sequences were generated using the MCM builder tool in VDDT (ESSA Technologies Ltd. 2007) for the desired number of model simulations.

Fire weather is a major driver of fire behavior (Agee 1993) and is one of the determinants of total land area burned in any year, as well as fire severity. We assumed fire weather was a random occurrence for this project, and generated an annual sequence of multipliers for each Monte Carlo simulation. Our multipliers were based on the MTBS fire occurrence data, assuming that 80 percent of years were normal fire years, 15 percent of years were high fire years, and 5 percent of years were severe, in terms of area burned.

Insect outbreaks were also modeled using MCMs but were assumed to occur as semirandom occurrences in OR/WA dry forest types and sagebrush arid land models. The onset of an outbreak was determined randomly or within a range of years since the previous disturbance, but once an outbreak occurred, its behavior was controlled by the boundaries established in the MCM multiplier streams. The MCM streams generated for insects were constrained by (1) the probability of a low to normal, high, and severe outbreak year occurring; (2) minimum and maximum lengths of outbreaks; (3) minimum and maximum length of time between outbreak peaks; (4) time since the peak of the previous outbreak; and (5) length of time it takes to develop and then diminish outbreak conditions. In this way, the initiation, length, and severity of an outbreak were randomized but informed by the previous outbreak occurrence. The OR/WA forest models included mountain pine beetle, western spruce budworm, and Douglas-fir tussock moth outbreaks, and multipliers varied among modeling zones. The OR/WA arid lands modeled sagebrush defoliating insect outbreaks (such as Aroga moth) and severe drought using MCMs. Interannual variation was not considered for insect outbreaks in the AZ/NM models, although insect/disease transitions were included in some models.

Spatial Data

The STM modeling framework required several spatial data layers, described below. These spatial layers were used to form modeling strata, the spatial basis of our modeling, and provided a means to quantify abundance and distribution of current vegetation to initialize the STMs. Modeling strata were formed by the overlay of three spatial layers, including potential vegetation, ownership-management, and watershed, and one STM was run for each. State-and-transition models were run for about 30,000 modeling strata across the entire project area. The initial vegetation conditions (vegetation composition and structure at time step zero) assigned to each modeling stratum came from the current vegetation layer attributes (see “Data Processing”).

Modeling Zones

We divided the four-state study area into 18 modeling zones (fig. 2.4), representing manageable spatial units for mapping and modeling. In AZ/NM, our mapping zone boundaries corresponded to the Multi-Resolution Landcover Consortium mapzones (Homer et al. 2004), modified to match watershed boundaries. In OR/WA, our modeling zones corresponded to level III ecoregions (Omernik 1987), with the boundaries modified to follow watershed boundaries (see "Watershed").

Potential Vegetation

Potential vegetation maps identified which STMs applied to each pixel in the modeled area. We used existing datasets where they were available and constructed our own where they were not. Our potential vegetation maps were categorical raster maps, and represented a vegetation classification that corresponded with the STMs. Within OR/WA, these vegetation concepts are called potential vegetation types, which represent environmental zones where vegetation tends to exhibit consistent dynamics through time (e.g., similar growth rates, species composition through succession, disturbance regimes). Within AZ/NM, the vegetation concepts illustrated in the potential vegetation map correspond with potential natural vegetation types refined by historic fire regimes or the equivalent of biophysical settings or ecological systems (Winthers et al. 2005).

In OR/WA, we used existing maps of potential vegetation for some locations, including maps of plant association groups (western OR), and vegetation zones (WA forests). For the parts of OR/WA that lacked potential vegetation data, and for all of AZ/NM, we mapped vegetation plots using a modeling technique called Random Forest Nearest Neighbor (RFNN) (Crookston and Finley 2008). RFNN links mapped pixels to forest inventory plots. It is one type of imputation, which is a general term for the technique of filling in missing values in a dataset with known values in the same dataset (Eskelson et al. 2009). RFNN uses a random forest model that illustrates the relationship between vegetation composition and explanatory environmental variables to match the best plot to each pixel. Explanatory variables fall within three broad categories: soil, climate, and topography. In OR/WA, we used Current Vegetation Survey plots from the USDA FS, LANDFIRE, plots from the USDI BLM, and other local supplemental datasets. In AZ/NM, we used all of the plots that were used for existing vegetation maps, plus supplemental plot data from the US Department of Defense. Each plot was assigned to a potential vegetation type through either a crosswalk or a keying process that assigned potential vegetation to plots based on plant community composition. Potential vegetation maps were drafted multiple times, based on expert review feedback. RFNN model accuracy was assessed through error matrices built comparing cross-validated model predictions with actual potential vegetation values for the plots. Map accuracy was further assessed by expert review. In some areas in AZ/NM, mapped potential vegetation data from national forests was layered on top of our modeled potential vegetation surface, a procedure recommended by our expert reviewers to improve local credibility and utility to forest managers on national forests.

Current Vegetation

Current vegetation maps provided information on state classes within each potential vegetation type. As with the potential vegetation maps, we used existing maps where available, otherwise we developed new maps. Current vegetation maps were raster layers, where each pixel value corresponded with a single vegetation survey plot identifier. Summarized variables from those plots were attached to the grid attribute table, yielding a multivariate map. These variables described vegetation composition and structure and included four variables for forests and 22 for arid lands.

For the forests of OR/WA, maps of current vegetation (map date 2006) were available through the Landscape Ecology Modeling Mapping and Analysis (LEMMA) team within the USDA FS (http://lemma.forestry.oregonstate.edu/). These maps were created using the gradient nearest neighbor imputation mapping technique (GNN) (Ohmann and Gregory 2002). GNN differs from RFNN only in the statistical method it uses to link plots to pixels, relying on an ordination model describing vegetation-environment relationships (canonical correspondence analysis). These maps were built from the same explanatory variables used for potential vegetation, with the addition of LANDSAT spectral reflectance. All maps from the LEMMA team are extensively assessed for accuracy. Within AZ/NM, we created maps of current forest vegetation RFNN. These maps were functionally identical to the GNN maps from R6 for the purpose of initializing STMs. The environmental explanatory variables used for our new maps were similar to those used by the LEMMA team, but our forest inventory plot data set was smaller and included only FIA annual plots, following procedures developed through the Nationwide Forest Imputation Study (Grossmann et al. 2009).

We also built current vegetation maps for the arid lands of AZ/NM and eastern OR/WA using the RFNN technique. The environmental explanatory data used to inform these models was the same as for forests, but the plot datasets differed. Within OR/WA, we obtained plot data primarily from the public LANDFIRE plot reference database, and the USDI BLM, with a few other minor sources. Within AZ/NM, we relied on LANDFIRE as well, supplemented by data from the USDI BLM and New Mexico Natural Heritage program. All new current vegetation maps were evaluated for statistical accuracy using cross-validated predictions for the plots and expert review. In AZ/NM, woodlands were mapped with arid lands instead of forests.

Ownership-Management

The ownership-management spatial layer differentiated landowners and management goals, allowing different treatment levels to be applied to different management areas within a single ownership. To derive the ownership-management allocation layer for OR/WA, we collected available management plans and layers from federal and state land management agencies. We used six ownership/stewardship categories: USDI BLM, USDA FS, other federal lands (e.g., National Park Service, Department of Defense, U.S. Fish and Wildlife Service), state, private, and tribal. Within each ownership category, management plans were matched to mapped management data. The management data were classified into visual resource classes that capture the visual management objectives for a particular portion of land. We defined our visual resource objective based on categories and definitions created by the USDI BLM, and added one category that reflected an additional management class (table 2.11). We intersected the geographic boundaries of the land ownership types with the boundaries of management categories to construct the comprehensive set of ownership and management intensity classes represented in the ownership-management layer. This process generated 30 combinations of ownership-management in OR/WA. The ownership-management layer for AZ/NM was compiled using the protected area database for the United States (www.protectedlands.net/padus/preview.php) stewardship codes combined into the same categories used in OR/WA.

Watershed

Watersheds divide the landscape into commonly used units for reporting model results. Watersheds were defined as 5^{th}-field or 10-digit hydrologic unit codes (HUCs), and were acquired from the standardized watershed boundary data set (WBD) (USGS and USDA NRCS 2011). The WBD uses the federal standards for delineation of hydrologic unit boundaries for delineating watershed boundaries. Watersheds generally represent drainage areas between 20 000 and 100 000 ha. In OR/WA, there are about 950 watersheds with an average size of 45 023 ha. In AZ/NM, there are about 1,020 watersheds with an average size of 60 560 ha.

Mask

A mask layer was applied to exclude lands not modeled, including riparian, barren, water, urban, and developed agricultural areas. The 2008 Pacific Northwest Gap Analysis Project (GAP) (available at www.oregon.gov/DAS/EISPD/GEO/sdlibrary.shtml) land cover map was used to create the mask as it was available across all four states, had been recently updated, and used a consistent classification system.

Table 2.11—Management categories used to define management objectives for the ownership-management spatial data layer[a]

Management category	Objectives	Description
Category 1	Protection/ preservation	VRC 1 protected lands (primarily national forests)—the objective was to preserve the existing character of the landscape with no management or very limited management activity. Examples include wilderness areas and congressional reserves.
Category 3	Retention	VRC 2 lands—the management objective was to retain the existing character of the landscape. The level of change to the landscape was low and management activities were minimal. Any changes repeated the basic elements of form, line, color, and texture found in the predominant natural features of the characteristic landscape. Examples include late-successional reserves and state natural areas.
Category 4	Partial retention	VRC 3 lands—the objective was to partially retain the existing character of the landscape. The level of change to the characteristic landscape was moderate, and changes repeated the basic elements found in the predominant natural features of the landscape. Examples include state wildlife management areas and federal adaptive management areas.
Category 5	Modification	VRC 4 lands—allows management activities that cause major modifications of the existing character of the landscape. The level of change to the characteristic landscape can be high. Examples include federal matrix lands, and general ecological management on state lands.
Category 6	Modification private	This category was outside of the scope of the USDI BLM definitions and was used to define management on private industrial lands. These lands typically are heavily managed with high volumes of timber harvest and other commercial activities.

[a] Visual Resource Class (VRC) definitions developed by the U.S. Department of the Interior, Bureau of Land Management (USDI BLM) are largely used as a basis to create the categories. USDI BLM categories 1 and 2 were combined to simplify the classification system.

After extracting vegetation categories that were not modeled, the grid was resampled from a 30- to a 300-m cell size to remove the excessive fragmentation caused by linear features, such as riparian areas and highways. Small polygons <405 ha were removed, then resampled back to 30 m.

Several steps were performed to integrate spatial layers, calculate initial vegetation conditions, and set up STM model runs in Path, a process we refer to as the "data rollup."

Data Processing

To run the STMs developed for the project area, we developed a process to handle large data sets and run many STMs simultaneously. The process was specifically designed to allow flexibility in updating any of the data layers that form the foundation of the modeling effort, as updated vegetation maps or new data become available.

Several steps were performed to integrate spatial layers, calculate initial vegetation conditions, and set up STM model runs in Path, a process we refer to as the “data rollup.” We developed several software tools to automate the data rollup process (available online at www.westernlandscapesexplorer.info). The first step employed a series of clipping, data format conversion (where needed), and combining operations customized in the GIS data rollup tool. This step assigned a modeling stratum, current vegetation cover, and current vegetation structure class to each raster cell. The second step used the Data Rollup Query Tool to transform the modeling strata and associated current vegetation conditions from separate spatial data sets into the tabular summary required by the modeling software. The tool used a rule set of if/else statements to assign all modeled pixels in the landscape to a model state class in the applicable STM based on vegetation cover, dominant species, and other criteria. The result was a current vegetation map classified into model state classes.

In OR/WA forest models, the process of assigning mapped pixels to model state classes involved an additional step not required for OR/WA arid land models or AZ/NM models. First, each pixel was assigned a cover type based on the importance value of the dominant tree species, then assigned a structural stage based on tree density. In many cases, not all the resulting cover and structure combinations occurred in the STMs. These pixels were subjected to a reclassification scheme until all pixels were assigned to existing STM state classes.

The intersection of watershed, ownership-management, and potential vegetation often yielded modeling strata with very little area. Because STMs do not yield reliable results when applied at very small spatial scales, some additional processing was needed to handle the smallest modeling strata. We set our minimum modeling stratum size to 405 ha and developed a two-tiered strategy to cope with small modeling strata. Wherever possible, we reassigned pixels from small modeling strata to larger, similar modeling strata within the same watershed. To reassign pixels to other modeling strata, we made decisions using two relative similarity indices, one for potential vegetation and another for ownership-management categories. The two indices ranked all pair-wise combinations of each category with respect to their relative similarity in multiple dimensions. The potential vegetation similarity index reflected overlap in elevation of the potential vegetation, species compositional similarity from the current vegetation map, and similarity in moisture regime (e.g., dry vs. mesic). The ownership-management similarity index was constricted in a similar fashion, scoring pair-wise similarity from information on management activities that may or may not occur within each ownership-management category. When there was no reasonable reassignment, we removed the small modeling strata from our analysis. This size constraint process is optional, and minimum size can be altered. All functionality necessary to create and use a similarity index for lumping modeling strata was contained within the LumpStrata script programmed using R statistical software (R Development Core Team 2008).

Once current vegetation pixels were properly assigned to state classes, there were logistical issues of moving initial conditions for hundreds or thousands of modeling strata into the Path software. To facilitate this process, we developed the Get into Path (GIP) tool to automatically upload the STMs, initial conditions, and transition multipliers for each modeling unit into the Path software. The result was a set of organized parameter scenarios customized for ILAP modeling that were ready for simulation (all of the tools described in this section are available at www.westernlandscapesexplorer.info).

Model Execution

All models were run using the baseline FSO scenario that represented vegetation dynamics under no management activities except continued fire suppression. In this scenario, all management transitions were deactivated. Grazing transitions were included in arid land and dry forest models, where applicable. In addition, some areas were analyzed with multiple scenarios, which included management actions such as timber harvesting, prescribed fire, seeding, and other treatments (see “Example Outputs”). Thirty Monte Carlo simulations were run for each modeling stratum, using a model resolution of 4 ha.

All models were run using the baseline FSO scenario that represented vegetation dynamics under no management activities except continued fire suppression. In addition, some areas were analyzed with multiple scenarios, which included management actions such as timber harvesting, prescribed fire, seeding, and other treatments.

Model Output

Output Summarization

Output from the Path model runs was exported outside of the Path interface as comma separated variable (csv) text files. Each of these files was quite large, sometimes having one or more rows for each combination of state class/transition, time step (150 or 300 years for arid lands and forests, respectively), and Monte Carlo simulation (30). We developed the SumPath summarization package programmed in the R statistical software package (R Development Core Team 2008) to calculate summary statistics (mean, median, minimum, and maximum area) for each state class and transition at each time step, summarized over the Monte Carlo runs. Summaries for each modeling stratum were unified in a single state class file and single transition file for each modeling zone.

Output Visualization

Model output can take the form of graphs, tables, and maps. For any output format, data can be summarized across the entire area of interest, by potential vegetation, or by ownership. We used several basic visualization tools to summarize output for all modeling strata.

Pivot charts—

If the data set is small enough, output from the model runs can be assessed using a Microsoft® Office Excel-based graphing tool developed for the project (available at www.westernlandscapesexplorer.info). The graphing tool allows an average Excel user to graph the results of the summarized text files in Excel based on modeling zone, potential vegetation, ownership-management, and watershed. It also allows the user to group state classes into fewer categories with meaningful interpretations, such as broad forest structural groups or levels of exotic grass invasion. The tool builds a pivot table in Excel, and a graph is automatically generated based on this table. This prebuilt graph can be changed to suit the user's needs based on any of the information available in the pivot table. The Excel tool currently graphs changes in state classes but not changes in transitions over time.

Maps—

Spatial display of model output can occur at a variety of scales, including the scales of modeling strata and whole watersheds (fig. 2.5). However, our process generated STM results that were spatially aggregated into modeling strata at a minimum; our maps were not meant to be used on a pixel-by-pixel basis (see "Limitations"). One method of displaying output spatially was in the form of risk maps, depicting conversion of areas to undesirable state classes such as dense, dry forest types or exotic-dominated grasslands. Risk maps were constructed to show the accumulation of cover or structural type(s) of interest through time. Maps also displayed temporal information in different ways, either depicting model results for a single year or indicating the magnitude and direction of change from the beginning to the end of the simulation.

Example Results

Although STM results are available across the four-state study area, the following examples illustrate STM projections and interpretations from a subset of the forest and arid land output in OR/WA. The forest example is taken from the Washington East Cascades (WEC) modeling zone, an area of approximately 2 million ha. The arid land example is from the southeastern Oregon modeling zone and spans about 5.3 million ha.

Washington Dry Forest Example

In this example (fig. 2.4), we ran two scenarios with 30 simulations each: (1) an FSO scenario where all types of management other than fire suppression did not occur, and (2) a resilience scenario where the intent was to actively manage drier forest types assuming fire suppression continued into the future. We were unable to obtain data on resilience-promoting activities in the area and therefore used our

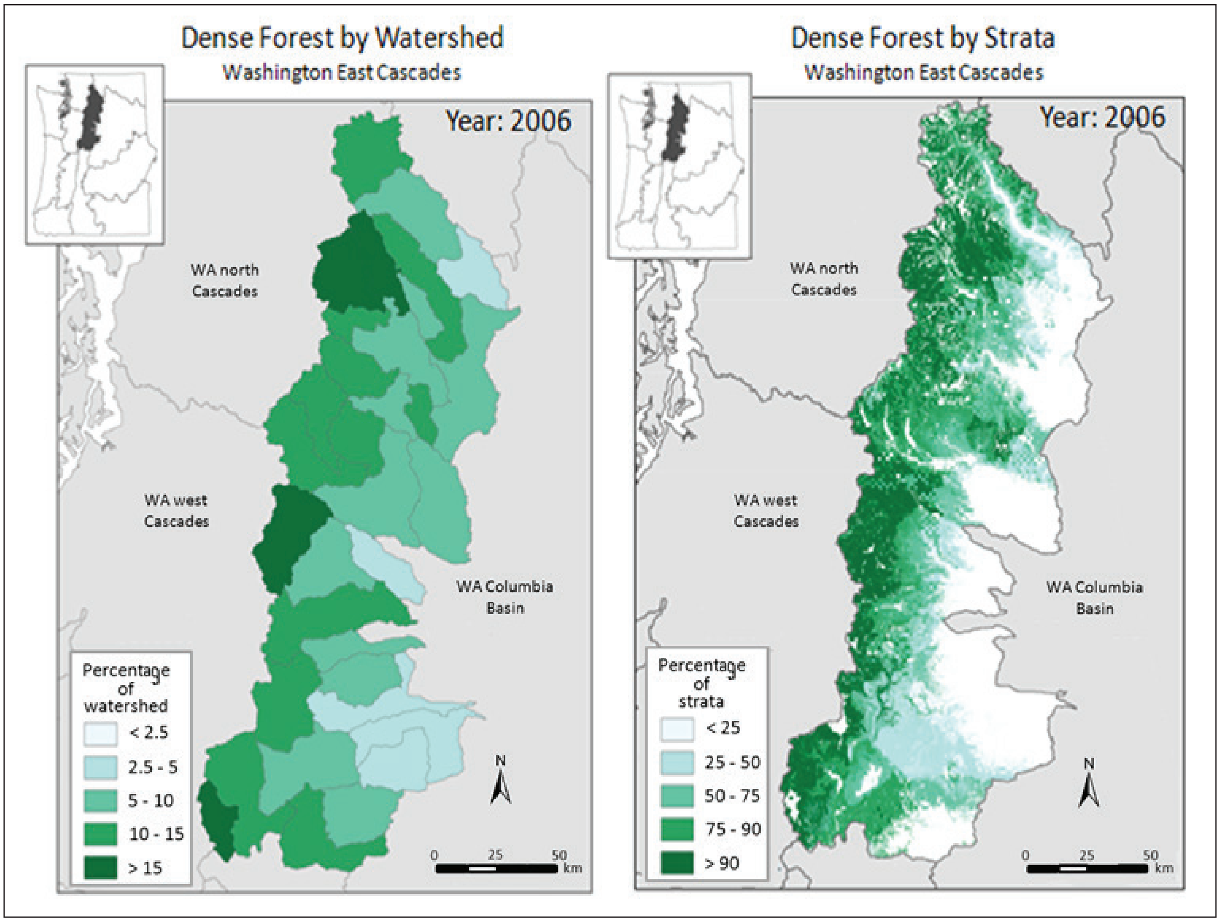

Figure 2.5—Example output (dense forest in eastern Washington [WA]), showing two spatial resolutions of output: the watershed level (left) and the modeling stratum level (right).

knowledge of the environment to generate example treatment rates. The resilience scenario largely utilized prescribed fire and thinning from below to try to create more resilient and open dry forests, although other treatments were also included. This scenario should not be viewed as an example of what land managers in the area might actually do in the area; rather, it demonstrates a small set of the kinds of analyses that can be done with our process.

Current forest conditions—

Model initial conditions, derived from a classification of the current vegetation map (see "Data Processing"), provided a means to understand current conditions across the landscape. Grouping the STMs and displaying the current vegetation data by ownership provided a unique perspective on the landscape. More than half of the landscape was in drier forest types like oak, ponderosa pine, or dry mixed conifer, and federal lands had the greatest proportion and area of these dry forest types (fig. 2.6).

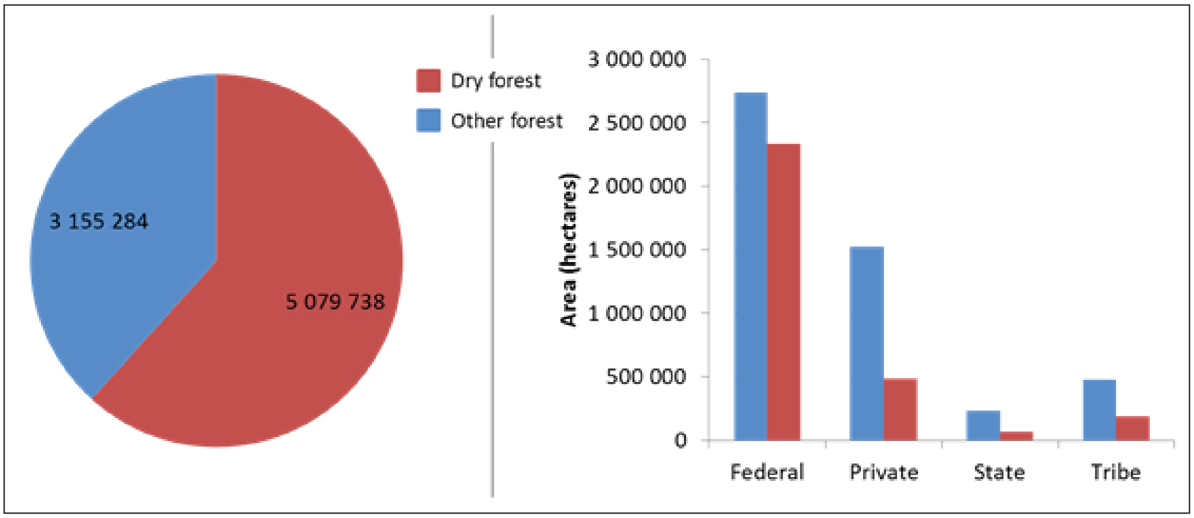

Figure 2.6—Hectares of dry forest across the landscape (left) and by ownership (right). Dry forests were composed of vegetation types such as oak, ponderosa pine, and dry mixed conifer. More mesic forests were assigned to the "other forest" category.

Trends in dense forests through time—
Because federal lands comprised such a large proportion of the landscape, we focused on these lands for the remainder of the forest example. In comparing the distribution of forest structures (i.e., tree diameter size classes and percentage canopy cover) across all federal lands, we found an increase in the average amount of open forest under the resilience scenario (fig. 2.7) relative to the FSO scenario. In contrast, forests became denser under the FSO scenario owing mostly to fire suppression effects. As we ran 30 simulations per scenario, we were also able to illustrate trends based on a range of potential future conditions and variation around those trends. Model results across all 30 model simulations suggested an increase in the amount of open forest with resilience activities, although the rate of increase varied among the simulations (fig. 2.8).

In this example, the FSO scenario generated declining amounts of dense forest in areas that are currently dense and prone to wildfire and insect outbreaks (fig. 2.9a). However, closed forests increased through time (green colors) in areas that were initially open, owing to recent active management, and where halting forest management resulted in the forests becoming more dense. In contrast, little of the landscape increased in density through time under the resilience scenario, owing to a high rate of active management across forested lands outside reserves and wilderness (fig. 2.9b).

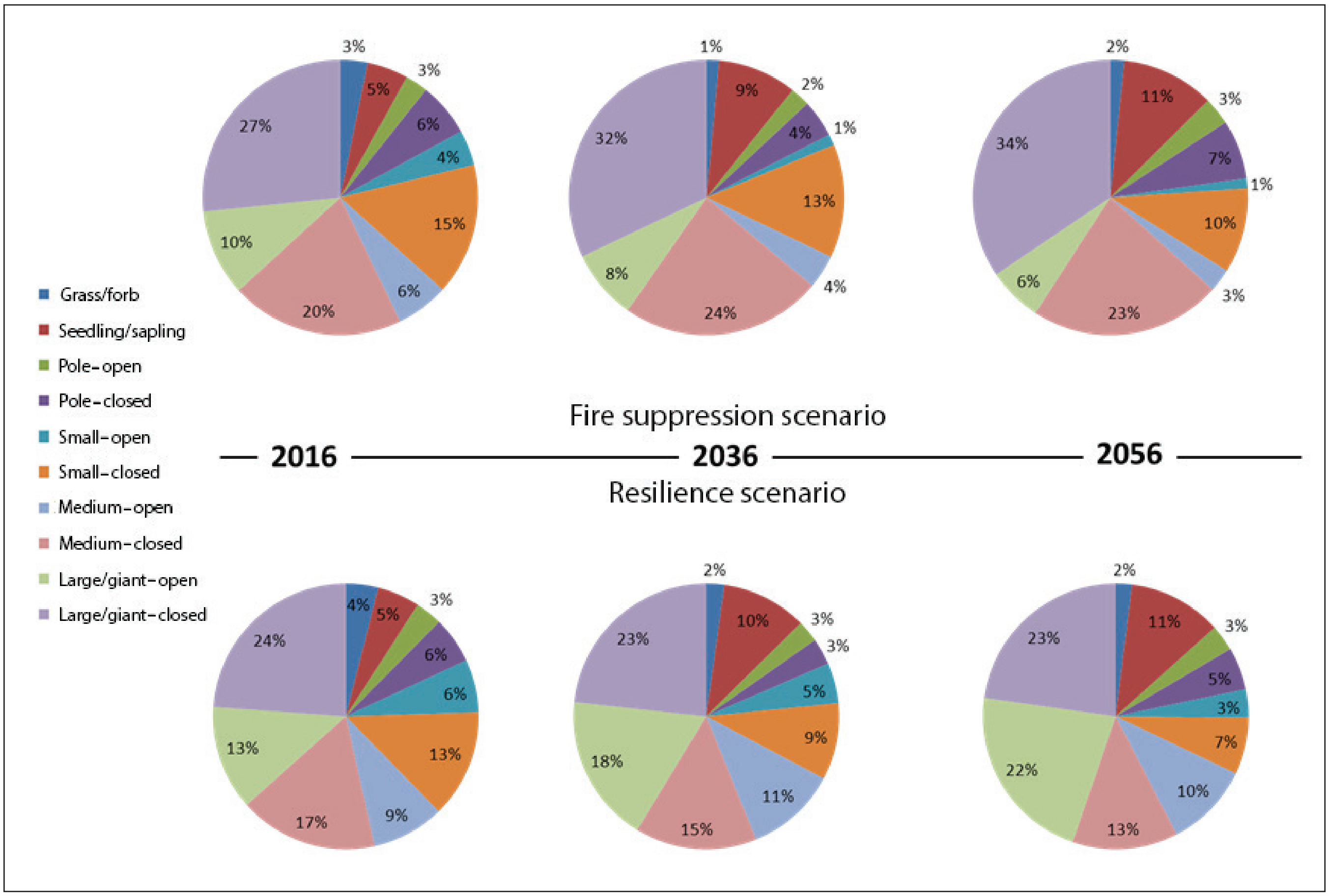

Figure 2.7—Projected trends in forest structure through time on federal lands under two different management scenarios. Structural stages represent the following tree diameter ranges in centimeters: sapling/shrub >0 to 13; pole >13 to 25; small >25 to 38; medium >38 to 51; and large/giant >51. Open and closed refer to canopy cover, where open structures have 10 to 40 percent canopy cover and closed structures have more than 40 percent canopy cover.

Although graphs can highlight overall trends, displaying output spatially provides the geographic context helpful in identifying areas likely to increase or decrease in open and dense forest through time (fig. 2.9). The use of potential vegetation, watershed, and ownership-management spatial layers allowed the creation of maps at the modeling stratum scale. Such maps should be used with some caution, however, as the information they depict may appear to have been generated by spatial rather than nonspatial modeling processes. Our nonspatial STM framework assumed that dynamics occur anywhere within the modeling strata, and the spatial maps illustrate results summarized to the scale of the modeling strata. Therefore, mapped results should be interpreted at the modeling strata scale, not at an individual pixel scale.

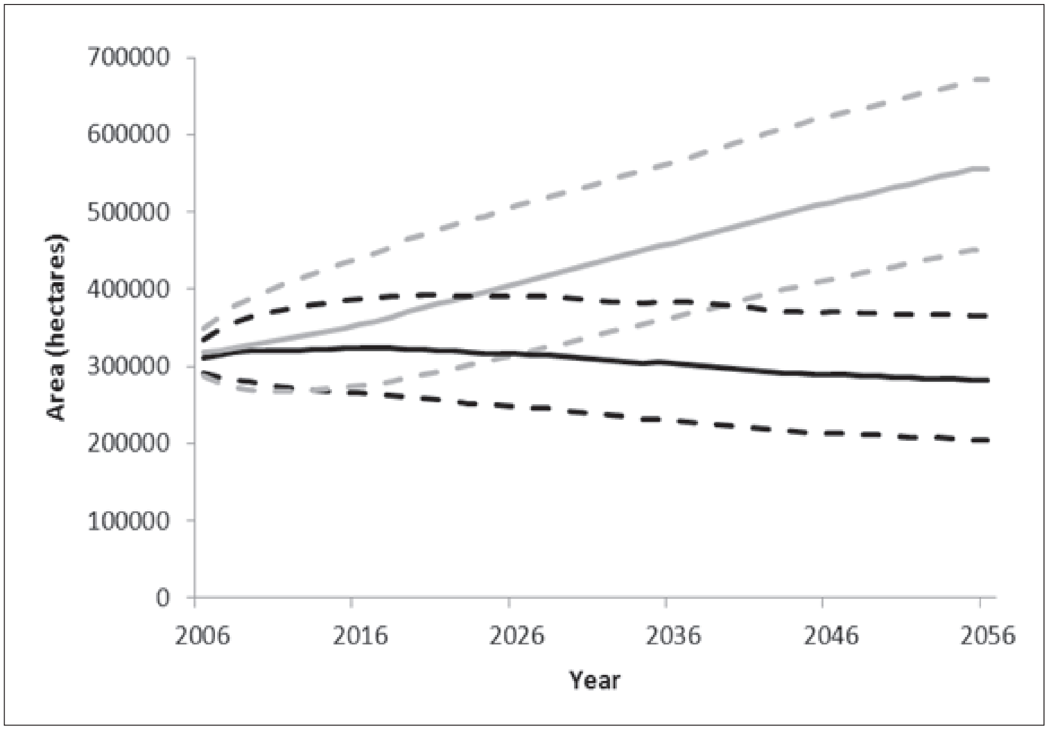

Figure 2.8—Projected trends in open (10 to 40 percent canopy cover) and closed (>40 percent canopy cover) forest under the resilience scenario on federal lands. Solid lines represent average trends; dashed lines indicate variation around the average. Lower dashed lines within a shade represent the minimum hectares for a given time step and simulation run. Upper dashed lines within a shade represent maximum hectares for a given time step and simulation run.

Arid Lands Example

We ran three scenarios to illustrate analysis using ILAP data in arid lands in southeastern Oregon: (1) a scenario with no grazing or active management (ungrazed/unmanaged), (2) a scenario where heavy livestock grazing occurred but no active management was implemented (grazed/unmanaged), and (3) a scenario with heavy livestock grazing and active restoration management that included postfire seeding of native grasses and juniper removal treatments (grazed/managed). These scenarios were intended to inform management by providing starting points for more refined scenarios with varying treatment levels and grazing intensities. Each scenario was run for 30 Monte Carlo simulations to project vegetation 50 years into the future (year 2050).

Comparison of management scenarios—

Under the unmanaged/grazed scenario, the warm, dry Wyoming big sagebrush types became dominated by exotic grasses (fig. 2.10). This resulted from our assumption that grazing (particularly in combination with fire and drought disturbances) reduced the presence of native grasses and provided a competitive

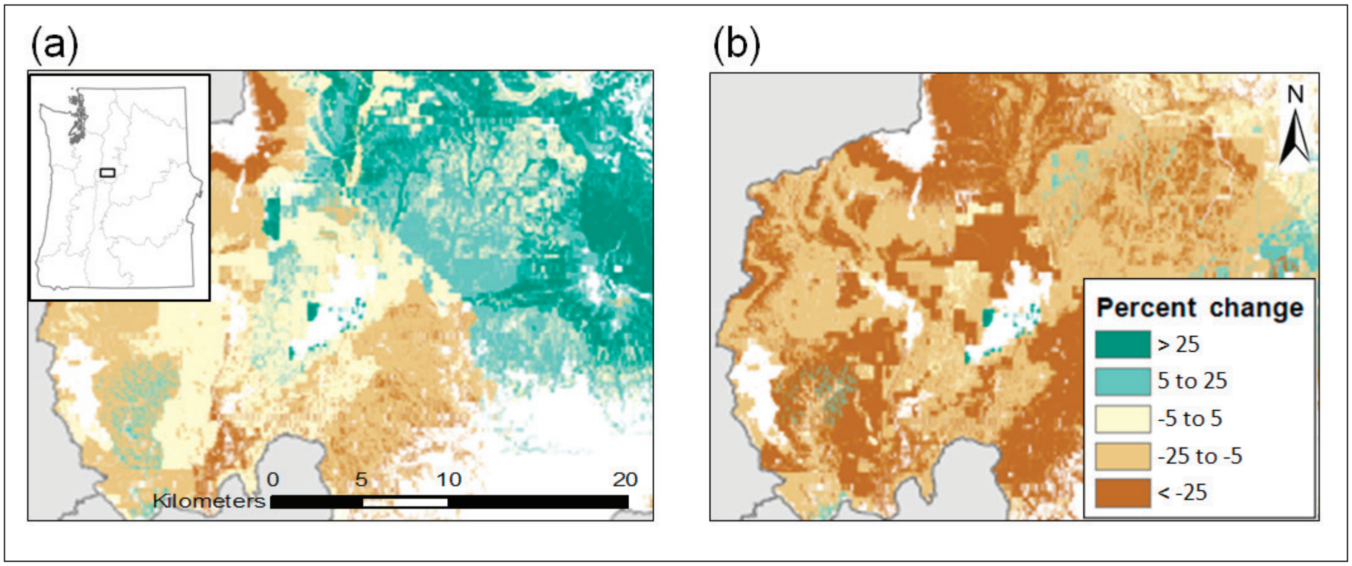

Figure 2.9—Trends in closed forest through time (after 50 years) in a portion of the Washington East Cascades modeling zone under the (a) fire suppression only scenario, and (b) resilience scenario. Colors represent those areas that either cumulatively increased or decreased in the amount of closed forest relative to current (2006) conditions.

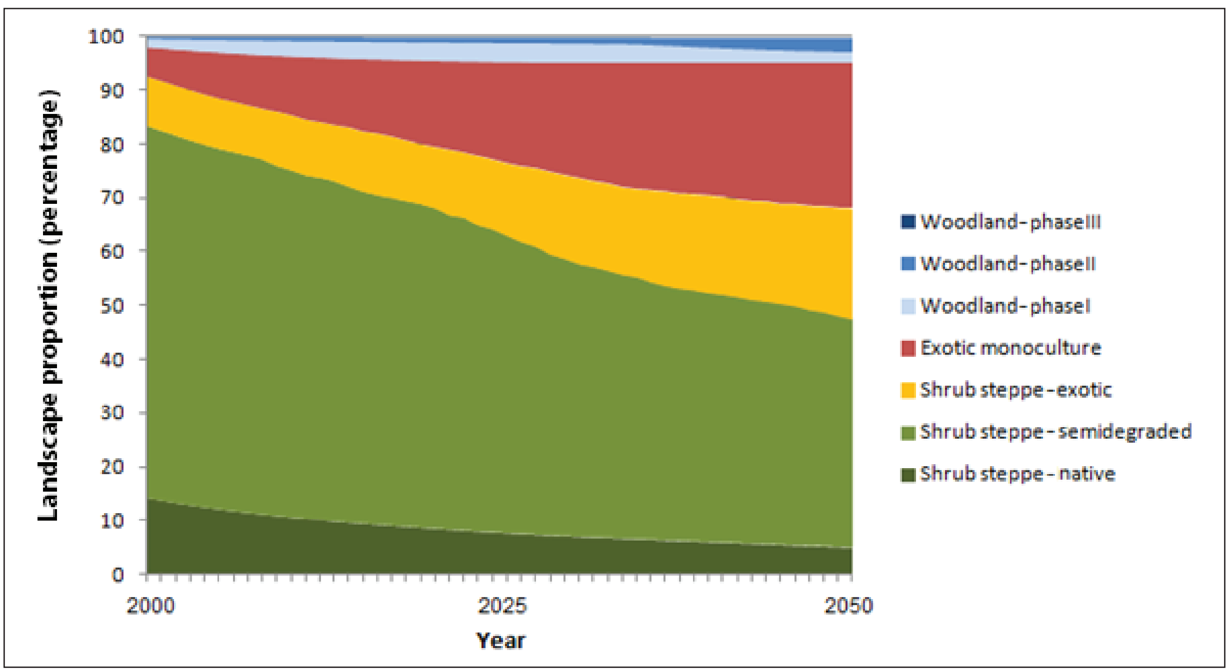

Figure 2.10—Trends in native and exotic vegetation in Wyoming big sagebrush under the grazed/unmanaged scenario in the Oregon southeast modeling zone.

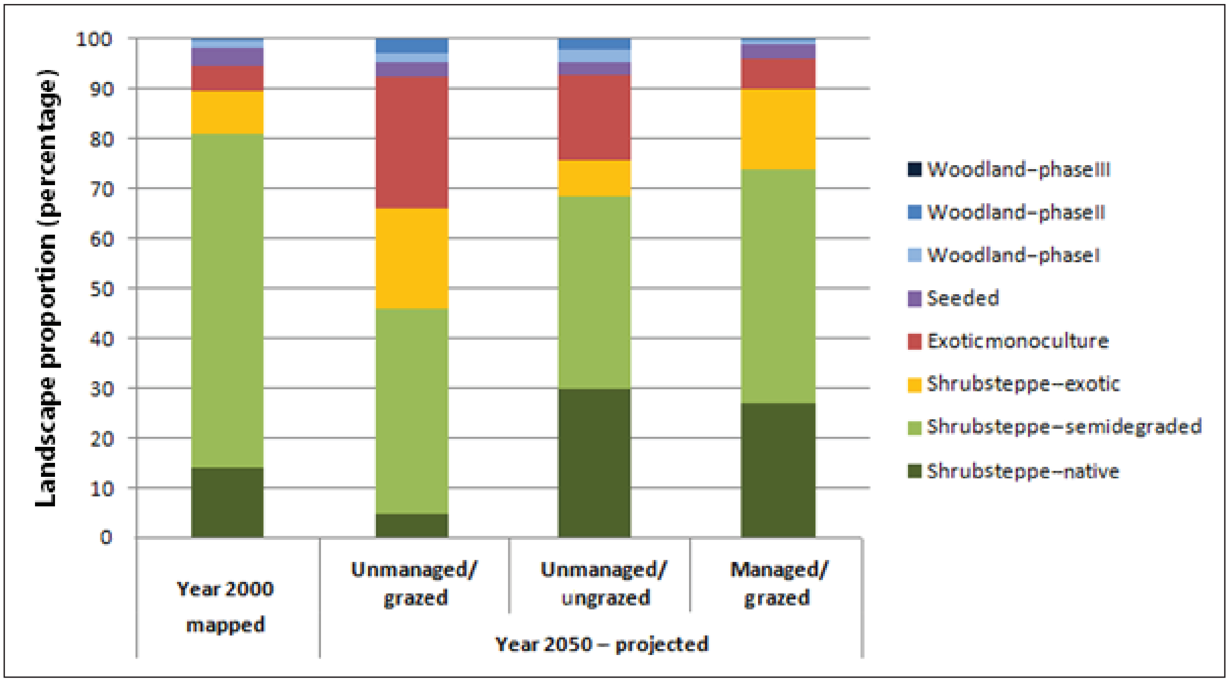

Figure 2.11—Generalized state class composition in Wyoming big sagebrush across three management scenarios in the Oregon southeast modeling zone. Columns show mapped landscape proportion in 2000 (left) and projected landscape proportion in year 2050 under three management scenarios (middle and right).

advantage for exotic grasses. A comparison of initial to future conditions across all scenarios suggested invasion by exotic grasses is likely to continue across all three management options, but the rate of invasion might vary depending on the amount of active management and grazing (fig. 2.11). By time step 50, about half of the landscape was projected to be dominated by exotic grasses (shrub steppe-exotic and exotic grass monocultures) under the unmanaged/grazed scenario. With the removal of grazing (unmanaged/ungrazed), vegetation condition still declined over time owing to conversion of already semidegraded shrub steppe to exotic grass state classes. Through active seeding of exotic grass encroached state classes, the managed/grazed scenario resulted in relatively constant levels of exotic grass monocultures over time, with slightly more area in exotic shrub steppe and native state classes, but less area in semidegraded shrub steppe by 2050. The model outcomes suggested that removal of grazing alone will not likely improve range condition in these warm, dry sagebrush systems, and that active management is required to restore already degraded systems.

Maps of model results by modeling stratum show the variability in invasion risk across the landscape (fig. 2.12) and may help managers identify locations likely to have high levels of invasion by exotic grasses and other conservation targets. Projected exotic grass invasion levels at year 2050 were highly heterogeneous across the warm, dry Wyoming sagebrush of southeastern Oregon. These maps can be useful to prioritize areas for treatment across large landscapes when budgetary constraints limit the ability to implement restoration treatments widely.

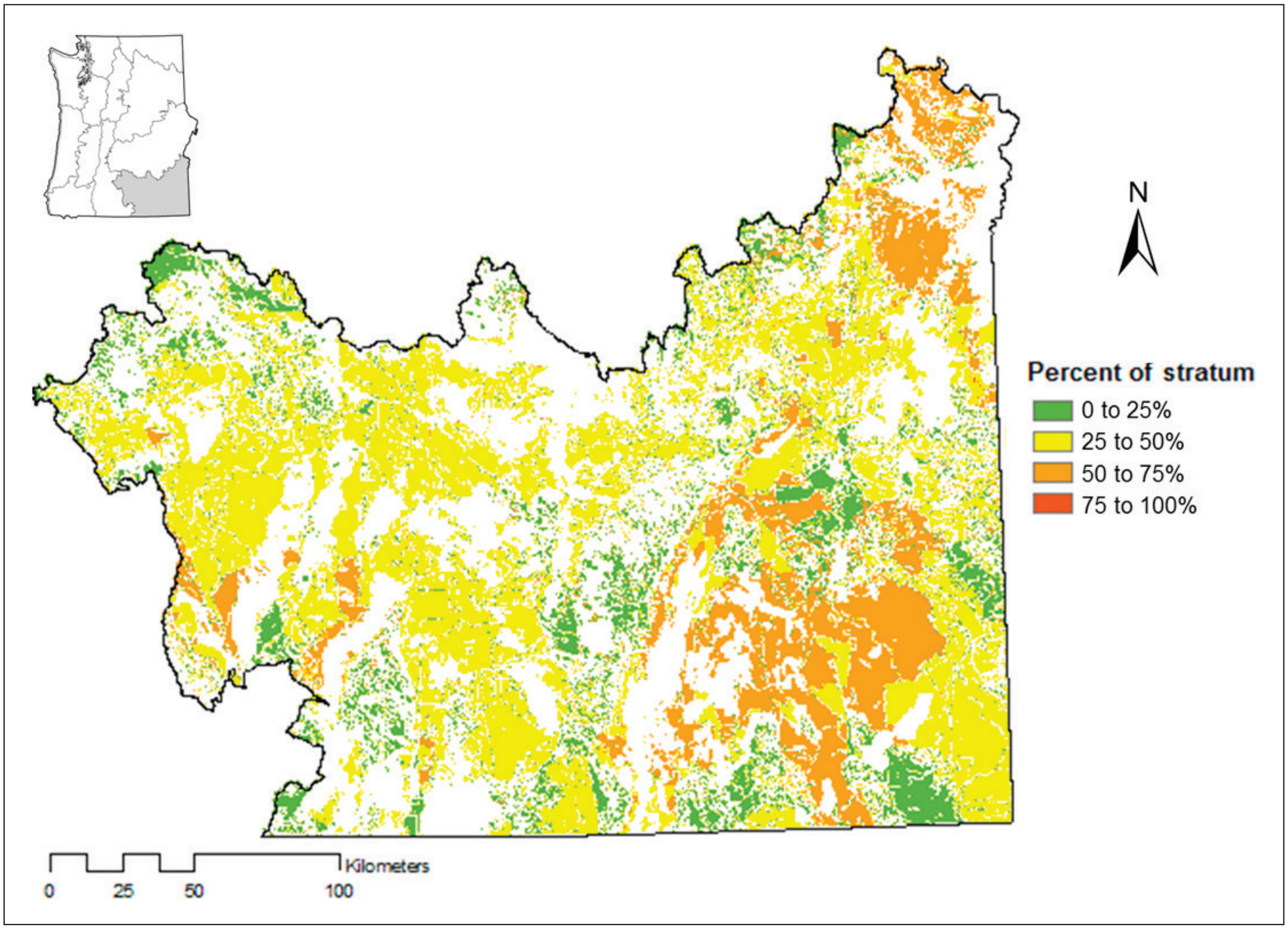

Figure 2.12—Exotic grass invasion risk map under the grazed/unmanaged scenario for Wyoming big sagebrush. The map identifies the percentage of each modeling stratum comprised of exotic grass at year 2050 in the Oregon southeast modeling zone.

Limitations

Although our STMs, model projections, and other products are useful tools for answering a variety of land management questions, they have several limitations. We identified three major types of constraints and limitations to our approach: spatial, logistical, and thematic.

Spatial Representation Constraints

The spatial resolution of our modeling outputs was limited by several interacting factors. First, our approach did not model interpixel or among-stand processes. Instead, our framework applied STM results to mapped modeling strata (intersection of potential vegetation, ownership-management, and watershed) to yield mapped stratum-scale output. Although maps of model projections appear to have a value for every modeled pixel on the landscape, our results are only applicable at the modeling stratum scale and larger aggregations. Thus, the spatial resolution of our modeling strata was constrained to a minimum modeling stratum size of 405 ha. In addition, constraints on modeling unit size reflected limitations in the spatial accuracy of the base data sets (especially potential and current vegetation, and their input data sets) that were available, but were not inherent limitations on our approach and process. Fine-scale analyses could be performed by substituting data with higher spatial resolution and accuracy.

In contrast to the size of our modeling strata, our input data were generally raster data with a 30-m pixel resolution. This difference in scale between inputs and outputs was a consequence of base data accuracy, our selection of modeling zone size, and logistical considerations. In general, our potential and current vegetation maps were reasonably robust at intermediate to broad scales, but limited in accuracy when examined on a pixel-by-pixel basis. Because of these spatial uncertainties at fine scales, our modeling framework is best adapted to addressing management-related questions at mid to broad scales (e.g., across multiple watersheds).

In addition, our projections incorporated error from each spatial layer used to define our modeling strata and initialize our models, in addition to error inherent in the STMs themselves. Although all of these maps were distributed as rasters with 30-m ground pixel resolution, they are all best used at broader scales. A more formal analysis on the relationship between summary scale and potential and current vegetation map accuracy performed by the LEMMA team showed that GNN maps of an old-growth structure index are quite robust when summarized over 8660-ha hexagons, and that accuracy increases with larger summary units (Gregory et al. 2011). Similarly, the ILAP variables are also likely to improve with coarser scales of summary. Map error was particularly prevalent at the forest/arid land margin, where low tree cover values were particularly challenging to map. Note that these errors were carried through the modeling process and contribute to modeling uncertainty, particularly at the boundaries between features within the spatial layers. For broad-scale analyses, the ramifications of boundary errors were likely minimal, but for analyses at intermediate and fine scales, additional assessments to quantify the error would be useful, and data developed for finer scale use should be substituted.

Logistical Constraints

Our approach had some practical limitations inherent to working with large data sets. To create, analyze, and display data from our models, users need a fairly high level of technical proficiency with databases and GIS software. The capacity to use and modify scripted procedures is also helpful if users plan to build new analyses to compare management options. We have packaged and documented our procedures to make them as user-friendly as possible, but there would still be a significant learning curve for anyone wishing to run the models and conduct further analyses. Improvements to the Path software are likely, and we hope that workflow and data management procedures will continue to improve.

An additional challenge stems from our models not being optimization models. Our process facilitated a gaming approach rather than producing “best” or “optimal” answers. The ILAP decision support tools currently being developed will help to fill some of this need.

Thematic Constraints

The simple structure of STMs is a strength of the modeling approach. The models are well-suited for both communication of results to nontechnical audiences, and for honing or building new models in collaboration with land managers, who may have deep ecological understanding but limited technical skills. However, this simplicity comes with some inherent limitations. These limitations are primarily thematic and relate to the definitions of the state classes and the transitions that are incorporated in each model. Definitions for state classes and transitions for vegetation were not standardized across STMs (Stringham et al. 2003), and the models adapted and developed for our project have important differences among the major vegetation types within the project and as compared to other projects.

The STMs do not explicitly model ecosystem processes, such as nutrient cycling, net primary productivity (NPP), seed dispersal, and interspecific competition. Vegetation dynamics may affect numerous ecosystem processes, and vice versa (Chapin et al. 1997). Although vegetation-ecosystem process interactions were not explicitly defined within ILAP STMs, they were implicit within the potential vegetation definitions, and the associated STM structures. For example, in the alpine STM, succession and growth-related transition rates were slow, reflecting low NPP common at high elevations where cold weather constrains nutrient cycling and the growing season. Because ecosystem processes were implicit within our models and not explicitly defined, they cannot be explored in and of themselves within the basic ILAP framework without additional efforts to link model state classes, transitions, and outputs to ecological processes. The data and information available to inform transition pathways and transition rates varies widely among STMs. Some transitions are based largely on empirical data (e.g., MTBS fire frequencies), some on models (e.g., FVS succession rates), and others almost entirely on expert judgment. Most STMs use a variety of information sources to parameterize transitions and use expert judgment to fill in the knowledge gaps. The STMs compiled and developed for ILAP can be easily updated and changed as new data and information become available.

The types of questions that can be addressed by our STMs were constrained to those that relate directly to the state classes and transitions contained within each model.

The types of questions that can be addressed by our STMs were constrained to those that relate directly to the state classes and transitions contained within each model. For example, the vegetation units within the forested potential vegetation state classes for OR/WA represented classified vegetation types that were labeled only with a dominant tree species. Subdominants were not explicitly labeled, although they were implicitly associated with each cover type, and understory species were not considered. Because of this setup, the behavior of individual species through time could only be extracted from a limited set of our ILAP potential vegetation types. However, arid land vegetation cover types usually contained more compositional detail than the forest cover types, including functional groups (perennial grass, annual grass) or individual species (e.g., Wyoming big sagebrush, western juniper). Still, the thematic resolution of the models was generally constrained to vegetation communities or groups of species, rather than individual species.

An additional constraint inherent to the basic ILAP framework is the implicit assumption that vegetation potential does not shift over time. Each modeling stratum was linked to a single potential vegetation type, which was linked to a single STM. This assumption may be robust enough on a short timeframe but becomes problematic in the context of climate change, which will likely shift site potential, and reorganize species assemblages. On the other hand, given the uncertainties embedded in current climate modeling, information on the historical range of variability (HRV) may be our most certain available guide as to how ecosystems will behave in the future (Keane et al. 2009) and what management actions may prevent irreversible, deleterious changes. Our potential vegetation types were based on

HRV, and our STMs were parameterized from HRV wherever data were available. This does not imply that modeling approaches that do incorporate climate change are not worth exploring, just that they do not yet necessarily yield a more reliable projection than HRV-based models. Other work as a part of ILAP explored how to extend the ILAP models to encompass shifts in site potential, linking multiple models to allow for transitions among vegetation types (chapter 7).

Another important driver of vegetation change through time is the legacy of past climate, disturbance, and land use (Chase 2003, Rhemtulla and Mladenoff 2007). Historical influences on vegetation composition operate at a variety of spatio-temporal scales. On a millennial timeframe, vegetation shifts related to historical climate change have been documented (Hotchkiss et al. 2007) and vegetation species composition and abundance is affected by past climate. On a decadal timeframe, or even over centuries, vegetation patterns can also reflect the legacy of human land use (Dupouey et al. 2002, Foster et al. 1998). Because the legacy of past climate, disturbance regime, and land use is reflected within our initial conditions (via our current vegetation layer), and also in our transition rates (via the fire regime), many important dimensions of legacy effects are accounted for implicitly within the ILAP framework. Because of this construction limitation, as with ecosystem processes, questions relating directly to legacy effects cannot be explicitly addressed with the data we have developed for our particular application.

Scenario Analysis

The baseline ILAP scenario with no management activities except for fire suppression only (FSO) is available across all lands. Within the focus areas, additional scenarios were also run to reflect timber harvesting, restoration activities, and other management. However, the range of management options is limited, and more work is needed to develop realistic management transition rates across ecological and administrative boundaries. The interpretation of model results would be greatly improved by a larger number of management scenarios, and development of alternative management scenarios will greatly improve the utility of ILAP models.

Data Products and Tools

Most of the data, models, and tools produced for ILAP are publically available on the Western Landscapes Explorer website, maintained by the Institute for Natural Resources (www.westernlandscapesexplorer.info). The STMs were run for all lands in AZ/NM and OR/WA, and input data, models, and summarized output for the default FSO scenario are available in files called rollout packages. Rollout packages each contain the Vegetation Dynamics Development Tool (VDDT) and Path modeling databases, model documentation, initial GIS data needed for the rollup process, post-rollup modeling strata and state class spatial data, and STM outputs (i.e., projected future landscape conditions). Rollout packages are separated by modeling zone, and separated between forests and arid lands in OR/WA, and forests, woodlands, and arid lands in AZ/NM.

In addition to the FSO scenario results for all lands, additional summarized results are available for focus areas to characterize alternative management actions and summarize across variables of interest (see "Example Results"). An alternative resilience scenario was run for the forested environments of eastern Washington and Oregon (called east-side forests), and two additional scenarios were run for arid lands of southeastern Oregon. In AZ/NM, results for various management scenarios are available for the Sky Islands focus area.

Conclusions

The original thrust of the project was to generate information for land management planning that could be used to support prioritization of management activities at mid to broad scales, and to advance efforts to integrate daunting issues such as including climate change in the planning process in a tractable manner.

The methods, models, and data presented here provide a means to perform integrated landscape assessments in support of mid- to broad-scale planning efforts that span multiple watersheds and affect vegetation, habitat, economics, and other processes. Similar models have been employed over smaller spatial extents to address shortcomings related to integrated metrics at the landscape scale, future conditions, alternative strategies, and cost-benefit analyses in commonly used approaches to conservation planning (Low et al. 2010). The original thrust of the project was to generate information for land management planning that could be used to support prioritization of management activities at mid to broad scales, and to advance efforts to integrate daunting issues such as including climate change in the planning process in a tractable manner. It has become exceptionally clear over the course of this project that when modeling at multistate scales, a wide range of complex questions can be studied using ILAP data and models. Examples in this chapter represent a small glimpse of the potential for graphs, maps, and other reporting that can be built from the VDDT/Path output. Linking STM output to other characteristics of interest can provide information about vegetation change integrated with fuels, wildlife, economic, and other data (chapters 3 through 7). Graphs, maps, and charts of economic, ecological, and social variables can be linked to the raw output, and summarized to further increase the breadth of questions that can be addressed across a landscape, watershed, ownership, or potential vegetation type. The framework can also be linked with other modeling systems (e.g., FVS) for validation or calibration. Although there are constraints to using STMs related to scale, and logistics, the ILAP framework is a demonstrably robust and flexible tool for assessing management alternatives and prioritizing management actions at regional scales, for a diversity of ecological and social contexts.

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Chapter 3: Simulating Fire Hazard Across Landscapes Through Time: Integrating State-and-Transition Models With the Fuel Characteristic Classification System

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Chapter Summary

Information on the effects of management activities such as fuel reduction treatments and of processes such as vegetation growth and disturbance on fire hazard can help land managers prioritize treatments across a landscape to best meet management goals. State-and-transition models (STMs) allow landscape-scale simulations that incorporate effects of succession, management, and disturbance on vegetation composition and structure. State-and-transition models have been used for many different types of landscape-scale assessments. However, STMs do not currently assess fuels and fire hazard for different vegetation state classes.

We integrated STMs with a software application called the Fuel Characteristic Classification System (FCCS) to enable assessment of fuel properties and fire hazard with succession, disturbance, and management across landscapes over time. We created FCCS fuel beds from inventory plots for each vegetation state class in STMs covering forests and woodlands in Arizona, New Mexico, Oregon, and Washington. We used FCCS to analyze each fuel bed for its potential fire behavior, and we linked results to STM simulation output to assess potential changes in fire hazard with management and natural disturbance regimes over time.

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The analysis across the four-state study area resulted in thousands of fuel beds that cover a broad range of fuel conditions, and the links between these fuel beds and STMs can be used to help develop successful fuel treatment regimes in fire-prone forests. We present a Washington East Cascades (WEC) case study that illustrates potential application of this work. We analyzed potential future fire hazard under fire-suppression-only and resilience scenarios for the WEC region. We found that crown fire potential was reduced under the resilience scenario; area of high crown fire potential was reduced by 13 percent by 2056. However, patterns in surface fire potential were obscured by variation in surface fuel characteristics within a vegetation state class. The fuels analysis in the WEC gives land managers information they need to prioritize areas for fuel treatments and help them to determine what types of activities will result in the greatest reduction in crown fire potential.

Introduction

Twentieth-century fire suppression policies have led to fuel accumulations and greater risk of high-severity fire in many dry forest types of western North America that were historically characterized by relatively high frequency and low- to moderate-severity fire regimes (Allen et al. 2002, Brown et al. 2004, Covington 2003, Hessburg et al. 2005). Fire area burned has increased in the Western United States over the past few decades (Westerling et al. 2006), and this trend is expected to continue with warmer and drier conditions associated with climate change (Littell et al. 2010, McKenzie et al. 2004). Climate change may also lead to fires becoming more difficult to control because of more frequent extreme burning conditions (Fried et al. 2004). To reduce stem densities and fire intensity and support suppression efforts, vegetation management treatments are often implemented in areas characterized by historically low- to moderate-severity fire regimes (Graham et al. 2004, Peterson et al. 2005). However, the effectiveness of these treatments varies by treatment type and treatment intensity within managed forest stands and by treatment type, intensity, and arrangement across landscapes (Finney et al. 2007, Johnson et al. 2011, Prichard et al. 2010a, Schmidt et al. 2008), making it difficult for managers to choose what type of treatments to conduct and where to prioritize treatments on a landscape.

Fire hazard, or the potential fire behavior for a fuel type (Hardy 2003), concerns fire and land managers because it gives an indication of the potential fireline intensity, flame lengths, crown fire activity, resistance to control, and potential physical and biological effects of fire in a given area of vegetation. Fire hazard also reflects the only element of fire behavior that can be affected by management—fuels. Information on the effects of management activities and forest succession and disturbances on fire hazard can help land managers prioritize vegetation management treatments on a landscape to meet management objectives.

State-and-transition models (STMs), which subsume vegetation dynamics into state classes (boxes) and transitions (arrows), are tools that provide landscape-scale information on the effects of management, forest growth and development, and natural disturbance on vegetation composition and structure (chapter 2). Thus, STMs can provide information that is useful to managers in prioritizing treatments across a landscape. The STMs have been used for many types of landscape-scale assessments that incorporate potential effects of management on vegetation composition and structure over time (e.g., Arbaugh et al. 2000; Forbis et al. 2006; Hemstrom et al. 2001, 2007; Merzenich et al. 2003; Merzenich and Frid 2005; Ryan et al. 2006; Weisz et al. 2009). However, STMs do not currently allow direct assessment of fuels and fire hazard for different vegetation state classes.

The Fuel Characteristic Classification System (FCCS) is a software application that allows users to analyze fuel properties and fire potential of wildland and managed vegetation.

The Fuel Characteristic Classification System (FCCS) (McKenzie et al. 2007; Ottmar et al. 2007; Riccardi et al. 2007a, 2007b; Sandberg et al. 2007a, 2007b; Schaaf et al. 2007) is a software application that allows users to analyze fuel properties and fire potential of wildland and managed vegetation. The FCCS analysis involves development of fuel beds (detailed descriptions of all burnable biomass, from the litter layer to the canopy), and the software evaluates those fuel beds for fire behavior potential (the intrinsic physical capacity of a fuel bed to support fire) (Sandberg et al. 2007a, 2007b). The FCCS is a flexible tool that allows fuel bed development and analysis for any relatively homogeneous unit. Providing an alternative to the categorization of fuel characteristics into standard fuel models (e.g., Scott and Burgan 2005), FCCS allows development of detailed fuel beds and analysis for any chosen unit of land. The flexibility and detailed analysis that characterize FCCS allowed us to integrate FCCS with STMs to enable assessment of fuel properties and fire hazard with succession, disturbance, and management across landscapes over time.

To integrate FCCS with STMs, we used inventory plot data to construct FCCS fuel beds that represent each vegetation state class in STMs covering forested and woodland ecosystems in Arizona, New Mexico, Oregon, and Washington. We then analyzed the potential fire behavior for each fuel bed and linked the results to STM simulation output to assess potential changes in fire hazard with management and natural disturbance regimes over time.

This project was conducted as a part of the Integrated Landscape Assessment Project (ILAP), which involved the examination of current and potential future dynamics of broad-scale, multiownership landscapes by integrating and evaluating information about current and future vegetation and related resources (see chapter 1 for further detail on the ILAP project). Linking our fuel bed analysis with output from the ILAP STM modeling effort (chapter 2) allowed us to address a number of research and management questions, including, (1) How do different forest management scenarios affect fuel conditions and fire hazard across a given landscape? and (2) To what extent can fuel treatment programs reduce fire hazard?

This chapter describes methods used to integrate STMs with FCCS in the four-state study areas and illustrates results of our process with a case study in the Washington East Cascades (WEC) modeling zone. We chose to use a case study to illustrate results because results are more clearly displayed and conceptualized at the scale of a region than at the scale of the four-state study area. We chose eastern Washington as a case study area for two main reasons: (1) the fire and fuels management questions on which ILAP was focused (see chapter 1) are highly relevant in this region; and (2) ILAP researchers worked with a land management collaborative in the WEC to get user input on models, output, and management scenarios. Thus, the vegetation models and management scenarios for WEC are likely to be used by land managers to answer management questions.

Methods

We integrated fuels information into STMs covering forest and woodland vegetation types in the states of Arizona, New Mexico, Oregon, and Washington (fig. 3.1). The integration of fuels information in STMs for the study area involved five main steps:

- Select field-measured inventory plots from existing data sets to represent each state class, or vegetation structure and cover combination, in STMs. Inventory plots were selected from the U.S. Department of Agriculture Forest Service (USDA FS) Forest Inventory and Analysis (FIA) program (USDA FS 2012) and Current Vegetation Survey (CVS) program data sets.
- Construct FCCS fuel beds (descriptions of burnable biomass extending from the forest floor to the canopy) for each plot.
- Use FCCS to analyze fuel beds for fire hazard (e.g., crown fire potential).
- Summarize fire potentials for all fuel beds representing each state class in STMs.
- Link summarized fuel beds and associated fire hazard to results of STM simulations.

Each of these steps is described in further detail below.

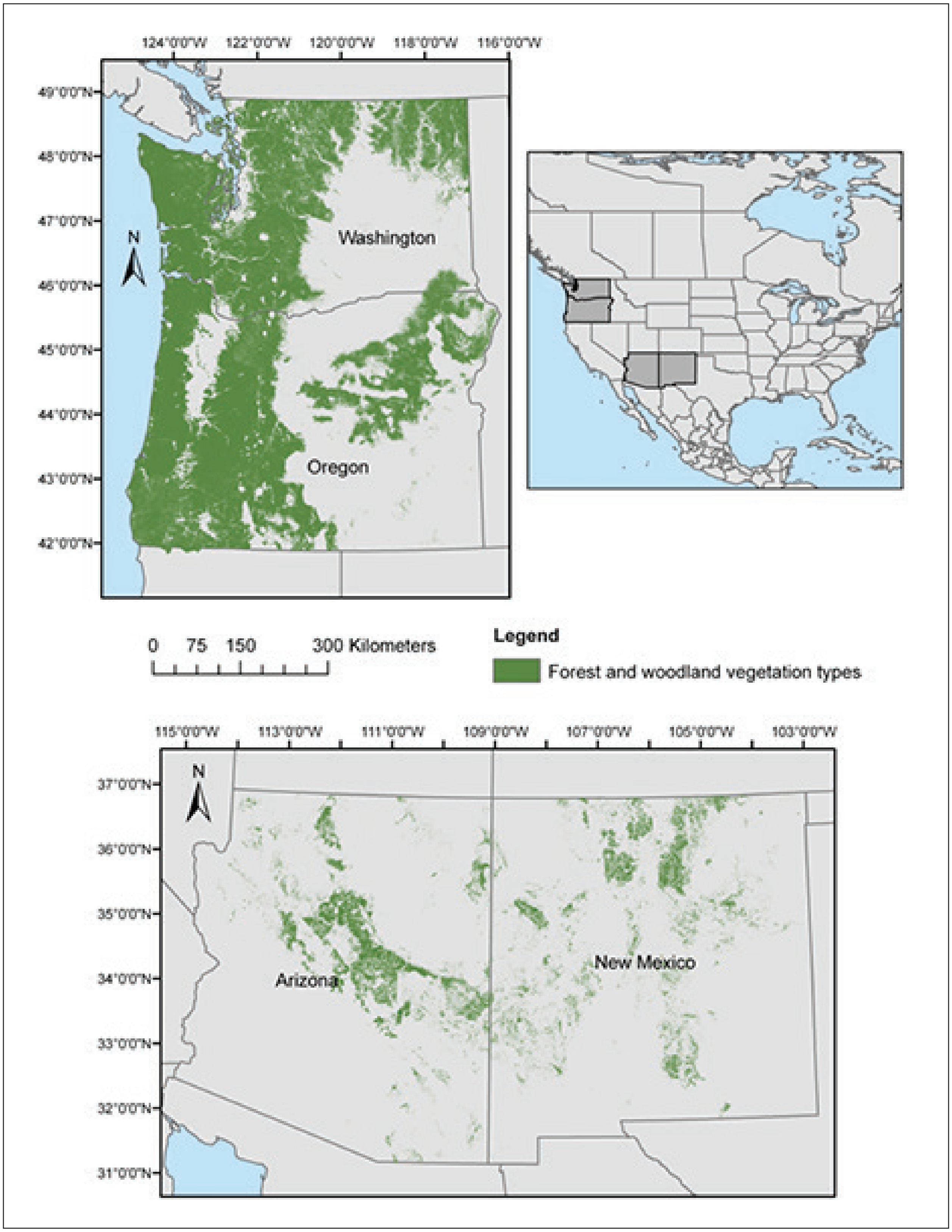

Figure 3.1—Area covered by forest and woodland vegetation types in the four-state study area.

Plot Selection and Classification Into State-and-Transition Model State Classes

State-and-transition models—

Inventory plots were selected to represent combinations of vegetation cover and structure within each STM. These combinations of vegetation cover and structure, called STM state classes, represent a subset of vegetative conditions found within the broader landscape. The STMs are represented by boxes (vegetation state classes) and arrows (transitions between state classes). Transitions between state classes are either deterministic (occurring with time, e.g., succession) or probabilistic (with a given probability of occurring at each time step, e.g., disturbance or management). The STM runs incorporate Monte Carlo simulations and track both the state of the landscape over time and the occurrence of transitions. The STMs were developed in the Vegetation Dynamics Development Tool (VDDT) framework (ESSA Technologies Ltd. 2007) and run using the Path Landscape Model platform (Apex Resource Management Solutions 2012; Daniel and Frid 2012). The VDDT and Path simulate vegetation dynamics by dividing the landscape into state classes, assigning probabilities to transitions between state classes, and simulating the state of the landscape over time using Monte Carlo methods (see chapter 2 for more detailed information on STMs).

Inventory plots were selected to represent combinations of vegetation cover and structure within each STM.

The ILAP STM modeling effort (chapter 2) encompassed all lands in Arizona, New Mexico, Oregon, and Washington. For modeling purposes, Oregon and Washington (OR/WA hereafter) were divided into 12 modeling zones, and Arizona and New Mexico (AZ/NM hereafter) were divided into six zones (see maps in chapter 2). These modeling zones represent Omernik ecoregions (Omernik 1987), with boundaries modified to coincide with hydrologic unit code 5 watershed boundaries (USGS and USDA NRCS 2011). One STM was built for every potential vegetation type (e.g., fig. 3.2) resulting in 7 to 22 models in each modeling zone. Potential vegetation type maps were downloaded from Ecoshare (http://ecoshare.info/). Each STM was characterized by 5 to 60 state classes.

Inventory plot data—

We used inventory plot data from the FIA and CVS programs to develop FCCS fuel beds for forested and woodland STM state classes. We limited our analysis to forests and woodlands (canopy cover >10 percent) because insufficient inventory plot data were available to characterize arid lands with canopy cover <10 percent. We used the most recent or comprehensive inventory plot data sets available for each state (the comprehensive CVS data set for OR/WA, and the most recent FIA data sets for AZ (annual) and NM (periodic)). Only the forested portions of CVS inventory plots were used in our analysis in OR/WA because our goal was to characterize

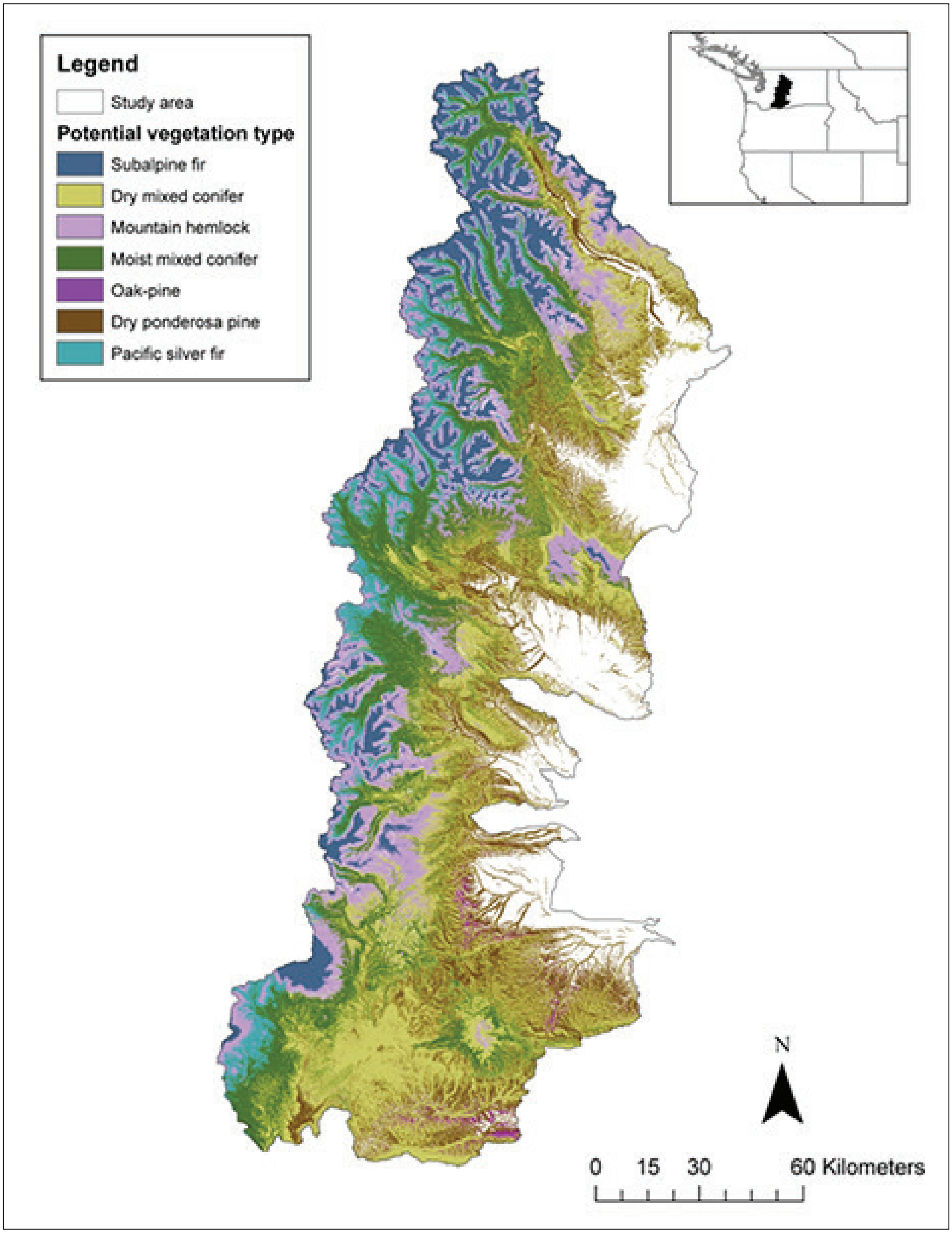

Figure 3.2—Potential vegetation types in the Washington East Cascades modeling zone. One state-and-transition model was built for each potential vegetation type.

fuel conditions in forested ecosystems and information was available at that scale for OR/WA. Owing to the unavailability of data at the forested condition level in AZ/NM, we used the entire FIA plot database in our analysis for AZ/NM.

Inventory plot classification—
The process used to select inventory plots to represent each state class in the STMs differed somewhat between AZ/NM and OR/WA. For AZ/NM, plots were first classified to one potential vegetation type, corresponding to one STM, by experts using plant association information associated with each plot. Once a plot was classified to a potential vegetation type, it was assigned a specific STM vegetation state class based on size class of the dominant cohort (defined by basal area), percentage canopy cover, number of canopy layers (1 or >1), and in some cases (i.e., aspen cover types) forest type importance value (Horn 1975). The rule-set for classification was vegetation type-specific. Owing to the relatively low number of FIA inventory plots for AZ/NM, inventory plots were used to represent state classes without regard for the geographic location of the plot. For example, an inventory plot from southwestern Arizona, classified into the dry pine vegetation type, could be used to represent a state class in the dry pine vegetation type in northeastern New Mexico. We classified a total of 1,734 inventory plots into 62 forest state classes and 1,870 inventory plots into 49 woodland state classes (state classes were consistent across modeling zones in AZ/NM).

Owing to a greater sample size, we further geographically constrained which inventory plots could be used to represent a given potential vegetation type in OR/WA. For each modeling zone in OR/WA, we considered all inventory plots within ECOMAP sections (Cleland et al. 2007) that fell within the modeling zone. For example, if four ECOMAP sections fell within a modeling zone boundary, we would consider all plots within those four ECOMAP sections, and not just the plots within the modeling zone boundary (see fig. 3.3 for illustration). We used plant association information (ecoclass codes from Hall 1998) for each plot to determine which plots to include in the analysis for each modeling zone. If plant association for a plot was determined to not occur in a modeling zone, the plot was dropped from the analysis for that region. Each plot was used only once (to represent a single state class within an STM) within a modeling zone, but plots could be used in more than one modeling zone because ECOMAP sections typically overlapped with multiple zones. Because one to five subplots were aggregated to represent a plot, it was possible for individual subplots to represent different potential vegetation types. When this occurred, the majority type (covering >60 percent of analysis area) was assigned to the plot.

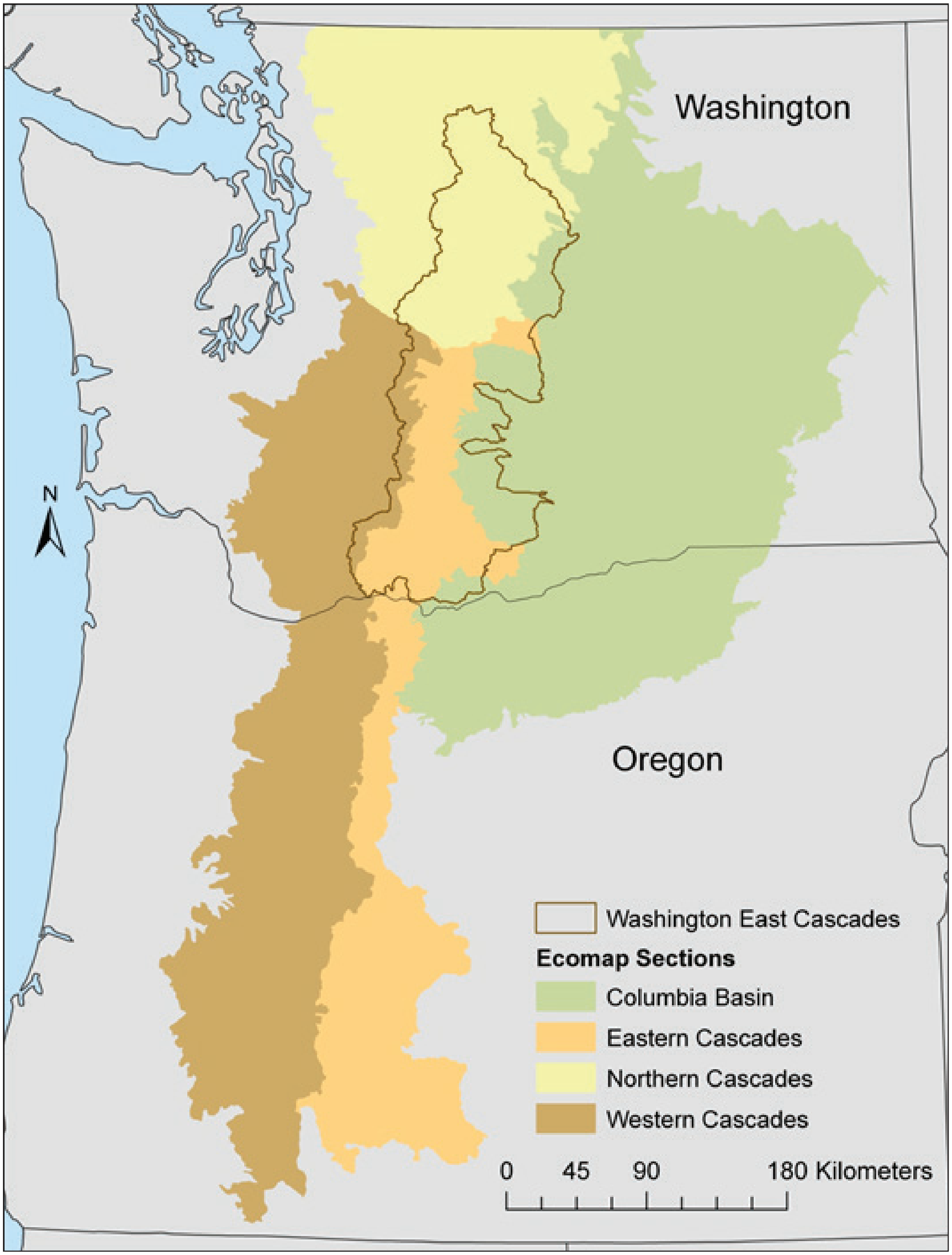

Figure 3.3—Ecomap sections in the Washington East Cascades (WEC) modeling zone. We used Ecomap sections to determine which inventory plots to use to represent vegetation state classes in state-and-transition models. For each modeling zone in Oregon and Washington, we considered all plots within Ecomap sections that were intersected by the modeling zone boundary. In this example, the analysis for the WEC modeling zone included all inventory plots that fell within the Columbia Basin, Eastern Cascades, Northern Cascades, and Western Cascades Ecomap sections.

We classified a total of 10,581 inventory plots into 3,716 state classes in Oregon and Washington.

In OR/WA, plots determined to be representative of each potential vegetation type were put into STM-specific cover categories based on forest type importance values as calculated by the Landscape Ecology, Modeling, Mapping, and Analysis (LEMMA) team (www.fsl.orst.edu/lemma/splash.php). Plots were then classified into structure categories based on calculated quadratic mean diameter (0 cm diameter at breast height (DBH) = grass/forb; <13 cm DBH = seedling/sapling; 13 to 25 cm DBH = pole; 25 to 38 cm DBH = small; 38 to 51 cm DBH = medium; 51 to 76 cm DBH = large; >76 cm DBH = giant), percentage canopy cover (<10 percent = grass/forb; 10 to 40 percent = low; 40 to 60 percent = medium; >60 percent = high), and number of canopy layers (1 = single or >1 = multiple) (these variables were also calculated by the LEMMA team). However, because a broader suite of conditions exist on a landscape than are modeled, many plots were reclassified to fit into one of the STM boxes. Our reclassification rules allowed plots with the same species cover, canopy cover, and canopy layers to either shift up or down one diameter size class. For example, a plot in the 25- to 38-cm DBH category could be reclassified into a state class in the 13- to 25-cm or 38- to 51-cm DBH category. Despite this potential reclassification by diameter, some plots still did not fit into any of the state classes included in the model, and these plots were dropped from the analysis. We classified a total of 10,581 inventory plots into 3,716 state classes in OR/WA (see table 3.1 for a modeling zone-specific list of number of state classes and number of inventory plots classified into state classes; see table 3.2 for a list of specific state classes and number of plot classified into each state class for the WEC modeling zone).

Once our classification of inventory plots into STM potential vegetation types and state classes was complete, we found some STM state classes had no representative plots (e.g., see table 3.2). In those cases, we chose plots with characteristics similar to the missing state class to represent the state class. We first selected plots from the size class above or below the missing state class (within potential vegetation and cover type). If there were no plots in the size classes above or below that of the missing state class, we selected plots with the same structural attributes in a similar potential vegetation type or species cover. If representative plots were still not found, we considered plots with a different number of layers but otherwise identical attributes to that of the missing state class. For example, in multiple zones, many medium canopy closure, single-storied state classes had no representative plots in the large and medium size classes. In such cases, we used plots with medium canopy closure and multiple layers within the same vegetation type, cover

Table 3.1—Number of state classes (combinations of vegetation cover and structure; excluding development state classes) that characterized state-and-transition models for each modeling zone in Oregon and Washington, and number of inventory plots classified into state classes in each zone[a]

Modeling zone	Number of state classes	Number of inventory plots classified into state classes
Oregon Blue Mountains	330	2,301
Oregon Coast Range	368	824
Oregon East Cascades	491	1,675
Oregon Southeast	134	321
Oregon Southwest	342	854
Oregon West Cascades	421	1,075
Washington Columbia Basin	477	1,117
Washington Coast Range	229	821
Washington East Cascades	207	3,000
Washington North Cascades	228	875
Washington Northeast	253	519
Washington West Cascades	236	1,008

[a] Total number of inventory plots classified into state classes was 10,581. Plots were used only once within a modeling zone but could be used in more than one modeling zone.

type and size class to represent the missing state class. Best judgment was used in the remaining cases. If no suitable plots were found to represent a state class, we did not include that state class in our analysis.

Although it is likely that some of the inventory plots used in our analysis were actively managed or experienced natural disturbance not long before the measurements were taken, we were not able to account for the management and disturbance history of the inventory plots in our analysis. Thus, one type of STM state class that was not covered by inventory data was postdisturbance state classes, which were included in all OR/WA STMs (but generally not in AZ/NM STMs). For the postdisturbance state classes in OR/WA, we used a set of expert-developed post-wildfire fuel beds from similar vegetation types (R. Ottmar, Central Oregon and Okanogan-Wenatchee Fuel Succession Pathways, http://www.fs.fed.us/pnw/fera/fccs/applications/oakwen.shtml) to represent the postdisturbance state classes (see table 3.3 for an example list of postdisturbance state classes and representative fuel beds for the WEC).

Table 3.2—State classes (combinations of vegetation cover and structure) included in state-and-transition models for the Washington East Cascades modeling zone (excluding postdisturbance state classes; see table 3.4), and number of inventory plots classified into and corresponding fuel beds built for each state class

Potential vegetation type	Cover type[a]	Size class[b]	Canopy density	Canopy layers	Number of fuel beds
Dry mixed conifer	Douglas-fir/ grand fir	Grass/forb	NA	NA	0
		Seedling/ sapling	Low	NA	1
		Pole	Low	Single	7
			Medium	Single	4
		Small	Low	Single	6
				Multiple	5
			Medium	Single	0
				Multiple	23
			High	Single	19
				Multiple	34
		Medium	Low	Single	10
				Multiple	7
			Medium	Single	0
				Multiple	25
			High	Single	0
				Multiple	46
		Large	Low	Single	13
				Multiple	9
			Medium	Single	0
				Multiple	61
			High	Single	0
				Multiple	94
		Giant	Low	Single	6
				Multiple	6
	Ponderosa pine	Grass/forb	NA	NA	50
		Seedling/ sapling	Low	NA	8
		Pole	Low	Single	9
			Medium	Single	1
		Small	Low	Single	10
				Multiple	4
			Medium	Single	2
				Multiple	7
		Medium	Low	Single	9
				Multiple	4
			Medium	Single	0
				Multiple	17
		Large	Low	Single	12
				Multiple	7
			Medium	Single	0
				Multiple	7
		Giant	Low	Single	2
				Multiple	5

Table 3.2—State classes (combinations of vegetation cover and structure) included in state-and-transition models for the Washington East Cascades modeling zone (excluding postdisturbance state classes; see table 3.4), and number of inventory plots classified into and corresponding fuel beds built for each state class (continued)

Potential vegetation type	Cover type[a]	Size class[b]	Canopy density	Canopy layers	Number of fuel beds
Dry pine	Ponderosa pine	Grass/forb	NA	NA	72
		Seedling/sapling	Low	NA	6
		Pole	Low	Single	33
			Medium	Single	25
		Small	Low	Single	39
				Multiple	41
			Medium	Single	3
				Multiple	61
		Medium	Low	Single	58
				Multiple	58
			Medium	Single	6
				Multiple	46
		Large	Low	Single	75
				Multiple	60
			Medium	Single	0
				Multiple	40
		Giant	Low	Single	21
				Multiple	8
Moist mixed conifer	Grand fir	Grass/forb	NA	NA	0
		Seedling/sapling	Low	NA	10
		Pole	Low	Single	10
			Medium	Single	14
			High	Single	27
		Small	Low	Single	11
				Multiple	0
			Medium	Single	0
				Multiple	14
			High	Single	0
				Multiple	66
		Medium	Low	Single	8
				Multiple	3
			Medium	Single	1
				Multiple	27
			High	Single	0
				Multiple	108
		Large	Low	Single	2
				Multiple	8
			Medium	Single	0
				Multiple	45
			High	Single	1
				Multiple	173
		Giant	Low	Single	1
				Multiple	1

Table 3.2—State classes (combinations of vegetation cover and structure) included in state-and-transition models for the Washington East Cascades modeling zone (excluding postdisturbance state classes; see table 3.4), and number of inventory plots classified into and corresponding fuel beds built for each state class (continued)

Potential vegetation type	Cover type[a]	Size class[b]	Canopy density	Canopy layers	Number of fuel beds
			Medium	Single	0
				Multiple	11
			High	Single	0
				Multiple	81
	Ponderosa pine	Grass/forb	NA	NA	23
		Seedling/sapling	Low	NA	9
		Pole	Low	Single	19
			Medium	Single	7
			High	Single	2
		Small	Low	Single	5
				Multiple	3
			Medium	Single	2
				Multiple	11
			High	Single	0
				Multiple	9
		Medium	Low	Single	5
				Multiple	4
			Medium	Single	0
				Multiple	7
			High	Single	0
				Multiple	10
		Large	Low	Single	6
				Multiple	2
			Medium	Single	0
				Multiple	14
			High	Single	0
				Multiple	9
		Giant	Low	Single	2
				Multiple	0
			Medium	Single	0
				Multiple	1
			High	Single	0
				Multiple	3
Mountain hemlock	Lodgepole pine	Grass/forb	NA	NA	0
		Seedling/sapling	Low	NA	0
		Pole	Low	Single	11
			Medium	Single	20
			High	Single	28
		Small	Low	Single	5
			Medium	Single	0
			High	Single	0
	Mountain hemlock	Grass/forb	NA	NA	23

Table 3.2—State classes (combinations of vegetation cover and structure) included in state-and-transition models for the Washington East Cascades modeling zone (excluding postdisturbance state classes; see table 3.4), and number of inventory plots classified into and corresponding fuel beds built for each state class (continued)

Potential vegetation type	Cover type[a]	Size class[b]	Canopy density	Canopy layers	Number of fuel beds
		Seedling/ sapling	Low	NA	9
		Pole	Low	Single	16
			Medium	Single	3
		Small	Low	Single	5
			Medium	Single	1
				Multiple	10
		Medium	Low	Single	5
			Medium	Single	0
				Multiple	22
			High	Multiple	378
		Large	Low	Single	4
			Medium	Single	1
				Multiple	14
Oak/pine	Grass/shrub	Open shrub	Low	NA	1
	Oregon white oak/ponderosa pine	Pole	Low	Single	2
			Medium	Single	0
		Small	Low	Single	4
			Medium	Multiple	0
		Medium	Low	Single	0
			Medium	Multiple	0
Pacific silver fir	Pacific silver fir mix	Grass/forb	NA	NA	11
		Seedling/ sapling	Low	NA	22
		Pole	High	Single	28
		Small	Medium	Single	18
			High	Single	0
		Medium	Medium	Single	0
			High	Single	1
				Multiple	113
		Large	Medium	Single	0
			High	Multiple	158
		Giant	Medium	Single	0
			High	Multiple	117
Subalpine parkland	Subalpine fir	Grass/forb	NA	NA	5
		Seedling/ sapling	Low	NA	2
		Pole	Medium	Single	5
		Small	Medium	Single	1
		Medium	Medium	Single	0

NA = not applicable.
[a] Cover type was determined for each plot using calculated forest type importance value.
[b] Size class was determined using quadratic mean diameter (0 = grass/forb; <13 cm DBH = seedling/sapling; 13 to 25 cm DBH = pole; 25 to 38 cm DBH = small; 38 to 51 cm DBH = medium; 51 to 76 cm DBH = large; >76 cm DBH = giant). Plots with canopy density of <10 percent were classified as grass/forb, while plots with canopy density of 10 to 40 percent were classified as low, 40 to 60 percent medium, and >60 percent high.

Table 3.3—Postdisturbance state classes in state-and-transition models for the Washington East Cascades modeling zone, and brief descriptions of expert-based postdisturbance fuel beds used to represent the postdisturbance state classes[a]

Potential vegetation type	Cover type	Tree size class	Representative fuel bed description
Dry mixed conifer	Douglas-fir/grand fir	Grass/forb	Douglas-fir, ponderosa pine, grand fir (dry site) (post-wildfire)
		Seedling/sapling	Douglas-fir, ponderosa pine, grand fir (dry site) (post-wildfire)
		Pole	Douglas-fir, ponderosa pine, grand fir (dry site) (post-wildfire)
		Small	Douglas-fir, ponderosa pine, grand fir (dry site) (post-wildfire)
		Medium	Douglas-fir, ponderosa pine, grand fir (dry site) (post-wildfire)
		Large	Douglas-fir, ponderosa pine, grand fir (dry site) (post-wildfire)
	Ponderosa pine	Grass/forb	Ponderosa pine (post-wildfire)
		Seedling/sapling	Ponderosa pine (post-wildfire)
		Pole	Ponderosa pine (post-wildfire)
		Small	Ponderosa pine (post-wildfire)
		Medium	Ponderosa pine (post-wildfire)
		Large	Ponderosa pine (post-wildfire)
		Giant	Ponderosa pine (post-wildfire)
Dry pine	Ponderosa pine	Grass/forb	Ponderosa pine (post-wildfire)
		Seedling/sapling	Ponderosa pine (post-wildfire)
		Pole	Ponderosa pine (post-wildfire)
		Small	Ponderosa pine (post-wildfire)
		Medium	Ponderosa pine (post-wildfire)
		Large	Ponderosa pine (post-wildfire)
		Giant	Ponderosa pine (post-wildfire)
Moist mixed conifer	Grand fir	Grass/forb	Douglas-fir, grand fir (post-wildfire)
		Seedling/sapling	Douglas-fir, grand fir (post-wildfire)
		Pole	Douglas-fir, grand fir (post-wildfire)
		Small	Douglas-fir, grand fir (post-wildfire)
		Medium	Douglas-fir, grand fir (post-wildfire)
		Large	Douglas-fir, grand fir (post-wildfire)

Table 3.3—Postdisturbance state classes in state-and-transition models for the Washington East Cascades modeling zone, and brief descriptions of expert-based postdisturbance fuel beds used to represent the postdisturbance state classes[a] (continued)

Potential vegetation type	Cover type	Tree size class	Representative fuel bed description
	Ponderosa pine	Grass/forb	Ponderosa pine, Douglas-fir, western larch (post-wildfire)
		Seedling/sapling	Ponderosa pine, Douglas-fir, western larch (post-wildfire)
		Pole	Ponderosa pine, Douglas-fir, western larch (post-wildfire)
		Small	Ponderosa pine, Douglas-fir, western larch (post-wildfire)
		Medium	Ponderosa pine, Douglas-fir, western larch (post-wildfire)
		Large	Ponderosa pine, Douglas-fir, western larch (post-wildfire)
		Giant	Ponderosa pine, Douglas-fir, western larch (post-wildfire)
Mountain hemlock	Lodgepole pine	Grass/forb	Lodgepole pine (post-wildfire)
		Seedling/sapling	Lodgepole pine (post-wildfire)
	Mountain hemlock	Grass/forb	Mountain hemlock
		Seedling/sapling	Mountain hemlock
Pacific silver fir	Pacific silver fir mix	Grass/forb	Western hemlock, Pacific silver fir, mountain hemlock (post-wildfire)
		Seedling/sapling	Western hemlock, Pacific silver fir, mountain hemlock (post-wildfire)
		Small	Western hemlock, Pacific silver fir, mountain hemlock (post-wildfire)
		Medium	Western hemlock, Pacific silver fir, mountain hemlock (post-wildfire)
		Large	Western hemlock, Pacific silver fir, mountain hemlock (post-wildfire)
		Giant	Western hemlock, Pacific silver fir, mountain hemlock (post-wildfire)

[a] All representative fuel beds for the Washington East Cascades were from the Okanogan-Wenatchee Fuel Succession Pathways project (http://www.fs.fed.us/pnw/fera/fccs/applications/oakwen.shtml).

We used inventory plot data to calculate the variables to build one or more FCCS fuel beds for each STM state class.

Building Fuel Beds

The FCCS defines a fuel bed as the inherent physical characteristics of fuels that contribute to fire behavior and effects (Riccardi et al. 2007a). The FCCS fuel beds are stratified into six strata that represent every fuel element that has the potential to combust, including canopy, shrubs, nonwoody fuels, woody fuels, litter, lichen, moss, and ground fuels (fig. 3.4). We used inventory plot data to calculate the variables to build one or more FCCS fuel beds for each STM state class (fuel bed variables listed in table 3.4). We chose to construct one fuel bed for each inventory plot and used one to many plots to represent each state class, rather than statistically summarizing data from multiple inventory plots to construct a single composite fuel bed for each state class, because we wanted to use real world fuels assemblages to represent the state classes instead of creating a composite fuel bed that may not exist under natural conditions.

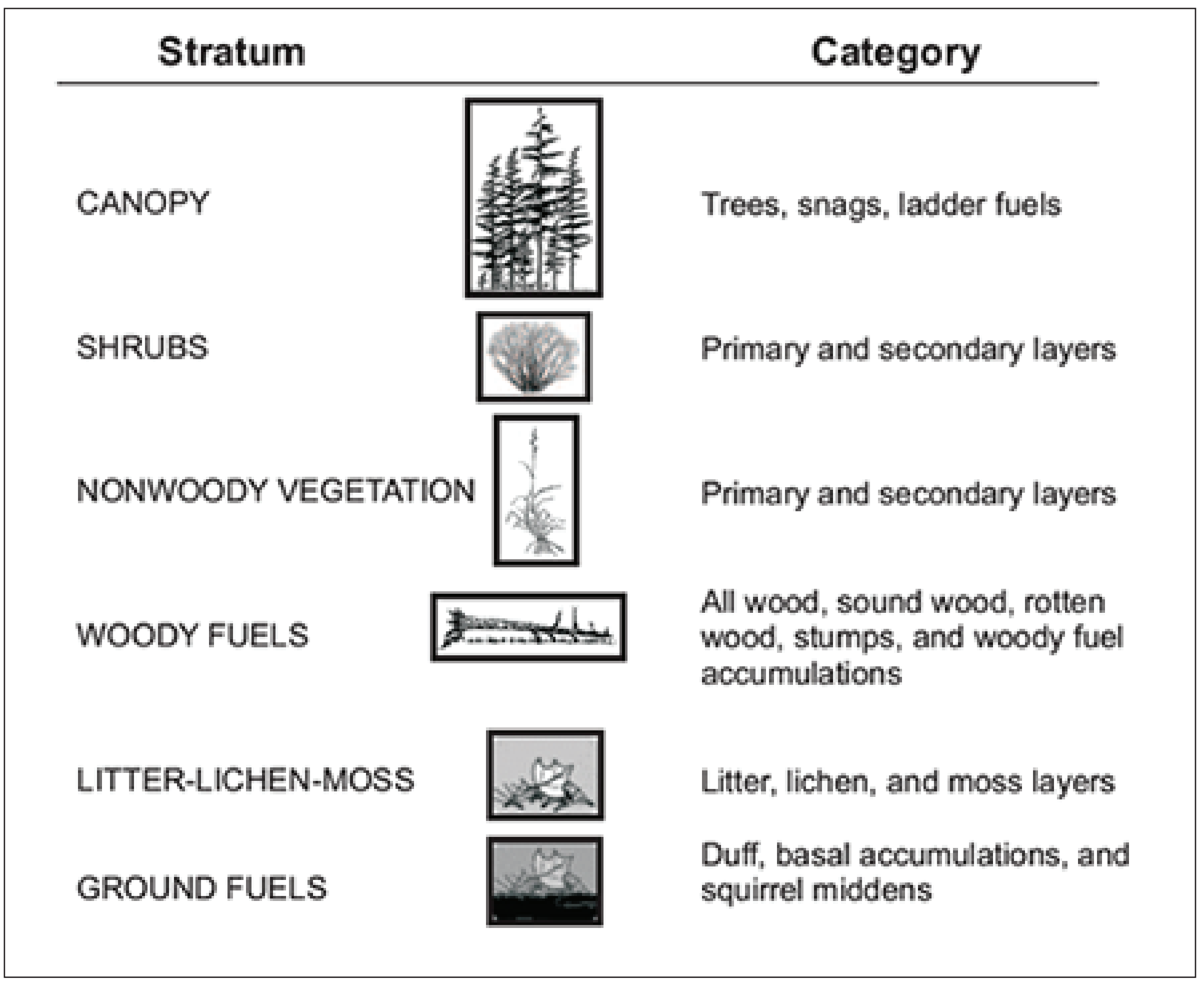

Figure 3.4—Strata and categories of vegetation information included in Fuel Characteristic Classification System (FCCS) fuel beds. We summarized FCCS fuel bed attributes for each state class (vegetation cover and structure combinations) in state-and-transition models of vegetation growth and dynamics. (Figure from Ottmar et al. 2007.)

Table 3.4—Fuel Characteristic Classification System fuel bed variables used in the calculation of physical characteristics and properties of wildland fuels[a]

Fuel strata	Category	Subcategory	Variable
Canopy	Total canopy		Percentage cover
	Trees	Overstory	Percentage cover
		Midstory	Height (m)
		Understory	Height to live crown (m)
			Density (number of stems/ha)
			Diameter at breast height (cm)
			Species and relative cover (%)
	Snags	Class 1 with foliage	Density (number of stems/ha)
		Class 1 without foliage	Diameter (cm)
		Class 2	Height (m)
		Class 3	Species and relative cover (%)
	Ladder fuels	Arboreal lichens and moss	Minimum height (m)
	`	Climbing ferns and other epiphytes	Maximum height (m)
		Dead branches	Is there vertical continuity sufficient to carry fire between the canopy and lower fuel strata? (yes/no)
		Leaning snags	
		Stringy or fuzzy bark	
		Tree regeneration	
		Vines–liana	
Shrub	Primary layer		Percentage cover
	Secondary layer		Height (m)
			Percentage live
			Species and relative cover (%)
	Needle drape		Is needle drape on shrubs sufficient to affect fire behavior? (yes/no)
Nonwoody fuels	Primary layer		Percentage cover
	Secondary layer		Height (m)
			Percentage live
			Loading (tons/ha)
			Species and relative cover (%)
Woody fuels	All woody		Total percentage cover
			Depth (m)
	Sound wood	All sound wood Size classes— <0.6 cm, >0.6 to 2.5 cm, >2.5 to 7.6 cm, >7.6 to 22.9 cm, >22.9 to 50.8 cm, >50.8 cm	For >7.6 cm sound wood— Species and relative cover (%) For size classes— Loading (Mg/ha)
	Rotten wood	All rotten wood Size classes— >7.6 to 22.9 cm, >22.9 to 50.8 cm, >50.8 cm	For all rotten wood— Species and relative cover (%) For size classes— Loading (Mg/ha)

Table 3.4—Fuel Characteristic Classification System fuel bed variables used in the calculation of physical characteristics and properties of wildland fuels[a] (continued)

Fuel strata	Category	Subcategory	Variable
	Stumps	Sound Rotten Lightered-pitchy	Density (number of stumps/ha) Diameter (cm) Height (m) Species and relative cover (%)
	Woody fuel accumulation	Piles Jackpots Windrows	Width (m) Length (m) Height (m) Density (number of accumulations/ha)
Litter-lichen-moss	Litter	Arrangement— Fluffy, normal, perched Type— Short needle pine, long needle pine, other conifer, broadleaf deciduous, broadleaf evergreen, palm frond, grass	For overall litter— Depth (cm) Percentage cover For each litter type— Relative cover (%)
	Lichen	None	Depth (cm)
	Moss	Type— Spaghnum, other moss	Percentage cover Depth (cm)
Ground fuels	Duff	Percentage rotten wood Upper layer— Partially decomposed dead moss and litter, partially decomposed sphagnum moss and sedge Lower layer— Fully decomposed dead moss and litter, fully decomposed sphagnum moss and sedge	For percentage rotten wood— Percentage cover For duff layers— Depth (cm) Percentage cover
	Squirrel middens	None	Depth (cm) Radius (m) Density (number of middens/ha)
	Basal accumulations	Type— Bark slough, branches, broadleaf deciduous, broadleaf evergreen, grass, needle litter, palm fronds	Depth (cm) Radius (m) Percentage of trees affected

[a] Adapted from Prichard et al. 2010b.

Inventory data did not include all of the potential inputs for FCCS, including information on ladder fuels, needle drape, fine (0 to 7.6 cm) woody fuel depth, woody fuel accumulations (piles, jackpots, windthrows), stumps, litter, lichens, moss, and ground fuels (duff, squirrel middens, and basal accumulations). With the exception of fine woody fuel depth, missing variables were not required by FCCS to calculate fire potentials for a fuel bed and were omitted. Fine woody fuel depth was estimated for each fuel bed as described below. Owing to a lack of inventory plot information that would indicate otherwise, we also assumed 100 percent of the shrub and nonwoody cover was live and assumed that only a primary shrub and nonwoody layer was present.

Fine woody fuel depth was estimated from formulas developed for the Forest Vegetation Simulator Fire and Fuels Extension (Reinhardt and Crookston 2003; see addendum). Formulas used fuel loading variables for which we had information (e.g., 10-hour fuels (diameter 0.64 to 2.54 cm), and 100-hour fuels (diameter 2.5 to 7.6 cm). The exception was 1-hour (diameter <0.64 cm) fuel loading information, which was estimated by matching inventory plots to timber-understory and timber-litter fire behavior fuel models in Scott and Burgan (2005) based on 10-hour, 100-hour, live herb, and live woody fuel loads, and using the 1-hour fuel load value from the matching fire behavior fuel model.

In some cases, categories for variables in FCCS did not match those of our information sources, so we had to reclassify accordingly. For example, FCCS requires information on snag decay class and uses a four-category system, including class 1 with foliage, class 1 without foliage, class 2, and class 3. The FIA (annual) and CVS programs use a five-class system and do not collect information on snag foliage. Therefore, based on the descriptions for both classification systems (Cline et al. 1980 for FIA/CVS classification, Prichard et al. 2010b for FCCS classification), we cross-walked the categories between the two systems (FIA/CVS class 1 with FCCS class 1 without foliage, FIA/CVS class 2 with FCCS class 2, and FIA/CVS class 3–5 with FCCS class 3). Similarly, FCCS differentiates between two categories of down wood (sound and rotten). The FIA protocol similarly differentiates between sound and rotten down wood. However, with CVS protocols, down wood was put into one of three decay classes. Therefore, based on the description of the CVS classification system in the metadata for the database, we classified decay classes 1 and 2 in the FCCS sound category and decay class 3 in the FCCS rotten category.

Inventory data were used to calculate many of the variables needed for FCCS. For example, percentage cover of fine woody fuels was calculated from inventory plot data using equations described in Woodall and Monleon (2009). Also, total canopy cover and average height of overstory, midstory, and understory canopy layers were calculated using equations from the Forest Vegetation Simulator (Crookston and Stage 1999).

Calculated variables from inventory plot data were used to build 10,581 fuel beds in OR/WA and 3,604 fuel beds for AZ/NM. These fuel beds were analyzed in FCCS as described below.

We used FCCS to analyze fire hazard of fuel beds constructed from inventory plot data.

Fire Hazard Analysis

We used FCCS to analyze fire hazard of fuel beds constructed from inventory plot data. Based on fuel characteristics, FCCS calculates fire potentials, including surface fire behavior potential (Sandberg et al. 2007a, 2007b), crown fire potential (Schaaf et al. 2007), and available fuel potential (Sandberg et al. 2007a), which rate the intrinsic physical capacity of a fuel bed to support surface fire, crown fire, and consume and smolder fuels, respectively (Ottmar et al. 2007). The FCCS fire potentials are indexed values, scaled between 0 and 9, and are based on default environmental conditions (6.4 km/h for midflame windspeed, and 0, 30, and 60 percent moisture content for the dead, herbaceous, and live moisture contents, respectively) (Sandberg et al. 2007a, 2007b). With user inputs of fuel moisture and windspeed values, FCCS also calculates surface fire behavior outputs including reaction intensity (kJ/m^2), rate of spread (m/s), and flame length (m) (Sandberg 2007b).

To calculate surface fire behavior outputs from FCCS fuel beds, we used fuel moisture and windspeed values that are typical during extreme fire weather in dry forest types (Agee and Lolley 2006, Ager et al. 2010; fuel moistures of 3 percent for 1-hour fuels, 4 percent for 10-hour fuels, 6 percent for 100-hour fuels, 7 percent for 1000-hour fuels (diameter 7.6 to 20.3 cm), 31 percent for nonwoody fuels, 90 percent for shrub and crown fuels, and 20 percent for duff, and a midflame windspeed value of 36 km/h). We did not account for topography within inventory plots (slope was set at zero for FCCS analysis) because we were using inventory plots to represent a general condition (an STM state class) with no defined topography. Increased slope leads to increased fireline intensity and surface fire rate of spread (Rothermel 1983), and thus setting slope at zero for FCCS analysis resulted in lower estimates of surface fire behavior potentials than those that would have resulted if slope were increased.

Summarizing Fuel Bed Information and Linking to State-and-Transition Model Output

We associated calculated FCCS fire behavior and fire potential variables for fuel beds with the appropriate STM state class. Because each STM state class could have multiple fuel beds to represent it, we calculated a mean and standard error for all fire behavior and potential variables for each state class. Once we had calculated means for fuel bed variables for each STM state class, we linked the mean fuel bed information to STM simulation output for each modeling zone to look at trends in fuels and fire potential over time. For spatial displays, we took the area-weighted average of each variable for each modeling stratum (a combination of watershed, potential vegetation type, and ownership-management; see chapter 2). Geodatabases used in this analysis are available at www.WesternLandscapeExplorer.com.

The STMs were run under a fire-suppression-only (FSO) scenario in all modeling zones in both OR/WA and AZ/NM. The FSO scenario was characterized by current levels of fire suppression (i.e., current fire frequency based on a 25-year Monitoring Trends in Burn Severity record for the study area; Eidenshink et al. 2007; see chapter 2 for further detail) but no other land management actions. For the WECs, Oregon East Cascades, Oregon Blue Mountains, and Washington Northeast modeling zones in OR/WA, STMs were also run under a resilience scenario, developed by ILAP to reflect management activities that could be undertaken to increase resilience of dry forests on the east side of the Cascades in Oregon and Washington. Thus, under the resilience scenario, management treatments were focused in the dry forest types, including oak-pine, dry pine, and dry mixed-conifer vegetation types. Prescribed fire was conducted on USDA FS and U.S. Department of the Interior Bureau of Land Management (USDI BLM) lands, excluding federally protected lands such as wilderness, with 1 to 4 percent of the available landscape treated with prescribed fire annually. Thinning from below was also conducted in dry forest types on USDA FS and USDI BLM lands, with annual area treated ranging from 0.005 to 5 percent of high- and medium-density stands and 0.25 to 1 percent of low density stands. On state and tribal lands, thinning from below was conducted on 1.25 to 5 percent of medium and dense forests, while on private industrial lands, 10 percent of available land was treated annually. Planting was conducted across ownerships and management allocations on 2.5 to 20 percent of available lands. Salvage logging was conducted on 5 to 20 percent of available federal lands and 12.5 to 50 percent of available state and tribal lands. See chapter 2 and www.WesternLandscapeExplorer.com for more detail on how scenarios were run.

Results

We built over 14,000 fuel beds characterizing approximately 3,800 vegetation conditions (state classes) in forests and woodlands of Arizona, New Mexico, Oregon, and Washington. A database with complete lists of inventory plots classified into each STM state class, and fuel bed input and output data is located at www.WesternLandscapeExplorer.com. Example fuel bed output for state classes in the WEC is shown in table 3.5, and results for the WEC are discussed further below.

Table 3.5—Mean flame length and crown fire potential (unit-less index) for state classes in the Washington East Cascades state-and-transition models[a]

Potential vegetation type	Cover type	Size class	Canopy density	Canopy layers	Flame length	Crown fire potential
					Meters	
Dry mixed conifer	Douglas-fir/ grand fir	Grass/forb	NA	NA	6.7	2.1
		Postdisturbance	Low	NA	8.2	1.8
		Seedling/sapling	Low	NA	1.3	1.9
		Pole	Low	Single	0.1	1.5
			Medium	Single	0.4	2.4
		Small	Low	Single	0.5	1.3
				Multiple	0.1	1.1
			Medium	Single	0.4	2.4
				Multiple	0.4	2.2
			High	Single	0.5	3.2
				Multiple	0.4	2.8
		Medium	Low	Single	0.4	1.4
				Multiple	0.4	1.3
			Medium	Single	0.4	2.3
				Multiple	0.4	2.3
			High	Single	0.5	3.2
				Multiple	0.6	3.1
		Large	Low	Single	0.7	1.7
				Multiple	0.5	1.6
			Medium	Single	0.6	2.6
				Multiple	0.6	2.6
			High	Single	0.9	3.5
				Multiple	0.4	3.2
		Giant	Low	Single	0.5	1.1
				Multiple	0.5	1.4
	Ponderosa pine	Grass/forb	NA	NA	0.7	0.4
		Postdisturbance	Low	Single	3.4	1.6
		Seedling/sapling	Low	NA	1.1	3.2
		Pole	Low	Single	0.7	1.8
			Medium	Single	1.3	4.0
		Small	Low	Single	0.7	1.8
				Multiple	0.4	1.1
			Medium	Single	0.7	2.8
				Multiple	0.4	2.0
		Medium	Low	Single	0.5	1.5
				Multiple	0.7	1.9
			Medium	Single	0.5	2.1
				Multiple	0.5	2.1
		Large	Low	Single	0.6	1.2
				Multiple	0.4	1.2
			Medium	Single	0.8	2.7
				Multiple	0.8	2.7
		Giant	Low	Single	0.2	0.5
				Multiple	0.5	1.3

Table 3.5—Mean flame length and crown fire potential (unit-less index) for state classes in the Washington East Cascades state-and-transition models[a] (continued)

Potential vegetation type	Cover type	Size class	Canopy density	Canopy layers	Flame length	Crown fire potential
Dry pine	Ponderosa pine	Grass/forb	NA	NA	0.6	0.9
		Postdisturbance	Low	Single	3.4	1.6
		Seedling/sapling	Low	NA	0.4	1.8
		Pole	Low	Single	0.4	1.8
			Medium	Single	0.4	2.2
		Small	Low	Single	0.5	1.6
				Multiple	0.4	1.4
			Medium	Single	0.3	1.6
				Multiple	0.5	2.3
		Medium	Low	Single	0.4	1.3
				Multiple	0.4	1.5
			Medium	Single	0.2	1.3
				Multiple	0.3	2.1
		Large	Low	Single	0.5	1.4
				Multiple	0.3	1.4
			Medium	Single	0.2	1.3
				Multiple	0.4	2.5
		Giant	Low	Single	0.4	1.4
				Multiple	0.4	1.6
Moist mixed conifer	Grand fir	Grass/forb	NA	NA	2.0	2.1
		Postdisturbance	Low	Single	2.4	2.3
		Seedling/sapling	Low	NA	0.4	2.5
		Pole	Low	Single	0.6	2.3
			Medium	Single	0.9	3.3
			High	Single	0.7	3.5
		Small	Low	Single	1.0	2.5
				Multiple	0.2	2.0
			Medium	Single	0.7	2.1
				Multiple	0.4	2.4
			High	Single	0.7	3.5
				Multiple	0.6	3.8
		Medium	Low	Single	0.3	1.4
				Multiple	0.2	2.0
			Medium	Single	0.7	2.1
				Multiple	0.5	2.8
			High	Single	0.9	3.5
				Multiple	0.5	3.5
		Large	Low	Single	1.1	1.4
				Multiple	0.6	1.4
			Medium	Single	0.6	2.8
				Multiple	0.6	2.8
			High	Single	0.9	3.5
				Multiple	0.5	3.7
		Giant	Low	Single	1.0	2.6
				Multiple	0.8	2.6

Table 3.5—Mean flame length and crown fire potential (unit-less index) for state classes in the Washington East Cascades state-and-transition models[a] (continued)

Potential vegetation type	Cover type	Size class	Canopy density	Canopy layers	Flame length	Crown fire potential
			Medium	Single	0.7	2.7
				Multiple	0.7	2.7
			High	Single	0.7	3.5
				Multiple	0.7	3.5
	Ponderosa pine	Grass/forb	NA	NA	0.7	0.6
		Postdisturbance	NA	NA	0.8	1.9
		Seedling/sapling	Low	NA	1.1	2.4
		Pole	Low	Single	0.8	2.5
			Medium	Single	0.5	2.7
			High	Single	0.1	3.7
		Small	Low	Single	1.0	2.4
				Multiple	1.2	2.1
			Medium	Single	1.0	2.7
				Multiple	0.7	2.4
			High	Single	0.2	2.9
				Multiple	0.2	2.9
		Medium	Low	Single	0.5	1.7
				Multiple	0.7	2.3
			Medium	Single	1.0	2.7
				Multiple	0.1	2.6
			High	Single	0.2	2.9
				Multiple	0.2	3.3
		Large	Low	Single	0.7	1.6
				Multiple	0.5	2.2
			Medium	Single	1.2	3.3
				Multiple	1.2	3.3
			High	Single	0.9	3.5
				Multiple	0.6	3.6
		Giant	Low	Single	0.5	0.9
				Multiple	0.5	2.2
			Medium	Single	1.2	3.3
				Multiple	0.5	1.8
			High	Single	0.9	3.5
				Multiple	0.5	2.4
Mountain hemlock	Lodgepole pine	Grass/forb	NA	NA	1.8	4.0
		Postdisturbance	NA	NA	1.8	4.0
		Seedling/sapling	Low	NA	0.1	1.0
		Pole	Low	Single	0.1	1.0
			Medium	Single	0.3	2.4
			High	Single	0.4	3.1
		Small	Low	Single	0.4	1.4
			Medium	Single	0.3	2.4
			High	Single	0.4	3.1

Table 3.5—Mean flame length and crown fire potential (unit-less index) for state classes in the Washington East Cascades state-and-transition models[a] (continued)

Potential vegetation type	Cover type	Size class	Canopy density	Canopy layers	Flame length	Crown fire potential
	Mountain hemlock	Grass/forb	NA	NA	0.5	0.6
		Postdisturbance	NA	NA	10.6	3.9
		Seedling/sapling	Low	NA	0.4	2.2
		Pole	Low	Single	0.7	2.0
			Medium	Single	0.8	3.4
		Small	Low	Single	0.4	1.7
			Medium	Single	0.8	4.1
				Multiple	0.4	2.5
		Medium	Low	Single	0.4	1.2
			Medium	Single	0.5	2.8
				Multiple	0.5	2.8
			High	Multiple	0.4	4.2
		Large	Low	Single	0.4	1.6
			Medium	Single	1.4	3.7
				Multiple	0.5	3.1
Oak/pine	Grass shrub	Open shrub	Low	NA	0.9	1.5
	Oregon white oak/ ponderosa pine	Pole	Low	Single	0.9	3.1
			Medium	Single	0.9	3.1
		Small	Low	Single	0.2	0.7
			Medium	Multiple	0.2	0.7
		Medium	Low	Single	0.2	0.7
			Medium	Multiple	0.2	0.7
Pacific silver fir	Pacific silver fir mix	Grass/forb	NA	NA	0.8	1.1
		Postdisturbance	NA	NA	5.8	3.5
		Seedling/sapling	Low	NA	0.7	2.7
		Pole	High	Single	0.4	3.7
		Small	Medium	Single	0.6	3.1
			High	Single	0.4	3.7
		Medium	Medium	Single	0.6	3.1
			High	Single	0.6	2.7
				Multiple	0.3	3.7
		Large	Medium	Single	0.6	2.6
			High	Multiple	0.4	3.8
		Giant	Medium	Single	0.6	2.6
			High	Multiple	0.6	4.0
Subalpine parkland	Subalpine fir	Seedling/sapling	Low	NA	0.4	2.5
		Pole	Medium	Single	0.5	2.9
		Small	Medium	Single	0.1	3.4
		Medium	Medium	Single	0.1	3.4
		Grass/forb	NA	NA	0.1	0.8

NA = not applicable.

[a] Inventory plots were classified into each state class, fuel beds were built from inventory plot data, and fuel beds were analyzed for potential fire behavior in the Fuel Characteristic Classification System. Each state class is represented by at least one inventory plot, and thus values represent the means for fuel beds classified into each state class.

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For the WEC modeling zone (figs. 3.2 and 3.3), we focused on two indicators of fire hazard: crown fire potential and flame length. Crown fire potential gives an indication of the potential for fire to spread into, and propagate through, the canopy of forests and woodlands. The FCCS crown fire potential index is based on whether the energy supplied by a surface fuel bed layer is sufficient to ignite and sustain fire spread in the canopy. Flame length, or the distance from the ground at the leading edge of the flame to the tip of the flame, is an indicator of surface fuels and potential surface fire behavior and can be interpreted to determine wildfire suppression strategies (Andrews and Rothermel 1982).

Crown fire potential—

Although the crown fire potential index was >7 (on a scale of 0 to 9) for some individual fuel beds in the WEC, the mean crown fire potential index for any state class in the WEC did not exceed 4.2 (table 3.5). To examine crown fire potential results in the region on a relative scale, we classified the state classes into low, moderate, and high crown fire potential categories by ordering the data based on crown fire potential and separating it into thirds. Although we cannot associate these categories with specific information on potential crown fire intensity and resulting mortality, we can assume that the sites in the high crown fire potential category would have a high likelihood of experiencing stand-replacing fire, while those in the low category would have a relatively low likelihood of experiencing stand-replacement fire.

For both the entire landscape and in the dry forest types (dry mixed conifer and dry ponderosa pine) crown fire potential increased over time under the FSO scenario but decreased over time under the resilience scenario.

For both the entire landscape and in the dry forest types (dry mixed conifer and dry ponderosa pine) crown fire potential increased over time under the FSO scenario but decreased over time under the resilience scenario (figs. 3.5 and 3.6). For the entire landscape under the FSO scenario, area in the high crown fire potential category increased by 61 163 ha (from 977 549 ha in 2007 to 1 038 711 ha in 2056), but under the resilience scenario, area in the high crown fire potential category decreased by 110 341 ha (from 966 336 ha in 2007 to 855 995 ha in 2056; fig. 3.5). In addition, in 2056, area in the low crown fire potential category was lower under the FSO scenario compared to the resilience scenario (141 936 ha versus 279 843; fig. 3.5).

In the dry mixed conifer and dry ponderosa pine types, which are the vegetation types in which fuel treatments such as thinning and prescribed fire are focused in the WEC, crown fire potential increased over time under the FSO scenario, with increases in area in the high crown fire potential category (from 211 087 ha in 2007 to 281 094 ha in 2056) and moderate crown fire potential category (from 132 484 ha in 2007 to 173 130 ha in 2056) and a decrease in area in the low crown fire potential category (from 228 247 ha in 2007 to 117 586 ha in 2056; fig. 3.6). Crown

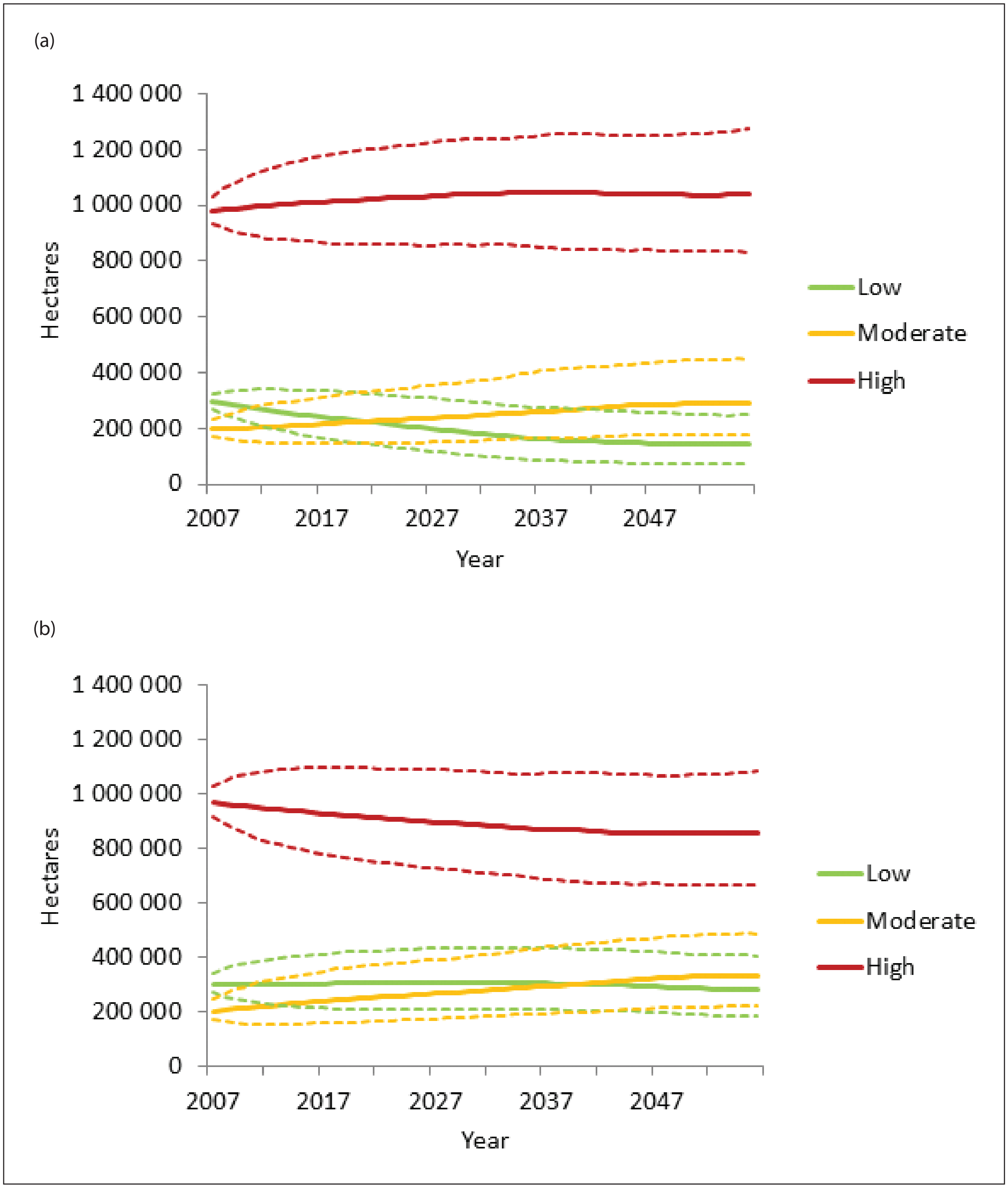

Figure 3.5—Area of the Washington East Cascades landscape in low, moderate, and high crown fire potential categories under (a) fire suppression only, and (b) resilience scenarios, as simulated by state-and-transition models. Dashed lines represent minimum and maximum area in each category across Monte Carlo simulations. Crown fire potential was assessed for model state classes by using inventory plots to represent each state class, building fuel beds with the inventory plot data, and analyzing the fuel beds in the Fuel Characteristic Classification System.

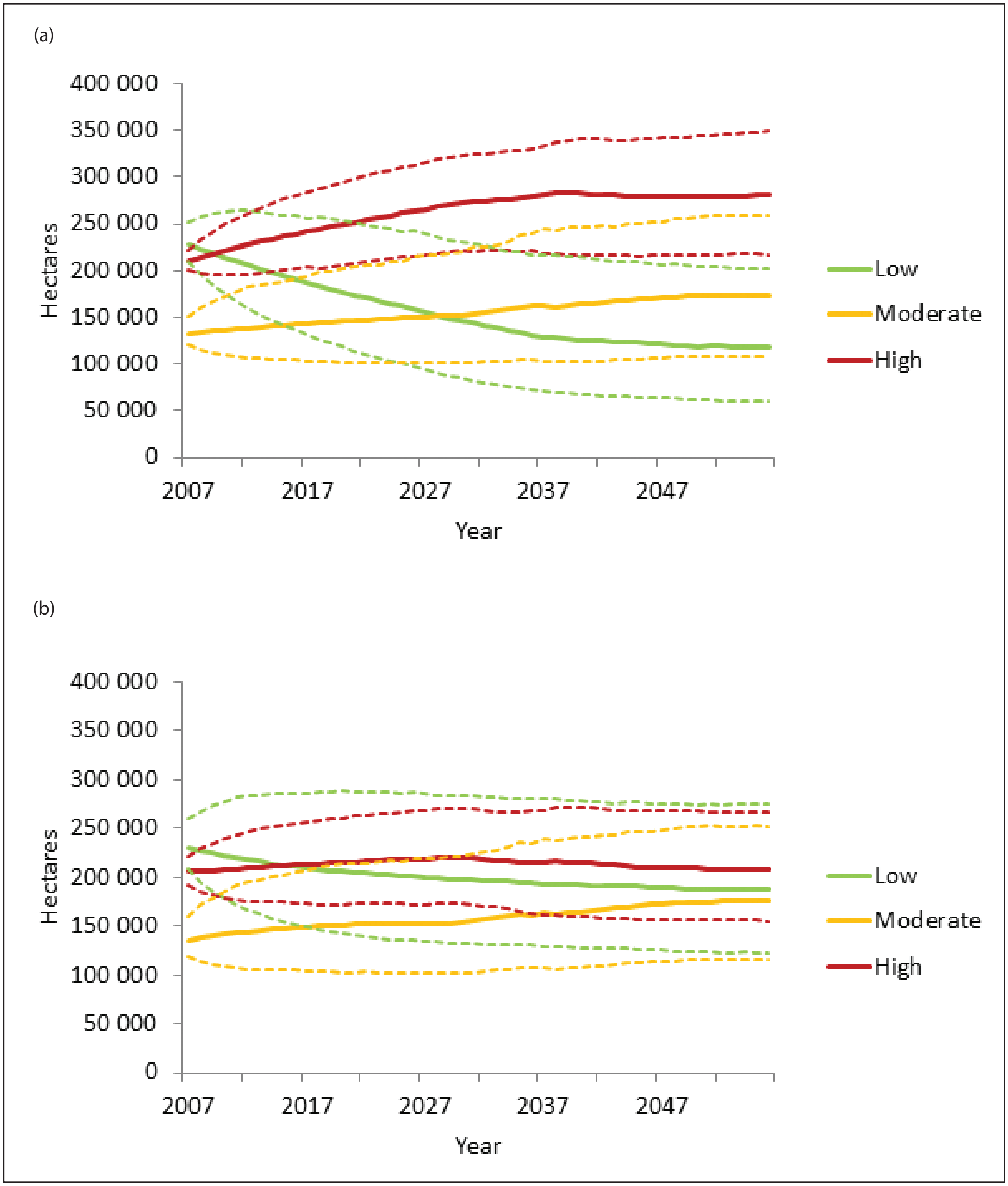

Figure 3.6—Area of the dry mixed conifer and dry pine potential vegetation types in low, moderate, and high crown fire potential categories under (a) fire suppression only, and (b) resilience scenarios, as simulated by state-and-transition models for the Washington East Cascades landscape. Dashed lines represent minimum and maximum area in each category across Monte Carlo simulations.

fire potential also increased over time under the resilience scenario, but not to the same degree as under the FSO scenario. Under the resilience scenario, area in the high crown fire potential category remained relatively constant (206 309 ha in 2007 versus 208 377 ha in 2056), while area in the moderate category increased (from 134 950 ha in 2007 to 175 384 ha in 2056), and area in the low category decreased (from 230 552 ha in 2007 to 188 055 ha in 2056; fig. 3.6).

These patterns were also apparent when results were examined spatially (fig. 3.7). Although crown fire potential was not substantially reduced compared to current conditions in the resilience scenario, crown fire potential was lower under the resilience scenario than under the FSO scenario by 2056, particularly in the lower elevation dry forest types where fuel treatments are focused (generally the southeastern and eastern portions of the study area; see dry pine, dry mixed-conifer, and moist mixed-conifer potential vegetation types in fig. 3.2).

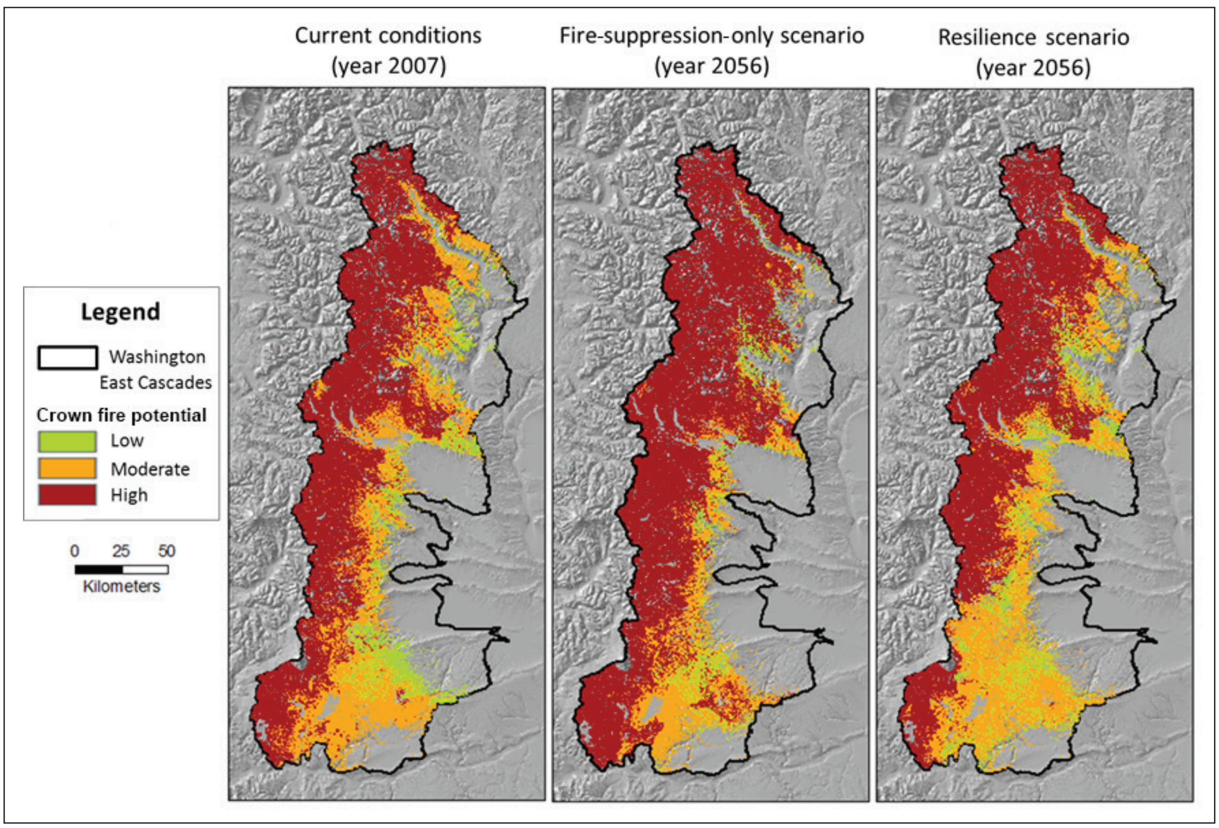

Figure 3.7—Crown fire potential for the Washington East Cascades region for current conditions (2007; left panel), simulated 2056 conditions under a fire-suppression-only (FSO) scenario (center panel), and simulated 2056 conditions under a resilience scenario (right panel). Crown fire potential was assessed for state-and-transition model state classes by using inventory plots to represent each state class, building fuel beds with the inventory plot data, and analyzing the fuel beds in the Fuel Characteristic Classification System. The FSO scenario was characterized by current levels of fire suppression (i.e., current fire frequency based on a 25-year Monitoring Trends in Burn Severity record for the study area) but no other land management actions. The resilience scenario was characterized by light to moderate levels of thinning and some prescribed fire in dry forest types. The area-weighted average for crown fire potential index was calculated for each modeling stratum (potential vegetation type, ownership, and land allocation within a watershed), and each modeling stratum was categorized into the low, moderate, and high crown fire potential category for the spatial display.

Flame length—
Patterns in potential flame length did not differ substantially between the FSO and resilience scenarios in the WEC. Under both scenarios, there was an increase in potential flame length over time, with a decrease in area in the low flame length category (<1.2 m), and slight increases in the area in the higher flame length categories (fig. 3.8). These patterns were similar between all lands (fig. 3.8) and the dry forest types (not shown).

Discussion

Management Applications

Our integrated FCCS-STM approach can give land managers information they need to determine the types, extent, and locations of management activities that will result in the greatest reduction in crown fire potential. Acquiring this information is a critical step in the development of successful fuel treatment regimes in fire-prone forests. Fire hazard information can be used alone or in conjunction with other types of information, such as wildlife habitat and economic potential, to prioritize areas for fuel treatments or other types of management activities. Analyses integrating fuels data with data on wildlife habitat, economic potential, and community economics are forthcoming products of ILAP.

The thousands of fuel beds developed for this work cover a broad range of fuel conditions in both OR/WA and AZ/NM. The link between these fuel beds and specific forest and woodland compositional and structural conditions has potential for other types of applications other than fire hazard analysis. For example, fuel beds generated from this work are being used in USDA FS Region 3 (AZ/NM) to assess potential emissions from wildfire using a software program that is complementary to FCCS, called CONSUME (Prichard et al. 2005). In addition, inventory data summaries could be used to assess wildlife habitat suitability in different STM state classes.

Results of the integrated FCCS-STM simulations in the WEC region indicated that a management regime characterized by targeted fuel treatments in dry forest types can lead to lower crown fire potential than a FSO management regime.

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Results of the integrated FCCS-STM simulations in the WEC region indicated that a management regime characterized by targeted fuel treatments in dry forest types can lead to lower crown fire potential than a FSO management regime. High crown fire potential is associated with stand-replacing fire events. Avoiding stand-replacement fire events and maintaining substantial live basal area is often a goal of vegetation management treatments (Agee and Skinner 2005), and our results suggest that the resilience scenario tested in this study would be successful in reducing the likelihood of stand-replacement fire events in dry forests compared to an FSO scenario.

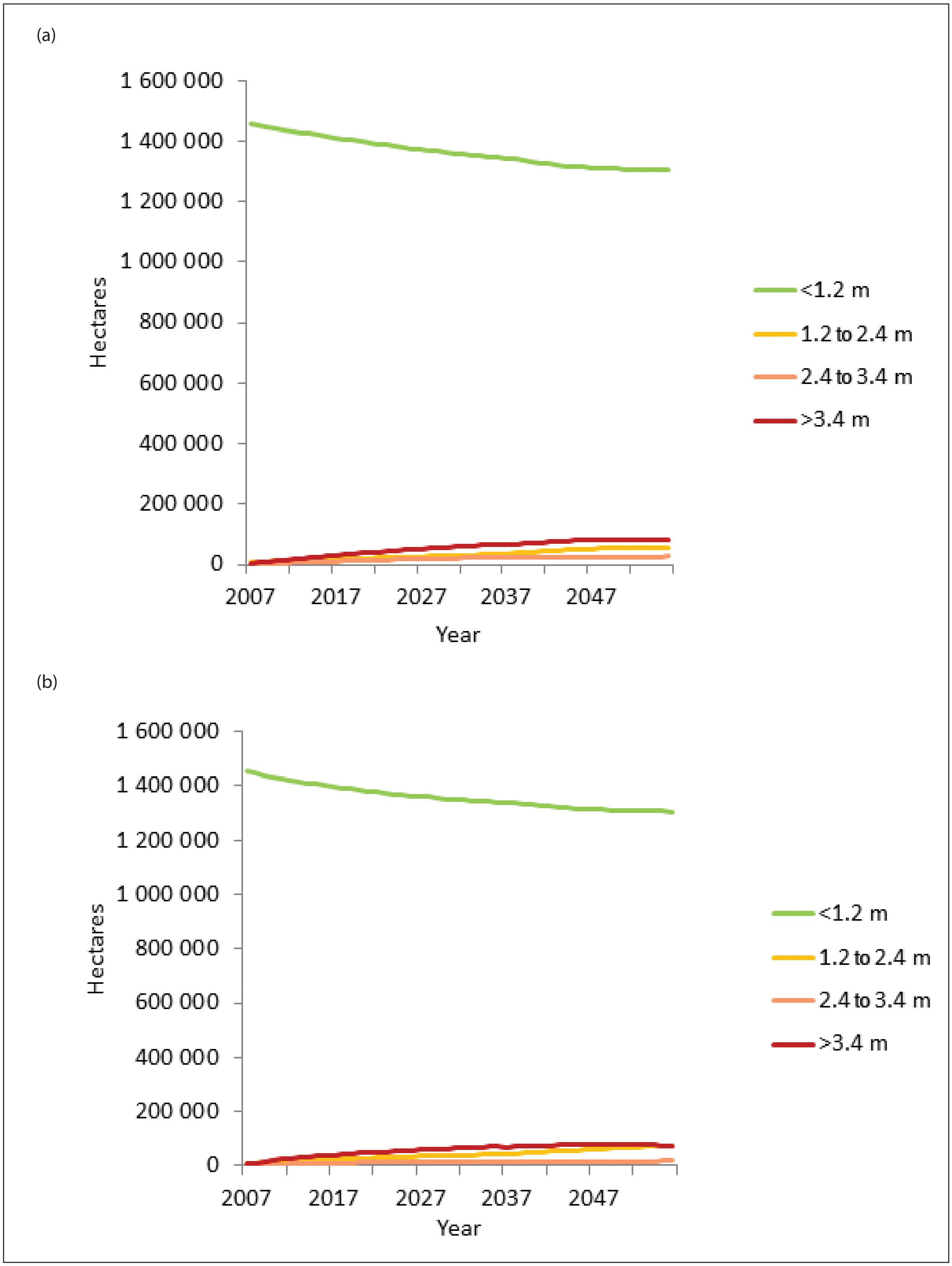

Figure 3.8—Area of the Washington East Cascades landscape in four flame length categories under (A) fire suppression only, and (B) resilience scenarios, as simulated by state-and-transition models. Each flame length category reflects thresholds in ease of fire suppression (Andrews and Rothermel 1982).

Although crown fire potential was reduced under the resilience scenario compared to the FSO scenario in the WEC, resilience activities did not decrease potential flame length in the study area. These results illustrate that forest thinning to reduce tree density, which was the focus of the resilience scenario evaluated in this study, does not necessarily reduce surface fuels and surface fire hazard, which is reflected in flame length (Agee and Skinner 2005). Instead of reducing surface fire hazard, thinning activities can increase surface fuel levels and surface fire intensity (Johnson et al. 2007, Raymond and Peterson 2005), and thus surface fuel treatments are essential to reduce surface fire hazard after thinning treatments (Agee and Skinner 2005).

The flame length results also illustrate a limitation to our approach. Although flame length potential did vary substantially among individual inventory plots used to represent the STM state classes (with flame lengths up to 10 m), mean flame lengths were much less variable (with most below 1.2 m). The state classes in STMs represent a general condition and are based primarily on forest and woodland overstory structure, whereas flame lengths are based largely on surface fuels. In addition, we could not account for the management and disturbance history of inventory plots used to develop fuel beds. Thus, representative plots for each state class had variable surface fuel levels, and this variation obscured patterns when surface fuel variables such as potential flame length were averaged by state class.

Future Improvements and Research Needs

There are several ways this analysis could be improved for future applications. First, although we used the most comprehensive inventory plot data set available in OR/WA (CVS), this data set only covered national forests in the two states. Thus, expanding the analysis to use more recent FIA annual data would help us to cover a broader range of vegetation/fuels conditions than what exists on national forests. We also discovered that inventory plot data for arid lands with sufficient detail for fuel bed characterization are lacking, particularly in Oregon and Washington. Further field-based data will be needed to integrate fuels information with arid lands STMs in the four states. We were also lacking information on postdisturbance fuel conditions, and thus more field-based data on postdisturbance conditions for different vegetation types and structural conditions would improve the estimates of fire hazard for postdisturbance state classes. Finally, to more accurately capture effects of management and disturbance on surface fuels and potential surface fire hazard, our approach would need to be modified. One way to do this would be to differentiate state classes based on surface fuel properties (e.g., include a ponderosa pine, large size class, open canopy, single layer, low surface fuels, and a ponderosa pine, large size class, open canopy, single layer, high surface fuels state class), and incorporate surface fuel characteristics in the plot classification process. Another way would be to do a post-hoc analysis of effects of specific treatments on surface fuel properties using the Forest Vegetation Simulator (Dixon 2003), and use estimates of area affected by those treatments to determine effects on surface fire hazard.

Acknowledgments

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Chapter 4: Overview of the Vegetation Management Treatment Economic Analysis Module in the Integrated Landscape Assessment Project

Xiaoping Zhou and Miles A. Hemstrom[1]

Chapter Summary

Forest land provides various ecosystem services, including timber, biomass, and carbon sequestration. Estimating trends in these ecosystem services is essential for assessing potential outcomes of landscape management scenarios. However, the state-and-transition models used in the Integrated Landscape Assessment Project for simulating landscape changes over time do not directly compute the variables necessary for volume and biomass estimation. This study illustrates a methodology for modeling timber production and biomass supply potential over time in central Washington, and gives an example on cost-benefit analysis of fuel treatment under different management scenarios to provide information on economic viability of potential vegetation management activities.

Introduction

The Integrated Landscape Assessment Project (ILAP) explores the dynamics of broad-scale, multiownership landscapes and develops information that can be used to prioritize management activities over time by evaluating and integrating information on fuel conditions, wildlife and aquatic habitats, economic values, and projected climate change in Arizona, New Mexico, Oregon, and Washington. Products from ILAP will help land managers, planners, and policymakers evaluate management strategies that reduce fire risk, improve habitat, and benefit rural communities.

This chapter documents the fuel treatment economic analysis completed as part of ILAP. This work involved analyzing economic feasibilities of proposed fuel treatments and addressed the following questions under alternative management scenarios: (1) Where might fuel treatments create biomass and timber products, and how much are they likely to create? (2) How much might it cost to harvest the

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timber and transport biomass from specific areas? How does transportation distance affect the financial outcome of fuel treatments? and (3) How might government subsidies and other related policies affect the financial viability of fuel treatments?

The major objectives of this chapter are to (1) document methodology that can be used to estimate the potential timber volume and biomass for alternative landscape management scenarios, (2) describe procedures for economic assessment of vegetation management treatments related to alternative land management scenarios, and (3) present some preliminary results of the fuel treatment economic analysis in a central Washington study area (tools developed and results from this work can be found at www.westernlandscapesexplorer.info).

Study Area

The study area is approximately 1 million ha of mostly forested lands in central Washington (fig. 4.1). It includes 25 hydrologic unit code (HUC) 5 watersheds (USGS and USDA NRCS 2011) and 16 combinations of ownership and land management allocations (see chapter 2). Forested vegetation ranges from very dry environments that support ponderosa pine stands to upper elevation forests of subalpine fir and mountain hemlock. Much of the forest consists of mixed-conifer stands dominated by Douglas-fir, grand fir, and ponderosa pine.

Methods

State-and-Transition Models

We used state-and-transition models (STMs) developed by ILAP (chapter 2) to simulate the effects of natural disturbances and management treatments on forested vegetation. The STM approach treats vegetation as combinations of cover type and structural stages (boxes) within different biophysical environments. Boxes are linked by arrows (transitions) that represent natural disturbances, management actions, or vegetation growth and development. For example, grass/forb herblands might become dominated by small trees and shrubs after a period of time or might remain as grass/forb communities following wildfire. This approach builds on transition matrix methods that represent vegetation development as a set of transition probabilities among various vegetative state classes (e.g., Hann et al.1997, Hemstrom et al. 2007, Keane et al.1996, Laycock 1991, Westoby et al.1989). The STMs were developed in the Vegetation Development Dynamics Tool framework, version 6.0.25, (VDDT; ESSA Technologies Ltd. 2007) and run using the Path modeling platform, version 3.0.4 (Apex Resource Management Solutions 2012, Daniel and Frid 2012).

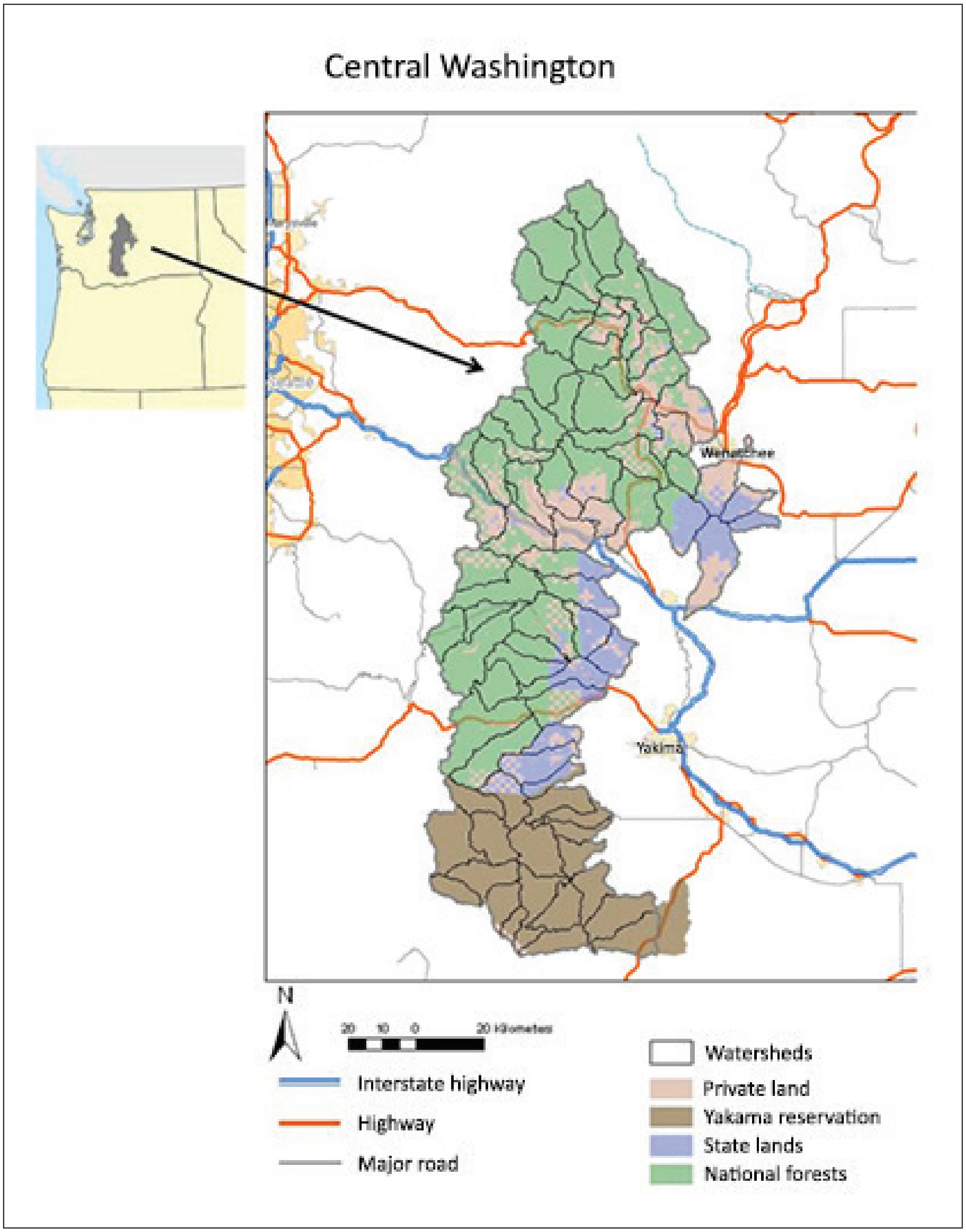

Figure 4.1—Central Washington study area.

Forest structural stages in the STMs were based on tree diameter, overstory canopy cover, and canopy layering (chapter 2). Forest diameter classes were based on the quadratic mean diameter of the dominant and codominant trees: (1) seedling/ sapling (>0 to 13 cm diameter at breast height [DBH]), (2) young (>13 to 25 cm DBH), (3) small (>25 to 38 cm DBH), (4) medium (>38 to 51 cm DBH), (5) large (>51 to 75 cmDBH, and (6) very large (>75 cm DBH). Canopy cover classes were open (0 to 40 percent), medium (41 to 60 percent), and closed (>60 percent). Canopy layering was either single or multiple. A grass-forb condition represented early-seral state classes before establishment of a significant tree canopy. Postdisturbance state classes were also included, representing conditions resulting from a stand-replacing disturbance (insect outbreak or wildfire) prior to any salvage logging and consequently containing abundant large dead wood.

A current vegetation map was developed for the study area (chapter 2). The vegetation was mapped using nearest neighbor imputation techniques based on random forest nearest neighbor (Crookston and Finley 2008) and gradient nearest neighbor (Ohmann and Gregory 2002) methods. A rule set was developed for classifying the current vegetation to STM state classes within each vegetation type. These data became the initial conditions for STM simulations. Burcsu et al. (chapter 2) used these initial conditions to project future vegetation conditions, management activities, and natural disturbances for alternative management scenarios. The resulting vegetation, disturbance, and management activity projections were summarized by time step, potential vegetation, and state class.

Two forest management scenarios were applied in this study. The fire-suppression-only (FSO) scenario assumed that no management treatments were practiced except continued current levels of wildfire suppression. The current management (CM) scenario was based on estimated forest management treatment rates currently in place by land ownership and land management allocation (ownership-management; see chapter 2). Management differed widely among ownerships and management allocations from no treatments (except wildfire suppression) in wilderness and similar reserved areas, to commercial timber harvest on private timberlands. The treatment rates by ownership-management were gathered as part of an ongoing local collaboration. Treatments in the CM scenario included regeneration harvests, commercial thinnings, precommercial thinning, planting, prescribed fire, and mechanical fuel treatments. Importantly, timber harvests from federal lands did not include trees over 53 cm in diameter. Wildfire probabilities were computed by potential vegetation groups (dry, moist, and cold forest) using data from the Monitoring Trends in Burn Severity project (Eidenshink et al. 2007; www.mtbs.gov), and reflect wildfire occurrence in the Washington East Cascades ecological region for the 1984–2008 time period (chapter 2).

Volume and Biomass Look-Up Tables

The STM simulation output does not carry detailed vegetation attributes over time such as tree species, diameter, and height, which are key variables for estimating timber volume and tree biomass. The challenge for users interested in information about timber products and biomass is how to quantify timber production and biomass supply from vegetation management treatments and other disturbances with STM simulation output. The U.S. Department of Agriculture Forest Service (USDA FS) Forest Inventory and Analysis (FIA; USDA FS 2012) plot and tree information were used to build look-up tables with volume and biomass attributes linked to each state class for each ILAP modeling zone (see chapter 2; fig. 2.4). The process involved the following major steps:

1. Gather detailed tree information from the FIA periodic and annual plots that occurred in the study area and in the same potential vegetation in adjacent ILAP modeling zones.
2. Assign each plot to a state class within the study area (chapter 2).
3. Calculate volume and biomass for each plot (Zhou and Hemstrom 2010).
4. Summarize volume and biomass attributes, as well as other desired variables, for each plot by product group or by total.
5. Compute attribute averages for each STM state class and build a look-up table with the computed attributes associated with each state class.
6. Fill in holes for those state classes with no plot samples by using a rule set that substituted the most similar state class with plot samples.

Look-up tables were built in step five for each modeling zone. Volume attributes included in look-up tables were total volume (DBH ≥2.5 cm), merchantable tree volume (DBH ≥12.7 cm), and sawtimber volume (DBH ≥22.9 cm for softwood and DBH ≥27.9 cm for hardwood); biomass attributes include biomass of stem, branch, bark, and leaf. The total biomass is the sum of all these parts. Merchantable biomass is computed separately. Five products were defined in this study based on tree diameters: (1) small tree with DBH <12.7 cm; (2) chip tree with DBH ≥12.7 cm, and <17.8 cm; (3) pole tree with DBH ≥17.8 cm and <22.9 cm for softwood and 27.9 cm for hardwood; (4) small sawtimber, a sawtimber tree with DBH ≥22.9 cm for softwood and 27.9 cm for hardwood and <50.8 cm; and (5) large sawtimber, a sawtimber tree with DBH ≥50.8 cm.

The attributes included in the look-up table were then linked to STM output, and the desired attributes were summarized for each modeling stratum by year.

Vegetation Management Treatment Economic Analysis

Timber products and biomass were estimated annually to examine potential economic outcomes from the two alternative land management scenarios. The process included the estimation of removed materials, harvesting costs, the transportation costs, and the market values of the removed timber, as well as the values of the woody biomass delivered to known mill locations.

Estimation of removals—

Various management activities, such as harvesting and prescribed fires, were applied to achieve ecological objectives and reduce fuel accumulations. The STM simulations forecasted annual area treated with different management activities for each year by modeling stratum and state class. In addition, model output included the transition pathway ("from" state class, and "to" state class) for every management transition that occurred in the simulations. Using this information, removed volume and biomass were calculated based on the difference of volume and biomass in the "from" state class and the "to" state class. Treatment was assumed to have occurred in the middle of the year, and growth was not considered during the treatment year. Merchantable timber volume was calculated for trees with DBH ≥12.7 cm, including pulpwood size trees (chip or pole trees) and sawtimber size trees. The sawtimber volume is the tree volume with DBH greater than sawtimber size (22.9 cm and above for softwood and 27.9 cm and above for hardwood). Aboveground biomass was calculated including stem, bark, branch, top, and leaves of all trees.

Harvesting cost—

The Fuel Reduction Cost Simulator (FRCS; Fight et al. 2006) software was modified and used to estimate harvesting costs, including removal of trees of mixed sizes in the form of whole trees, logs, or chips. In FRCS, equipment production rates were developed from existing studies, and cost assumptions for labor and equipment were modified with the most current available information. The FRCS includes four ground-based systems, four cable systems, and two helicopter systems. The helicopter systems were not included owing to high costs. Harvesting estimates generated by FRCS were in U.S. dollars per 100 ft^3, or per 2.83 m^3. The harvesting costs included five major components: felling (and bunching), skidding, processing, loading, chipping (if applicable), and machine move-in cost.

Stumpage and product prices—

Stumpage price differs among regions and ownership. The average stumpage price for eastern Washington on public timberland was about $147 per thousand board feet (MBF) from 2003 to 2010, with an average of $218/MBF for the Washington state-owned timberland and $76/MBF for the Forest Service-owned timberland

(Warren 2011). The average log export price from port districts in Washington and Oregon to major Asian countries (China and South Korea) was used as a ceiling log price. The averaged saw-log price to these two Asian countries from 2006 to 2010 was about \$560/MBF from Seattle and Columbia-Snake districts, \$619/MBF to China, and \$500/MBF to South Korea (Warren 2011). The pulpwood size timber was valued as average chip export prices from Seattle and Columbia-Snake districts, with an average of \$75/tonne (t) from 2006 to 2010, \$56/t from Seattle and \$93/t from the ports of Columbia-Snake districts. The potential woody biomass supply included the removed trees with diameter at breast height <12.7 cm, plus the tops and branches from the larger removed trees. We assumed \$33 per bone dry tonne of delivered biomass.

Transportation cost—
The transportation cost of biomass substantially affected the economic feasibility of the biomass supply. The distance from each watershed to all the biomass facilities was estimated, and only the facilities with transportation distance <402 km were considered for biomass supply. The total transportation cost depended on hauling distance, the truck load, labor, and fuel costs. Travel time for transporting biomass from watershed source areas to various mills was calculated based on posted road speed limit. Railroad transportation was not considered for this study. We assumed 22.7 t per truck load and \$4.40 per hour and per tonne truck for transportation costs.

Results

Standing Biomass and Timber Volume

Total aboveground live tree biomass peaked at year 44 for both scenarios, at 214 billion dry tonnes for the FSO scenario and at 204 billion dry tonnes under the CM scenario; it declined after that time and then stabilized or slightly increased near the end of the simulation period (fig. 4.2a). Total standing merchantable tree inventory followed the same trajectory as biomass; it increased rapidly for both the CM and FSO scenarios during the first three decades, stabilized for 10 to 20 years, and then decreased. The timber inventory under the CM scenario peaked at 0.31 billion m^3 around year 33, about 13.5 percent higher than the initial inventory, and then declined. The inventory under the FSO scenario reached its maximum of 0.33 billion m^3 at year 40 and remained steady for about a decade, then declined (fig. 4.2b). In both scenarios, timber inventories at the end of simulation period were above their initial level. The merchantable tree inventory under the FSO scenario ended about 10 percent higher than under the CM scenario.

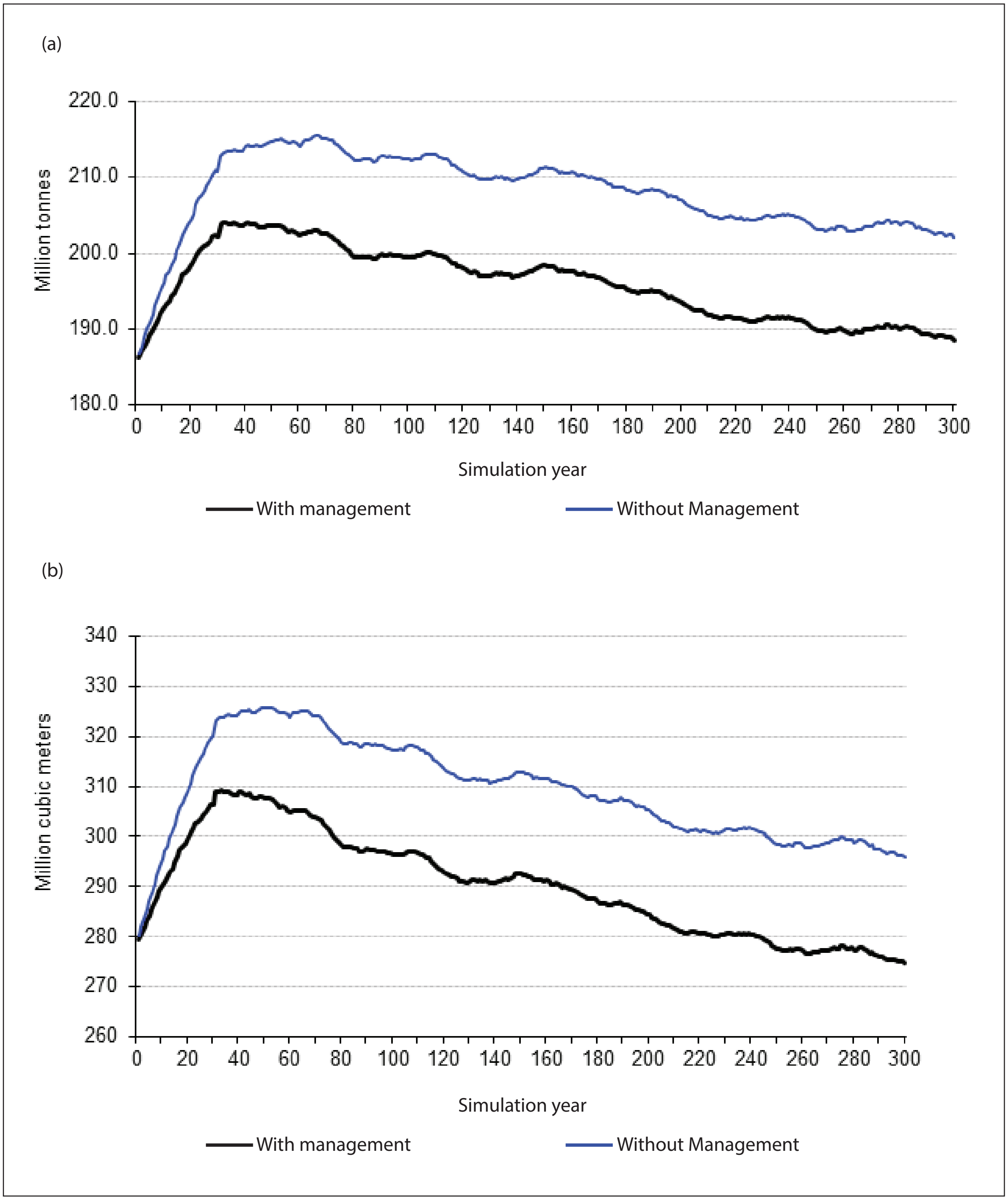

Figure 4.2—(a) Total aboveground live tree biomass in the central Washington landscape, and (b) total merchantable volume in the central Washington landscape.

Removals From Vegetation Management Treatments

We examined the first 50 years of simulation results for the economic analysis. The removals of merchantable tree volume from USDA FS and Yakama Nation tribal lands increased gradually for the first two to three decades and then stabilized, while the removals from private lands were relatively low during the entire simulation period owing to the small area of privately owned forest land in the study area (fig. 4.3a). Removals of merchantable tree volume from the Washington state-owned forest lands were about 68 percent of the total removals at the beginning of the simulation period, decreased dramatically to less than 40 percent of the total after three decades, then stabilized at 20 to 25 percent of the total at the end of the simulation (fig. 4.3a). These patterns reflect differences in management treatments and initial forest conditions on different ownership-management strata. Management treatments were more likely, given appropriate stand conditions, on Washington state and tribal lands compared to those on lands managed by the USDA FS. Conversely, much of the forested land in the Yakama reservation had been treated following extensive, recent insect damage, while state lands had a mixed history and many had not been recently managed. Lands managed by the USDA FS were generally not recently managed, and there was considerable area in reserves that are not subject to much active management. As a result, removals rates reflected (1) relatively stable levels on tribal lands, (2) initially high then declining levels on state lands, as most of the available area became managed, and (3) stable and relatively low levels on USDA FS lands constrained by lower management probabilities and relatively extensive reserves. The same general patterns emerged for biomass removals.

The woody biomass supply from USDA FS and Yakama tribal lands increased gradually over the first two to three decades and then stabilized, while the removals from private land remained at relatively low levels during the simulation period (fig. 4.3b). Total biomass removals from USDA FS lands averaged 12 percent of the total biomass removals initially, and reached 25 percent of the total at the end of 50 years. Biomass removals from tribal lands during the same period increased from 19 to 35 percent. The biomass supply potentials from Washington state-owned forest were about 65 percent of the total initially, and decreased dramatically for the first half of the century to 35 percent of the total potential biomass supply. The contribution of woody biomass from private forest land in the study area was relatively small, but increased from 3.8 to 5.4 percent of the total over 50 simulation years.

The quantity of potential biomass supply varied substantially by watershed and ownership-management (fig. 4.3c). Four major watersheds produced woody biomass over 181,000 dry tonnes each over the 50-year simulation period, while other watersheds produced much less. Most of the Yakama Tribe contributions were from two watersheds. The biomass removals in four watersheds were all from Yakama Tribal lands. Washington state lands produced most of the biomass from three watersheds.

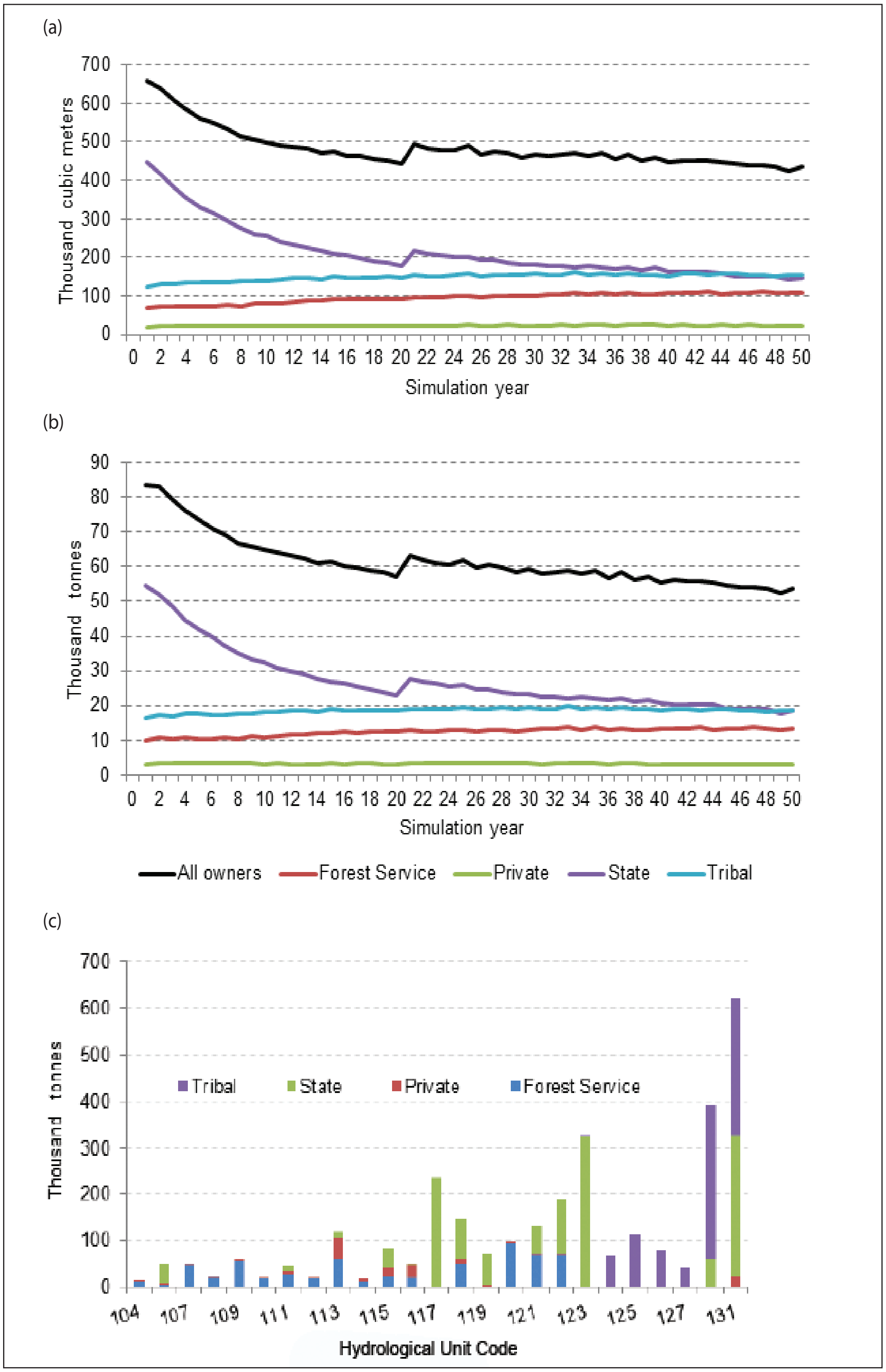

Figure 4.3—(a) Merchantable volume removals (trees with diameter at breast height [DBH] >12.7 cm) from fuel treatments by ownership under the current management scenario, (b) potential biomass (trees with DBH <12.7 cm and residues) supply from fuel treatments by ownership under the current management scenario, and (c) total biomass (DBH <12.7 cm plus branch and top) of removals over 50 years by watershed and ownership.

Economic Analysis

The economic outcome of vegetation management treatments was determined by the total cost of the treatment and the possible revenues generated from the removed products, including commercial timber and woody biomass. For this study, woody biomass supply included all removed trees <12.7 cm DBH, plus the branches and tops from the larger removed trees, estimated in bone-dry tonnes. Trees above 12.7 cm in diameter were considered as timber products. The biomass value was also affected by the distance from the supply point to the processing facility.

The analyses in this section addressed two major questions under the CM scenario: (1) For given timber prices, would selling only commercial timber products be profitable? (2) Would adding woody biomass as a commercial product for fuel energy improve the economic outcome?

We estimated the total stumpage value from potential treatments under the CM scenario to be over $15 million annually. To simplify the calculation process, and for the purposes of this paper, we assumed an average stumpage price of $218/MBF for all ownerships (fig. 4.4a), likely an overestimate for stumpage on USDA FS land. The average harvesting cost for all ownership-management ranges from $159/MBF to $170/MBF, with USDA FS averaging from $169 to $191/MBF (fig. 4.4b).

Potential profit varied considerably among owners including woody biomass as biofuel made a considerable difference to profits.

Potential profit varied considerably among owners. Assuming an average stumpage price of $218/MBF, figure 4.5a shows the potential treated hectares with positive profits at different log prices of $360, $380, $400, $420, and $450. When log prices averaged $360/MBF, about 10 percent of the area treated generated profits, mostly Yakama and private forest land. Almost all treated area on USDA FS land lost money at this log price. However, almost all treated hectares generated profit when log prices were above $450/MBF, regardless of ownership. Figure 4.5b maps the profitability for the first 30 years for each watershed at the saw-log price of $400/MBF and pulpwood log price of $75 per dry tonne, with timber as the only product for different ownerships. In this case, profits occurred for USDA FS managed lands in 4 out of 19 watersheds, Washington state owned lands in 2 of 13 watersheds, private lands in 8 of 21 watersheds, and Yakama tribal lands in five of seven watersheds.

Including woody biomass as biofuel made a considerable difference to profits (fig. 4.6a). For example, at a log price of $380/MBF, selling the woody biomass added 1 to 7 percent to the area that could be treated at a profit. However, the marginal benefit of selling woody biomass decreased as the log price increased. When log price was $420/MBF and more, the marginal benefit of selling the woody biomass was insignificant (fig. 4.6a).

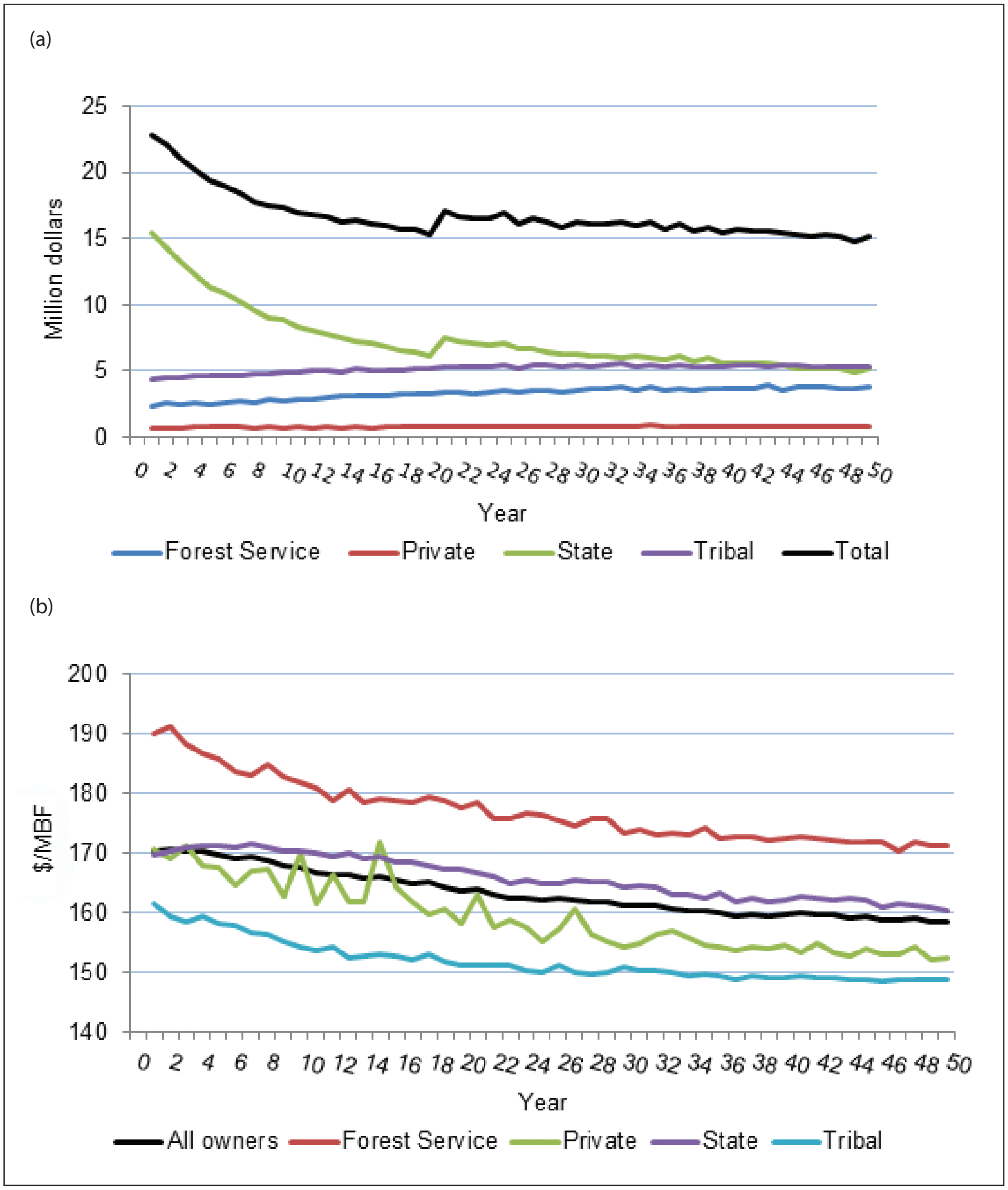

Figure 4.4—(a) Stumpage value by owner (at a price of $218 per thousand board feet [MBF]), and (b) average harvesting cost by owner.

The quantity of potential biomass supply from watersheds and the distance or transportation time considerably changed the economic outcome of using woody biomass as fuel. Depending on where the biomass facilities were located, the contribution of selling woody biomass as products increased or decreased the total revenue. At a saw-log price of $400/MBF, pulpwood log price of $75 per dry tonne, a biomass price of $33 per dry tonne, and delivery to the processing facility with cheapest transportation costs, all the ownerships in all watersheds benefitted

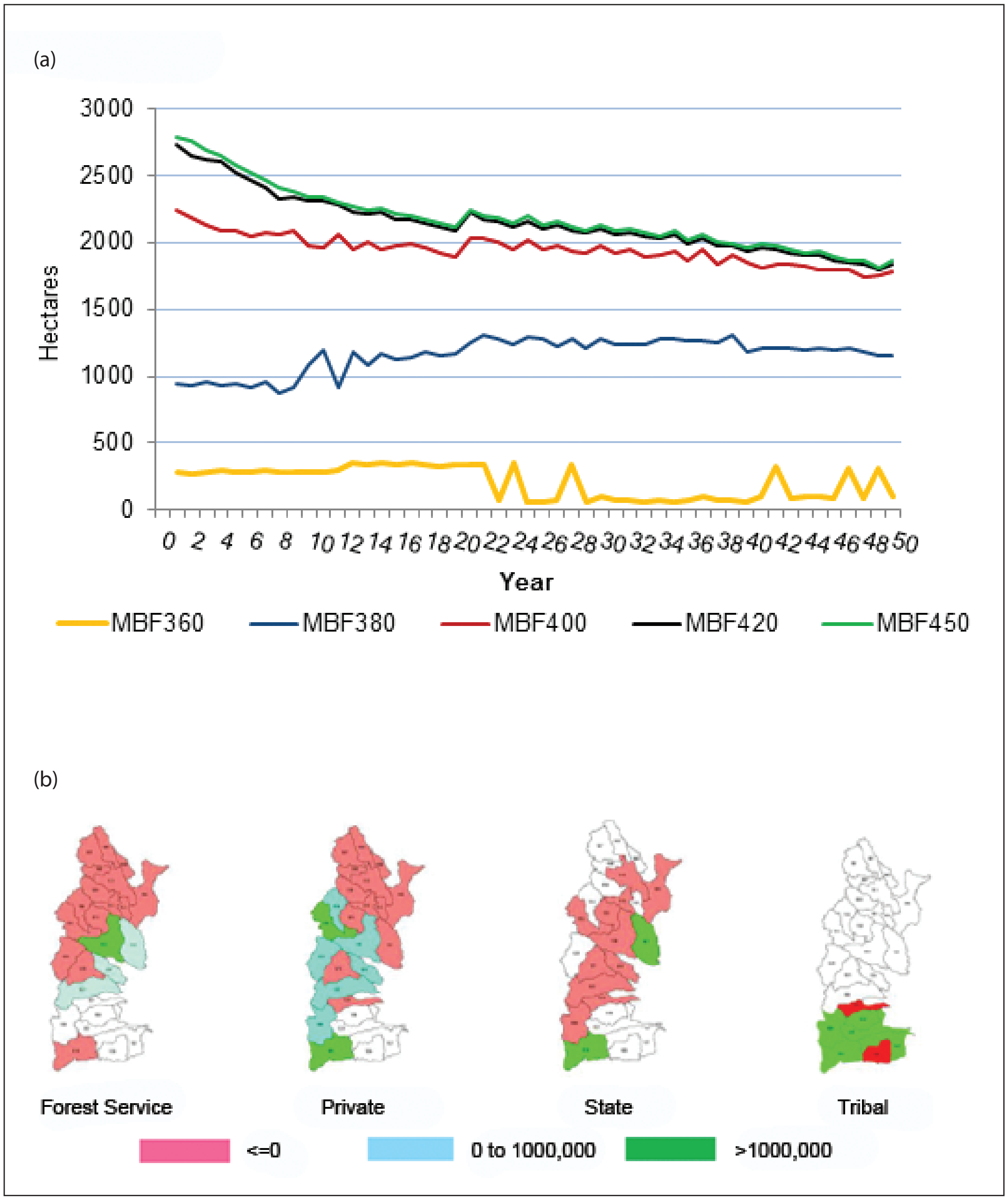

Figure 4.5—(a) The potential treated hectares with positive profits at different log prices of \$360, \$380, \$400, \$420, and \$450 (including timber volume only), and (b) the profitability of timber only by owner and watershed. Profits are 30-year totals in dollars.

from selling the woody biomass. In addition, compared to selling timber only, all owner groups increased the number of watersheds with positive economic returns when biomass sales were included (fig. 4.6b). In this case, profits came from USDA FS lands in 5 of 19 watersheds, Washington state owned lands in 3 of 13 watersheds, private lands in 13 of 21 watersheds, and Yakama tribal lands in six of seven watersheds.

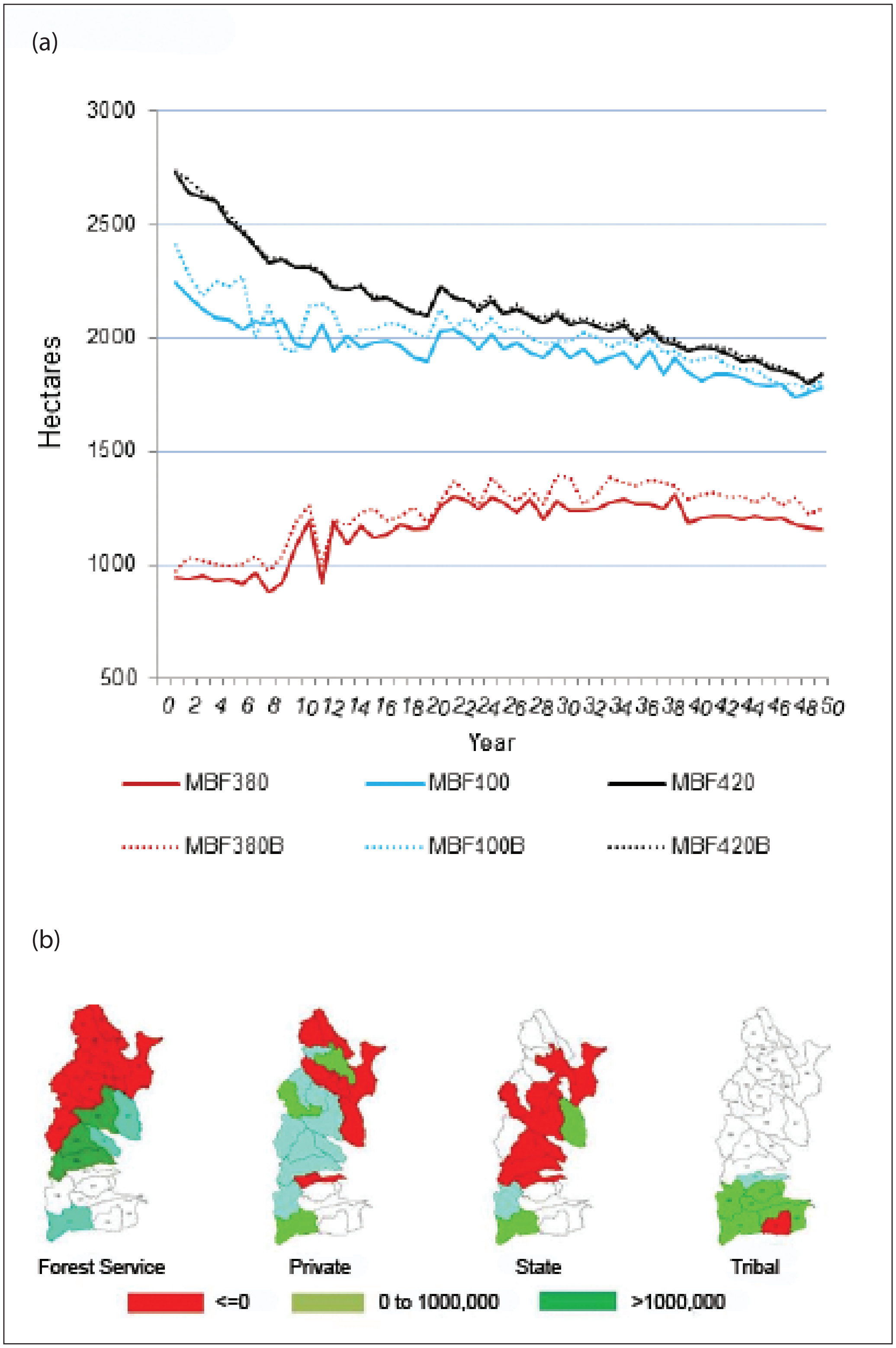

Figure 4.6—(a) The potential treated hectares with positive profits at different log prices of $360, $380, $400, $420, and $450 (timber and timber plus biomass (dotted lines), and (b) profitability of both timber and biomass by owner and watershed. Profits are 30-year totals in dollars.

Discussion

Management to achieve forest health and restoration objectives can be expensive. Many management treatments could produce a variety of potentially valuable forest products, but economic costs and benefits differ widely depending on the kind of product, transportation distances, product prices, and other factors. Although the ecological value of treatments is often debated, the fact is, treatments cost money. The realities of treatment economics play a critical role in the selection of a set of treatments to achieve objectives. In a setting where ecological restoration and forest health are primary considerations, the value of forest products generated by management treatments may be relatively low. This suggests that several land managers and organizations in a large landscape may need to coordinate treatments to generate enough product value to make overall forest objectives economically viable. In many cases in the Western United States, federal lands make up a large portion of the area that might generate forest products in the course of restoration, but declining budgets, transportation costs, product values, treatment costs, and available infrastructure can severely limit the ability of federal land managers to implement restoration treatments.

Our study illustrates procedures that can be used for economic analysis of STM simulations across large landscapes. Vegetation management treatment may effectively treat dense stands to create a desirable mix of future stand conditions to meet forest health and habitat objectives and to reduce fire hazard. However, the cost of harvesting and the limited use of small trees removed from treatment can directly affect the economics of management practices. The study presented in this paper for the central Washington landscape reveals that the economic outcome of fuel treatments differs significantly among owners and watersheds, depending on the quality of the timber and the market value of logs and woody biomass. Although we focused on vegetation management treatments, any combination of treatments to achieve a wide variety of forest conditions could be addressed with our methods.

We found that adding the woody biomass as biofuel could add value to the treatment if the cost of processing small trees and branches onsite is low and the facility is relatively nearby. In general, we found that when the log prices were low, the marginal benefit of adding biomass as a commercial product was higher, but that marginal benefit diminished as log price increased. In our CM scenario, the area that could be profitably treated annually increased by 5 to 10 percent when log prices were $400/MBF or less (fig. 4.6a). Economic values associated with biomass production were primarily limited by transportation costs. Even though this scenario did not harvest trees over 53 cm diameter on federal lands, the production of timber products drove most of the economic value derived from treatments.

This does not necessarily imply that biomass harvest could not provide benefits to local communities. It does imply that, in this area and under this scenario, biomass generated by the proposed treatments is of relatively small marginal value.

Conclusions

The estimation of the materials or products removed from treatments, product market projections, and bioenergy policies all affected the economic outcome of the CM scenario. When considered across all lands, the CM scenario generated net positive revenue if the price of log was above the sum of harvesting cost and stumpage value, and the cost of transporting woody biomass was lower than that value. But revenues differed widely by ownership, treatment type, and general landscape location. In general, Yakama tribal and privately owned lands with lower harvesting cost generated more net positive revenues, while public lands with higher harvesting costs generated net negative revenues. By working together in a collaborative fashion, the local land managers might generate additional positive benefits. Our process, as part of ILAP, allows land managers, policymakers, and others to evaluate potential economic benefits in conjunction with the effects that alternative scenarios might have on wildlife habitat conditions, wildfire and fuel hazards, and other important resource values.

Acknowledgments

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Chapter 5: Application of State-and-Transition Models to Evaluate Wildlife Habitat

Anita T. Morzillo, Pamela Comeleo, Blair Csuti, and Stephanie Lee[1]

Summary

Wildlife habitat analysis often is a central focus of natural resources management and policy. State-and-transition models (STMs) allow for simulation of landscape-level ecological processes, and for managers to test "what if" scenarios of how those processes may affect wildlife habitat. This chapter describes the methods used to link STM output to wildlife habitat to determine how estimated habitat varies across the landscape, how habitat is affected by different land management scenarios, and how management might enhance estimated habitat for particular species. Using the Washington East Cascades as an example, we provide sample output of habitat analysis for the American marten and western bluebird under two management scenarios. Wildlife habitat assessments based on the methods illustrated here will differ greatly based on habitat characteristics important to individual species, and the ability to interpret wildlife information accurately.

Introduction

Wildlife habitat analysis and conservation often are a central focus of natural resources management for both ecological and social objectives. Ecologically, wildlife is an indicator of ecosystem conditions. Each species depends on a range of ecosystem features for life activities such as foraging, roosting, nesting, denning, and hiding from predators (Bolen and Robinson 1999). Therefore, presence or absence of a species in a location assumed to contain habitat characteristics linked to that species may provide clues about both habitat condition and integrity of ecological processes (Grimm 1995). Wildlife also has social value (Decker et al. 2001). This value is illustrated by hunting and fishing fees paid, wildlife viewing (e.g., birdwatching), and the wealth of existing nonprofit groups that focus on wildlife (e.g., World Wildlife Fund, National Audubon Society, Boone and Crockett Club). Ultimately, wildlife plays an important role in both ecological and social systems.

Wildlife habitat is relevant to both ecological and social natural resource objectives.

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The importance of wildlife within both ecological and social contexts has resulted in many wildlife-related policies. Among the most notable federal policies is the Endangered Species Act (ESA) which was established "to protect and recover imperiled species and the ecosystems upon which they depend" (USFWS 2011). Regional and state-level policies, such as the U.S. Department of Agriculture Forest Service (USDA FS) Northwest Forest Plan (USDA FS 1997) and state land harvest regulations, respectively, may complement or supplement federal guidelines for wildlife conservation, or focus on regional priorities. However, mismatches between geopolitical boundaries, land use, and geographic ranges of wildlife species often result in inconsistent management needs and conflict related to management strategies for species (Morzillo et al. 2012). Such conflict can become even more prolific when wildlife management is placed within a context of broader natural resource policy goals. A need exists to provide a useful interpretation of available knowledge at a defined scale of analysis for decisionmaking about wildlife in the context of other management objectives.

State-and-transition models allow for landscape-level evaluation of simulated ecological processes. The STMs can be used to project changes in future vegetation condition and allow managers to test "what if?" scenarios about how landscape change may affect natural resources. The STMs divide the landscape into state classes that characterize the cover type (dominant species or functional group) and structural stage (percentage cover, canopy layers, etc.). The STM state classes do not measure wildlife habitat directly, but rather particular vegetation features that may be related to species-specific habitat variables. Transitions between state classes simulate dynamic processes such as succession, disturbance, and management activities. The STMs have been used in many coarse-scale analyses of vegetation dynamics, as well as those that include examples of management questions focused on wildlife (Barbour et al. 2005, Evers et al. 2011, Wisdom et al. 2002, Wondzell et al. 2007). In this study, we linked STM state classes to habitat relationships for selected wildlife species.

This research was part of the Integrated Landscape Assessment Project (ILAP), the objective of which was to prioritize land management actions based on fuels conditions, wildlife habitat, economic values, and climate change across Arizona, New Mexico, Oregon, and Washington. This chapter describes the methods used to link STM output with wildlife habitat information. Specific research and management questions pursued with that linkage include:

- How does estimated habitat for focal species differ across the study landscapes?
- How do different land management scenarios affect estimated habitat?
- What management efforts are necessary to enhance estimated habitat for focal species?

As follows, we describe the use of STMs to answer these questions. Using the Washington East Cascades modeling zone (chapter 2; fig. 2.4) as an example, we provide example output of habitat analysis from STM modeling results for two species and two management scenarios.

Methods

We integrated species-habitat relationships with STM output for two regional analyses: the USDA FS Pacific Northwest Region 6 (Oregon and Washington; OR/WA hereafter), and the USDA FS Southwestern Region 3 (Arizona and New Mexico; AZ/NM hereafter). See chapter 2 for details on STMs. Our analysis included all forested and arid land areas, and our observational unit of analysis was habitat. Construction of the wildlife habitat module consisted of three phases:

- Phase 1: Identify data capabilities and limitations
- Phase 2: Select focal species
- Phase 3: Build wildlife habitat models

Phase 1: Identify Data Capabilities and Limitations

There are many approaches to modeling estimated wildlife habitat, which we classify here into two categories: "bottom-up" and "top-down" models. Bottom-up models are constructed using ecological variables known to be important to a species. For example, a habitat suitability index (HSI) is a mathematical index based on known habitat characteristics that represents the estimated ability for a location to support a particular species (US EPA 2008). Many agencies, including the U.S. Geological Survey, U.S. Environmental Protection Agency, U.S. Fish and Wildlife Service, and U.S. Army Corps of Engineers use HSIs as tools for landscape evaluation. In contrast, top-down models are constructed using information derived from models used for observation of ecological processes (e.g., vegetation community succession or response to disturbance), but were not built primarily for evaluation of wildlife habitat dynamics. The species-habitat relationships described here are top-down models, such that they are constructed based on output variables from STMs.

We used STM state classes, comprised of vegetation cover type and structural stage, as a basis for evaluating potential wildlife habitat (see chapter 2 for a detailed description about the construction and application of STMs). For forest STMs, cover types were based on forest type importance value (Horn 1975), and structural stages included tree size (quadratic mean diameter), canopy cover (percentage closure), and canopy layers. Because we used a top-down modeling approach, wildlife habitat models were constructed solely from species-habitat relationships that could be derived from STM state classes. As a result, STM state classes do not align perfectly with key wildlife habitat features that might be identified as important when using a bottom-up habitat modeling approach. Therefore, as also suggested by other researchers (e.g., Shifley et al. 2008), we recommend that the utility and precision of our wildlife habitat models be perceived as limited to mid- to coarse-scale evaluation, the state classes defined by STMs, and the available scientific information that aligns habitat features with those state classes.

Phase 2: Select Focal Species

Our objective was to construct wildlife habitat models for approximately two-dozen focal species across both study regions. Identifying focal species consisted of a three-step process.

Step 1: Focus on terrestrial vertebrate species—
We included only terrestrial vertebrates in this analysis. Terrestrial vertebrates were more likely than other species to be related to habitat characteristics that could be evaluated using ILAP STM state classes. During our investigation, we determined that wildlife habitat models for many amphibians and small mammals required consideration of fine-scale habitat features, such as water, soil moisture, and ground nest or burrow sites that are not associated with ILAP STM state classes (but could be associated with STM state classes if models were designed differently). Although we initially sought to include a wide variety of terrestrial vertebrate species, or even invertebrates, this step resulted in a general focus on mammals and birds because the habitat features important to those species are best represented in the coarse-scale ILAP STMs.

Step 2: Identify species of management concern—
Species of management concern include endangered and threatened species under the ESA, those species proposed to be listed under ESA, state-level listed species, and management indicator species (i.e., species identified by agencies to be representative of particular land cover characteristics such as forest types). We compiled information about species of management concern for several organizations, including federal and state agencies and nongovernment organizations. We sought further

guidance for OR/WA from sources such as the USDA FS Region 6 focal species list[2] and Washington Department of Fish and Wildlife Comprehensive Wildlife Conservation Strategy (WDFW 2005). For AZ/NM, we used USDA FS Region 3 focal species list,[3] U.S. Department of the Interior Bureau of Land Management (USDI BLM) sensitive species lists, and New Mexico State Game and Fish BISON-M database.

Use of listed and sensitive species was advantageous for several reasons. First, those species already are recognized by federal agencies or other land management organizations. For example, the northern and Mexican subspecies of the spotted owl are listed as federally threatened in northern and southern portions of their range, and therefore, were selected as candidate focal species in both study areas. Second, lists of species of management concern already exist, and mechanisms are in place for consideration of them in project and land use planning. Third, there already exists a statutory, regulatory, and policy framework for designation and consideration of those species. Fourth, there already is substantial "buy-in" for those species within agencies and nongovernment organizations. Thus, species of management concern already are part of the planning process.

Step 3: Match species-habitat relationships with STM output—
After identifying species of management concern, we attempted to match habitat requirements for those species to STM state classes (see chapter 2 for a detailed description of state classes). This step greatly reduced the potential number of focal species, as many critical habitat associations were beyond the scope of STM models. For example, the mule deer is a species of economic interest in both study regions. Important habitat characteristics of mule deer include thermal cover (i.e., canopy cover), hiding cover (i.e., shrub and stem density), and forage quality (i.e., forbs and shrubs; Feldhamer et al. 2003). Assessment of thermal cover characteristics was possible from canopy information derived from STM state classes. However, we were unable to associate hiding cover and forage quality to STM state classes with acceptable precision, and as a result, we were unable to include mule deer as a focal species. Therefore, focal species were limited to those with habitat characteristics that could be matched to the thematic detail of vegetation data provided by STM models. Our final lists included 24 focal species for OR/WA and 13 focal species for AZ/NM (table 5.1).

Focal species were limited to those with habitat characteristics that could be matched to the thematic detail of vegetation data provided by STM models.

[2] Terrestrial species assessments: Region 6 forest plan revisions. Unpublished document. on file with: U.S. Department of Agriculture, Forest Service, Portland, OR.

[3] Federally listed threatened and endangered (including critical habitat) found within national forests in the USDA Forest Service Southwestern Region. Albuquerque, NM: U.S. Department of Agriculture, Forest Service. Unpublished document.

Table 5.1—Focal species for the two study regions

Oregon and Washington	Arizona and New Mexico
American marten	Desert bighorn sheep
Ash-throated flycatcher	Giant spotted whiptail
Black-backed woodpecker	Gray vireo
Cassin's finch	Grey checkered whiptail
Fisher	Lesser prairie chicken
Flammulated owl	Mountain plover
Gray wolf	Northern goshawk
Greater sage-grouse	Northern sagebrush lizard
Lark sparrow	Mexican spotted owl
Lewis's woodpecker	White Sands woodrat
Loggerhead shrike	White-sided jackrabbit
Northern goshawk	Yellow-nosed cotton rat
Northern harrier	Zone-tailed hawk
Northern spotted owl	
Olive-sided flycatcher	
Pileated woodpecker	
Pygmy rabbit	
Red tree vole	
Sharp-shinned hawk	
Snowshoe hare	
Swainson's hawk	
Western bluebird	
Western gray squirrel	
White-headed woodpecker	

Phase 3: Build Wildlife Habitat Models

We used simple tools and decision rules to build wildlife habitat models, which allowed for transparency, transferability, and consistency in methods and between regions. The model-building process consisted of three parts:

Part 1: Build species-habitat relationships—

We developed data sheets in Microsoft Excel to construct habitat models for focal species identified in Phase 2 above. For each focal species, we constructed two spreadsheets. One spreadsheet was used to develop a matrix linking STM state classes to species-habitat relationships. The other spreadsheet was used to summarize the data matrix (i.e., first spreadsheet) into lists of state classes that were identified as "habitat" for each species. All remaining state classes not identified as habitat for an individual species were considered "nonhabitat." This summary of habitat relationship information was used as a foundation for step two of this section.

To create a matrix linking STM state classes to species habitat information, we reviewed available scientific literature for each focal species. Literature reviewed included peer-reviewed publications, general technical reports, reference books, theses and dissertations, gray literature from dependable sources (e.g., habitat management plans), and Internet references from reliable sources (e.g., "The Birds of North America," distributed by the Cornell Laboratory of Ornithology and the American Ornithologist's Union; http://bna.birds.cornell.edu/bna/). We extracted any information that would allow for matching STM state classes to important features of a species' habitat, and recorded that information on our spreadsheet matrix. Our focus was primarily on quantitative data (e.g., certain size class categories of trees), but we used best judgment to interpret qualitative data (e.g., "large trees") into STM categories, as appropriate.

During this matching process, we developed decision rules for useful yet incongruent information related to particular situations in order to maintain consistency among selections of cover types and structural stages for all species. In some cases, a number of variations were nested within a dominant cover type in the STMs. For example, lodgepole pine and lodgepole pine/mixed conifer represented variations of cover types nested within lodgepole pine-associated cover types. Thus, we developed a decision rule to select all cover types that contained variations of the cover type of interest. In this situation, focal species habitat that was matched to lodgepole pine would also be matched to lodgepole pine/mixed conifer. Because our observational unit of habitat (and projected potential habitat) was based on habitat characteristics rather than current species occurrence, similar decision rules were adapted for actual versus potential geographic range of each species. In other words, cover type and structural stage combinations within a modeling zone were classified as habitat if that cover type and structural stage combination would provide suitable habitat for the species, even if the species does not currently occur in that modeling zone. For example, the gray wolf is a habitat generalist for which forest cover types and structural stages in western Oregon and Washington provide potential habitat because of potential presence of prey species (e.g., deer and elk). Although gray wolves currently do not occupy western Oregon and Washington, suitable habitat exists and was identified in those locations. Conversely, the black-backed woodpecker is a habitat specialist that occupies more specific dry-forest cover types and structural stages particularly in eastern Oregon and Washington, and identified habitat closely mirrors current distribution of the species.

Decision rules remained intact with very few exceptions, but we recorded those exceptions on the habitat summary Excel spreadsheet. Exceptions included completely illogical matches, and additions or deletions recommended by external reviewers during the quality assurance process (see "Step Three" of this section). For example, Douglas-fir is one cover type associated with spotted owl habitat. Following our decision rule, all cover types containing Douglas-fir would be considered spotted owl habitat. However, one reviewer suggested that Douglas-fir/oak should not be considered habitat because of increased probability of contact with the more aggressive barred owl. Therefore, even though the Douglas-fir/oak cover type contains Douglas-fir, it was not considered a suitable cover type for the spotted owl. Similarly, cover types that included mountain big sagebrush initially were included as lark sparrow habitat based on the described decision rules and information from the literature. However, the mountain big sagebrush cover type (and combinations thereof) was later removed because an external reviewer noted that lark sparrows only are observed among it during migration.

Part 2: Build wildlife habitat database—

We constructed a database in Microsoft Access to link information from the wildlife habitat models to STM simulation output. This database served as an external module that can "dock" to the STMs. This module approach allows for flexibility for additional focal species, linkages with STM output from future study areas, and addition of new wildlife habitat research and information that becomes available. Because of limited ability to derive species-habitat relationships from STM state classes, we used a conservative binomial numbering method to transfer information from the Excel worksheets to Access. In the Access query for each focal species, we entered a "1" to indicate state classes that were identified as habitat, whereas those state classes that were not identified as habitat were left blank.

Part 3: Quality assurance—

Our quality assurance process consisted of an external review of wildlife habitat models and intragroup proofreading of Access data. We solicited expert reviews of habitat models to identify errors created during model-building, and clarify information from the scientific literature that did not match regional observations. At least one external review was completed for each species. Reviewers were sent the Excel datasheets and directions for interpreting habitat information. Although information provided by reviewers was extremely helpful for fine-tuning the habitat models, several reviewers commented that STM state classes were not well-suited for capturing key habitat features related to particular focal species (see "Identify Data Capabilities and Limitations").

We completed an intragroup proofreading process to error check transformations of data from wildlife habitat models to the Access database. For each species, this process was led by a different individual than the one who constructed the habitat model for the same species. The Excel worksheet containing summarized STM state class information was used as the basis for proofreading efforts. Any discrepancies between the Excel worksheet and Access database were recorded by the proofreader, and the worksheet or database were then modified by the lead for that species as appropriate. A second proofreading was completed, if needed.

Example Results and Discussion

The objective of this example is to evaluate estimated changes in area of wildlife habitat in the Washington East Cascades (WEC) modeling zone (see fig. 2.4 in chapter 2) during a 50-year period (2006 to 2056). Our focal species of interest are the American marten and western bluebird. The American marten is a small- to medium-sized carnivore, typically associated with a variety of forests that contain large trees and closed canopy (Feldhamer et al. 2003). This species is a USDA FS management indicator species for late-successional conditions for all forests in the Pacific Northwest. The western bluebird is a medium-sized songbird typically associated with a variety of open forests (Guinan et al. 2008). It is a focal species for open forest communities across all forest types (USDA FS 2006). We used the process described in preceding sections to construct habitat models for both species (tables 5.2 and 5.3). Then, we applied the habitat models to STM output for two management scenarios within forested areas of the WEC. For the STM runs, 30 Monte Carlo simulations were completed for each management scenario, and thus results are reported as average values across those 30 simulations. Results of habitat analysis were visualized at the 5th field HUC scale (USGS and USDA NRCS 2011), as both hectares per watershed and percentage of area modeled within each watershed.

The marten is a carnivore typically associated with large trees and closed canopy. The western bluebird is a songbird typically associated with open forests.

The first scenario was a fire-suppression-only (FSO) scenario, in which no management prescriptions, such as logging or prescribed burning, are implemented. Under this scenario, it is assumed that fire suppression continues at the same rates as it has under the management regimes of the last few decades. With STM simulations for this scenario, change was minimal in moist and cold forest types where there was a long fire-return interval compared to dry forests. However, under the FSO scenario, dry forests became denser and more susceptible over time to stand-replacing fires.

In the FSO scenario, about 1.83 million ha of estimated marten habitat existed initially (currently), as well as 415 700 ha of estimated bluebird habitat (fig. 5.1). During the course of the simulations, estimated marten habitat decreased gradually to 1.68 million ha in 2056 (net loss = 8 percent). Estimated bluebird habitat remained relatively constant during the first 25 years of simulations, but then increased to approximately 516 900 ha in 2056 (net gain = 24 percent; fig. 5.1). Loss of marten habitat appears to be concentrated in the northern portions of the study area, as well as at relatively lower elevations (fig. 5.2). Gains in bluebird habitat are well-distributed across the study area (fig. 5.3). Thus, the small net decrease in

Table 5.2—Habitat model for the American marten, as derived from state-and-transition model state classes for Oregon and Washington

Suitable size classes:	Size description:
Grass/forb	0 or nonstocked
Small tree	25.4 to 38.1 cm quadratic mean diameter (10 to 15 inches)
Medium tree	38.1 to 50.8 cm quadratic mean diameter (15 to 20 inches)
Large tree	50.8 to 76.2 cm quadratic mean diameter (20 to 30 inches)
Giant tree	>76.2 cm quadratic mean diameter (>30 inches)
Suitable canopy closure:	
Medium	40 to 60 percent
Closed	>60 percent
Suitable canopy layers:	
Multiple layers	
Suitable cover types:	
Alaska cedar	
Barren	
Black spruce	
Cedar swamp	
Douglas-fir	
Engelmann spruce	
Grand fir	
Grass/forb	
Jeffrey pine	
Lodgepole pine	
Montane chaparral	
Montane hardwoods	
Montane riparian	
Mountain hemlock	
Pacific silver fir	
Ponderosa pine	
Red fir	
Riparian lodgepole pine	
Sitka spruce	
Spruce	
Subalpine fir	
Western hemlock	
Western larch	
White fir	
White bark pine	
White spruce	

Table 5.3—Habitat model for the western bluebird, as derived from state-and-transition model state classes for Oregon and Washington

Suitable size classes:	Size description:
Grass/forb	0 or nonstocked
Open shrub	NA
Seedling/sapling	12.7 to 25.4 cm quadratic mean diameter (5 to 10 inches)
Pole tree	<12.7 cm quadratic mean diameter (<5 inches)
Shrub	NA
Small tree	25.4 to 38.1 cm quadratic mean diameter (10 to 15 inches)
Medium tree	38.1 to 50.8 cm quadratic mean diameter (15 to 20 inches)
Large tree	50.8 to 76.2 cm quadratic mean diameter (20 to 30 inches)
Suitable canopy closure:	
Open	10 to 40 percent
Postdisturbance	
Suitable canopy layers:	
Single	
Suitable cover types:	
Douglas-fir	
Grand fir	
Noble fir	
Ponderosa pine	
Ponderosa pine/Gambel oak	
Pacific silver fir	
Western hemlock	
Western hemlock/Douglas-fir	

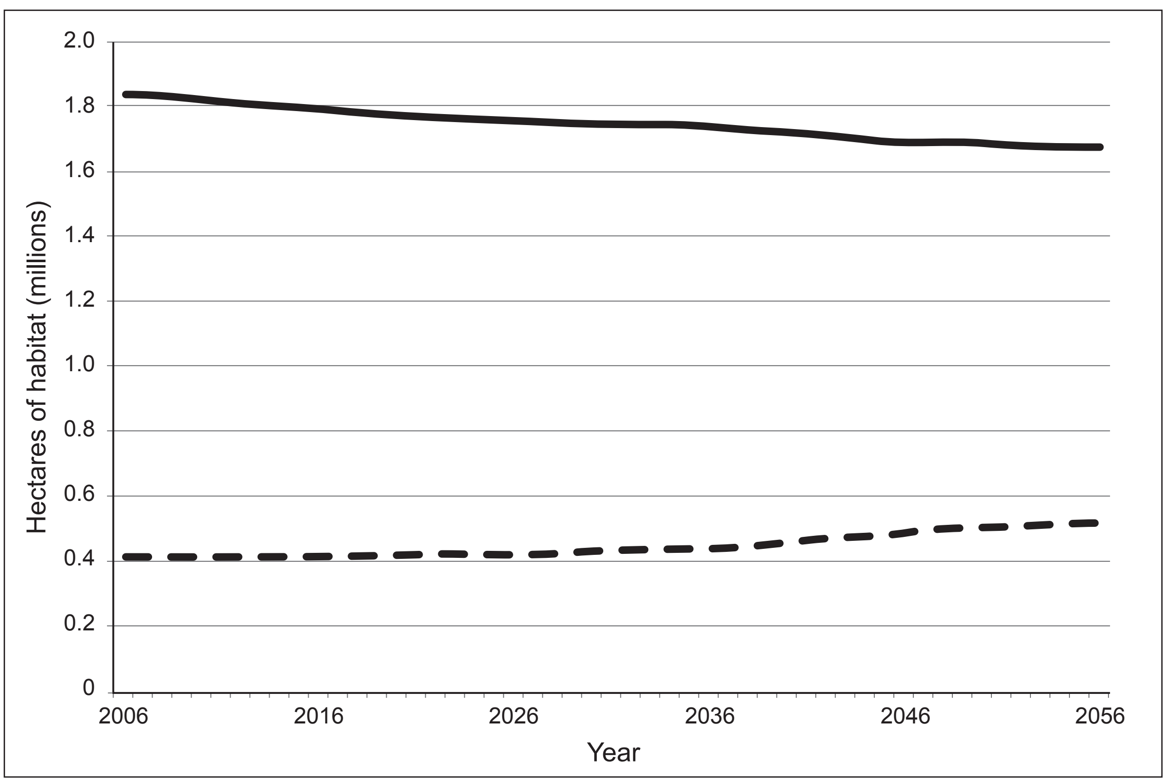

Figure 5.1—Estimated American marten (solid line) and western bluebird (dashed line) habitat for the fire suppression only scenario.

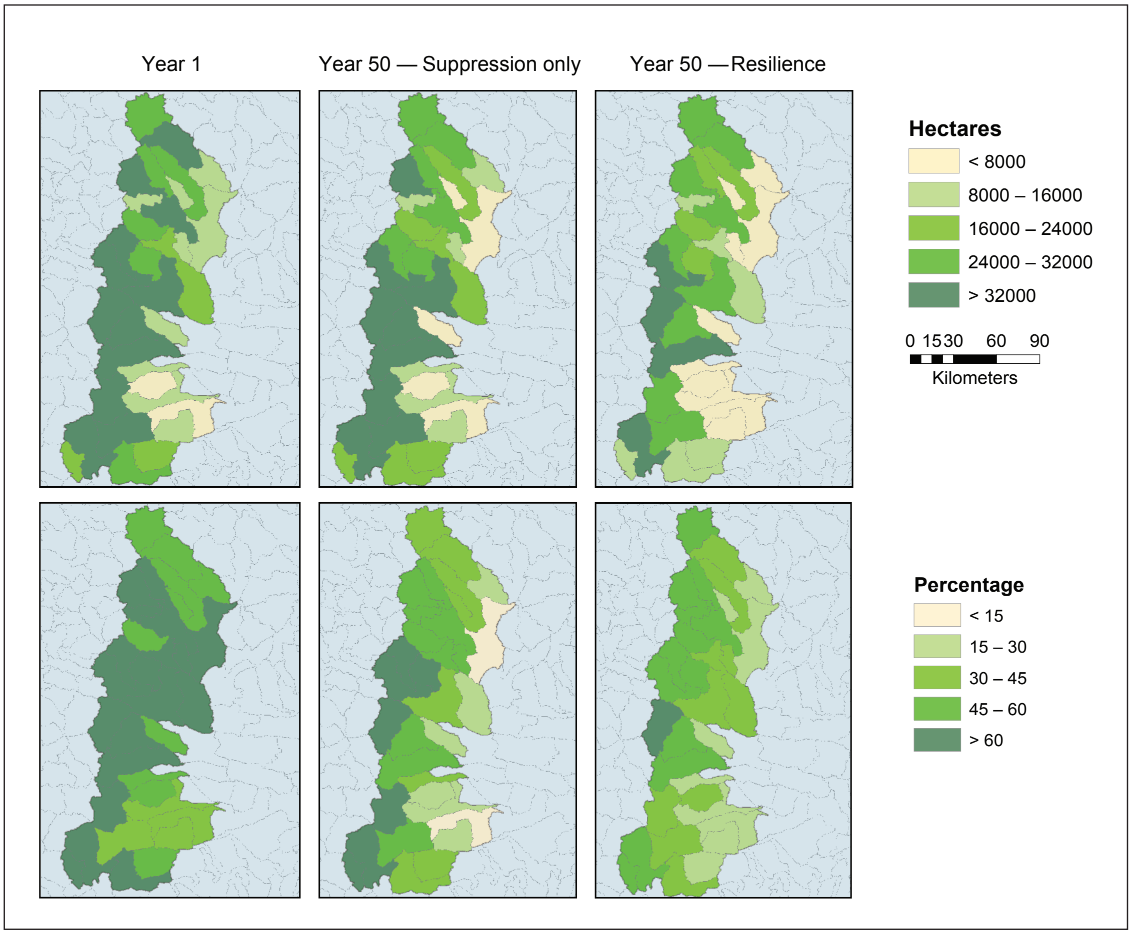

Figure 5.2—Projected habitat for the American marten at the watershed scale under fire-suppression-only and resilience scenarios. The top row illustrates area (hectares) of habitat within each watershed. The bottom row illustrates percentage of area of each watershed that is classified as habitat.

estimated area of marten habitat likely corresponds to an initial increased amount of closed forest, yet is concurrent with losses of big trees and closed forest as a result of an increase in high-severity and stand-replacing fires. Conversely, the gradual then more-pronounced increase in bluebird habitat likely is influenced by an initial closing of the forest followed by forest opening as a result of projected stand-replacing fire.

The second scenario is a resilience scenario, the objective of which is to create more fire-resilient dry forest while also maintaining or even increasing the number of large trees in the dry forest landscape. Treatments included prescribed fire and thinning from below on USDA FS and USDI BLM lands, excluding wilderness areas. We assumed that more area was treated annually on private and tribal versus public land. We also assumed that some prescribed fire took place, particularly on public lands. With STM simulations for this scenario, there was an increase in open forest and reduction in closed forest over time, even for the large tree category.

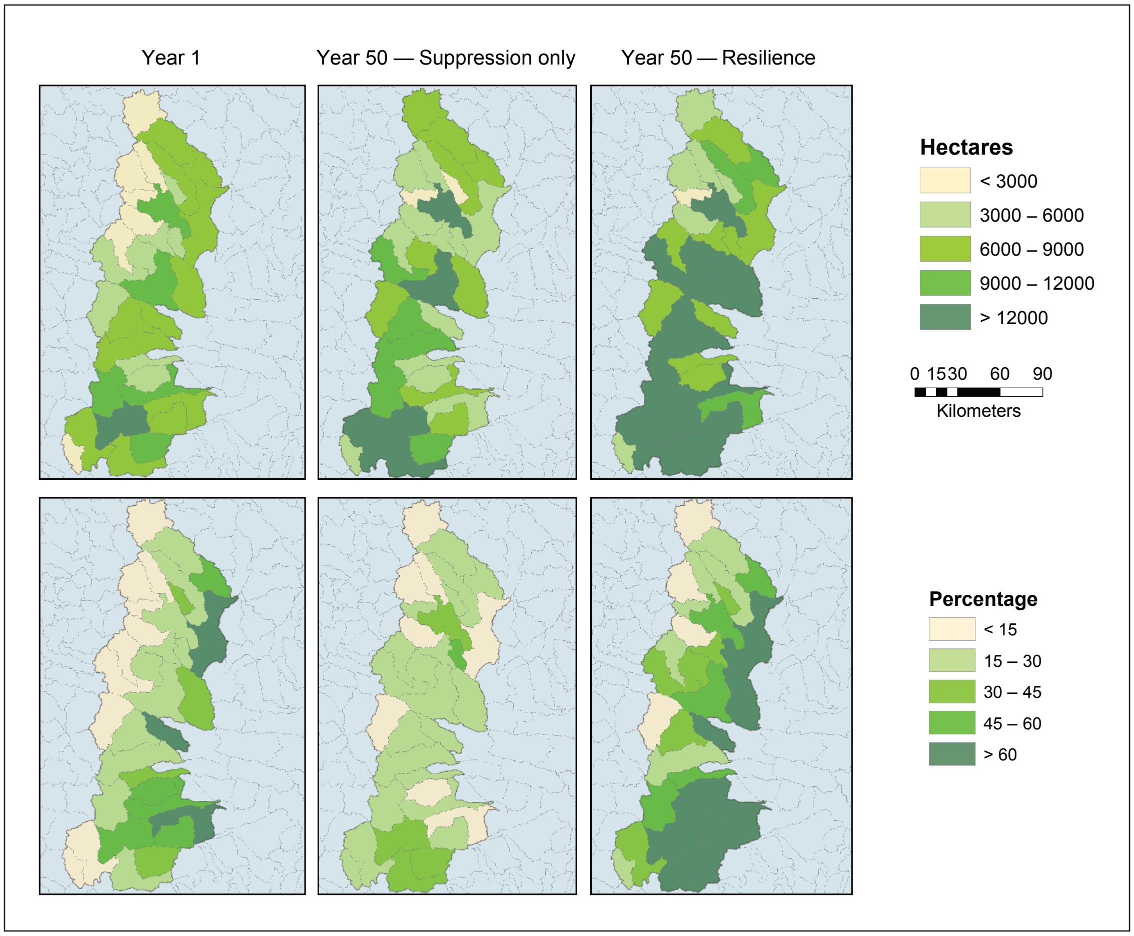

Figure 5.3— Projected habitat for the western bluebird at the watershed scale under fire-suppression-only and resilience scenarios. The top row illustrates area (in hectares) of habitat within each watershed. The bottom row illustrates percentage of area of each watershed that is classified as habitat.

In the resilience scenario, approximately 1.83 million ha of estimated marten habitat existed initially (fig. 5.4), as well as 415 700 ha of estimated bluebird habitat (i.e., same initial conditions as the FSO scenario; fig. 5.4). During the simulation, projected estimated marten habitat decreased to 1.26 million ha in 2056 (net loss = 31 percent). Estimated area of bluebird habitat doubled during the first 25 years of simulations; this rate then slowed, but estimated habitat continued to increase to 865 600 ha in 2056 (net gain = 108 percent). Losses of marten habitat appear to be concentrated in the northern portions of the WEC study area, as well as at relatively low elevations (fig. 5.2). Although similar spatially to the FSO scenario, marten habitat losses are greater in magnitude in the resilience scenario. The greatest gains in bluebird habitat are projected to take place among the central and southern portions of the area (fig. 5.3). Thus, although efforts to maintain or increase large trees initially may seem beneficial for promoting marten habitat, thinning and burning to

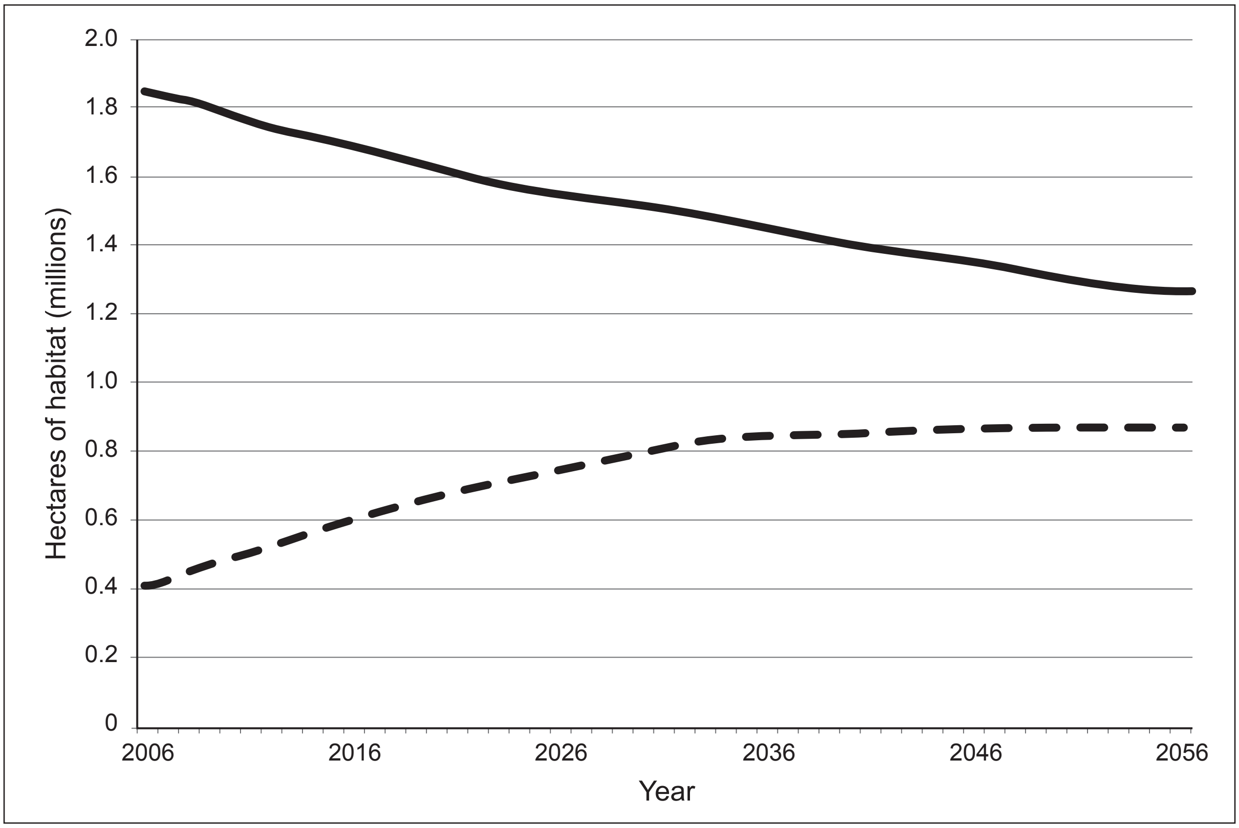

Figure 5.4—Estimated American marten (solid line) and western bluebird (dashed line) habitat for the resilience scenario.

open the forest may outweigh benefits of maintaining large trees. However, opening of the forest is expected to be beneficial for bluebird habitat across all size classes of trees. Simulations using additional management scenarios will allow users to further evaluate estimated changes as a result of management treatments and disturbance.

The methods and habitat models described here are not intended to be used for site-level management, but rather to provide managers with a vision for broader land-use planning. General trends in wildlife habitat across the study area (figs. 5.1 and 5.4) provide information about how decisions at the regional level may affect net habitat across the landscape within the context of other management objectives. Further assessment at the watershed scale (figs. 5.2 and 5.3) provides spatial information about where within the study area changes in habitat are projected to occur. Hypothetically, the resilience scenario may not be a viable option if the primary management goal is to increase the area of marten habitat in the eastern portions of the study area. However, management actions often have positive effects on some species and negative effects on others. In this case, management actions had a positive effect on bluebird habitat. Ultimately, coarse-scale analysis can provide a broad illustration of potential impacts on wildlife habitat as a result of management actions, but further site-specific evaluation will enhance this approach by providing more detailed assessment of habitat characteristics.

Conclusions

Wildlife habitat is an important component of land management and related policy. Using habitat as the unit of observation, the approach described and illustrated here can aid managers with evaluating potential impacts of management activities on estimated wildlife habitat across landscapes at mid to coarse scales. A benefit of such analysis is the ability to assess both spatial and temporal effects of land use decisions, and where and when those effects might be most beneficial or harmful to estimated wildlife habitat in the context of overall management objectives. Caveats of using this approach to address wildlife habitat management objectives, particularly the use of STM models, are that information reported at mid to coarse scales does not account for site level distribution of wildlife, and the ability to build wildlife models is limited to those variables used to evaluate vegetation dynamics. For example, this assessment at the watershed scale cannot account for management impacts on habitat distribution, and related life history traits such as dispersal. Therefore, confidence in wildlife habitat assessments based on the methods illustrated here will vary greatly based on ability to address habitat characteristics important to individual species, and ability to interpret wildlife information created by these models accurately.

Acknowledgments

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Chapter 6: Incorporating Rural Community Characteristics Into Forest Management Decisions

Mindy S. Crandall, Jane L. Harrison, and Claire A. Montgomery[1]

Chapter Summary

As part of the Integrated Landscape Assessment Project, we developed a methodology for managers to include potential community benefits when considering forest management treatments. To do this, we created a watershed impact score that scores each watershed (potential source of wood material) with respect to the communities that are likely to benefit from increased wood supply. The communities we consider are census county subdivisions that correspond to all rural places in four states (Arizona, New Mexico, Oregon, and Washington). Incorporated into the watershed impact score are indices of community characteristics that may be of concern (indicators of socioeconomic well-being, business capacity, and effects of forest policies) and potential biomass supply. A gravity index combined the watershed-level biomass supply with the community-level information and weighted it by the distance between watershed and community, producing an impact score that indicated the potential of each watershed to supply material to communities. Our process allows managers to consider the potential for community benefits when choosing where on the landscape to act.

Introduction

United States federal forest fire policy during the last century has been to aggressively suppress wildfires as rapidly as possible. Consequently, forest conditions are well outside the natural range of variation in many of the forest types of the Western United States (Agee 1993, Franklin and Agee 2003). According to the U.S. Department of Agriculture Forest Service (USDA FS) and the Department of the Interior, about 263 million ha are susceptible to wildland fire (U.S. General Accounting Office 2003). Massive accumulation of forest fuels and changes in the species composition and structure of forests that historically experienced frequent,

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low-intensity fires create conditions in which wildfire, when it does occur, is far more likely than in the past to be catastrophic and expensive, and if not impossible, to contain. Average annual USDA FS expenditures on fire suppression in the 2000s was three times what it was in the previous three decades (Abt et al. 2009). Forest ecologists have become increasingly aware in recent decades, however, that wildfire plays an important role in maintaining healthy forests in fire-adapted forest ecosystems

Recognition of the effects of fire suppression on future fire risk led to the passage of the Healthy Forest Restoration Act (HFRA) in 2003, which was designed to expedite hazardous fuel reduction and forest restoration projects on federal lands. Mechanical fuel removal and restoration thinning have been proposed as potential remedies to this forest health crisis. But because these treatments are costly, resource constraints prevent managers from treating as many hectares as need treating. For example, even after the passage of the HFRA, only 8 million of the 263 million ha at risk were planned for treatment to reduce hazardous fuels between 2000 and 2007 (Rey 2006). Prioritizing the placement of fuel treatments in order to use limited resources efficiently is critically important.

Limited funding means that managers must pick and choose where across the landscape to place treatments. The human benefits of these treatments—the possible increases in mill processing activity, jobs, and other economic activities that may arise from an increase in wood material harvested—will also be located in specific places across the landscape.

The Integrated Landscape Assessment Project (ILAP) was a watershed-level prioritization of land management actions based on vegetation and fuel conditions, wildlife and aquatic habitats, and economic values across all lands in Arizona, New Mexico, Oregon, and Washington. One of the goals of ILAP was to provide land managers, policy analysts, and others with tools to enable them to determine how to place fuel treatments across the landscape in a way that maximizes diverse benefits—such as wildfire risk reduction and wildlife habitat improvements—while still ensuring the treatments are as financially feasible as possible. Limited funding means that managers must pick and choose where across the landscape to place treatments. The human benefits of these treatments—the possible increases in mill processing activity, jobs, and other economic activities that may arise from an increase in wood material harvested—will also be located in specific places across the landscape. That means that managers and others may also have a very real opportunity to consider which communities may benefit from harvest action.

Why Should We Care About Rural Communities When Considering Fuel Treatment Location Decisions?

Throughout the United States, timber harvests from federal lands have declined dramatically since the 1980s. Harvest declines on federal lands, as well as greater domestic and international competition and technological advances, have contributed to fewer employment opportunities in the timber industry. Mechanization, consolidation, and closures of mills continue to concern rural forest communities. Harvest decline was exacerbated in Oregon and Washington with injunctions prohibiting logging on USDA FS and U.S. Department of the Interior Bureau of Land Management (USDI BLM) lands within the range of the northern spotted owl, which was listed as threatened under the Endangered Species Act in 1990. By the late 2000s, national forest harvests in Oregon and Washington declined to about 10 percent of what they were in the late 1980s (USDA FS 2010). The listing of the Mexican spotted owl in 1993, and the subsequent temporary injunction against logging on national forests in 1995, had a similar effect in the southwest. By the late 1990s, national forest harvests had declined to about 19 percent and 13 percent of late 1980s levels in Arizona and New Mexico, respectively (Keegan et al. 2001a, 2001b). These harvest figures include fuelwood harvest, still a significant proportion of use in the southwest, so the decline in useable material for mills was even more pronounced; in 1998, less than half of the cut from national forests in Arizona was available for processing in local community mills.

Federal timber harvests have also contributed directly to county revenues in all four states, through a revenue sharing program from timber sales designed to compensate counties for the loss of tax revenue associated with federal ownership status (Gorte 2010). Beginning in the early 20th century, the USDA FS paid 25 percent of gross timber receipts to the states where the timber was harvested to provide and maintain roads and schools in those counties where national forests are located, while the USDI BLM shared 50 percent of revenues from harvests on the Oregon and California revested grantlands. These lands cover 970 000 ha of forest land, originally granted to railroads, in 18 western Oregon counties (Corn and Alexander 2012). In counties with considerable public lands, these receipts contributed substantially to county budgets after World War II, when harvest activity on public lands accelerated substantially (Corn and Alexander 2012). Although the initial goal of the revenue-sharing was compensation for foregone tax revenue, concern over effects on county budgets of declines in timber harvest resulting from the spotted owl listing led to the appropriation of payments to Oregon, Washington, and California in the early 1990s, and to an adjustment of the revenue-sharing program through the Secure Rural Schools and Community Self-Determination Act of 2000 (USDA FS 2011). Still, adjusted for inflation, Oregon county payments declined by 32 percent on average from the late 1980s to the late 2000s, while Washington county payments declined by 23 percent (USDA FS 2011). Payments from the Secure Rural Schools ended in 2011, which created further uncertainty for rural communities.

Changing public forest values mean that communities near public forest lands can no longer depend on federal agencies to promote rural economic growth and provide sustained-yield employment through timber harvest. Kennedy et al. (2001) suggested that federal agencies have shifted their rural development roles to help facilitate effective rural community adaptation to broader socioeconomic change. The USDA FS has maintained the goal to, “help states and communities to wisely use the forests to promote rural economic development and a quality rural environment” (USDA FS 1994). The USDI BLM maintains that the “most effective, realistic role for public forest resource agencies is to assist rural economies and communities in adapting to changing regional and global forces” (USDI BLM 1994).

Vegetation management (which can include the removal of small trees, excessive brush, and larger trees on overstocked sites to reduce fuels) has the potential to meet the diverse goals of these federal agencies. Treatments can mitigate future wildfire damage, restore forest ecosystem health, and provide many benefits to rural communities, including local employment opportunities, wood supply for mills, and biomass supply for emerging energy markets.

For this study, we have compiled information to be included in a decision support system that allows managers who are considering fire-hazard reducing treatments to include potential community benefits in their considerations. The decision support system consists of layers provided by ILAP teams that address future effects on wildlife, wildfire hazard, and economic feasibility of fuel treatments at the watershed level in the four states. The support system uses Ecosystem Management Decision Support (EMDS; Reynolds 1999, Reynolds and Hessburg 2005), an application framework that runs within a geographic information system, allowing for spatially explicit information on multiple topics to be analyzed simultaneously to aid decisionmaking. The goal of the community economics team was to provide information that enables land managers to consider the current and historical state of communities that may be affected by increased harvest activity. We did this by creating a watershed impact score (WIS) that rates each watershed (potential source of wood material) with respect to the communities that are likely to benefit from an increase in locally delivered wood supply resulting from forest health treatments.

Methods and Data

To give land managers the ability to consider potential effects on communities when choosing where to place fuel treatments, we sought a method to combine disparate information into a single, easy-to-interpret index. Input data included indicators of community conditions, estimates of potential biomass that each watershed could produce, and the distance between sources of material and potential places that can utilize material. The WIS is based on the idea of a gravity index: that the transfer or flow of products is related to both the characteristics of the origin (the push factors) as well as the destination (the pull factors). Although distance is clearly important in determining where material goes, it is not the only consideration. To calculate the WIS, we needed to establish the following:

- Criteria for an origin location
- Criteria for a destination location
- Community characteristics that may be of importance to decisionmakers
- Estimates of potential biomass supply
- Weights and structure of the gravity index

The WIS is based on the idea of a gravity index: that the transfer or flow of products is related to both the characteristics of the origin (the push factors) as well as the destination (the pull factors). Although distance is clearly important in determining where material goes, it is not the only consideration.

Gravity index models are a widely used spatial interaction model; they have applications in market analysis (e.g., placing and optimizing shopping center size), predicting migration patterns, or even analyzing university enrollment (Haynes and Fotheringham 1984). For example, shoppers from a rural area faced with two cities to shop in will take many things into consideration when deciding which city to go to, not just the distance involved. Certain mixes of shops or amenities may be more attractive to consumers. Destinations that offer more or have more weight can attract input from a wider radius than destinations that have less weight. Gravity indices have also been used in landscape-level assessments to model the urbanization potential of undeveloped land as a function of population and proximity (Kline et al. 2001). The WIS in the ILAP project built directly on this use of a gravity index in service of both ecological and social issues, but represented, to our knowledge, the first use of a gravity index to include community concerns in planning land management activities.

Following Haynes and Fotheringham (1984), we considered a single sending center gravity index:

$$\mathrm{WIS} = \sum_i \frac{(\mathrm{v}_i \mathrm{w}_j)}{\mathrm{d}_{ij}} ,$$

where

WIS_j = watershed impact score assigned to j^{th} watershed,

v_i = drawing attribute of i^{th} community (the "pull" factors, e.g., mill presence or poverty, of each destination),

w_j = sending attribute of j^{th} watershed (the "push" factors, e.g., biomass potential, of each origin), and

d_{ij} = travel time between i^{th} community and j^{th} watershed.

Criteria for an Origin Location

The base layer for origins was all watersheds (5^{th} field hydrologic units; USGS and USDA NRCS 2011) in USDA FS Pacific Northwest Region 6 (Oregon and Washington; OR/WA hereafter) and USDA FS Southwestern Region 3 (Arizona and New Mexico; AZ/NM hereafter). Watersheds average about 44 000 to 56 000 ha. We created a centroid point for each watershed—a geographically-weighted center latitude and longitude point for each unique watershed in the four states, excluding masked urban areas (masked areas represented nonmodeled areas for the purposes of determining future potential vegetation, wildlife habitat potential, fire hazard, and economic feasibility of fuel treatments). The result was 945 unique potential origin points in OR/WA and 1,015 in AZ/NM. In the end, many of these watersheds had no biomass that could potentially be supplied to local communities and were not given scores (the process for determining biomass potential is discussed below).

Criteria for a Destination Location

Potential destinations for material removed in fuel treatments are likely to be existing mills but may also include future processing development in places currently without facilities. Thus, we included both those places that are currently processing wood products and those that may potentially be a site for future capital investment as destination locations. Capital investment (both past and future) is likely to be concentrated in existing communities and places with infrastructure and population. In addition, any discrete place also draws workers in from a surrounding area that may be much larger than one physical town or city. Both of these levels of geography are accounted for in our destinations.

For places with infrastructure and population (potential destinations for material), we used the census designated place (CDP) geography from 2000. Places included designated places, consolidated cities, and incorporated places. Designated places are the census-defined counterparts of incorporated places. They are delineated to provide data for settled concentrations of population that are identifiable by name but are not legally incorporated. Boundaries usually coincide with visible features of the boundary of an adjacent incorporated place or other legal entity boundary and are defined in cooperation with local or tribal officials. To focus on rural places, we omitted urban places (those with populations over 50,000) and urban-linked places (those with populations from 10,000 to 50,000 and that are within 32 km of an urban place). Recognizing that both material and people may cross state borders for work, we included those rural places within a 96-straight-line-km distance of the border in each two-state region.

These rural places, along with census county divisions (CCDs), defined community in our study. Census county divisions are delineated by the U.S. Census Bureau to follow visible features, have a community orientation, and coincide with census tracts where applicable. Defining communities as subcounty units helps to mitigate concerns over the use of larger geographies, which may not reflect local peoples' sense of community boundaries. Production, marketing, consumption, and local institutions factored into the delineation of CCDs by the census. The locality on which a CCD is centered usually is an incorporated place or an unincorporated community, which might be identified as a census designated place. In some cases, the CCD may center on a major area of significantly different topography, land use, or ownership, such as a large military installation or American Indian reservation. The name of each CCD is based on a place, county, or well-known local name that identifies its location (USDC Bureau of the Census 1998). We considered CCDs that contain a rural place, as defined above, to be **rural communities**.

To our knowledge, this is the first use of combined census CDP and CCD geographies to delineate communities. More often, single geographic layers are used that are either smaller or larger, such as census tracts or block groups for the former, and counties for the latter. Although geographies smaller than CCDs may better represent a person's sense of place, the use of census tracts or block groups is problematic. Both census tracts and block groups are based on population and neighborhoods, but do not necessarily correspond to recognizable communities. In addition, both block groups and tracts change boundaries between censuses. In more densely populated areas, aggregation is necessary to represent communities using these geographies. This type of aggregation is a time-consuming process, which requires key informants from all regions to determine the appropriate aggregation (Donoghue 2003). Counties are frequently used for ease of data gathering from multiple sources, availability of historical data, and ease of recognition. However, counties are large, heterogeneous areas with multiple recognizable communities contained within. We defined communities in terms of the CCDs that contain rural CDPs in order to maximize the strengths and minimize the individual weaknesses of these geographical definitions as units of analysis, recognizing that this limited the community-level data available to that produced by the U.S. Census Bureau.

The distance between each watershed centroid point and place centroid point was calculated by creating an origin-destination matrix in ArcGIS. This matrix contained distance and travel time estimates for all community-watershed pairs. We added 5 minutes to each distance to account for startup time.

Community Characteristics That May Be of Importance to Decisionmakers

To calculate v_i, the drawing attributes of the i^{th} community, we considered indicators of (1) community distress, (2) business capacity, and (3) USDA FS policy impacts (table 6.1). Our gravity index can, thus, prioritize communities experiencing distress, those with business capacity ready to utilize new supply, those that have been adversely affected by USDA FS policy, or those that share combinations of characteristics. All drawing attributes and their data sources are defined in table 6.1. The rationale for inclusion of each drawing attribute follows.

Indicators of communities in distress (SES)—

The fates of rural communities highly dependent on natural resources has been a concern of land managers for decades. The Sustained Yield Forest Management Act of 1944, for example, had as part of its stated purpose to promote the stability of communities (Hibbard 1999). It is possible that managers may want to target areas of high distress as potential recipients of forest material. Inputs of supply may be able to help stimulate positive economic activity within the community. Although

Table 6.1—Drawing attributes for communities

Indicators	Description	Data source
Communities in distress:		
Poverty	Percentage of population at 185 percent poverty level or below	Census 2000
Income inequality	Gini index: measures inequality of household income distribution	Census 2000
Unemployment	Percentage of labor force unemployed	Census 2000
Unattached youth	Percentage of youth (16 to 19 years old) not in school and unemployed	Census 2000
Population decline	Percentage of decline in population from 1990 to 2000	Census 2000
Business capacity:		
Education	Percentage of population 25 years and over with associates degree or higher	Census 2000
Labor force	Percentage of population in labor force	Census 2000
Mill capacity	Presence of a mill in a census county division since 2000	Various sources (see text)
Specialization	Percentage of employment in forestry, wood products, or paper	Bureau of Land Management 2000
Federal forest policy impact:		
Forest Service impacts	Percentage change in U.S. Department of Agriculture, Forest Service county payments from 1986–90 to 2003–07 average	USDA FS 2010

increasing levels of community distress act as an increasing draw within the WIS, it is worth noting that higher values of these indicators correspond to lower socioeconomic status, and overall, more negative conditions within the communities. Community distress indicators used in the WIS include poverty, income inequality, unemployment, unattached youth, and population decline. A measure of overall economic diversity was considered, but rejected owing to lack of variability in the calculated data.

Community distress indicators used in the WIS include poverty, income inequality, unemployment, unattached youth, and population decline.

Poverty—Poverty is a ubiquitous measure that indicates community distress (Charnley et al. 2008, Donoghue and Haynes 2002, Isserman et al. 2009, Kusel and Fortmann 1991, Overdevest and Green 1995, Reeder and Brown 2005, Stedman et al. 2004, 2005). The most widely used definition of poverty is the U.S. Census definition, in which a family is considered poor if its annual pretax money income is less than the poverty threshold for a family of that composition and size. Although the official definition of poverty generally considers only those below the threshold as poor, agencies that provide housing subsidies, food assistance, and low-income health insurance often use income thresholds of 185 percent of poverty or greater. These families may be working enough to keep them above the threshold, but are at great risk of slipping into poverty should their job be lost or work hours reduced. Additionally, studies exploring alternative measures of poverty have generally concluded that a threshold of 150 to 200 percent of the current poverty threshold would be a better measure to adequately capture the modern costs of living (Boushey et al. 2001). For poverty, we used the percentage of households at 185 percent of the poverty level or below, a measure reported in the Decennial Census.

Income inequality—Timberlands have rapidly changed hands in the last decade with millions of hectares of forested land being acquired by real estate investment companies and timber investment management organizations. Many of these parcels of land have been developed into high-end residential properties, which have attracted wealthy retirees and newcomers to forest communities (Bliss et al. 2008). Inmigration of wealthy suburbanites searching for recreation and retirement has greatly altered the socioeconomic structure of amenity-rich communities in the west. While the new residents bring skills, experience, and extra-local connections to these communities, they tend to support low-paying menial jobs instead of employment opportunities with large multiplier effects like wood processing. Nelson (1997) and Brown (1995) found socioeconomic differences between long-time (less wealthy) and newer (more wealthy) residents in rural counties in the Pacific Northwest. Wilkinson (1991) found that income inequality is often a source of social tension.

We measured income inequality using the Gini index, a standard measure of inequality (Allison 1978). To calculate the Gini index for inequality of household income, households are ranked by income, and the aggregate share of income held by each group of households is divided by the total aggregate income. For example, the bottom 10 percent of households may hold 1 percent of the total household wealth, and the next 10 percent of households may hold 5 percent of the total household wealth, and so on until all households are included. The Gini index compares this distribution to a perfectly proportional distribution and incorporates all of this information into a single measure with a value of [0, 1] (DeNavas-Walt et al. 2009). A zero Gini index expresses total equality, where all households receive the same income, while a value of 1 expresses complete inequality (one household receives all income, the others receive none). The Gini index was calculated using household income data from the U.S. Decennial Census.

Unemployment—Unemployment is a common metric in calculating socioeconomic well-being (Ashton and Pickens 1995, Charnley et al. 2008, Feser and Sweeney 2003, Isserman et al. 2009, Stedman et al. 2004, 2005). At an individual level, access to work is a basic economic need and is typically a necessary, though not sufficient, condition for generating a livable income. At a regional level, a high level of unemployment constitutes wasted economic potential, a drain on public resources, and a limitation on public service capacity. The U.S. Census Bureau defines civilians as unemployed if they (1) were neither "at work" nor "with a job but not at work" during the reference week, (2) were looking for work during the last 4 weeks, and (3) were available to start a job. The percentage of the population unemployed was taken directly from this Decennial Census information.

Unattached youth—young adults not working or in school represent both the lack of opportunity in an area, as well as the potential for greater social disruption. This cohort tends to contribute to community distress, as youth who move away from the school and work trajectory are associated with lower lifetime earnings, increased poverty, homelessness, and criminal activity (Montalvo and O'Hara 2008). Such youth are often found in black and Hispanic populations, among teen mothers, in rural areas, and in the youth criminal justice system (Snyder and McLaughlin 2008). To calculate the percentage of unattached youth in a community, we used the population of people ages 16 to 19 not in school, not in the armed forces, nor currently employed from the U.S. Decennial Census data.

Population decline—Dramatic changes in population demographics likely decrease the level of social capital in a community and contribute to community distress, at least in the short term. Measuring population turnover is problematic, though, in that a stable population does not necessarily imply a stable community. Inmigration and outmigration occurring in tandem may result in a stable population, but a very different demographic.

We used percentage of decline in population from the 1990 to 2000 Decennial Censuses as one indicator of community distress. Communities that remained stable or increased in population were given a value of zero. Although increases in population may indicate both positive and negative impacts on community well-being, population decline is less ambiguous. Population adjustment in areas of high unemployment or low income could be considered a positive shift in labor resources regionally, but most certainly indicate negative conditions in the community experiencing this loss (Feser and Sweeney 2003). At a minimum, an absolute loss of population can result in a shrinking work force, lower tax revenues, and smaller school enrollments. When we consider that migration tends to be a highly selective process, with younger, better educated, and higher skilled workers moving first in response to economic decline, the negative effects of population change become more pronounced (Bronars and Trejo 1992, Kwok and Leland 1982, Sjaastad 1962).

Indicators of business capacity (BC)—

Harvest activities may also be focused with a goal of helping communities that are positioned for success, but lacking only in timber supply. In contrast to the idea of assisting communities that are struggling, managers under this scenario would want to target fuels reduction treatments in watersheds that may benefit communities with high business capacity, where small increases in supply may have the potential to secure mill longevity and wage income for the immediate future. In all cases, higher values of each indicator used represent positive factors of the community that contribute to higher business capacity. Business capacity indicators for the WIS include education, the size of the labor force, presence of a mill, and employment specialization in forestry, wood products, or paper.

Business capacity indicators for the WIS include education, the size of the labor force, presence of a mill, and employment specialization in forestry, wood products, or paper.

Educational attainment—Education is a common proxy for human capital, another variable that contributes to socioeconomic well-being and community resilience (Donoghue and Haynes 2002, Reeder and Brown 2005, Stedman et al. 2004). In the past, family-wage jobs in forest communities traditionally did not require much education past high school. However, the decline of the forest products industry requires new skills on the part of community residents if they are to adapt successfully to this change. We calculated the percentage of population with an associate's degree or higher to represent educational attainment, one indicator of business capacity. These data are available through the U.S. Decennial Census.

Labor force size—The percentage of the population in the labor force, also calculated from Decennial Census data, indicates the relative number of people available for employment. Communities with large proportions of youth or older residents are unlikely to take advantage of new business and employment opportunities. Retirement income and transfer payments account for one quarter or more of economic activity in some rural areas (Rasker 1994). Although some retirees may possess financial capital to invest in new enterprises, a robust labor force is needed to attract new investment and take advantage of new jobs.

Recent mill presence—The presence of a mill indicates whether the physical infrastructure is available to process new supplies of wood. A community with a mill currently operating has the most capacity to easily utilize raw materials supplied as a result of forest treatments, but mills that have recently closed may also be re-opened with capital investment. We drew on several sources to obtain data on current mill locations in the four states. The methodology and specific sources used are described below.

Random Lengths publishes a buyers' and sellers' directory of the forest products industry annually (Random Lengths 2010, 2000). This publication, titled "The Big Book," is compiled from information voluntarily provided by mills and secondary processers across the country. Mills listed in the 2000 or 2010 editions of The Big Book were considered as open in those years. The USDA FS periodically publishes profiles of softwood sawmills by state, along with recent capacity data (Spelter and Alderman 2005, Spelter et al. 2009); these reports were used to add to or verify open mills listed in the Big Book. In addition, data from industry consultants of open mills in OR/WA in 2011 (Ehinger 2011) were used to further verify currently open mills.

A community with evidence of a mill open in 2011 or 2010 was given a mill capacity value of 1. If a mill was present prior to or in 2005, the mill capacity was 0.5; if there was a mill present prior to or in 2000, the mill capacity was 0.25. In all cases, all reported mills were counted, regardless of processing capacity or size of logs used in the mill. Communities with no evidence of a mill since 2000 were given a capacity of zero.

Specialization—We used the percentage of the labor force currently employed in forestry, wood products, or the paper industry to indicate whether the labor force in a given community has the appropriate skills and experience to take advantage of employment and business opportunities from fuel treatments. Percentage of specialization in relevant industries was calculated as the number of employees in three industries (three-digit North American Industry Classification System Codes): forestry and logging (113), wood product manufacturing (321), and paper manufacturing (322), as a proportion of all employees. County-level percentage of specialization was calculated in this way from U.S. Bureau of Labor Statistics employment data and applied to all CCDs in that county.

Indicators of federal forest policy impact (FP)—
In addition to focusing on communities with high distress or those with high business capacity to target through fuels reduction treatments, an additional motivation for public land managers may be to assist communities that have experienced changes in fortune as a result of shifting federal forest management policies. If federal land management policies have affected some communities more than others, these communities may have more weight for managers hoping to impact specific places through harvest activities. Forest Service policy impacts are represented by the percentage of decline in county payments from the late-1980s to the mid-2000s.

Forest Service policy impacts are represented by the percentage of decline in county payments from the late-1980s to the mid-2000s.

Decline in county payments—It is difficult to quantify the effect of changes in public forest management on rural communities. One very real effect, however, of declines in public timber harvest has been the decline in contributions to county budgets related to the revenue-sharing agreements that date back to the early 1900s. Although initially payments were small (along with public timber harvest), payments to counties were still linked to harvest receipts in the late-1980s and peaked at that time in terms of the amount of money distributed to rural counties. Legislation that replaced the timber harvest county payments in the 1990s altered the relative payment amounts to each county and increased some counties' payments, while decreasing others. The large declines in public timber harvest in the late 1990s led to alternative payments designed to ease the strain on county budgets. USDA FS Policy impacts of the USDA FS were indicated by the percentage of decline in county payments from the late-1980s to the mid-2000s. These two time points represent recent overall highs and lows, respectively, in payments to counties from all sources, and were used to proxy negative effects of changes in land management policy. Communities in counties that experienced a decline in county payments were represented by their percentage of decline, whereas communities in counties that experienced no change or an increase in county payments were given a value of 0.

We originally included harvest decline as a USDA FS policy impact variable, but discarded it because of lack of variation between communities. Because all national forests in OR/WA witnessed a comparable decline in harvest rates, inclusion of this variable did not add to our analysis. Although the cut and sold reports were not as easily available at the level needed to check similar trends for AZ/NM, evidence of the decline in overall harvest in AZ/NM indicated that the same was likely true for each national forest there (Keegan et al. 2001a, 2001b). Another indicator considered but not included owing to data gathering difficulties was the change in USDA FS employment.

Estimates of Potential Biomass Supply

Biomass supply is a key component of the possible benefits of fuel treatments for rural communities. Through the use of secondary data, primarily from the U.S. Census Bureau, the WIS captures the need (the Socioeconomic Status, Business Capacity, and Forest Policy indices) at the community level, but also must incorporate the potential for each watershed to actually supply material to nearby places. To quantify the potential biomass, we relied on two different methods: one for OR/WA and one for AZ/NM (where a lack of data precluded utilization of the method used for OR/WA). Both methods are detailed below.

Potential biomass in the Pacific Northwest (OR/WA)—

In OR/WA, potential biomass estimates for each watershed began with maps of current vegetation and vegetation models created for ILAP. Current vegetation conditions were imputed using a nearest neighbor modeling method to match USDA FS Forest Inventory and Analysis plot-level data to remotely sensed imagery (chapter 2). Vegetation state classes were defined using state-and-transition models (STMs). The STMs are used to represent alternative vegetation state classes (combinations of vegetation cover and structure) within potential vegetation types (chapter 2). Transitions link state classes and represent processes such as succession, disturbance, and management activities. The current vegetation condition map was classified into STM state classes, which were used to calculate potential biomass (this analysis is described in detail in chapter 4).

For the eastern forests in OR/WA and the southwestern forests in Oregon,[4] ecologists from the USDA FS identified the desired ending state classes for each current state class and the transition required to get there given a goal of forest restoration. For example, within the dry mixed-conifer potential vegetation type

[4] West-side forests in OR/WA have a very infrequent fire regime and forest conditions are generally not out of historical range (Agee 1993), so these regions are unlikely to be targeted for large-scale vegetation management or restoration treatments and were omitted from the analysis.

in the Oregon East Cascades modeling zone, there is a state class characterized by Douglas-fir/white fir trees of medium size and medium canopy cover. A restoration target state class in this vegetation type and region is Douglas-fir/white fir trees of medium size with open canopy. To move from the former state class to the latter, a partial harvest of small- to medium-sized trees at a medium density was selected as the transition.

An estimate of standing biomass was assigned to each state class using methodology documented in Zhou and Hemstrom (2010) and chapter 4. To obtain an estimate of potential biomass removal for each state class that was a candidate for treatment, we subtracted the standing biomass estimate for the ideal ending state class from the standing biomass estimate for the current state class. Using the example above, the total nonleaf biomass for the more dense state class is estimated at 73.9 oven-dry metric tons (odt) per hectare, while the total non-leaf biomass of the more open state class is an estimated 42.9 oven-dry metric tons per hectare. The difference between these two values (73.9 – 42.9 = 31 odt of material removed per hectare of land in that initial state class) represents the maximum amount of biomass material that could be removed under a restoration scenario, where all areas that are currently in a more dense condition are harvested to create more open, fire-resistant conditions. To calculate total potential removals per watershed, we simply multiplied the per-hectare amount of potential biomass removed under resilience goals by the amount of that state class currently in each watershed.

Because we calculated biomass based on the potential removed from each and every hectare of land, this estimate represented an upper bound on the quantity of biomass that might be removed and supplied to communities for processing from each watershed as a result of resilience activities.

Because we calculated biomass based on the potential removed from each and every hectare of land, this estimate represented an upper bound on the quantity of biomass that might be removed and supplied to communities for processing from each watershed as a result of resilience activities. These estimates included removal of all available biomass, whether stems, branches, or leaves, without leaving any material behind for wildlife or nutrient concerns. Byproducts of current management (logging slash or mill residues) are not included. In addition, different types of biomass destined for specific forest products (e.g., sawtimber, hog fuel chips) were not identified. This estimate was intended only to indicate maximum potential supply relative to other watersheds and was not meant to represent a likely management scenario. It also did not assess financial feasibility.

Given the total biomass potential in each watershed, all watersheds in OR/WA were then ranked by their contribution to total potential biomass supply in the region. The biomass supply variable, as incorporated into the WIS, weighted each watershed relative to all others in the region in terms of its potential biomass supply on a scale of zero to one.

Potential biomass in the Southwest—
Biomass estimates in AZ/NM, where the more detailed resilience strategies and biomass amounts by state class information was not available, were calculated differently from OR/WA. For AZ/NM, biomass estimates were based on the Forest Biomass Resources Supply Study (Cook and O'Laughlin 2011a, 2011c). Researchers at the University of Idaho provided county-level estimates of biomass that might be available at different roadside prices (from $10 to $40 an odt) for all Western States for the Western Governors' Association, following a method established in an earlier, state-level analysis of biomass supply potential. These biomass estimates incorporated several potential sources: forest thinning to reduce fuel loads or improve forest health, logging slash, and mill residues (Cook and O'Laughlin 2011b). Most of this material is, by definition, unmerchantable poles, small and low-value sawtimber, and chip material. As such, it represents opportunities for either use in energy production, or in sawmills with small-diameter stem processing capability.

To portion out the county-level biomass estimates to the watershed level necessary for the WIS, we used the current vegetation map (chapter 2) to identify areas of each watershed with at least 20 percent forest canopy cover. County and watershed layers were combined to create modified counties that coincided with watershed boundaries. County-level biomass total estimates were portioned out to each watershed, with forest canopy cover based on the watershed's contribution to the total county area. For example, if a county had a potential biomass of 25,000 odts per year, and a watershed with complete forest cover comprised 20 percent of the modified county's forested area, then the watershed would be allocated a proportional amount of biomass (20 percent of 25,000 = 5,000 odt/yr). At the modified county level, the total estimates of the biomass for all watersheds in a given county equal the totals given in the Forest Biomass Supply Analysis at a roadside price of $40 per dry ton.

Similar to the process used following the calculations of actual biomass estimates in OR/WA, watersheds were ranked by their contribution to total regional biomass that might be provided. The variable used in the WIS represented the potential contribution of each watershed relative to other watersheds in the region that have forest cover. However, the different methodology used in the two regions, along with the use of a regional baseline in ranking the watersheds, means that the resulting WIS indices are not directly comparable between OR/WA and AZ/NM. For example, a watershed with a high ranking of biomass potential in OR/WA is not the same as a watershed with a high ranking in AZ/NM. Each considered different pools of material as potential biomass, was developed with different motivations, and each was ranked against a different regional total potential supply.

Weights and Structure of the Gravity Index

The component data for each of the community-level indices—the socioeconomic status variables, the business capacity variables, and forest policy variable in OR/WA—were all scaled to lie in the interval [0, 1], inverted when necessary, and truncated in cases when only some of the data values were of interest (e.g., we were only interested in communities with population decline). Adding the component data together resulted in a range of values for the two regions and three indices (table 6.2). Higher values of the SES and FP indices indicate greater distress, a negative measure, while higher values of the BC index indicate greater capacity, a positive measure. High values for all three increase the priority ranking of the watershed.

The indices were inversely weighted by travel time between each watershed-community pair. We constructed the WIS for each watershed by summing the weighted indices for the five nearest communities. This resulted in a score for each watershed consisting of the attributes of the communities most likely to receive biomass supply from that watershed. By limiting the analysis to include only the nearest five communities, we avoided creating a WIS that is dominated by population; it is more responsive to the attributes of the communities. For inclusion in the final calculations, the biomass rankings in tenths based on each watershed's contribution to the total regional biomass potential were scaled to lie in the interval [0, 1].

The resulting comprehensive WIS for watershed *j* was calculated as:

$$\mathrm{WIS}_j = \sum_{i=1}^{n} \frac{\mathrm{BS}_j\,(\mathrm{SES}_i + \mathrm{BC}_i + \mathrm{FP}_i)}{\mathrm{d}_{ij}},$$

where

WIS_j = watershed impact score for the j^{th} watershed,

BS_j = relative biomass supply for the j^{th} watershed (a push factor),

SES_i = socioeconomic status index for the ith community (a pull factor),

BC_i = business capacity index for the i^{th} community (a pull factor),

FP_i = forest policy impact index for the i^{th} community (a pull factor), and

d_{ij} = travel time between the j^{th} watershed and the i^{th} community.

Table 6.2—Ranges of nonstandardized community indices for Oregon and Washington (OR/WA), and Arizona and New Mexico (AZ/NM)

Region	Range of values of socioeconomic status index	Range of values of business capacity index	Range of values of forest policy impacts Index
OR/WA	0 to 2.82	0.30 to 2.58	0 to 1
AZ/NM	0 to 3.00	0.04 to 2.54	NA

NA = not applicable.

The implicit weights on the indices are 5, 4, and 1 for SES, BC, and FP, respectively, based on the number of variables they each contain. These weights can be adjusted to reflect the values of the policymaker.

Resulting Datasets

We produced nine gravity indices. Each gravity index represented a different combination of data on the numerator; all summed up the values at the watershed level for the nearest five communities and inversely weighted the total by the travel time between watershed and community. Our nine gravity indices included:

- Three simple WIS that incorporated only the standardized community condition variables (SES, BC, and FP indices).
- A WIS that summed these three variables together.
- A WIS that comprised only the relative biomass supply ranking.
- Three WIS that combined each of the community characteristics with the biomass supply ranking (e.g., SES weighted by biomass ranking).
- A comprehensive WIS that combined all information.

We created maps to illustrate how current community conditions translate into WIS, shown in figures 6.1, 6.2, and 6.3. While the WIS is designed for use in a geographic information system-based decision support system for full flexibility, static maps allow the resulting data sets to be displayed in multiple media. In addition, maps allow any stakeholder to visually identify areas where treatments may be able to have the largest positive impact on communities. A full set of maps will be presented in subsequent reports that will describe and interpret our results for both regions.

Current conditions at the community level (the nonstandardized SES, BC, and FP index values) are shown for OR/WA (fig. 6.1). Values are grouped into quintiles, where each quintile represents one-fifth of the observations. In all cases, the first quintile (shown in yellow) are the communities making up the lowest fifth of all the possible values. The dark blues (4^{th} and 5^{th} quintiles) are communities with characteristics that may be targeted by land managers as places to benefit from increases in supply.

We also mapped the relative potential biomass supply for OR/WA (fig. 6.2). The figure shows the volume removed for each watershed per treated acre, relative to the mean amount removed, and the contribution of each watershed to the total potential biomass (the value used in the WIS). The map displays the relative rankings in fifths, so the lowest group (watersheds in yellow) contributes the lowest amounts per acre or the last 20 percent of the total potential biomass supplied in the region. The watersheds in blue contribute either a high amount of biomass per treated acre

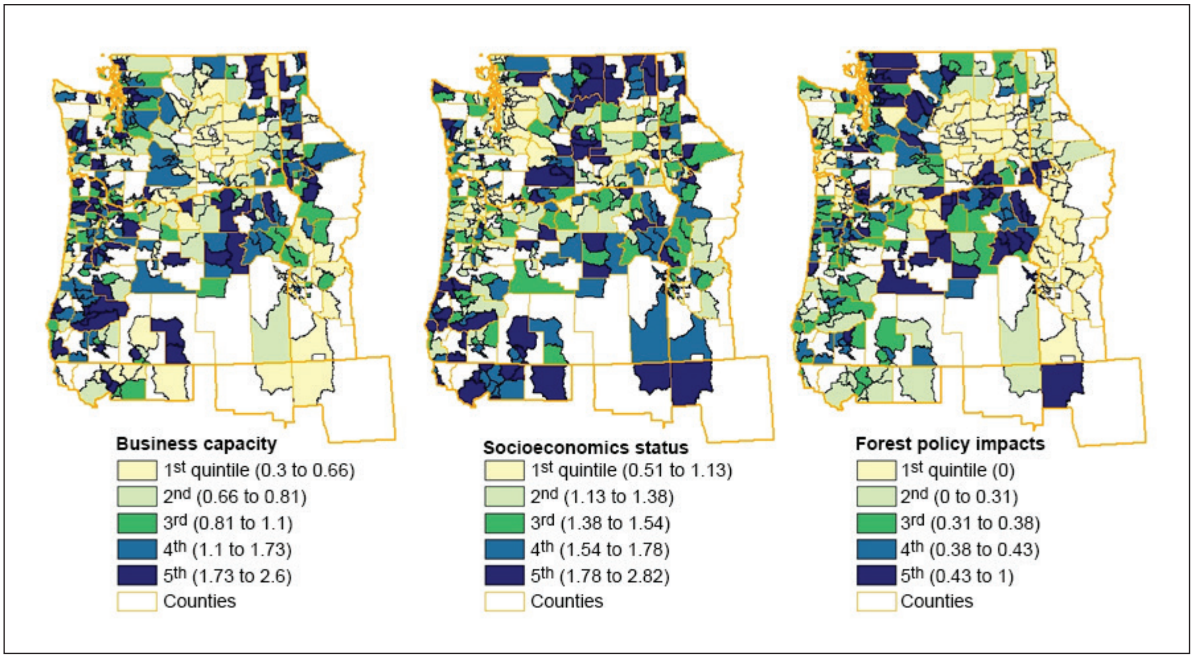

Figure 6.1—Community characteristics in the Pacific Northwest (Oregon and Washington). Data are grouped by quintile. Yellow indicates the lowest values of the indices. Communities in shades of blue are areas where managers may want to prioritize action, either because they have a high business capacity to utilize material, have low socioeconomic status and may benefit from stimulus, or because they have been impacted by changes in federal forest policy.

or a high amount of the total regional biomass potential. These watersheds have the potential to supply a large amount of material relative to other watersheds in the region.

Because the biomass calculation methodology was quite different between the OR/WA and AZ/NM, and the rankings were of each watershed relative to others in the region, maps and values are not comparable between regions in the same way as community conditions. In OR/WA, there was wide variation in potential biomass supply estimated between the heavily stocked east side of the Cascades in both states to the minimally forested Columbia Basin and southeastern Oregon. In AZ/NM, conditions are much less variable across the landscape, but OR/WA has the potential to produce far more overall biomass.

One possible configuration of the gravity index for OR/WA is shown in figure 6.3. In this case, the scores for each community index and the biomass ranking are added together and weighted by the distance between the watershed and the nearest five communities, as given in the formula above. The data are, however, flexible enough that a manager could compute a different gravity index by combining all community characteristics together or incorporating more nearby communities in the score.

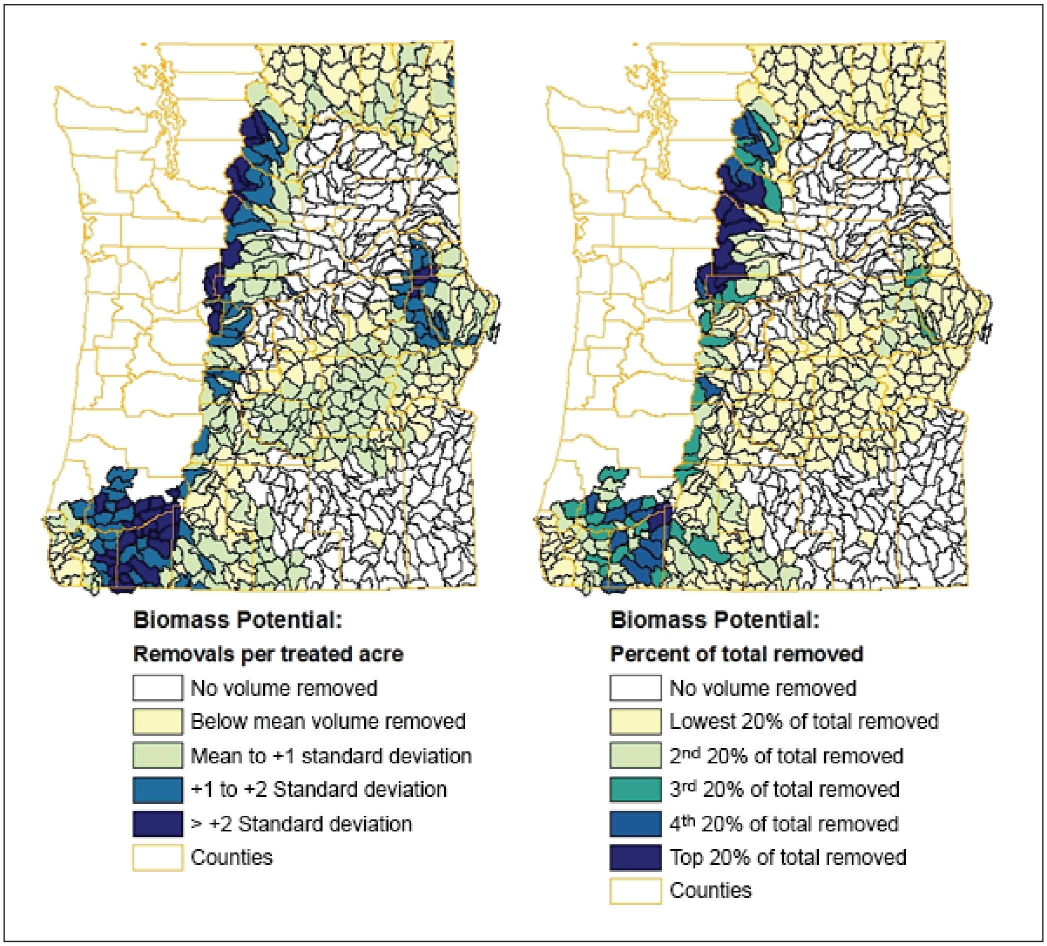

Figure 6.2—Biomass potential in the Pacific Northwest (Oregon and Washington). The map on the left shows the volume removed for each watershed per treated acre, relative to the mean amount removed. Watersheds in blues have a high amount of biomass potential per treated acre. The map on the right shows the contribution of each watershed to the total potential biomass. The watersheds in blue, again, contribute a high amount of the total regional biomass potential, and these watersheds have the potential to supply a large amount of material relative to other watersheds in the region.

Conclusions

Our goal was to provide a spatial data layer for inclusion in an EMDS system, similar to those provided by other teams of ILAP (see chapter 1). This knowledge-based system operates within ArcGIS and allows land managers or policy analysts to incorporate disparate information across landscapes and prioritize activities to meet management goals easily and flexibly. For example, layers showing the future condition of the forest in terms of wildlife habitat, fire hazard, and financial feasibility of treatments can allow managers to highlight watersheds for treatment that can meet all goals of reducing hazard, improving habitat, and having treatments pay for themselves.

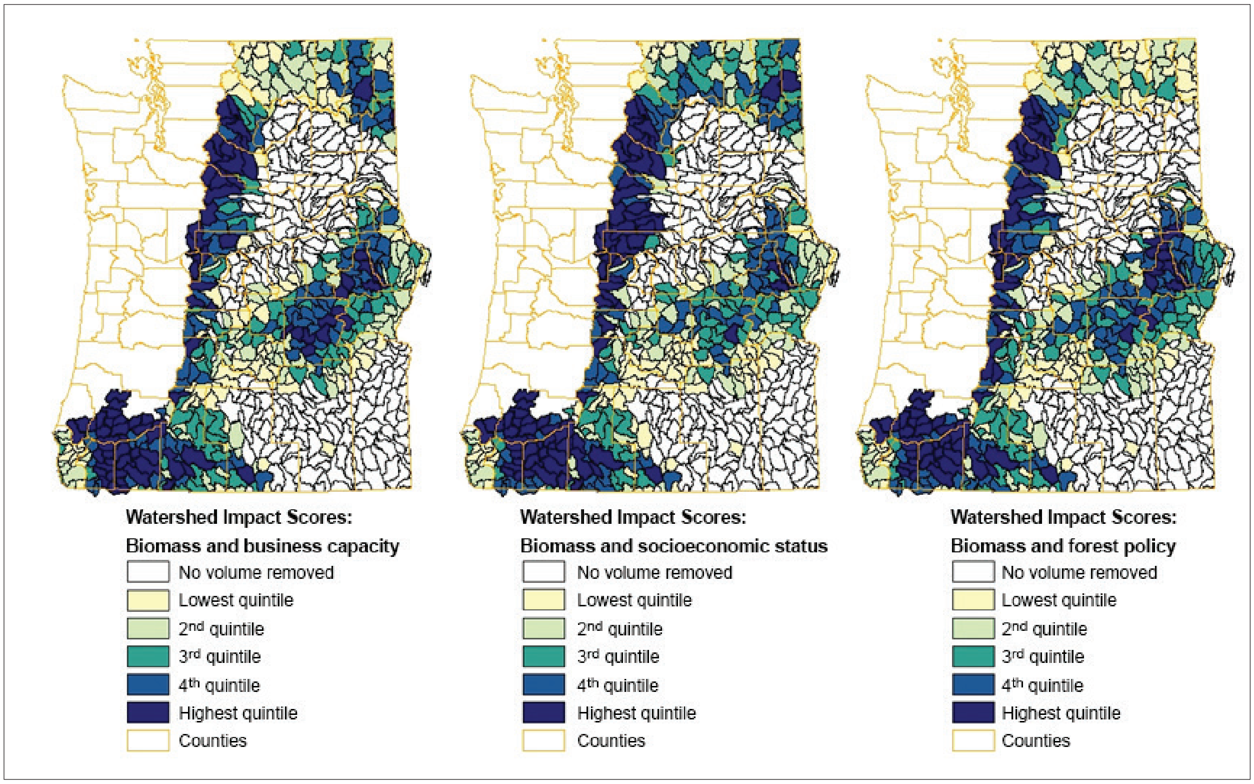

Figure 6.3—Watershed Impact Scores (WIS) for the Pacific Northwest (Oregon and Washington). These maps combine biomass potential with business capacity (left), socioeconomic status (center), and forest policy effects (right). Watersheds in yellow offer the lowest values for each combination, while watersheds in blues offer the highest potential in terms of both biomass and potential to contribute to communities with each of the characteristics. In all cases, the east side of the Cascades in both states along with southwestern Oregon consistently combine high biomass potential in watersheds near to communities who are in socioeconomic distress, have high business capacity, and have been affected by U.S. Forest Service policy. Areas of central Oregon (Grant and Union Counties) and northeastern Washington (Stevens, Pend Oreille, and Spokane Counties) show mixed potential, depending on which community characteristic is of importance.

The community economics information produced in this study allows managers or analysts to add an additional criterion that reflects potential effects of treatments on rural communities in the two regions. The data layer we provided for the EMDS system included nine gravity indices that can be mapped at the watershed level, along with other watershed-level information. Gravity indices are a useful tool for weighting conflicting factors that are spatially related. Distance matters to managers when considering where to place fuel treatments, as they must weigh the costs of removing the material against the benefits created by removing it. The material removed has an opportunity to benefit nearby communities, and the gravity index allows us to incorporate possible community weighing factors into the decision-making process. The compilation of already-existing data in this way allows managers to consider communities when choosing where to place fuel treatments on the landscape.

Distance matters to managers when considering where to place fuel treatments, as they must weigh the costs of removing the material against the benefits created by removing it.

However, there are limitations to this work. Because we want this tool to be replicated and updated in the future, we chose to use census data in many cases. This also allowed the use of a smaller level of geographic aggregation than counties. However, the use of census data resulted in a tradeoff between timeliness of the information for more geographically specific and readily available information. Most of the data were obtained from the 2000 Census, and yet we can expect that the fortunes of many communities, particularly rural, timber-reliant ones, have changed since then, as the United States entered a prolonged recession, and the reduction in demand for wood products was particularly felt in rural OR/WA. In addition, our WIS provides only a snapshot view of community conditions. With the release of the 2010 census data at the subcounty level (assuming no major changes in CCD boundaries occur), both updating and incorporating trend information into the community characteristic indices would increase the relevance of the WIS.

Although we tried to balance considerations of simplicity and accuracy when determining what to use as a community in this study, our boundaries and definitions are clearly somewhat subjective. Reliance on preexisting data limits the true accuracy with which researchers can define communities.

For this study, we made every effort to keep the methodology simple enough to be easily updated and expanded (updated when the new census information is released or expanded to other states). Nonetheless, there were more steps and complicating factors than we had anticipated, particularly in incorporating biomass estimates for watersheds. Still, it is our hope that versions of this work can be adapted to other locations with modifications as needed.

Across the dry forest regions of the west, forest health has declined and the risk for catastrophic wildfire has increased. At the same time, rural communities in the region have struggled in the face of declining harvests from public lands and losses in employment in traditional industries. Fuel treatments have been widely promoted as a way to improve forest health while providing jobs for rural residents skilled in forestry, but the fact that budget limitations will force managers to choose priority forest areas to treat is sometimes neglected. The ILAP project was developed to provide land managers with a way to account for multiple goals by creating a decision support system capable of combining disparate information about wildlife habitat, wildfire hazard reduction, and financial feasibility of treatments in order to inform their on-the-ground prioritization. The potential increase in supply material will not be dispersed evenly across the landscape, but will by necessity have the potential to benefit specific locations and not others, with concurrent benefits for particular rural communities. We developed the watershed impact scores as a way of incorporating information about communities into the prioritization of watersheds to treat. Our primary goal is that this analysis will help managers to understand the implications of actions in specific locations with respect to local communities, and to make intentional decisions when choosing where to act on the landscape.

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Chapter 7: Developing Climate-Informed State-and-Transition Models

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Chapter Summary

Land managers and others need ways to understand the potential effects of climate change on local vegetation types and how management activities might be impacted by climate change. To date, climate change impact models have not included localized vegetation communities or the integrated effects of vegetation development dynamics, natural disturbances, and management activities. We developed methods to link a dynamic global vegetation model to local state-and-transition models. Our methods allow examination of how climate change effects might play out in localized areas given a combination of vegetation dynamics, natural disturbances, and management actions. Initial results for the eastern Cascades of central Oregon suggest that ponderosa pine forests are likely to remain the dominant forest vegetation type while moist and subalpine forests decline. Sagebrush shrublands will likely also remain the dominant arid land vegetation type, but could undergo substantial fluctuations and begin to give way to shrublands dominated by warm-season species.

Our methods allow examination of how climate change effects might play out in localized areas given a combination of vegetation dynamics, natural disturbances, and management actions.

Introduction

Resource planners and managers are often inundated with information on climate change and potential corresponding trends in wildfire, drought, insect outbreaks, and silvicultural treatment effects. These factors are frequently discussed independently of each other without considering how one may affect the other. Incorporating and utilizing various pieces of information can be difficult, because information on these factors is often generated at different spatial and temporal scales with both known and unknown uncertainties.

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One source of confusion is the interpretation of projections from general circulation models (also referred to as global climate models [GCMs]). The GCMs reflect the current state of knowledge and understanding of how the planet's climate system works. However, our climate system is very complex, and different GCMs incorporate different assumptions about the physical drivers of the Earth's climate, resulting in varied projections of future temperature and precipitation. It is difficult for managers and planners to translate these various climate projections into plausible scenarios of future ecosystem change and to develop robust adaptation strategies, because there are many models and assumptions about the current understanding of local feedbacks.

Downscaling climate projections from different GCMs to drive projections of vegetation growth, and corresponding changes in drought and wildfire, is a necessary step to begin to make GCM results useful to natural resource managers and planners. Dynamic global vegetation models (DGVMs) such as MC1 (Bachelet et al. 2001, Daly et al. 2000) have been developed to integrate climate information and grow vegetation through time by simulating fundamental ecological processes like carbon and water uptake and losses, and competition for light, water, and nutrients. The required climate information to run the DGVMs, including minimum and maximum temperatures, precipitation, and vapor pressure deficit, is provided by interpolated climate observations for both the historical period and climate scenarios for the future. However, because DGVMs simulate changes in broad functional vegetation types, the output is not readily usable by managers and planners and is often not appropriately scaled to address many management and planning questions. Moreover, many DGVMs omit important drivers of vegetation dynamics, such as species-specific fire tolerance and human-driven landscape change, and generalize processes such as plant competition and succession.

State-and-transition models (STMs) incorporate effects of succession, management, and disturbance on relatively fine-scale vegetation composition (i.e., communities) and structure. The STMs have been used for many different types of landscape-scale assessments (e.g., Arbaugh et al. 2000; Forbis et al. 2006; Hemstrom et al. 2001, 2007; Merzenich et al. 2003; Merzenich and Frid 2005; Weisz et al. 2009). However, most STMs do not currently incorporate potential effects of climate change. Because STMs do not consider a changing climate, site potential in these models currently does not change through time, resulting in static potential vegetation for any modeled location. Fire frequency and severity is also currently unchanged through time in STMs. Integrating vegetation models such as MC1 with STMs would help make the information provided by MC1 about climate change effects on vegetation and disturbance regimes more useful for natural resource managers.

The Integrated Landscape Assessment Project (ILAP) was a $6 million investment funded by the American Recovery and Reinvestment Act to inform watershed-level planning, assessment, and prioritization of land management actions across wildlands in Arizona, New Mexico, Oregon, and Washington. The ILAP has partnered with many organizations across the Western United States, including state and federal agencies, universities, nonprofit organizations, and tribes. As part of ILAP, we used the MC1 DGVM to inform changes in vegetation potential and changing trends in fire probabilities in a suite of local STMs. This process was used in two focus areas (central Oregon and eastern Arizona). The outcome of our process was a collection of climate-informed STMs, which are connected to each other, allowing climate-driven shifts in potential vegetation to occur. These climate-informed STMs can be used to weigh potential benefits or tradeoffs associated with alternative land management approaches in mid- to broad-scale landscapes. In this chapter, we describe the methods we used to develop climate-informed STMs in central Oregon. Although the approach was nearly identical in eastern Arizona, we use central Oregon here to illustrate the process and results that can be produced.

Climate-informed STMs can be used to weigh potential benefits or tradeoffs associated with alternative land management approaches in mid- to broad-scale landscapes.

Study Area

The study area is a landscape of forested and arid vegetation that covers approximately 1 023 808 ha in eastern central Oregon (fig. 7.1). The study area lies along a sharp rain shadow generated by the north-south trending Cascade Mountain range. Elevations vary from about 1200 m to above 2400 m. The climate is transitional between moist, maritime conditions west of the Cascade crest and continental conditions to the east. Annual precipitation varies from over 200 cm along the Cascade crest to less than 35 cm along lower timberline, and 25 cm at the lowest elevations in shrub-steppe environments. Most of the precipitation falls as rain and snow during the winter months, with snowpacks of more than 2 m common in upper elevations. Summers are warm and dry, often with several weeks of very low or no precipitation and warmest temperatures at lower elevations exceeding 30 ˚C. Thunder storms and lightning commonly occur from April through September.

A variety of mostly coniferous tree species dominate the forested lands. We defined seven potential vegetation classes (PVCs) by aggregating plant associations described by Simpson (2007), Johnson and Simon (1987), Johnson and Clausnitzer (1992), and Johnson and Swanson (2005), and added an additional eighth type that the DGVM indicates might be common in the future:

1. Temperate needle-leaved forests. Ponderosa pine is the dominant early early-seral tree species at the lowest forested elevations. Dry forests containing a mix of ponderosa pine, Douglas-fir, and grand fir are abundant at middle elevations.

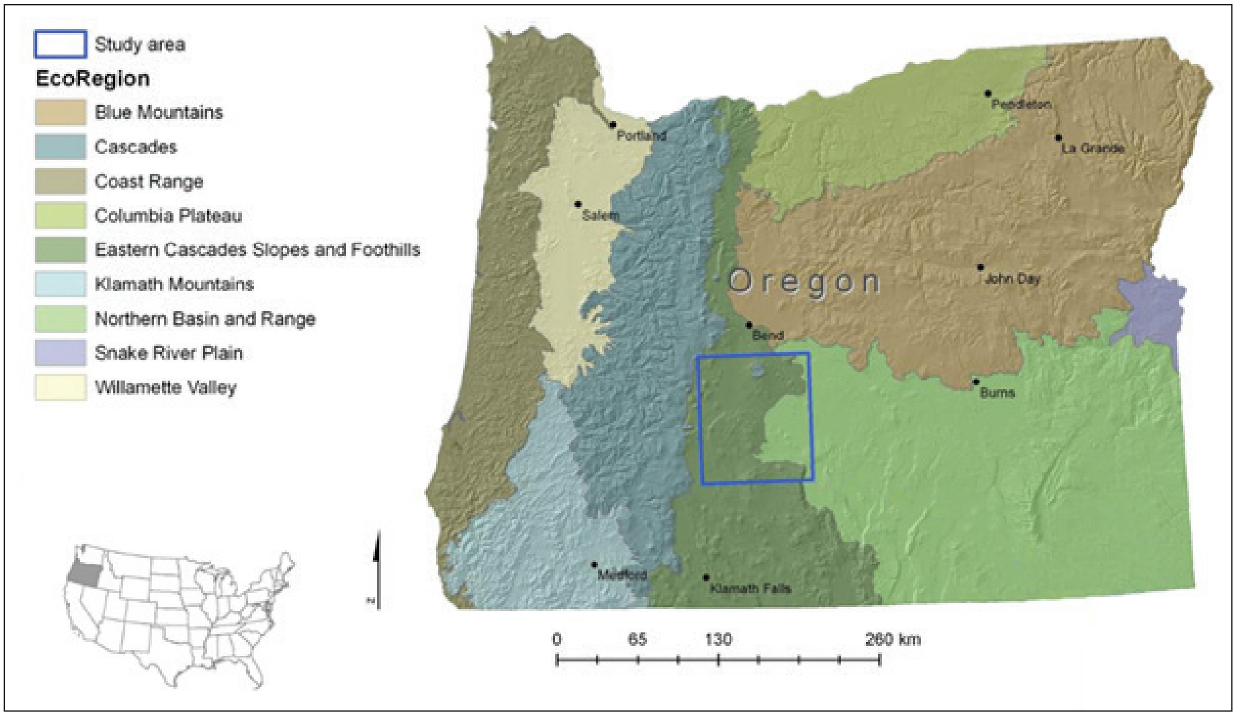

Figure 7.1—Study area and ecological (Omernik level III) regions in central Oregon.

2. Cool needle-leaved forests. Upper montane forests are a variable mixture of conifer species, including Douglas-fir, white fir, Shasta red fir, ponderosa pine, and other species. In very moist environments, several conifers more common to the western Cascades may be abundant, including Pacific silver fir, western hemlock, and western redcedar.
3. Subalpine forests. Mountain hemlock dominates upper elevation cold forests in moist environments, while whitebark pine, Engelmann spruce, and subalpine fir dominate in drier cold forests. Lodgepole pine is common in disturbed, especially burned, areas at upper elevations.
4. Temperate needle-leaved woodland. Western juniper and mountain big sagebrush, with several other shrubs, grasses, and forbs, occupy mid- to upper elevation woodlands and shrublands that do not currently support forests.
5. Temperate shrubland. Much of the lower elevation, warm, dry temperate shrubland is dominated by Wyoming big sagebrush communities that have little or no potential for invasion by western juniper. Cheatgrass is a prevalent invasive annual grass in many Wyoming big sagebrush types.
6. Xeromorphic shrubland. Salt desert shrub communities with greasewood, saltbush, and Wyoming big sagebrush are of somewhat limited distribution in low-elevation areas, particularly those with saline soils. Cheatgrass is often invasive in these shrublands.

7. Temperate grassland. Various bunchgrass-dominated grasslands, often including Idaho fescue and bluebunch wheatgrass, form scattered grasslands dominated by cool-season (i.e., C3) species. Cheatgrass can be invasive in these grasslands.
8. Warm-season grassland. Warm-season grasslands do not occur to any significant extent in the study area at present, but do occur in warmer areas to the south. We included a warm-season grassland potential vegetation type because MC1 model projections indicated they might occur in our area in the future.

Methods

MC1 Modeling

MC1 (Bachelet et al. 2001) is a DGVM that reads soil and monthly climate data, and calls interacting modules that simulate biogeography, biogeochemistry, and fire disturbance. The biogeography module simulates life form or plant functional type mixtures, classifying them into PVCs; the biogeochemistry module simulates fluxes and pools of carbon, nitrogen, and water through ecosystems; and the fire module simulates natural fire disturbance. Each module is described in the following paragraphs. MC1 routinely generates century-long, regional-scale simulations (e.g., Bachelet et al. 2000, 2003, 2005; Lenihan et al. 2008).

The MC1 biogeography module simulates mixtures of needle-leaved or broad-leaved evergreen and deciduous trees, as well as temperate (C3) and warm-season (C4) grasses. The tree life form mixture is determined at each annual time-step as a function of annual minimum temperature and growing season precipitation. The temperate/warm-season grass mixture is determined by reference to their relative potential productivity during the three warmest consecutive months. Tree and grass life form mixtures, their biomass levels as simulated by the biogeochemistry module, and growing degree-day sums are used to determine which of several dozen possible PVCs occurs in each simulated grid cell each year (grid cell size was 800 m in this study). Shrubs are simulated as short-stature trees.

The MC1 biogeochemistry module is a modified version of the CENTURY model (Parton et al.1993), which simulates plant growth, organic matter decomposition, and the movement of water and nutrients through the ecosystem. Plant growth is determined by empirical functions of temperature, moisture, and nutrient availability. In this model, plant growth was assumed not to be limited by nutrient availability. The direct effect of an increase in atmospheric carbon dioxide is simulated using a beta factor (Friedlingstein et al.1995) that increases maximum potential productivity and reduces the moisture constraint on productivity. Grasses compete with woody plants for soil moisture and nutrients in the upper soil layers, where both are rooted, while the deeper rooted woody plants have sole access to resources in deeper layers. The growth of grasses may be limited by reduced light levels in the shade cast by woody plants. The values of model parameters that control woody plant and grass growth are adjusted with shifts in the life form mixture determined annually by the biogeography module.

The MC1 fire module simulates the occurrence, behavior, and effects of fire. The module simulates the behavior of a fire event in terms of the potential rate of fire spread, fireline intensity, and the transition from surface to crown fire (Cohen and Deeming 1985, Rothermel 1972, van Wagner 1993). Several measurements of the fuel bed are required for simulating fire behavior, and they are estimated by the fire module using information provided by the other two MC1 modules. The current life form mixture is used to select factors that allocate live and dead biomass into different classes of live and dead fuels. The moisture content of the two live fuel classes (grasses and leaves/twigs of woody plants) is derived from soil moisture provided by the biogeochemical module. Dead fuel moisture content is estimated from climatic inputs to MC1 using different functions for each of four dead fuel size-classes (Cohen and Deeming 1985).

Fire events are triggered in the model when the fine fuel moisture code (FFMC) and the buildup index (BUI) simultaneously exceed prescribed thresholds. The FFMC is a numerical rating of the relative ease of ignition and flammability, used in the Canadian Forest Fire Danger Rating System. The BUI is a number that reflects the combined cumulative effects of daily drying and precipitation in fuels, part of the 1964 National Fire Danger Rating System in the U.S. Sources of ignition (e.g., lightning or anthropogenic) are assumed to be always available. Area burned is not simulated explicitly as fire spread within a given cell. Instead, the fraction of a cell burned by a fire event is estimated as a function of set minimum and maximum fire return intervals for the dynamically simulated PVC and the number of years since the last simulated fire event.

The fire effects simulated by the model include the mortality or consumption of vegetation carbon, which is removed from (or transferred to) the appropriate carbon pools in the biogeochemistry module. Live carbon mortality and consumption are simulated as a function of fireline intensity and tree canopy structure (Peterson and Ryan 1986). Dead biomass consumption is simulated using functions of fuel moisture that are fuel class specific (Prichard et al. 2005).

Climate Data—

Climate data sets required for running the MC1 model consisted of monthly values of four variables: precipitation, the monthly means of diurnal extreme temperatures (minimum and maximum), and a measure of atmospheric water content. We used the Parameter-elevation Relationships on Independent Slopes Model (PRISM) climate data set at 30 arc-second spatial grain for the historical period (Daly et al. 2008). This data set contains monthly data for the years 1895 to 2008, for the conterminous United States on a 30 arc-second grid (about 800 m grain). PRISM estimates spatial means of temperature and precipitation for grid cells based on records from nearby weather stations, assigning weights to the station data according to the similarity of the station location to the grid cell location. PRISM, although not the only source for gridded historical climate data for the United States, is well established and has been widely used (Daly et al.1994, Guentchev et al. 2010).

Future climate projections from GCMs included in the Intergovernmental Panel on Climate Change (IPCC) fourth assessment report (Solomon et al. 2007) were used to drive MC1 projections for the 21st century. In this respect, our study is similar to a number of other studies using MC1 at various scales on a variety of landscapes (Bachelet et al. 2008, Conklin 2009, Gonzalez et al. 2010, Lenihan et al. 2008, Rogers 2011). We obtained complete time series of the four necessary climate variables from the data repository at Lawrence Livermore Laboratories produced by the Hadley CM3 (Gordon et al. 2000), MIROC 3.2 medres (Hasumi and Emori 2004), and CSIRO Mk3 (Gordon et al. 2002) GCMs (hereafter Hadley, MIROC, and CSIRO, respectively) for the IPCC Special Report on Emissions Scenarios (SRES) A2 emissions scenario (Nakićenović and Swart 2000). For the future 2070-2099 period, CSIRO projects a relatively cool and wet U.S. Pacific Northwest (+2.6 °C and +176 mm mean annual precipitation), MIROC projects a relatively hot and wet U.S. Pacific Northwest (+4.2 °C and +82 mm mean annual precipitation), and Hadley projects a relatively hot and dry U.S. Pacific Northwest (+4.2 °C and −78 mm mean annual precipitation) (Rogers et al. 2011). For all three GCMs, temperature increases are generally greater in summer than in winter, while precipitation increases in winter and decreases for summer (Rogers et al. 2011). Thus, the seasonality in temperature and precipitation that characterized the Pacific Northwest historically is increased (i.e., there is a greater difference in temperature and precipitation between seasons) in future projections by all three GCMs.

The spatial grids used by the Hadley, MIROC, and CSIRO GCMs are coarse (1.9 x 1.9 degrees for CSIRO, 2.8 x 2.8 degrees for MIROC, and 2.5 x 3.75 degrees for Hadley). We down-scaled the coarse grid GCM data to a 30 arc-second grid using a "delta" or "anomaly" method (Fowler et al. 2007). We first calculated a historical climate baseline (mean values for each month) for 1971-2000, our reference period, using PRISM data. We also calculated a coarse-scale historical baseline for the same reference period using the GCM projections. Secondly, for each month of each year of the future period, we calculated the difference (for temperature and vapor pressure) between or the ratio (for precipitation), of the future value and the reference historical baseline climate (the “deltas”). We then estimated the values of the deltas on the fine (30 arc-second) grid using bilinear interpolation between deltas from the coarse GCM grid. Finally, we calculated the fine grid climate values for each month of each year of the future period using the PRISM historical baseline modified by the fine-scale deltas. Difference deltas were added to the baseline, while ratio deltas were multiplied with the baseline.

State-and-Transition Models

We used a set of STMs that included both previously developed models that were modified and standardized in ILAP, and some new models built for ILAP (chapter 2). The STMs divide vegetation into state classes (boxes), representing combinations of cover type and structure stage within different biophysical environments (potential vegetation types). Boxes are linked by transitions (arrows) that represent natural disturbances, management actions, or vegetation growth and development. This approach builds on methods that represent vegetation development as a set of transition probabilities among various vegetative state classes (within, but not between biophysical environments; e.g., Hann et al.1997, Hemstrom et al. 2007, Keane et al.1996, Laycock 1991, Westoby et al.1989). For example, grass/forb lands might become dominated by small trees and shrubs after a period of time or might remain as grass/forb communities following wildfire. Our STMs include major natural disturbances, such as wildfire (low-severity [<25 percent canopy mortality], mixed-severity [25 to 75 percent canopy mortality], and high-severity [>75 percent canopy mortality]), insect outbreaks, wind disturbance, drought mortality, and others as appropriate to individual ecological systems. Wildfire probabilities in base STMs came from an analysis of empirical wildfire data for the 1984-2008 time period (Monitoring Trends in Burn Severity [MTBS]; Eidenshink et al. 2007; www.mtbs.gov) in the Oregon East Cascades modeling zone (chapter 2). The STMs were developed in the Vegetation Development Dynamics Tool (VDDT) framework, version 6.0.25, (ESSA Technologies Ltd. 2007) and run in the Path modeling platform, version 3.0.4 (Apex Resource Management Solutions 2012, Daniel and Frid 2012).

With the STM approach, a given area can transition from one state class to another within a potential vegetation type with growth, succession, natural disturbance, or management. However, the environmental conditions, and thus site potential, of an area do not change with time. For example, an area classified as having ponderosa pine potential cannot become a juniper woodland over time with drought. However, by adding climate information from the MC1 DGVM, we are able to simulate changes in the distribution of vegetation types over time, allowing us to create more realistic projections of future conditions under changing climate.

Vegetation maps—

We used a set of potential and current vegetation maps as spatial inputs for ILAP (chapter 2). Previously developed current and potential vegetation maps were used for portions of the four-state study area, including most of the forests in Oregon and Washington (with the exception of southeastern Oregon). These maps were developed by the U.S. Department of Agriculture, Forest Service Landscape Ecology Monitoring Mapping and Analysis team (http://www.fsl.orst.edu/lemma/splash.php). However, in most other regions of the study area, existing maps contained insufficient detail or were not sufficiently accurate to inform the STMs for ILAP. Where existing maps were insufficient to initialize the STMs (arid lands in Oregon and Washington and all lands in Arizona and New Mexico), ILAP built new maps using imputation modeling techniques (Crookston and Finley 2008, Ohmann and Gregory 2002). For potential vegetation maps, pixels were essentially assigned the plant association identified in inventory plots as a statistical function of grids of environmental variables, such as topography, slope, and climate. A rule set was developed that classified the plant association identified on each imputed pixel into one of the potential vegetation types. Current vegetation was also mapped using nearest neighbor imputation, but uses plot data (cover by species) instead of plant association and incorporates LANDSAT-TM imagery in addition to environmental variables to characterize current vegetation at the site. A rule-set was used to classify the current vegetation from each mapped pixel to one of our STM state classes. The resulting maps of potential and current vegetation were used to determine which STM to run in each geographic location and the initial starting conditions of the STMs (chapter 2).

Developing Climate-Informed State-and-Transition Models

We made several basic assumptions in connecting MC1 output with the STMs. First, we assumed that the general vegetation dynamics of major potential vegetation types near the study area adequately represented the general dynamics of similar vegetation in the future. We did not assume that the species mix in future vegetation types is static, only that the kinds of disturbances, successional patterns, and reactions to management in current plant communities sufficiently represent future communities well enough to be useful for examining vegetation and disturbance trends with climate change.

Second, we assumed that possible changes in vegetation potential from future climate change will be actuated by disturbances that return vegetation to early-seral conditions. Our supposition was that vegetation existing on a site uses resources and impedes development of altered plant communities unless a disturbance makes site resources available. We assumed that mixed- and high-severity wildfire, some management activities, some kinds of insect outbreaks, direct drought-related mortality in arid lands, and a few other major disturbances were of a sufficient magnitude to facilitate vegetation type change. We accounted for potential changes in fire regimes with climate change in our STMs through integration with MC1, but potential changes in insect disturbances with climate change are not incorporated in MC1 and thus were not incorporated in our STMs.

Third, we assumed that we could use a common, overlapping time period (1984–2008) to compare current vegetation and MC1 modeled vegetation to develop scaling factors that might allow us to attach trends and variation from MC1 projections to the vegetation potential and dynamics in the study area.

Step 1: Match local vegetation types to MC1 potential vegetation classes—
The connection between MC1 and STMs required cross-walking recent historical vegetation as mapped by MC1 to maps of ILAP potential vegetation types. MC1 produced an output grid for every simulation year showing the expected PVC for each grid cell (800 m). We compared the modeled MC1 PVC grid for the recent historical period (1984-2008) to the potential vegetation grid for the study area using a geographic information system (GIS) (fig. 7.2). The resulting GIS data were used to compute the area of ILAP potential vegetation types within each of the MC1 PVCs and for visual comparisons of PVC with potential vegetation locations for the recent historical period. The ILAP included more than two dozen potential vegetation types, substantially more spatial detail than that contained in the MC1 PVC maps. Consequently, we selected one ILAP potential vegetation type to represent each of the PVCs that occurred in MC1 simulations for either the recent historical or future time periods (table 7.1). We generally chose the most common potential vegetation to represent MC1 PVCs. For simplicity, we dropped any MC1 PVC that never exceeded 1 percent of the study area in either time period. This process resulted in seven potential vegetation types that we related to seven MC1 PVCs. In addition, MC1 simulated more than 1 percent of the study area in warm-season (i.e., C4) grasslands under some climate scenarios in the future. The map of potential vegetation from ILAP did not include any warm-season grasslands, so we added an eighth type that only occurred in future simulations. Our warm-season grassland STM was modified from a similar potential vegetation type that occurs in adjacent Nevada.

Step 2: Tune MC1 potential vegetation classes—
MC1 used a set of rules that assign combinations of biomass, carbon, climate factors, and other data to a PVC. We examined the rules and PVCs simulated for the historical period (1971–2000) to better understand how they related to other maps of historical vegetation (Kuchler 1964). We changed the interpretation of two PVCs

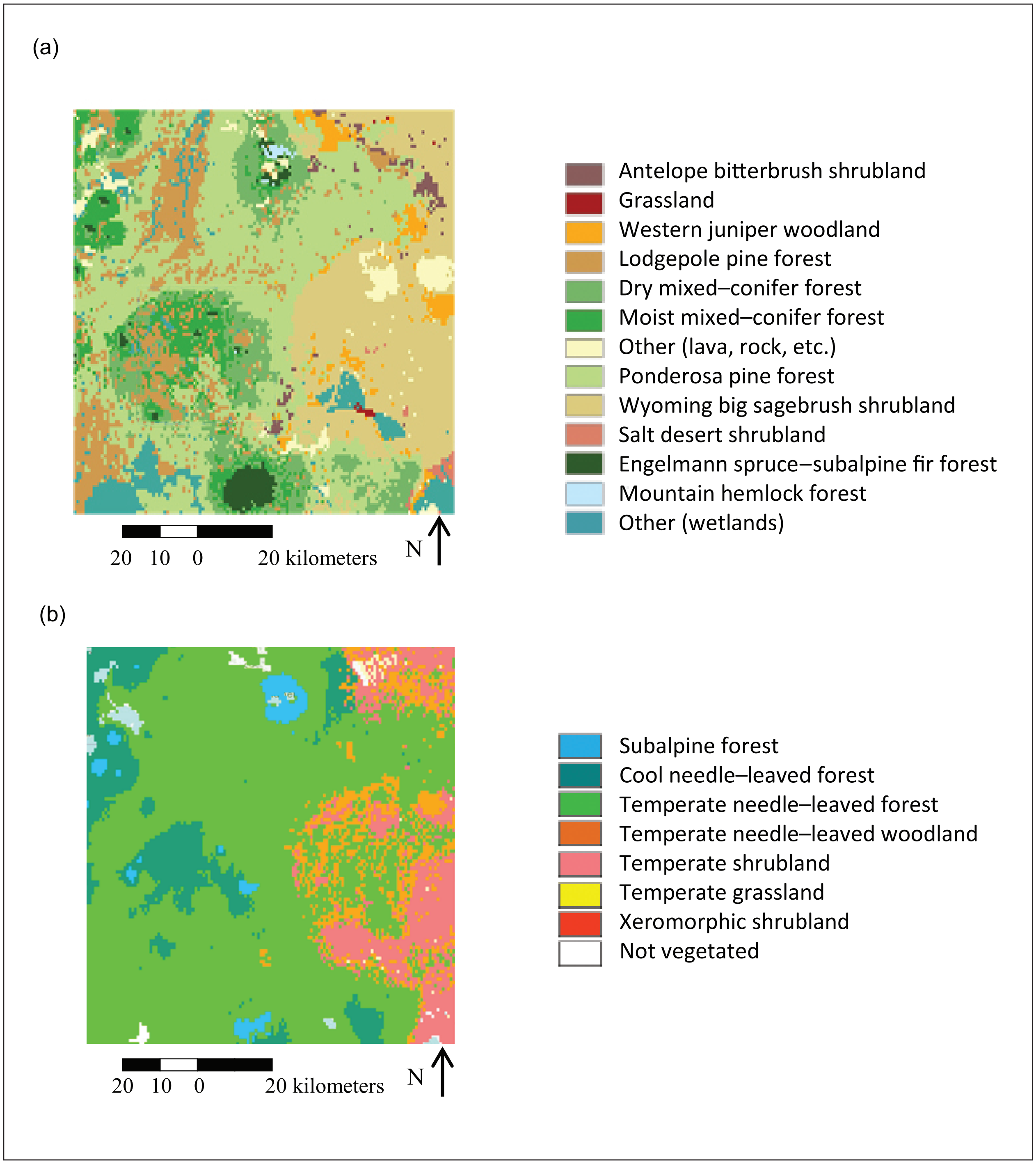

Figure 7.2—Potential vegetation types mapped from (a) imputation of inventory plots and (b) recent historical (1971–2000) vegetation simulated by MC1.

Table 7.1—MC1 potential vegetation classes and their corresponding potential vegetation types in the study area[a]

MC1 potential vegetation classes	State-and-transition model potential vegetation type	Mapped current extent	MC1 current extent
		Percent	
Subalpine forest	Mountain hemlock forests	2	2
Cool needle-leaved forest	Moist mixed conifer forests	11	4
Temperate needle-leaved forest	Ponderosa pine–lodgepole pine forests	59	67
Temperate needle-leaved woodland	Mountain big sagebrush–with juniper	14	6
Temperate shrubland	Wyoming big sagebrush	12	18
Xeromorphic shrubland	Salt desert shrub	2	<1
Temperate grassland	Bluebunch wheatgrass–Sandberg bluegrass grassland	<1	3
Warm-season grassland	Warm-season grassland	0	0

[a] The "Mapped current extent" of forested potential vegetation types was developed from the U.S. Department of Agriculture Forest Service's Landscape Ecology Monitoring Mapping and Analysis team (http://www.fsl.orst.edu/lemma/splash.php), and the maps of arid land potential vegetation types were developed as a part of the Integrated Landscape Assessment Project.

to better match maps of historical vegetation. MC1 simulated substantial areas of maritime needle-leaved forests in middle- and upper-elevation forested areas in the study area, where the historical vegetation map indicated cool white fir and moist mixed-conifer conditions. MC1 suggested that the diurnal temperature fluctuation in these areas was less than that in most inland forests. Rather than label these as maritime forests, we labeled them as cool and moist needle-leaved forests. Likewise, MC1 simulated some areas in very warm, dry environments as coniferous xeromorphic woodland. Comparisons to the historical vegetation map suggested they should be labeled xeromorphic shrubland instead.

Step 3: Run MC1 and summarize for future climate scenarios—
We ran MC1 for the three future climate scenarios using down-scaled (800-m grid) climate input data for the 2010 to 2100 time period. We analyzed the annual output from each scenario and developed a set of tables that depict the yearly total proportion of the landscape in each PVC, the yearly directional change in area among PVCs, and the yearly proportion of the landscape burned. Yearly directional change among PVCs was the pair-wise proportion of area moving from one PVC to another over the 1-year time step. The resulting tables, when combined with output from the historical run of MC1, allowed us to examine the historical and possible future PVC trends and wildfire trends for all the selected PVCs during the 1895 to 2100 time period.

Step 4: Construct a state-and-transition megamodel—

We developed an STM "mega-model" by combining all eight individual STMs representing the selected potential vegetation into one large, interconnected STM. MC1 output determined which climate change transitions among potential vegetation types to include in the STM megamodel, but we also had to determine when those changes were going to occur. Our assumption that climate change effects could be expressed only from early-seral vegetation conditions meant that climate change transitions connected early-seral state classes in STMs losing area to parallel early-seral state classes in STMs receiving area.

Step 5: Compute climate change and wildfire trends—

We identified PVC changes simulated by MC1 between 2010 and 2100 for three climate change scenarios (MIROC-A2, CSIRO-A2, and Hadley-A2). We then calculated the average annual probability for each of those changes by dividing the area in which the PVC change occurred each year by the total area of the study region, and averaging it across all years. Average annual probabilities for PVC changes were incorporated in STMs as transitions that led from postdisturbance and early-seral state classes of a source STM toward a different STM state class.

We assumed that these new climate change transitions would occur either following stand-replacing disturbances of any kind that affected any state class in the STMs or when vegetation was in an open condition. Essentially, change in vegetation potential could happen when a shift in vegetation potential was simulated by MC1 and when vegetation was in an open condition or when disturbance removed most of the current vegetation. We then computed the annual departure of PVC change from the overall mean change for each climate change transition. For each year simulated by MC1, a scalar indicated how much the PVC trend was above or below the long-term average trend. This method produced a set of trend multipliers for our STM that tracked the year-to-year changes in PVCs simulated by MC1. These trend multipliers were applied as annual multipliers to scale the proportional annual movement of area among the relevant state classes within and between the STMs.

In addition to vegetation type trends, MC1 projects wildfire dynamics into the future. We analyzed the wildfire trends from MC1 and applied them in similar fashion as trend multipliers to the wildfire transitions in our STM. Since our MC1 runs assumed no fire suppression, but the MTBS data were from a period with fire suppression (1984–2008), we could not use the MC1-derived probabilities directly. Instead, we used the overlap in wildfire data from the MTBS sampling period and the MC1 outputs for the 1984 to 2008 time period to develop a base relationship between MC1 proportion of cells burned to the average wildfire probabilities for the study area from MTBS data. We then scaled future wildfire probabilities in our STM according to trends from MC1.

The MTBS does not capture small fires (<405 ha) and may underestimate fire frequency. Thus, using the MTBS record could lead to relatively low estimates of fire area burned for the future. However, MC1 assumes that ignitions are not limiting. In other words, fire always occurs in a cell when fuel and weather conditions are in a certain range. Thus, MC1 may overestimate future fire area burned.

Step 6: Run and check state-and-transition megamodel—
The process of summarizing and computing climate change transitions from MC1 data was performed as a set of database and linked spreadsheet operations. Three individuals checked the databases and spreadsheets for errors. Once the STM megamodel was developed, we ran at least 10 simulation tests to check for logic errors and model sensitivity. Test simulations were carefully reviewed and the STM output compared to raw MC1 projections. We ran multiples of climate change trends, each ramping up climate change trends by two, four, and eight times. Similarly, we ran multiples of wildfire trends. We looked at figures of proportion of the landscape in different PVCs over time (e.g., fig. 7.3) to qualitatively assess model sensitivity. We found that the STM megamodel was highly sensitive to changes in climate change trends, but not as sensitive to changes in wildfire, especially in forested types. Increased fire frequency in several forest types merely reinforced development of open forests with large, fire-resistant trees (e.g., ponderosa pine). On the other hand, increased wildfire rates had larger effects in arid lands, where frequent wildfire favors increases in exotic annual grasses.

Results

Although we ran MC1 and our climate-informed STM for three future climate scenarios, we limit our results here to the MIROC-A2 scenario. We chose this scenario for illustration purposes because it is moderate in terms of temperature and precipitation change for the Pacific Northwest. In addition, although spatial results are possible with our coupled model process, we did not examine them for the purposes of this report, but plan to include them in forthcoming reports.

Historical Period Potential Vegetation and Wildfire—MC1 Simulation Results

Except for a short period in the early 1900s, temperate needle-leaved forest (TNF) occupied over half the study area for the entire historical period, and at present, TNF potential existed across 50 to 70 percent of the study area (fig. 7.3). This agreed reasonably well with our potential vegetation maps for the study area, in

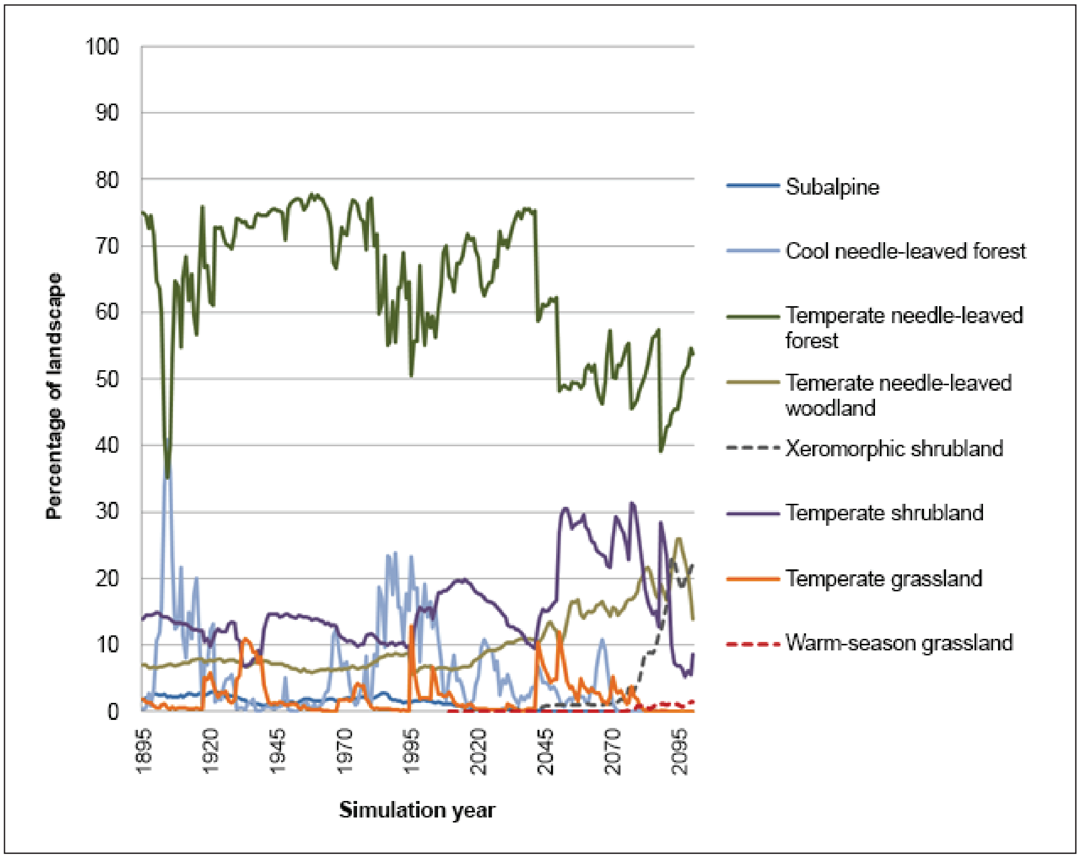

Figure 7.3—MC1 projections of annual extent of selected vegetation types for historical (1895–2008) and future (2009–2100) periods (using MIROC general circulation model climate projections under the A2 emissions scenario).

which the corresponding ponderosa pine/lodgepole pine forest type occupied about 59 percent of the landscape area. In addition to temperate needle-leaved forests, MC1 projected significant but variable amounts of cool needle-leaved forests and small amounts of subalpine forests. These also compared well with our estimates of current amounts. Temperate shrubland dominated arid lands in simulated historical conditions and generally occupied 10 to 15 percent of the landscape area. Temperate needle-leaved woodlands were also prominent, and there were minor amounts of temperate grasslands, as was the case in our current vegetation map. Our historical simulations did not show significant areas of xeromorphic shrublands or warm-season grasslands.

Overall, wildfire simulations from MC1 for the historical period (1895–2008) varied considerably from year to year, especially in arid lands (fig. 7.4). There were high fire years in the early 1930s and in the 1994 to 2008 period. Relatively less arid land burned in other years, and forested lands generally experienced little wildfire. The hot and wet MIROC-A2 scenario produced highly variable wildfire in arid lands, with spikes to over 80 percent of cells burned in 2041, 2049, 2069, 2087, and

2099. Wildfire also increased in forested lands under MIROC-A2, but amounts were much lower (<40 percent burned) compared to arid lands.

Future Vegetation Conditions—MC1 Simulation Results

Under the MC1 DGVM simulation results alone (without the STMs), temperate needle-leaved forests rose to >70 percent of the study area by 2039, and then dropped to 40 percent to 55 percent of the area until 2100 (fig. 7.3). Cool needle-leaved forests occupied between zero and 20 percent of the landscape, then area of climatically suitable habitat was nearly zero by 2075. Climatically suitable habitat for subalpine forests declined to nearly zero by 2020. Temperate shrublands were variable, initially declining somewhat from current conditions, then rising sharply to more than 25 percent of the area, and ultimately declining to <10 percent of the landscape by 2010. Area of temperate needle-leaved woodlands rose slowly from

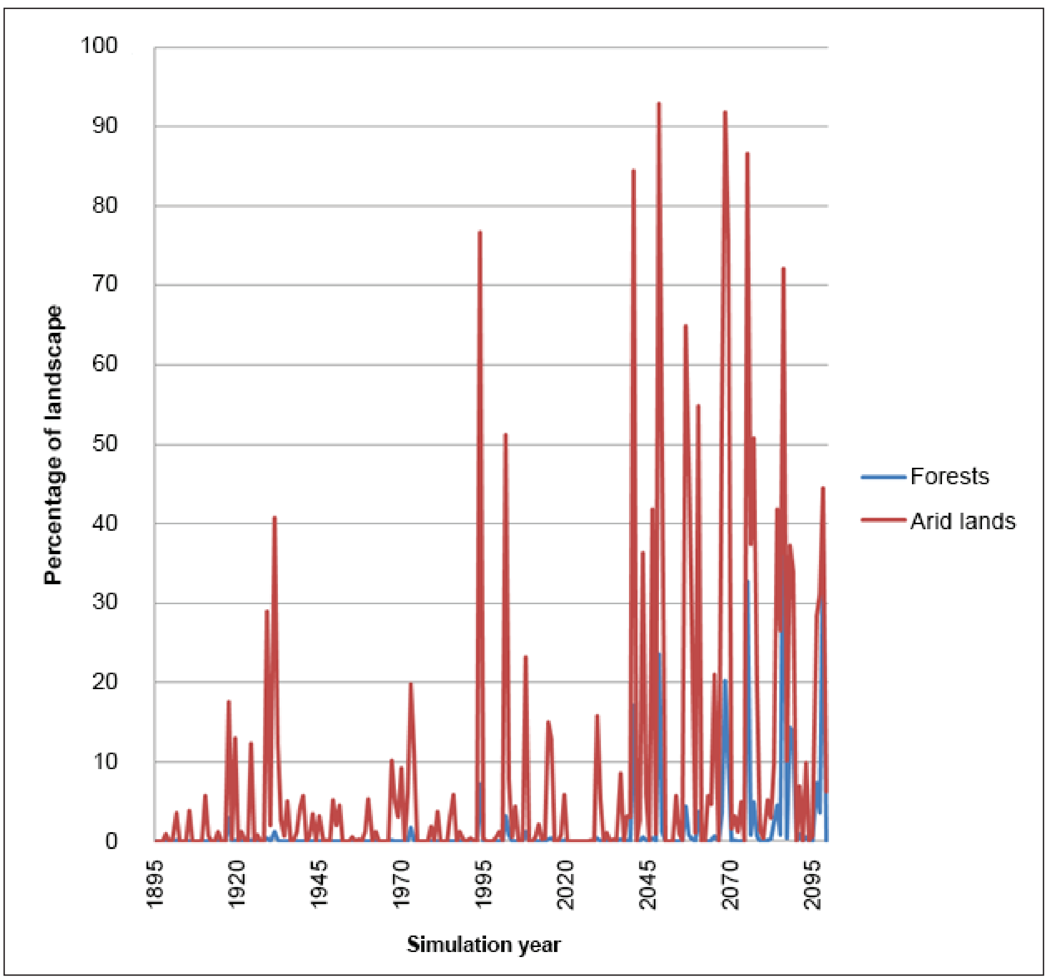

Figure 7.4—Simulated historical (1895–2008) and future (2009–2100) annual area burned as a proportion of the study area simulated by the MC1 model. Future projections used MIROC general circulation model climate information under the A2 emissions scenario.

about 5 percent to >25 percent from 2010 to 2100, but variation in simulated area also increased, especially in the last decade. Climatically suitable habitat for xeromorphic shrublands was absent until 2045, then rose slowly to nearly 20 percent of the landscape by 2100. Climatically suitable habitat for temperate grasslands varied from nearly zero to >10 percent over the 2010-2080 period, and then declined to almost zero. Climatically suitable habitat for warm-season grasslands was absent until 2080, and then never increased to more than 1 percent of the study area.

Our combined DGVM-STM indicated continued temperate needle-leaf forest dominance, though with increasing temporal variation in area.

Future Vegetation Conditions—State-and-Transition Model Simulation Results

Our combined DGVM-STM indicated continued temperate needle-leaf forest dominance, though with increasing temporal variation in area (fig. 7.5). Area of cool needle-leaf forests was relatively constant, at slightly over 10 percent of the landscape until 2085, and then dropped to <5 percent. Subalpine forests remained at about 2 percent of the study area for most of the simulation period, then declined. Area of temperate needle-leaf woodlands was generally steady at about 10 to 15 percent of the landscape, but temporal variation in simulated area increased, especially in the last 20 years. Area of temperate shrublands declined slowly to about 5 percent by 2040, then rose to vary around 15 percent of the landscape. Climatically suitable habitat for xeromorphic shrublands was absent until the last three decades, then increased to about 20 percent of the area. Climatically suitable habitat for temperate grasslands was nearly absent until 2040, rose sharply to vary between 1 and 6 percent of the landscape, and then declined to near zero by 2080. Warm-season grasslands were never present in any significant amount.

Discussion

Effects of Model Linkage

Future vegetation conditions projected by our climate-informed STM differed from those generated by MC1 alone. MC1 included basic ecosystem processes such as carbon sequestration, wildfire disturbance, and a biogeography rule-base defining PVCs based on climate and biomass thresholds. However, the combined DGVM-STM model also incorporated vegetation successional trends, lag times owing to fire-resistant species, and other local vegetation dynamics. The coupled MC1-STM model showed, for example, significantly less variation in the area of temperate needle-leaved forest, which corresponds to ponderosa pine forest in the study area, over the next century compared to MC1 alone. We suggest that including species resistance to disturbance and ecological constraints on climate-driven vegetation change dampens episodic shifts in vegetation conditions and results in greater vegetation resilience than a DGVM alone might indicate.

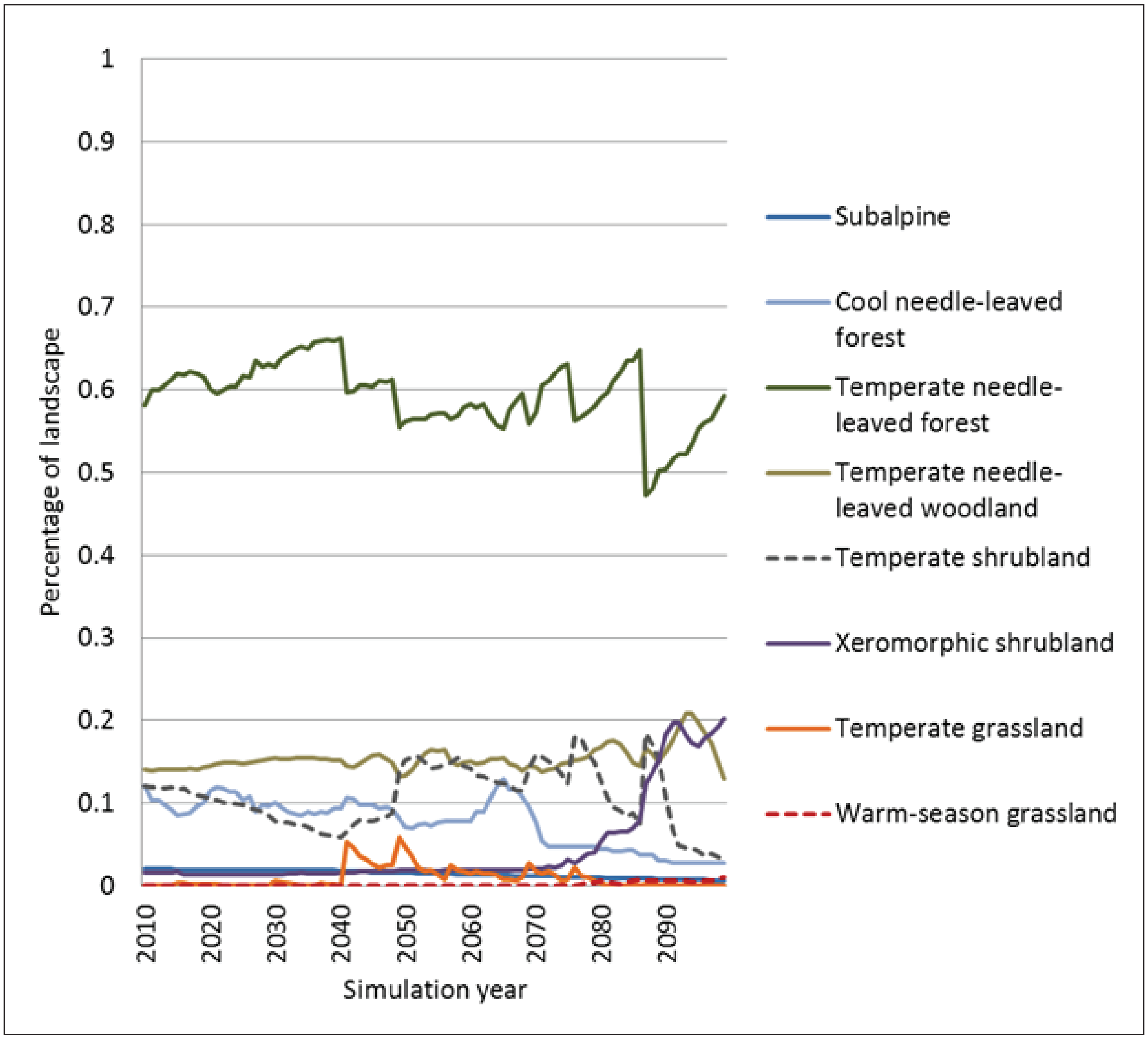

Figure 7.5—State-and-transition model simulation of potential future potential vegetation type trends using vegetation type trends from MC1 for MIROC general circulation model climate projections under the A2 emissions scenario.

The biogeography rule-base in MC1 estimates how changes in the carbon cycle can affect vegetation distribution given these future climate projections. For example, if the climate is generally cool and temperatures do not fluctuate widely seasonally, a given amount of carbon in a simulation pixel might be classified as cool needle-leaved forest. The same amount of carbon in a different climate zone that experiences warmer, drier summers might be called temperate needle-leaved forest. Although there is some lag built into MC1, PVCs can and do fluctuate sharply over relatively short timeframes as climate conditions change. The vegetation dynamics simulated by the model do not reflect a lag between changes in environmental conditions and the replacement of the current vegetation, especially when plant species are resistant to disturbance and long lived. Consequently, DGVM results should be used as a guide to what might happen if large-scale disturbances such as extensive insect outbreak, a stand-replacing fire, an extensive windthrow or heatwave mortality, wipe the landscape slate clean and facilitate quick replacement of plant communities. Even more importantly, DGVM results alone should not be associated with specific years, as natural climate variability is poorly represented by climate models in the future and decreases the temporal accuracy of climate trends in the future.

On the other hand, STMs simulate detailed vegetation dynamics, including growth, disturbance resistance, species competition and succession, natural disturbance and management impacts, and others, but have not traditionally allowed for changes in climate. Our results show that simulating detailed vegetation dynamics and including factors such as specific growth rates and resistances to disturbance, can produce decade- to century-long lags to climate change and associated disturbance impacts.

Woodlands, shrublands, and grasslands did not show the same level of resistance as dry forests to climate change in our coupled model simulations, largely because the species that comprise these communities in our study area are generally not as long lived or fire resistant as many of the early-seral forest tree species.

Woodlands, shrublands, and grasslands did not show the same level of resistance as dry forests to climate change in our coupled model simulations, largely because the species that comprise these communities in our study area are generally not as long lived or fire resistant as many of the early-seral forest tree species. Rather, the tree, shrub, and grass species that characterize arid-land communities are generally killed or top killed by fire, and then recolonize via sprouting, stored propagules, or dispersed propagules. Thus, when fire occurs in these communities, they often move to early-successional or open condition state classes, at which point available resources and lack of competition facilitate potential vegetation shifts. While the ponderosa pine forests in our area resisted change and dampened climate-induced fluctuations because of the fire resistance of mature ponderosa pine, the woodlands, shrublands, and grasslands all showed climate-mediated fluctuations over relatively short timeframes. Thus, we might expect similar landscapes dominated now and in the future by woodlands, shrublands, and grasslands to be less stable than dry forests in the face of climate variation.

Limitations

Our approach has several assumptions and limitations:

- We assume that wildfire suppression remains as effective in the future as it is now. However, extreme fire weather may become more frequent (Westerling et al. 2006), and thus fire frequency and fire size may increase more than our coupled model reflects.

- MC1 does not incorporate insect outbreaks. The STMs do incorporate insect outbreaks, but our approach assumes current roles and probabilities of insect outbreaks will be similar in the future. This may well not be the case. The severity and distribution of some recent insect outbreaks differ from what can be inferred from historical records, and higher temperatures associated with climate change are believed to be a significant factor in these differences (Aukema et al. 2008, Carroll et al. 2004, Logan and Powell 2001). For example, the large mountain pine beetle outbreak in British Columbia has expanded into northern areas and into areas east of the Rocky Mountains in Alberta, where mountain pine beetles were not successful in the past because of cold winter temperatures (Carroll et al. 2004). Our process could accommodate changes in insect or other disturbances if we included changing trends in future frequency and severity of insect outbreaks in our STMs.
- We did not build in lag times for species migration in the STMs. Paleoecological records indicate that migration rates for tree species are approximately 200 to 300 m/yr (Fischlin et al. 2007), but there is great uncertainty in that estimate, lags in response may vary considerably, and landscape fragmentation will also affect migration. All the PVCs projected to occur in the three climate scenarios examined in this study exist in or near to our study area at present, making incorporation of migration lags in our models less critical.
- We assume that the general dynamics of current vegetation and potential vegetation types adequately reflect similar vegetation conditions in the future under different climatic conditions, and that the types we chose to represent broader PVCs are indeed representative. Growth rates, succession, and other factors and interactions could change in the future, even in generally similar vegetation conditions. Past species response to changing climate observed in the paleoecological record shows that the abundance and distribution of plant species shift individualistically in response to climate fluctuations (Davis and Shaw 2001, Delcourt and Delcourt 1991, Whitlock 1992), leading to new species assemblages and communities. Increasing temperatures associated with climate change, and corresponding increases in summer drought stress and fire frequency in the Pacific Northwest, will probably lead to changing species distribution in the region, resulting in vegetation types different from those we see today (Zolbrod and Peterson 1999).

- Our MC1 simulations did not account for complex topography, owing to the scale of available climate projections, which may be important in mountainous regions such as that of our study area.
- Our coupled model results should not be associated with specific years, as natural climate variability is poorly represented by climate models in the future. Rather, our projections should be considered as potential trends across many years to decades.

Despite these limitations, the results of our coupled model exercise can help in understanding potential interactions between coarse-scale vegetation processes and finer scale vegetation dynamics in a changing climate.

Conclusions

Our modeling process links a DGVM to STMs to localize potential climate change effects on vegetation. With this linkage, we used information from climate-vegetation-disturbance interactions in DGVM simulations and used it to inform vegetation change and fire parameters in local STMs, which model finer scale vegetation dynamics. The addition of finer scale vegetation dynamics resulted in lags in climate effects and increased vegetation resilience to climate change than what was indicated by DGVM output alone.

Managers and others can use our process to better understand potential climate-induced shifts in local vegetation, and through linkage with other ILAP information, fuel conditions, potential wildlife habitats, biomass, silvicultural treatment economics, and other important vegetation-related trends and values. Managers wishing to retain moist forests with large, old trees, for example, could examine the various kinds of stand management activities (thinnings from below, fuel treatments, prescribed fire, no vegetation treatment, and others) included in our STMs and determine what set of treatments results in the conservation of moist forests across all three climate scenarios. Expanding this work to other areas could help natural resource planners and others more easily incorporate climate change into their decisionmaking process.

Managers and others can use our process to better understand potential climate-induced shifts in local vegetation, and through linkage with other ILAP information, fuel conditions, potential wildlife habitats, biomass, silvicultural treatment economics, and other important vegetation-related trends and values.

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English Equivalents

When you know:	Multiply by:	To find:
Millimeters (mm)	0.03934	Inches
Centimeters (cm)	0.394	Inches
Meters (m)	3.28	Feet
Meters per second (m/s)	2.24	Miles per hour
Kilometers (km)	0.621	Miles
Hectares (ha)	2.47	Acres
Cubic meters (m^3)	35.3	Cubic feet
Tonnes (t)	1.102	Tons
Kilojoules per square meter (kJ/m^2)	0.08806	British thermal units per square foot
Degrees Celsius	1.8 °C + 32	Degrees Fahrenheit

Appendix: Common and Scientific Names

Common name	Scientific name
Acacia	*Acacia* spp.
Alaska cedar	*Callitropsis nootkatensis* (D. Don) Oerst. ex D.P. Little
Alderleaf mountain mahogany	*Cercocarpus montanus* Raf.
Alkali sacaton	*Sporobolus airoides* Torr.
Alligator juniper	*Juniperus deppeana* Steud.
American marten	*Martes americana* Turton
Apache pine	*Pinus engelmannii* Carrière
Apache plume	*Fallugia paradoxa* (D. Don) Endl. ex Torr.
Arizona fescue	*Festuca arizonica* Vasey
Arizona orange	*Choisya dumosa* (Torr.) A. Gray var. *arizonica* (Standl.) L.D. Benson
Arizona white oak	*Quercus arizonica* Sarg.
Aroga moth	*Aroga websteri* Clarke
Ash-throated flycatcher	*Myiarchus cinerascens*
Barred owl	*Strix varia* Barton
Big galleta	*Pleuraphis rigida* Thurb.
Big leaf maple	*Acer macrophyllum* Pursh
Big sacaton	*Sporobolus wrightii* Munro ex Scribn.
Bitterbrush	*Purshia tridentata* (Pursh) DC.
Black grama	*Bouteloua eriopoda* (Torr.) Torr.
Black spruce	*Picea mariana* (Mill.) Britton, Sterns & Poggenb.
Black-backed woodpecker	*Picoides arcticus*
Blackbrush	*Coleogyne ramosissima* Torr.
Blue grama	*Bouteloua gracilis* (Willd. ex Kunth) Lag. ex Griffiths
Bluebunch wheatgrass	*Pseudoregeneria spicata* (Pursh) A. Löve
Bluestem	*Andropogon* spp.
Bobcat	*Lynx rufus*
Border pinyon	*Pinus discolor* D.K. Bailey & Hawksw.
Buckwheat	*Eriogonum* spp.
Buffalograss	*Bouteloua dactyloides* (Nutt.) J.T. Columbus
Buffelgrass	*Pennisetum ciliare* (L.) Link
Bullgrass	*Muhlenbergia emersleyi* Vasey
Burrograss	*Scleropogon brevifolius* Phil.
Bush muhly	*Muhlenbergia porteri* Scribn. ex Beal
Cassin's finch	*Carpodacus cassinii*
Ceanothus	*Ceanothus* spp.
Cheatgrass	*Bromus tectorum* L.
Chihuahuan pine	*Pinus leiophylla* Schiede & Deppe
Corkbark fir	*Abies lasiocarpa* (Hook.) Nutt. var. *arizonica* (Merriam) Lemmon
Creosote bush	*Larrea tridentata* (DC.) Coville
Crucufixion thorn	*Canotia holacantha* Torr.
Curl-leaf mountain mahogany	*Cercocarpus ledifolius* Nutt.
Curlyleaf muhly	*Muhlenbergia setifolia* Vasey
Desert bighorn sheep	*Ovis canadensis* nelsoni
Desert ironwood	*Olneya tesota* A. Gray

Appendix: Common and Scientific Names (continued)

Common name	Scientific name
Douglas-fir	*Pseudotsuga menziesii* (Mirb.) Franco
Douglas-fir tussock moth	*Orgyia pseudotsugata* McDunnough
Dropseed	*Sporobolus* spp.
Emory oak	*Quercus emoryi* Torr.
Englemann spruce	*Picea engelmanii* Parry ex Engelm.
Fisher	*Martes pennanti*
Flammulated owl	*Otus flammeolus*
Fluffgrass	*Dasyochloa pulchella* (Kunth) Willd. ex Rydb.
Gambel oak	*Quercus gambelii* Nutt.
Giant spotted whiptail	*Aspidoscelis burti stictogrammus*
Grama	*Bouteloua* spp.
Grand fir	*Abies grandis* (Douglas ex D. Don) Lindl.
Gray oak	*Quercus grisea* Liebm.
Gray vireo	*Vireo vicinior*
Gray wolf	*Canis lupus* L.
Greasewood	*Sarcobatus vermiculatus* (Hook.) Torr.
Greater sage-grouse	*Centrocercus urophasianus*
Green fescue	*Festuca viridula* Vasey
Grey checkered whiptail	*Cnemidophorus dixoni*
Idaho fescue	*Festuca idahoensis* Elmer
Incense cedar	*Calocedrus decurrens* (Torr.) Florin
Indian ricegrass	*Achnatherum hymenoides* (Roem. & Schult.) Barkworth
James' galleta	*Pleuraphis jamesii* Torr.
Jeffrey pine	*Pinus jeffreyi* Balf.
Jointfir	*Ephedra* spp.
Juniper	*Juniperus* spp.
Kentucky bluegrass	*Poa pratensis* L.
Lark sparrow	*Chondestes grammacus*
Lesser prairie chicken	*Tympanuchus pallidicinctus*
Lewis's woodpecker	*Melanerpes lewis*
Limber pine	*Pinus flexilis* James
Lodgepole pine	*Pinus contorta* Dougl. ex Laud.
Loggerhead shrike	*Lanius ludovicianus*
Low sagebrush	*Artemisia arbuscula* Nutt.
Mallow ninebark	*Physocarpus malvaceu* (Greene) Kuntze
Manzanita	*Arctostaphylos* spp.
Medusahead	*Taeniatherum caput-medusae* (L.) Nevski
Mesquite	*Prosopis* spp.
Mexican blue oak	*Quercus oblongifolia* Torr.
Mexican spotted owl	*Strix occidentalis lucida*
Mountain big sagebrush	*Artemisia tridentata* ssp. Vaseyana (Rydb.) B.
Mountain hemlock	*Tsuga mertensiana* (Bong) Carr.
Mountain muhly	*Muhlenbergia montana* (Nutt.) Hitchc.
Mountain pine beetle	*Dendroctonus ponderosae* Hopkins
Mountain plover	*Charadrius montanus*
Mountain snowberry	*Symphoricarpos albus* (L.) S.F. Blake
Muhly	*Muhlenbergia* spp.

Appendix: Common and Scientific Names (continued)

Common name	Scientific name
Needle-and-thread	*Hesperostipa comata* (Trin. & Rupr.) Barkworth
New Mexico feathergrass	*Hesperostipa neomexicana* (Thurb. ex J.M. Coult.) Barkworth
Noble fir	*Abies procera* Rehder
North American black bear	*Ursus americanus* Pallas
Northern goshawk	*Accipiter gentilis*
Northern harrier	*Circus cyaneus*
Northern sagebrush lizard	*Sceloporus graciosus graciosus*
Northern spotted owl	*Strix occidentalis caurina* Merriam
Northwestern salamander	*Ambystoma gracile* Baird
Oak	*Quercus* spp.
Olive-sided flycatcher	*Contopus cooperi*
Oneseed juniper	*Juniperus monosperma* (Engelm.) Sarg.
Oregon white oak	*Quercus garryana* Douglas ex Hook.
Pacific madrone	*Arbutus menziesii* Pursh
Pacific silver fir	*Abies amabilis* (Douglas ex Louden) Douglas ex Forbes
Paloverde	*Parkinsonia* spp.
Pileated woodpecker	*Dryocopus pileatu* L.
Pinchot's juniper	*Juniperus pinchotii* Sudw.
Pointleaf manzanita	*Arctostaphylos pungens* Kunth
Ponderosa pine	*Pinus ponderosa* Dougl. ex C. Laws.
Prairie dog	*Cynomys* spp.
Prairie junegrass	*Koeleria macrantha* (Ledeb.) Schult.
Pygmy rabbit	*Brachylagus idahoensis*
Quaking aspen	*Populus tremuloides* Michx.
Rabbitbrush	*Ericameria* or *Chrysothamnus* spp.
Red brome	*Bromus rubens* L.
Red tree vole	*Arborimus longicaudus*
Rigid sagebrush	*Artemisia rigida* (Nutt.) A. Gray
Rubber rabbitbrush	*Ericameria nauseosa* (Pall. ex Pursh) G.L. Nesom & Baird
Sagebrush	*Artemisia* spp.
Saguaro	*Carnegiea gigantea* (Engelm.) Britton & Rose
Sahara mustard	*Brassica tournefortii* Gouan
Salt grass	*Distichlis spicata* (L.) Greene
Saltbush	*Atriplex* spp.
Sand sagebrush	*Artemisia filifolia* Torr.
Sandberg bluegrass	*Poa secunda* J. Presl
Serviceberry	*Amelanchier* spp.
Sharp-shinned hawk	*Accipiter striatus*
Shasta red fir	*Abies magnifica* A. Murray bis var. *shastensis* Lemmon
Shinnery oak	*Quercus havardii* Rydb.
Sideoats grama	*Bouteloua curtipendula* (Michx.) Torr.
Sitka spruce	*Picea sitchensis* (Bong.) Carrière
Snowshoe hare	*Lepus americanus*
Sonoran scrub oak	*Quercus turbinella* Greene
Subalpine fir	*Abies lasiocarpa* (Hook.) Nutt.

Appendix: Common and Scientific Names (continued)

Common name	Scientific name
Swainson's hawk	*Buteo swainsoni*
Switchgrass	*Panicum virgatum* L.
Tan oak	*Lithocarpus densiflorus* (Hook. & Arn.) Rehder
Tarbush	*Flourensia cernua* DC.
Three-tip sagebrush	*Artemisia tripartita* Rydb. ssp. *tripartita*
Tobosagrass	*Pleuraphis mutica* Buckley
Tourney oak	*Quercus toumeyi* Sarg.
Twinflower	*Linnaea borealis* L.
Vine mesquite	*Panicum obtusum* Kunth
Western bluebird	*Sialia mexicana*
Western gray squirrel	*Sciurus griseus*
Western hemlock	*Tsuga heterophylla* (Raf.) Sarg.
Western juniper	*Juniperus occidentalis* Hook.
Western larch	*Larix occidentalis* Nutt.
Western redcedar	*Thuja plicata* Donn ex D. Don.
Western spruce budworm	*Choristoneura occidentalis* Freeman
Western white pine	*Pinus monticola* Douglas ex D. Don
White bursage	*Ambrosia dumosa* (A. Gray) Payne
White fir	*Abies concolor* (Gord. & Glend.) Lindl. ex Hildebr.
White Sands woodrat	*Neotoma micropus leucophaea*
White spruce	*Picea glauca* (Moench) Voss
Whitebark pine	*Pinus albicauli* Engelm.
White-headed woodpecker	*Picoides albolarvatus*
White-sided jackrabbit	*Lepus callotis gaillardi* Mearns
Willow	*Salix* spp.
Wyoming big sagebrush	*Artemisia tridentata* Nutt. ssp. wyomingensis Beetle & Young
Yellow-nosed cotton rat	*Sigmodon ochrognathus* V. Bailey
Zone-tailed Hawk	*Buteo albonotatus* Kaup

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