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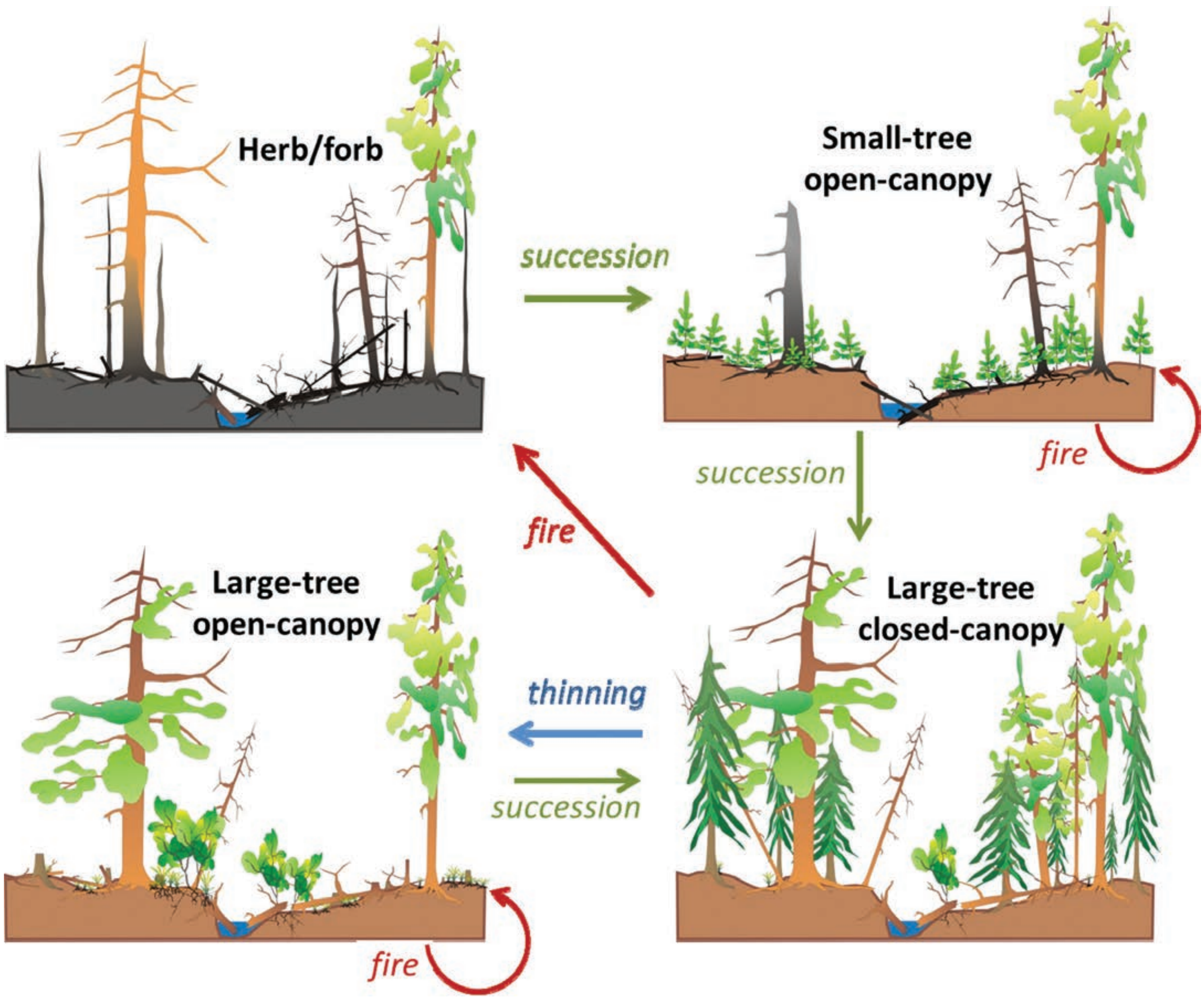
Proceedings of the First Landscape State-and-Transition Simulation Modeling Conference, June 14–16, 2011



Technical Editors

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Cover

A simplified schematic diagram of a state-and-transition simulation model (STSM) for a dry mixed-conifer forest, illustrating a subset of a vegetation successional states and natural and human-caused transitions. The papers in these proceedings provide more detail about STSM mechanics and specific applications across a range of ecosystem types. Cover design and drawings by Steven M. Wondzell.

Proceedings of the First Landscape State-and-Transition Simulation Modeling Conference, June 14–16, 2011

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Abstract

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The first ever Landscape State-and-Transition Simulation Modeling Conference was held from June 14–16, 2011, in Portland Oregon. The conference brought together over 70 users of state-and-transition simulation modeling tools—the Vegetation Dynamics Development Tool (VDDT), the Tool for Exploratory Landscape Analysis (TELSA) and the Path Landscape Model. The goal of the conference was to (1) provide opportunities for sharing experiences with different applications of the tools, (2) identify major existing conceptual or technological gaps, and develop goals for future state-and-transition simulation model (STSM) development, and (3) start building an international network of STSM users. Eighteen oral presentations and thirteen posters were presented. This proceeding includes thirteen papers that build on some key STSM concepts, applications, and innovations from that conference, and shares them with a wider audience. The goal of these proceedings is to provide a state-of-the-science reference for STSM modelers and users. All papers were peer-reviewed by two blind reviewers and one editor. The presentation of these papers reveals that the STSM approach has been applied to a wide range of management and land-use questions and ecosystems, with an equally wide variation in the amounts of scientific data and expert knowledge available for model parameterization.

Keywords: Climate change, FVS, ILAP, LANDFIRE, Path Landscape Model, state-and-transition simulation model, TELSA, vegetation dynamics, vegetation ecology, VDDT.

Conference sponsors:

Apex Resource Management Solutions
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Oregon University System’s Institute for Natural Resources
LANDFIRE project

Contents

Introduction

Becky K. Kerns, Ayn J. Shlisky, and Colin J. Daniel

Authors

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Introduction

The first ever Landscape State-and-Transition Simulation Modeling Conference was held from June 14–16, 2011, in Portland, Oregon. The conference was hosted by Apex Resource Management Solutions and ESSA Technologies, in collaboration with the U.S. Forest Service, The Nature Conservancy, the Oregon University System's Institute for Natural Resources, and the LANDFIRE project. The conference brought together over 70 users of state-and-transition simulation modeling tools–the Vegetation Dynamics Development Tool (VDDT), the Tool for Exploratory Landscape Analysis (TELSA), and the Path Landscape Model. The goal of the conference was to (1) provide opportunities for sharing experiences with different applications of the tools, (2) identify major existing conceptual or technological gaps, and develop goals for future state-and-transition simulation model (STSM) development, and (3) start building an international network of STSM users. Eighteen oral presentations and thirteen posters were presented. This proceedings includes thirteen papers that build on some key STSM concepts, applications, and innovations from that conference, and shares them with a wider audience. Our goal is to provide a state-of-the-science reference for STSM modelers and users. As part of that goal, all papers were peer-reviewed by two blind reviewers and one editor. At least one reviewer was external to the USDA Forest Service.

The proceedings begin with a paper by Daniel and Frid (2012) that outlines how STSMs can be used to project changes in vegetation over time across a landscape, explaining the theory and concepts behind the approach. Developing consistency in terminology was one goal identified at the conference; differences in terminology are apparent across the papers presented in these proceedings. Daniel and Frid include definitions and terminology for some of the most commonly used STSM features, setting the groundwork for more consistent usage of terminology in the future.

The following two papers focus on using STSMs in regional planning efforts and projects. Shlisky and Vandrendriesche (2012) describe how STSMs are used in national forest planning in the Pacific Northwest (PNW). They illustrate the use of STSMs to examine current and desired forest conditions, and the development of Forest Plan Environmental Impact Statement (EIS) alternatives, and evaluations of environmental effects, using examples from PNW forests. Blankenship et al. (2012) describe 10 lessons learned from the LANDFIRE project. The LANDFIRE project developed a consistent and comprehensive set of STSMs for all major ecosystems of the United States.

The following five papers are all focused on a large regional assessment project—The Integrated Landscape Assessment Project (ILAP). Hemstrom et al. (2012) describe the overall ILAP project, which spanned four U.S. states, and produced information, models, data, and tools to help land managers, policy makers, and others examine mid- to broad-scale projections of land management actions, perform landscape assessments, and estimate potential effects of management actions for planning and other purposes. As part of the ILAP project, Creutzburg et al. (2012) used an STSM approach to project changes in sagebrush steppe vegetation across the landscape of southeastern Oregon. Duncan and Burcsu (2012) explore the ecological consequences of rural residential development and management on the spatial patterns of mule deer habitat in central Oregon. They used the spatially explicit STSM tool TELSA. Morzillo et al. (2012) present an analysis of tradeoffs in assessments of wildlife habitat within a multiple objectives based STSM framework. Finally, Zhou and Hemstrom (2012) use STSMs to model timber volume, biomass, and forest carbon over time in central Washington under two alternative management scenarios.

The next three papers are largely focused on methodological issues. Robinson and Beukema (2012) compare output from both VDDT and the Forest Vegetation Simulator (FVS), two modeling frameworks commonly used in land management planning. VDDT is a landscape-level model and FVS was developed as a stand-level model. Their comparison helps to identify differences in the assumptions of the two models and will hopefully result in more consistent and compatible use across the models. Weisz and Vandrendriesche (2012) expand on how FVS can be used to provide or calibrate rates of natural growth transitions under endemic conditions in STSMs, and also present methods that can be used to capture resultant vegetation stages following disturbance activities (natural or human caused). Kerns et al. (2012) present approaches for incorporating climate change effects in STSMs. Up until recently, most STSMs did not explicitly include climate considerations. Yet incorporating climate change into vegetation management for land managers and others has become increasingly critical.

The last two papers in the proceedings are focused on two STSM applications in very different environments. Wondzell et al. (2012) model the effects of various management and restoration practices on conditions of riparian forests, channel morphology, and salmonid habitat. Results are from the Wilson River in the Oregon Coast Range. Strand et al. (2012) developed a spatially explicit STSM to assess effects of current and historic wildfire regimes and prescribed burning programs on landscape vegetation composition and quaking aspen (*Populus tremuloides*) habitat in the Owyhee Mountains in Idaho. Quaking aspen habitat has been declining across the western United States, and is of particular management concern.

The presentation of these papers reveals that the STSM approach has been applied to a wide range of management and land use questions and ecosystems, with an equally wide variation in the amounts of scientific data and expert knowledge available for model parameterization. Over the years, an extensive suite of features have been added to STSMs that allow them to represent a range of dynamics important to landscape modeling (Daniel and Frid 2012), while maintaining a relatively user-friendly modeling platform. In part this is shown by many of the papers in this volume that highlight current development work and new applications in diverse locations. Compared to linear programming or optimization models, STSMs are proving to be a more realistic way to describe ecosystem dynamics and management interactions, and they have the potential to substantially improve communication between scientists, planners, land managers, and the general public about complex ecological processes and alternative solutions to natural resource problems. At the conference, attendees developed a list of possible future directions for the recently released Path Landscape Model, and the vast majority expressed interest in continuing to network and share lessons learned through future conferences, virtual meetings, and peer-to-peer learning. Presently, developers are working on adding more features from VDDT, developing a spatially explicit raster simulator (to complement the polygon-based TELSA approach), creating tools for collaboration (e.g. Web portal, model library), exploring performance enhancements, and building the STSM community. The STSM framework will continue to evolve as a management tool and increasingly as a research tool. State-and-transition simulation modeling tools are being actively developed and improved, and are currently applied to new ecological questions every day.

References

Blankenship, K.; Smith, J.; Swaty, R.; Shlisky, A.; Patton, J.; Hagen, S. 2012. Modeling on the Grand Scale: LANDFIRE Lessons Learned. In: Kerns, B.K.; Shlisky, A.J.; Daniel, C.J., tech. eds. Proceedings of the First Landscape State-and-Transition Simulation Modeling Conference, June 14–16, 2011, Portland, Oregon. Gen. Tech. Rep. PNW-GTR-869. Portland, OR: U.S. Department of Agriculture, Forest Service, Pacific Northwest Research Station: 43–56.

Creutzburg, M.K.; Halofsky, J.S.; Hemstrom, M.A. 2012. Using state-and-transition models to project cheatgrass and juniper invasion in southeastern Oregon sagebrush steppe. In: Kerns, B.K.; Shlisky, A.J.; Daniel, C.J., tech. eds. Proceedings of the First Landscape State-and-Transition Simulation Modeling Conference, June 14–16, 2011, Portland, Oregon. Gen. Tech. Rep. PNW-GTR-869. Portland, OR: U.S. Department of Agriculture, Forest Service, Pacific Northwest Research Station: 73–84.

Daniel, C.J.; Frid, L. 2012. Predicting landscape vegetation dynamics using state-and-transition simulation models. In: Kerns, B.K.; Shlisky, A.J.; Daniel, C.J., tech. eds. Proceedings of the First Landscape State-and-Transition Simulation Modeling Conference, June 14–16, 2011, Portland, Oregon. Gen. Tech. Rep. PNW-GTR-869. Portland, OR: U.S. Department of Agriculture, Forest Service, Pacific Northwest Research Station: 5–22.

Duncan, J.A.; Burcsu, T. 2012. Landscape development and mule deer habitat in Central Oregon. In: Kerns, B.K.; Shlisky, A.J.; Daniel, C.J., tech. eds. Proceedings of the First Landscape State-and-Transition Simulation Modeling Conference, June 14–16, 2011, Portland, Oregon. Gen. Tech. Rep. PNW-GTR-869. Portland, OR: U.S. Department of Agriculture, Forest Service, Pacific Northwest Research Station: 85–101.

Hemstrom, M.A.; Salwasser, J.; Halofsky, J.S.; Kagan J.; Comfort, C. 2012. The Integrated Landscape Assessment Project. In: Kerns, B.K.; Shlisky, A.J.; Daniel, C.J., tech. eds. Proceedings of the First Landscape State-and-Transition Simulation Modeling Conference, June 14–16, 2011, Portland, Oregon. Gen. Tech. Rep. PNW-GTR-869. Portland, OR: U.S. Department of Agriculture, Forest Service, Pacific Northwest Research Station: 57–71.

Kerns, B.K.; Hemstrom, M.A.; Conklin, D.; Yospin, G.; Johnson, B.; Bachelet, D.; Bridgham, S. 2012. Approaches to incorporating climate change effects in state and transition simulation models of vegetation. In: Kerns, B.K.; Shlisky, A.J.; Daniel, C.J., tech. eds. Proceedings of the First Landscape State-and-Transition Simulation Modeling Conference, June 14–16, 2011, Portland, Oregon. Gen. Tech. Rep. PNW-GTR-869. Portland, OR: U.S. Department of Agriculture, Forest Service, Pacific Northwest Research Station: 161–171.

Morzillo, A.T.; Halofsky, J.S.; DiMiceli, J.; Hemstrom, M.A. 2012. Balancing feasibility and precision of wildlife habitat analysis in planning for natural resources. In: Kerns, B.K.; Shlisky, A.J.; Daniel, C.J., tech. eds. Proceedings of the First Landscape State-and-Transition Simulation Modeling Conference, June 14–16, 2011, Portland, Oregon. Gen. Tech. Rep. PNW-GTR-869. Portland, OR: U.S. Department of Agriculture, Forest Service, Pacific Northwest Research Station: 103–114.

Robinson, D.C.E.; Beukema, S.J. 2012. Through a glass, darkly—comparing VDDT and FVS. In: Kerns, B.K.; Shlisky, A.J.; Daniel, C.J., tech. eds. Proceedings of the First Landscape State-and-Transition Simulation Modeling Conference, June 14–16, 2011, Portland, Oregon. Gen. Tech. Rep. PNW-GTR-869. Portland, OR: U.S. Department of Agriculture, Forest Service, Pacific Northwest Research Station: 123–141.

Shlisky, A.J.; Vandendriesche, D. 2012. Use of state-and-transition simulation modeling in national forest planning in the Pacific Northwest, U.S.A. In: Kerns, B.K.; Shlisky, A.J.; Daniel, C.J., tech. eds. Proceedings of the First Landscape State-and-Transition Simulation Modeling Conference, June 14–16, 2011, Portland, Oregon. Gen. Tech. Rep. PNW-GTR-869. Portland, OR: U.S. Department of Agriculture, Forest Service, Pacific Northwest Research Station: 23–41.

Strand, E.K.; Bunting, S.C.; Vierling, L.A. 2012. Landscape composition in aspen woodlands under various modeled fire regimes. In: Kerns, B.K.; Shlisky, A.J.; Daniel, C.J., tech. eds. Proceedings of the First Landscape State-and-Transition Simulation Modeling Conference, June 14–16, 2011, Portland, Oregon. Gen. Tech. Rep. PNW-GTR-869. Portland, OR: U.S. Department of Agriculture, Forest Service, Pacific Northwest Research Station: 197–214.

Weisz, R.; Vandendriesche, D. 2012. Use of the Forest Vegetation Simulator to quantify disturbance activities in state and transition models. In: Kerns, B.K.; Shlisky, A.J.; Daniel, C.J., tech. eds. Proceedings of the First Landscape State-and-Transition Simulation Modeling Conference, June 14–16, 2011, Portland, Oregon. Gen. Tech. Rep. PNW-GTR-869. Portland, OR: U.S. Department of Agriculture, Forest Service, Pacific Northwest Research Station: 143–160.

Wondzell, S.M.; Przeszlowska, A.; Pflugmacher, D.; Hemstrom, M.A.; Bisson, P.A. 2012. Modeling the dynamic responses of riparian vegetation and salmon habitat in the Oregon Coast Range with state and transition models. In: Kerns, B.K.; Shlisky, A.J.; Daniel, C.J., tech. eds. Proceedings of the First Landscape State-and-Transition Simulation Modeling Conference, June 14–16, 2011, Portland, Oregon. Gen. Tech. Rep. PNW-GTR-869. Portland, OR: U.S. Department of Agriculture, Forest Service, Pacific Northwest Research Station: 173–196.

Zhou, X.; Hemstrom, M.A. 2012. Forecasting timber, biomass and tree carbon pools with the output of state and transition models. In: Kerns, B.K.; Shlisky, A.J.; Daniel, C.J., tech. eds. Proceedings of the First Landscape State-and-Transition Simulation Modeling Conference, June 14–16, 2011, Portland, Oregon. Gen. Tech. Rep. PNW-GTR-869. Portland, OR: U.S. Department of Agriculture, Forest Service, Pacific Northwest Research Station: 115–121.

Predicting Landscape Vegetation Dynamics Using State-and-Transition Simulation Models

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Abstract

This paper outlines how state-and-transition simulation models (STSMs) can be used to project changes in vegetation over time across a landscape. STSMs are stochastic, empirical simulation models that use an adapted Markov chain approach to predict how vegetation will transition between states over time, typically in response to interactions between succession, disturbances and management. With STSMs a landscape is divided into a set of simulation cells, each cell is assigned to an initial vegetation state, and the model then predicts how each cell may change from one vegetation state to another over time. Over the years an extensive suite of features have been added to STSMs that allow them to represent a range of dynamics important to landscape modeling, including tracking age-structure, triggering transitions based on past events, setting targets for certain transitions, and varying transition rates over time. STSMs are also now able to represent spatial variability in two different ways: by dividing the landscape into spatial strata, typically defined by one or more important drivers of vegetation change, or alternatively by developing a spatially-explicit STSM, whereby transition events, such as fire or invasion by non-native vegetation, can be simulated to spread across the landscape.

Since their introduction in the early 1990s, STSMs have been applied to a wide range of landscapes and management questions, including forests, rangelands, grasslands, wetlands and aquatic communities, over spatial extents ranging from thousands to millions of hectares. Several software tools currently exist to support the development of STSMs; the most recent of these products, called the Path Landscape Model, is the latest in a lineage of STSM development tools that includes both the Vegetation Dynamics Development Tool (VDDT) and the Tool for Exploratory Landscape Analysis (TELSA).

Keywords: state-and-transition simulation model, STSM, ecological model, ecological restoration, ecosystem management, landscape ecology, vegetation dynamics, Path Landscape Model, TELSA, VDDT.

Introduction

Models are often used to predict vegetation conditions across a landscape over time. Since the early 1970’s a wide range of models have been developed for this purpose. As one might expect these models vary considerably in their approach; over the years several authors have attempted to classify these models, each using a slightly different set of criteria for distinguishing between approaches (Baker 1989; Keane et al. 2004; Scheller and Mladenoff 2006; Xi et al. 2009). Common criteria that emerge from these reviews for distinguishing between models include:

1. Degree to which ecosystem processes, such as succession and disturbance, are simulated mechanistically (as opposed to being developed empirically);
2. Whether or not the models are deterministic (i.e. predict a single future) or stochastic (i.e. predict a distribution of possible futures);
3. Scale at which ecosystem processes are represented—e.g. gap (m^2), stand (ha), region (km^2);
4. Extent to which the spatial dynamics of ecosystem processes are represented explicitly (e.g. disturbance spread over time);

5. Range of ecosystems to which the models can be applied (the majority of landscape models have been developed for forest ecosystems).

One technique for predicting landscape-level vegetation change is to use *state-and-transition models*, a term first introduced by Westoby et al. (1989) in reference to conceptual models describing the successional dynamics of rangeland vegetation over time. Through the use of box-and-arrow diagrams, these models describe a series of discrete states in which a parcel of land can find itself at any point in time, along with transitions, both natural and anthropogenic, that can move land between these states. Conceptual state-and-transition models provide a simple, flexible approach for describing and documenting one's understanding of the vegetation dynamics associated with a particular ecosystem (for example, see Grant 2006; Bestelmeyer et al. 2009; Knapp et al. 2011).

A second form of state-and-transition models exists that extend the conceptual models described above by assigning probabilities to each of the transition pathways, leading to models that can simulate the states and transitions that might occur over time across a landscape. To distinguish these models from their conceptual counterparts we refer to them more specifically as *state-and-transition simulation models* (STSM).

Using the criteria outlined above, STSMs can be considered:

1. Strongly empirical: model relationships are typically fitted to data and/or understanding (including output from other models);
2. Stochastic: model dynamics involve probabilities, allowing for predictions that are distributions, rather than single values;
3. Stand or regional scale: models are typically developed to represent processes occurring at either a stand or regional scale;
4. Optionally spatially-explicit: a unique feature of STSMs is that models can be developed and run either with or without explicitly representing spatial processes;
5. Suitable for any ecosystem: due to their empirical nature, STSMs can be developed for any vegetation community.

In this article we provide an introduction to the STSM approach for simulating vegetation change across a landscape. We first explain the theory and concepts behind the approach, including definitions and terminology for some of the most commonly used STSM features. We then review software currently available for constructing and running STSMs, including a simple example of how the software can be used. Finally we provide an overview of some of the questions and landscapes to which STSMs have been applied.

Modeling Approach

STSMs are stochastic, empirical simulation models, whereby the vegetation across a landscape is classified into states, probabilities are assigned to possible transitions between states, and the landscape is then simulated through time using Monte Carlo simulation methods. Technically an STSM can be considered a Markov chain, whereby the probability of transitioning from one state to another at any given time depends only on the present state (Baker 1989). However, as we shall see later in this section, current STSMs tend to push the Markov chain definition well beyond its usual "textbook" formulation.

For simple STSMs, a landscape is first divided into a set of *n* simulation cells; these cells can be any geometric shape and size (e.g. either raster pixels or polygons). The model then defines a discrete set of *r* states, $S = \{s_1, s_2, ..., s_r\}$, in which each simulation cell can be found over time, and a discrete set of *m* transition types, $U = \{u_1, u_2, ..., u_m\}$, through which each state s_i can transition to s_j. Transition probabilities, $p_{m,i,j}$, specify the probability that state s_i transitions to s_j via transition type *m* in a single timestep of the simulation. Each non-zero transition probability is referred to as a *transition pathway.*

A simulation then tracks the state of each simulation cell, $C = \{c_1, c_2, ..., c_n\}$, over a series of discrete timesteps; the duration of each timestep is user-defined (e.g. day, year, decade). A simulation begins by assigning each cell to an initial state for the first timestep. Each cell's state is then

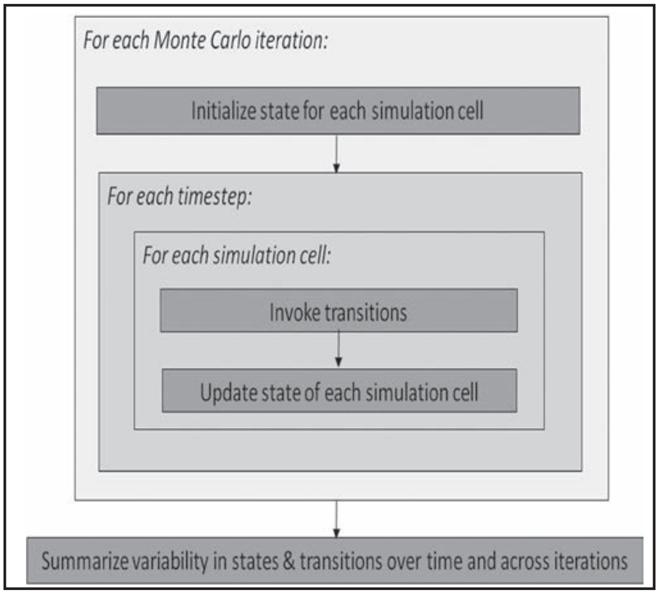


Figure 1—Flowchart showing the STSM algorithm.

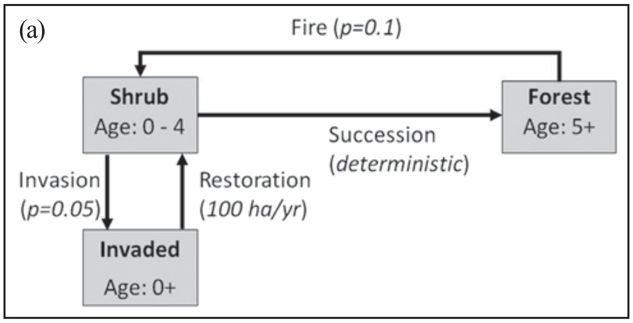
subject to change from one timestep to the next according to the transition probabilities defined between states; a maximum of one transition can occur per cell in each timestep. A single iteration of a simulation is complete once the state of every simulation cell is calculated for a specified number of successive timesteps. Because the model uses probabilities to determine when and where transitions occur, the model's predictions are stochastic, and the fate of any one simulation cell can vary from one run of the model to the next. Consequently, model runs are repeated for several iterations (i.e., Monte Carlo simulations), with the result being a distribution of projected outcomes for the state of each simulation cell over time (fig. 1).

There is a common misconception that Markov chains are too simple to represent many of the processes important to landscape vegetation modelling. However, as described in detail by Baker (1989), it is possible to represent a rich suite of dynamics using Markov chains by simply defining the state space appropriately. For example if transition probabilities for a particular state depend upon the time since a previous event, one can expand the state space to include a new state for each possible time since the event; with enough states any number of preceding events can be represented as a Markov chain. Markov chains can also be non-stationary, allowing probabilities to change over space and time; in this way external drivers, such as climate change or harvest demand, can be represented through changing probabilities; any non-stationary Markov chain can be made stationary by including enough new states to represent all the conditions across which the probabilities vary. Spatial processes, such as contagion of transitions between neighbouring simulation cells, can also be represented by allowing the transition probabilities to vary according to both the cell location and the previous state of neighbouring cells. Again, with a sufficiently large state space, these dynamics can ultimately be represented as a Markov chain: in the limit a model could be disaggregated such that each simulation cell and year is defined as a unique "state," allowing complete control over how the transition probabilities vary across both space and time.

So while it is possible, in theory, to model quite complex dynamics using a Markov chain, the number of states required rises geometrically once the model includes one or more of the features outlined above; formulating a model as a traditional Markov chain can quickly overwhelm even the most proficient modeller. STSMs have been designed specifically to overcome these challenges: since the development of the first STSMs in the early 1990's, features have been steadily added to the basic Markov chain formulation to shield modellers from this complexity. Over the years a number of features have been added to STSMs to allow users to rapidly and simply incorporate many of the complex dynamics outlined above, without the need to explicitly specify and track the full state space; some of these key features are described below.

State Age and Deterministic Transitions

In addition to tracking the state (S) of each simulation cell, STSMs can also track a second state variable: the age, $A = \{a_1, a_2, ..., a_n\}$, of each cell. Each simulation cell is assigned an age at the start of the simulation (in addition to its state); by incrementing this age by one for each timestep of the simulation, the age of each simulation cell can be tracked over time. In addition, an upper and lower age limit can be specified for each state. Once the age for a simulation cell reaches the upper limit for its current state, the age is no longer incremented in future timesteps. A special form of



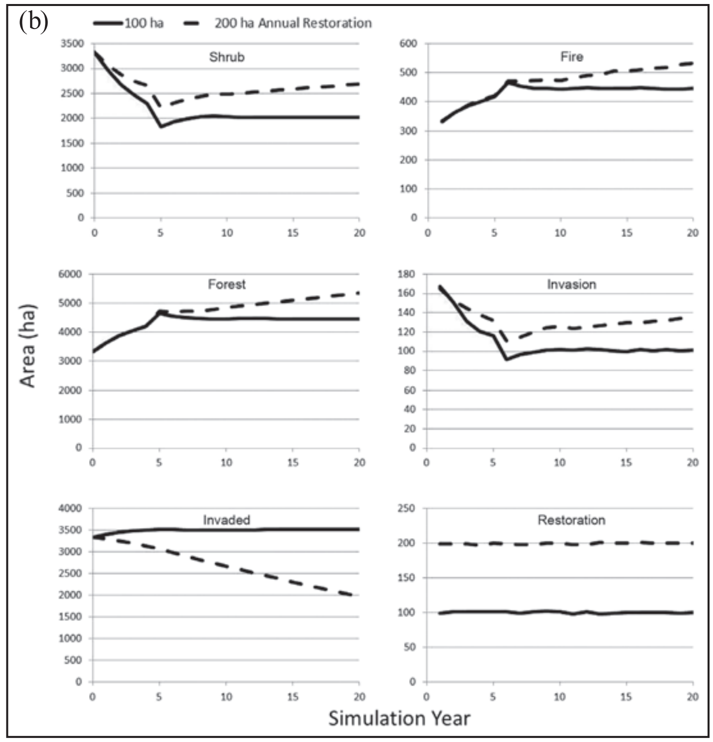


Figure 2—Example of a simple STSM, illustrating (a) Pathway diagram and (b) Simulation results showing area over time in each state class (Shrub, Forest, Invaded) and area over time undergoing transitions (Fire, Invasion, Restoration) for 2 scenarios: 100 ha/yr restoration (solid) and 200 ha/yr restoration (dashed). Model was run for 20 years and 100 Monte Carlo iterations over a 10,000 ha landscape, divided into 1ha simulation cells, assuming equal area in each state and age class initially; graphs display the average values over all iterations.

transition pathway, referred to as a *deterministic transition pathway*, can be specified for each state, causing a transition to occur automatically to a new state once the age of the cell exceeds the state's upper age limit; only one deterministic transition pathway is allowed per state. The lower age limit of the destination state associated with a deterministic transition pathway is then used to reset the age for the cell after a deterministic transition. Deterministic transitions are typically used to represent age-based vegetation succession in STSMs.

Transition Targets

Not all transition pathways are best represented in STSMs using probabilities. One example of this is the deterministic transition pathways that handle the dynamics of ageing between states. A second situation occurs frequently for management-oriented transition pathways, where the level of a transition is more appropriately expressed as a fixed target for the area treated each timestep, rather than as a probability of occurrence. Transition targets are typically specified by transition type for some portion of the landscape, and can vary over time (e.g. target for area harvested each year). To reformulate these transitions as a Markov chain, STSMs dynamically convert these targets to an equivalent probability for each transition pathway and timestep, based on the number of simulation cells eligible for the transition pathway during each timestep of the simulation.

Figure 2a shows an example of a very simple STSM. The pathway diagram defines the suite of possible states and transitions between states for the modelled system, where boxes represent states and arrows represent transition pathways. In this example "Fire" and "Invasion" are represented as probabilistic transitions (with annual probabilities), "Succession" is modelled as a deterministic transition (i.e. occurring with a probability of 1 once the age of the Shrub state reaches 4), and "Restoration" is set to have a transition target of 100 ha/yr. To begin the simulation the landscape is divided into simulation cells, with each cell being assigned an initial state and age. A single iteration of the model involves predicting the state of each simulation cell for a series of timesteps. Repeating the simulation for multiple iterations generates a distribution of predicted results for the state of each cell (fig. 2b).

While already more complicated than a basic Markov chain, the model shown in fig. 2 would still be considered a very simple STSM; additional advanced features, which are commonly found in most applied STSMs, are described below.

Transition Pathway Age Range

By default a transition pathway specified for a particular state acts on all simulation cells in that state, regardless of the cell's age. One exception is a deterministic transition, which is triggered only for cells that have exceeded the state's upper age limit (and does so with a probability of 1). For all other probabilistic transition pathways, however, it is possible in STSMs to change the default settings to constrain a particular transition pathway such that it only applies to those simulation cells that fall within a specified age range. For example, in a forest-based landscape one might specify a transition pathway called "thinning" that applies to a particular state (e.g., Forest), but only to those simulation cells within a specified age range (e.g., 70–90 years old).

Post-Transition Age

When a probabilistic transition occurs for a simulation cell, both the state and the age of the cell are updated according the associated transition pathway. By default the age of the cell is set to the start age of the pathway's new destination state; however it is possible to specify alternative behaviours for the post-transition age: transition pathways can be specified to retain the age of the cell prior to the transition; they can also shift the age forward or backward. For example a transition pathway called "stand-replacing fire" might reset the age of the simulation cell to age 0, while a "surface fire" transition pathway might maintain the current age of the simulation cell.

Time-Since-Transition

In addition to tracking the current state (S) and age (A) of each simulation cell over the course of the simulation, STSMs can optionally track a third state variable: the number of timesteps since each type of transition last occurred for each simulation cell. This state variable is referred to as the *time-since-transition* (TST) for the cell, $tst_{m,n}$, which is tracked for every simulation cell n and transition type m. Similar to the option to specify age ranges for transition pathways, TST can also be used to constrain which simulation cells are eligible for each transition pathway. For example, STSMs can represent the changing fire dynamics that occur on a landscape through the implicit build-up of fuels using the TST feature: simulation cells burned in a high severity wildfire can be modelled to not experience another high severity wildfire for the following 20 years, the assumed time needed for sufficient fuels build up.

Temporal Heterogeneity

Because STSMs are stochastic, using probabilities to determine when and where transition occur, there will always be some variability between timesteps in the number of transitions that occur during a particular simulation. However there are often situations in which additional temporal variability is required in order to adequately capture landscape dynamics. For example, weather conditions and/or climate change may result in different patterns or trends of high and low fire risks from one year to the next, or insect outbreaks may show a cyclical pattern over many years. From an STSM perspective, capturing this kind of temporal variability involves varying the probabilities for certain transition pathways over time. This can be accomplished using *transition multipliers*, which scale the base probabilities associated with one or more transition pathways up and down over the course of a simulation according to an externally driven pattern.

Spatial Heterogeneity

Often there is a requirement for STSMs to capture the spatial heterogeneity across a landscape. Biophysical factors, such as climate and soils, are often key determinants of vegetation dynamics; many disturbances, both natural and anthropogenic, tend to be aggregated spatially. Depending upon the requirements of any particular analysis, the amount of spatial variability that the model must capture will vary.

STSMs can represent three different forms of spatial variability. Firstly, as illustrated in fig. 2, a single set of pathway diagrams can be developed for the landscape as a whole—we refer to this as a *whole landscape* STSM.

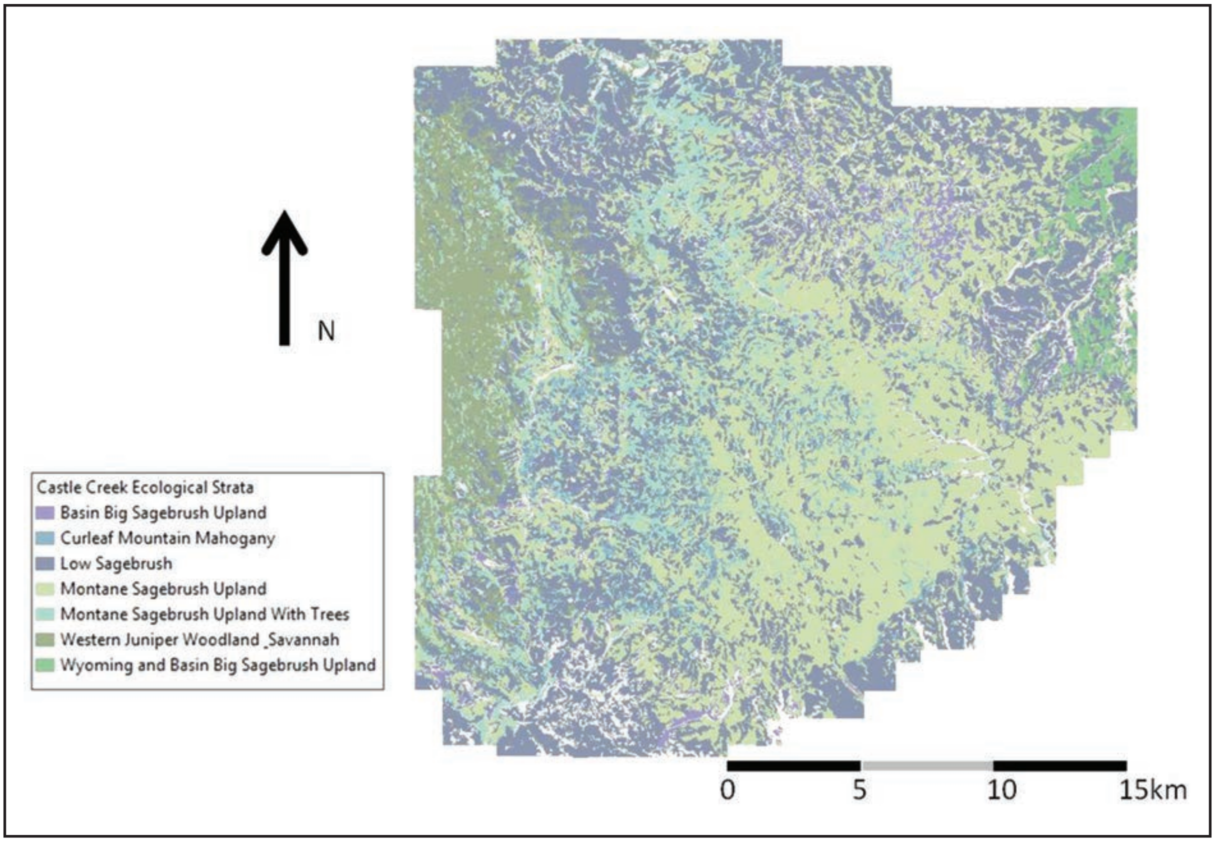


Figure 3—Map showing ecological strata for the 46,000 ha Castle Creek landscape in Idaho. In a spatially stratified STSM, each stratum can be represented by a different pathway diagram.

A model in this form does not capture any of the spatial variability across the landscape; rather, it provides only predictions for the total area in each state over time. Information on the location and configuration of the simulation cells is not used: each cell is simulated independently of all others, essentially acting as a spatial "replicate" in a Monte Carlo simulation.

A second and more commonly used approach for STSM development is to stratify the landscape according to one or more criteria that are considered to be important external drivers of vegetation change, and then to develop a separate pathway diagram for each of these strata—we refer to this as a *spatially-stratified* STSM. As with whole landscape STSMs, each simulation cell within a stratum is simulated independently of its neighbours; however each simulation cell is assigned to a particular stratum at the start of the simulation. Strata are typically defined according to one or more important drivers of vegetation change. Biophysical drivers, such as soils, climate and topography are the most common: here the strata often follow existing ecological classification systems, examples of which include potential vegetation types (Chiarucci et al. 2010), biophysical settings (Long et al. 2006), ecological sites (Bestelmeyer et al. 2009) and biogeoclimatic zones (MacKenzie 2012). Stratification also often reflects differences in anthropogenic activity across the landscape, by dividing the landscape into zones with differing management practices (e.g., protected vs. unprotected, private vs. public).

Stratifying the landscape allows the modeller to specify different assumptions–such as states, transitions, pathway probabilities and management targets—for each of the landscape strata, thus capturing some of the spatial variability in vegetation dynamics across the landscape (fig. 3). Model projections, in turn, can be displayed spatially across the entire landscape, classified according to the original strata polygons. Recently, modellers have begun to relax the assumption that the stratum to which each simulation cell is assigned is static, and instead allow simulation cells to shift from one stratum to another over the course of a simulation (e.g., Provencher and Anderson 2011). This option has become an increasingly important tool for representing shifts over time in biophysical boundaries (e.g. due to climate change) that were historically considered fixed.

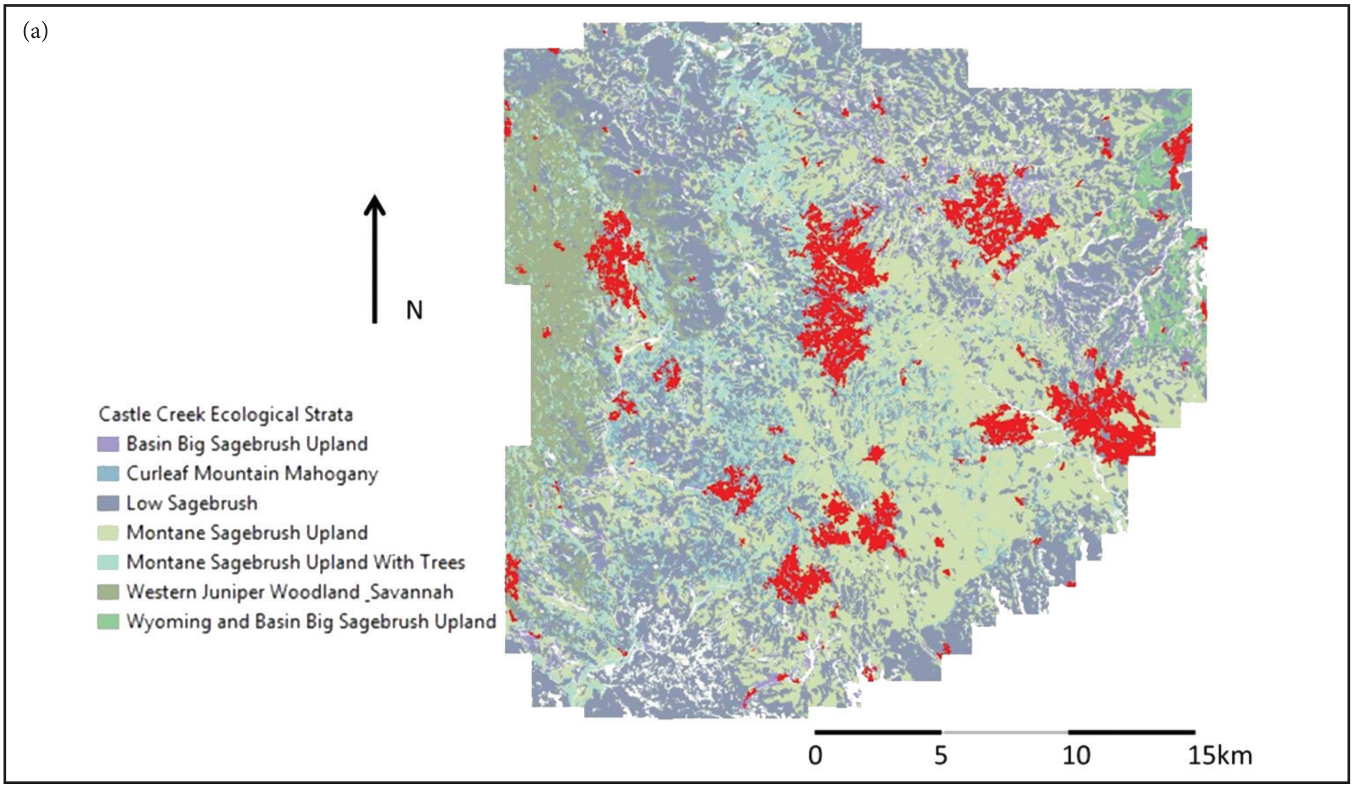
The third and final approach for representing spatial variability is to develop what we refer to as a *spatially-explicit* STSM. These models typically begin as a spatially-stratified STSM with a number of important additions. Firstly, transition probabilities can be made specific to each simulation cell; for example an external driver, such as topography or climate, could influence the probability of a particular transition (e.g., fire) for each cell on the landscape. Secondly, in a spatially-explicit STSM each simulation cell is aware of the location and state of other cells; as a result the transition probabilities for any cell can be influenced by the past and present state of its neighbours (i.e. simulation cells are no longer independent). This feature is commonly used to "spread" transitions across a landscape, both within a timestep (e.g., fire) or between timesteps (e.g., invasives). Finally, a target frequency distribution of sizes for transition events on the landscape can be specified as a model input, guiding the number and size of contiguous transition "patches" that occur across the landscape. Figure 4 shows the typical output of a spatially-explicit STSM: the result of a single Monte Carlo iteration is a prediction for the state of each simulation cell in every timestep; repeating simulations for multiple iterations results in distribution of predictions for the state of each cell.

On the surface it would seem that spatially-explicit models would always be a modeller's first choice, as they appear to produce the most ecologically relevant predictions. However there are important trade-offs to consider when deciding on the approach to use for capturing spatial variation (Mladenoff 2004). Firstly, the time and resources required to prepare and run a spatially-explicit model are generally much greater than that of a simpler spatially-stratified model. Deciding to use a spatially-explicit model often means making sacrifices in other areas of an overall modeling project: extra time spent preparing, running and analysing these models takes away from time available for exploring model behaviour in greater depth, particularly over larger landscapes. Secondly, the data and additional model parameters required to run a spatially-explicit STSM are not always available. Finally, many modelling projects do not require predictions be spatially disaggregated to the level of a simulation cell, in which case the added complexity of a spatially-explicit model may be of no value to the modeller.

Software

While many landscape vegetation models include some capabilities to represent Markov chains (Keane et al. 2004), over the past 20 years three software tools have been developed to support the STSM features described in this paper. The first of these is the Vegetation Dynamics Development Tool (VDDT; ESSA Technologies Ltd. 2007). Originally developed to support landscape modeling for a project in the Interior Columbia River Basin of the U.S. in the early 1990's (Hann et al. 1997), VDDT was the first software tool designed specifically to develop and run STSMs, and includes a simple visual editor for STSM pathway diagrams, a simulation engine for running models, and a graphics module for viewing model output. VDDT was originally designed to support the development of whole landscape STSMs, although with some effort technically adept users are able to adapt the software to run spatially-stratified STSMs. While the software is still available for free download, and support for using the software is still available, development of the product was discontinued in 2011.

A second software tool available for developing STSMs is the Tool for Exploratory Landscape Analysis (TELSA; Kurz et al. 2000; ESSA Technologies 2008). Originally developed to support forest management in British Columbia, Canada, TELSA provides the capability to run polygon-based spatially-explicit STSMs. TELSA was designed to work in conjunction with VDDT—typically users develop their STSM pathway diagrams first using the visual editor in VDDT, then import this information into TELSA. Within the TELSA software, users are able to prepare additional model inputs (including GIS maps using an ArcGIS extension), run Monte Carlo simulations, and view model



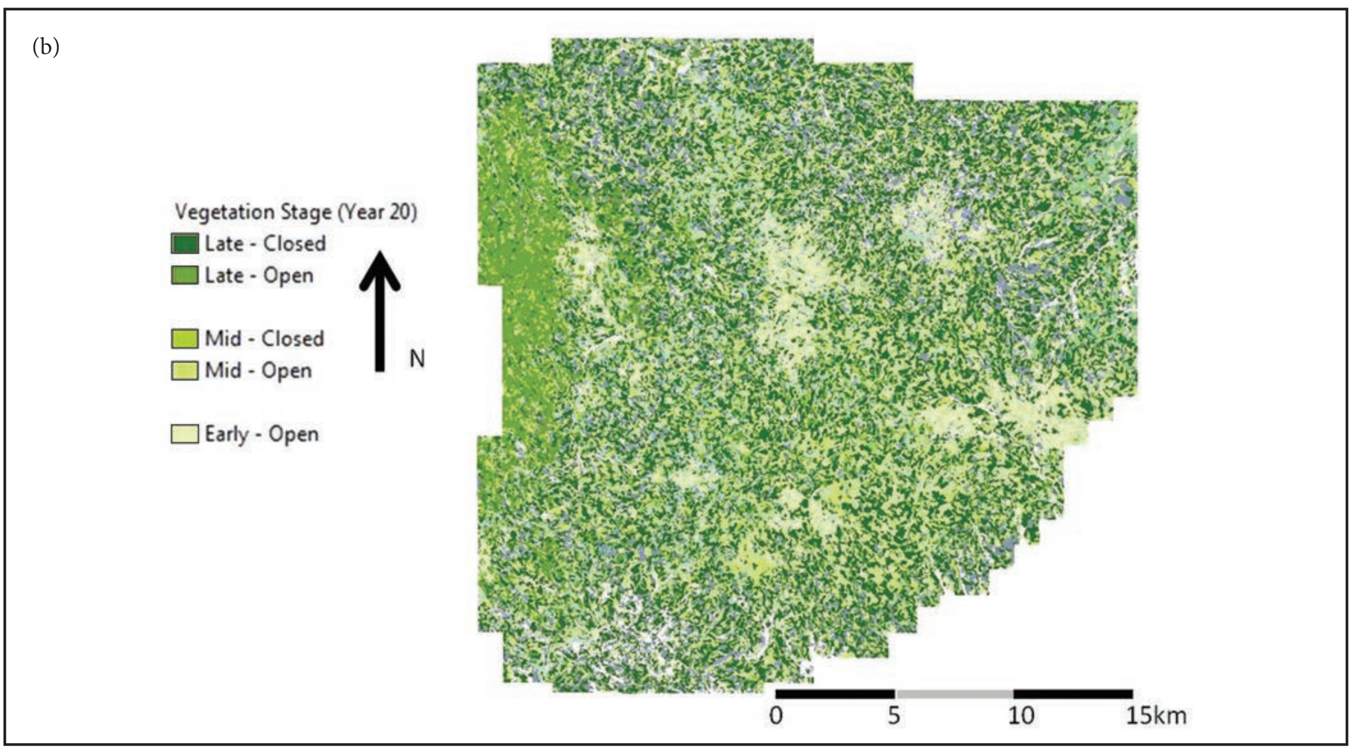


Figure 4—Sample output from a spatially-explicit STSM for the Castle Creek landscape of fig. 3. Results are shown for a single Monte Carlo iteration of a 20-year simulation. (a) Map showing cumulative area burned (in red) from year 0-20, superimposed upon ecological strata. (b) Map showing projected vegetation state in year 20.

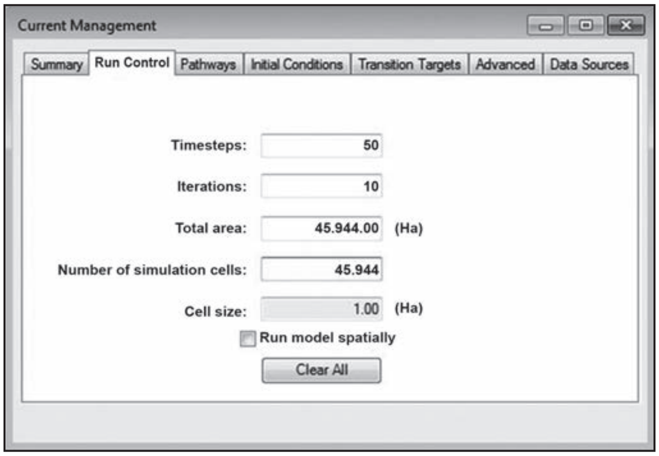


Figure 5—Sample run control settings for a STSM, as specified using the Path Landscape Model.

outputs (including maps in ArcGIS). Like VDDT, TELSA is available for free download, and continues to be supported and developed.

The last software tool for developing STSMs is the Path Landscape Model (Apex Resource Management Solutions 2012). First released in 2009, Path was designed to merge the functionality of both VDDT and TELSA, allowing users to develop and run whole landscape, spatially-stratified, and spatially-explicit STSMs all from a single platform. One of the major objectives of Path was to simplify the process of developing models and analyzing the results. Some of the important new features found in Path include the ability to develop and run spatially-stratified STSM models, including an option to specify transitions between strata, and the ability to create and run spatially-explicit STSMs using an automated connection to the TELSA simulation engine. At present, Path is being actively developed and supported, and is available for free download.

Using a software tool such as the Path Landscape Model to develop and run a spatially-stratified STSM is relatively straightforward, and typically involves the following steps:

1. Specify a number of run control parameters for the simulation, including the total number of timesteps, the number of Monte Carlo iterations, the total area of the landscape and the number of simulation cells (fig 5).
2. Divide the landscape into one or more ecological and/or land management strata and define a suite of possible states and transitions for each stratum. The list of possible strata, states and transitions is fully configurable by the user. Possible transitions between states are typically displayed as a pathway diagram for each strata. As discussed previously, there are three types of transition pathways that can be defined in STSMs: deterministic, probabilistic, and targets (fig. 6).
3. Specify the proportion of the landscape in each stratum and state (and optionally age) at the beginning of the simulation (fig. 7). For spatially-explicit simulations this information is derived directly from a user-specified polygon GIS map layer.
4. Run the simulation to generate model output. There are two basic outputs generated for every run: the area in each state over time, and the area undergoing each type of transition over time (fig. 8). If multiple Monte Carlo iterations are simulated then a range of variability around each of these model outputs can also be calculated.

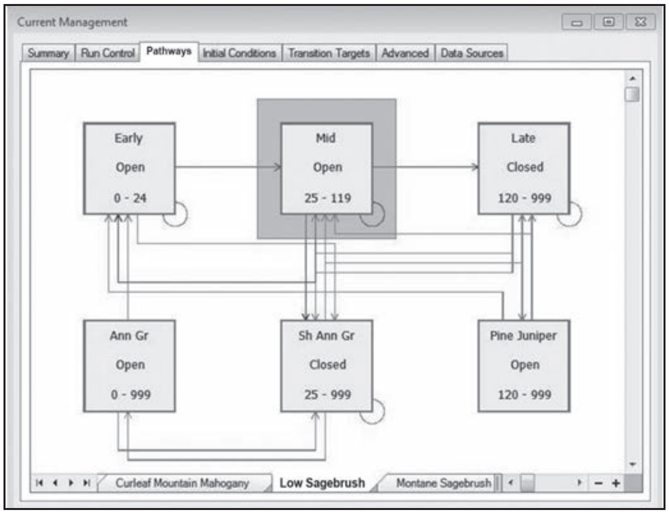
In addition to the model inputs described above, running a spatially-explicit STSM in Path involves specifying the following:

1. A polygon GIS shapefile dividing the landscape into simulation cells, with an initial STSM stratum, state and age assigned to each polygon;
2. A frequency distribution for the size of STSM transition events.

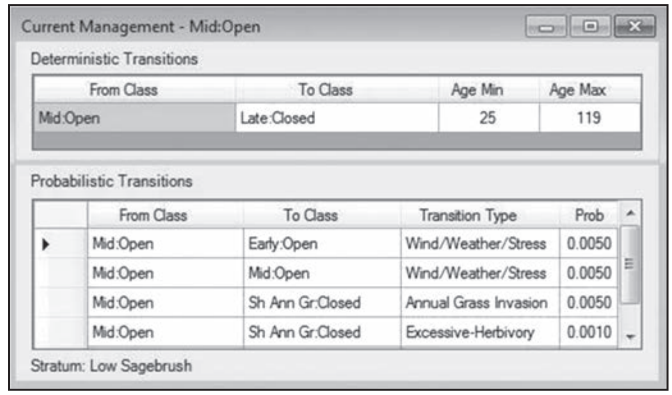
With these additional model inputs, Path is then able to automatically configure and run a spatially explicit TELSA simulation. Additional details on the TELSA model algorithms can be found in Kurz et al. (2000) and ESSA Technologies (2008).

Applications

STSMs have been applied to a wide range of questions and ecological systems. While most other landscape simulation models are designed to work with only forested ecosystems, due to their empirical nature STSMs have no predetermined



(a)



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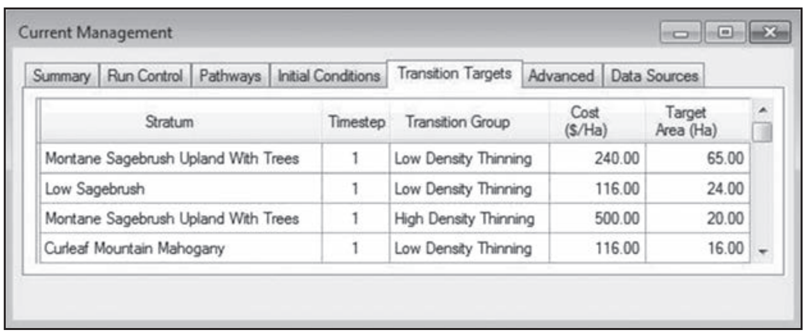


(c)

Figure 6—Sample transition inputs associated with a STSM, as specified using the Path Landscape Model. (a) Transition pathway diagram showing state classes (boxes) and transitions (arrows). (b) Deterministic and probabilistic transitions associated with a particular state class. (c) Transition targets and costs associated with management transitions.

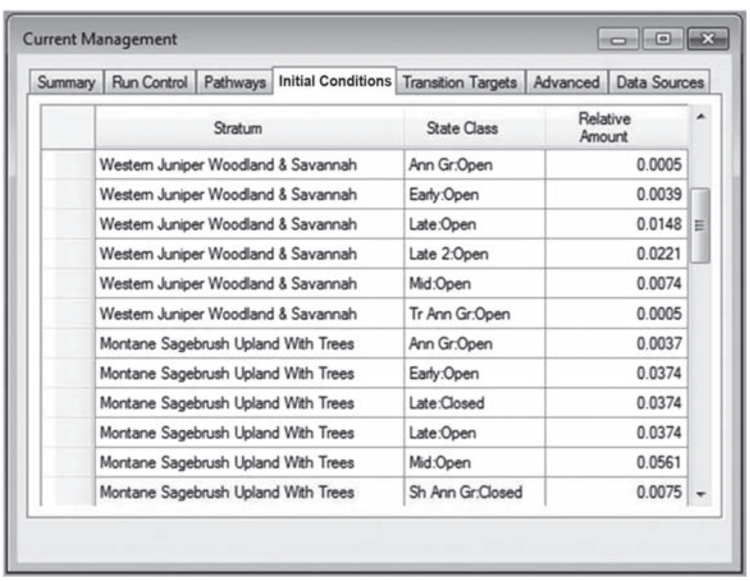
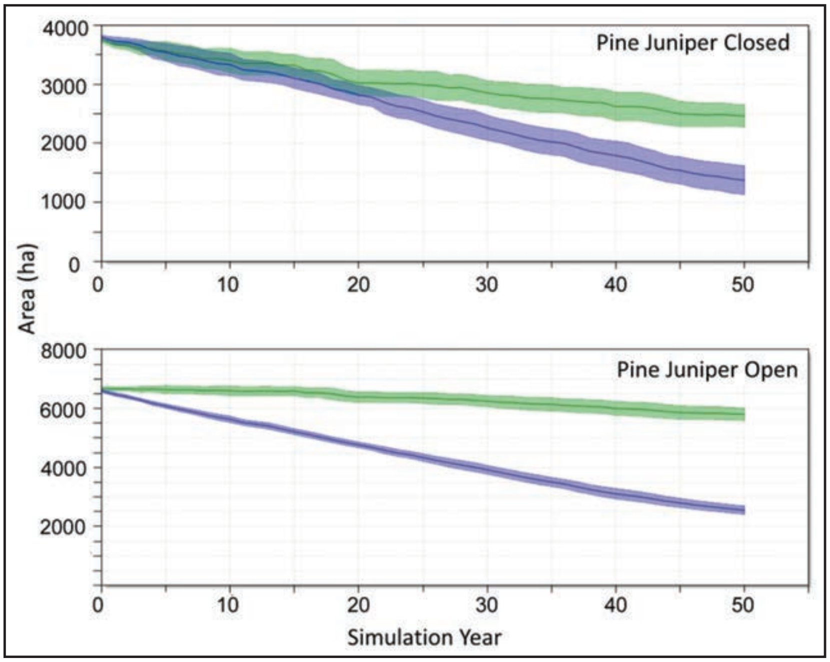


Figure 7—Sample STSM initial conditions, as specified using the Path Landscape Model.

relationships and as such can be parameterized for any suite of vegetation communities. STSMs are often used in situations where landscapes include some non-forest vegetation types; they are also often used in situations where landscapes include non-native exotic vegetation (e.g., Forbis et al. 2006; Provencher et al. 2007; Frid and Wilmshurst 2009).

(a)



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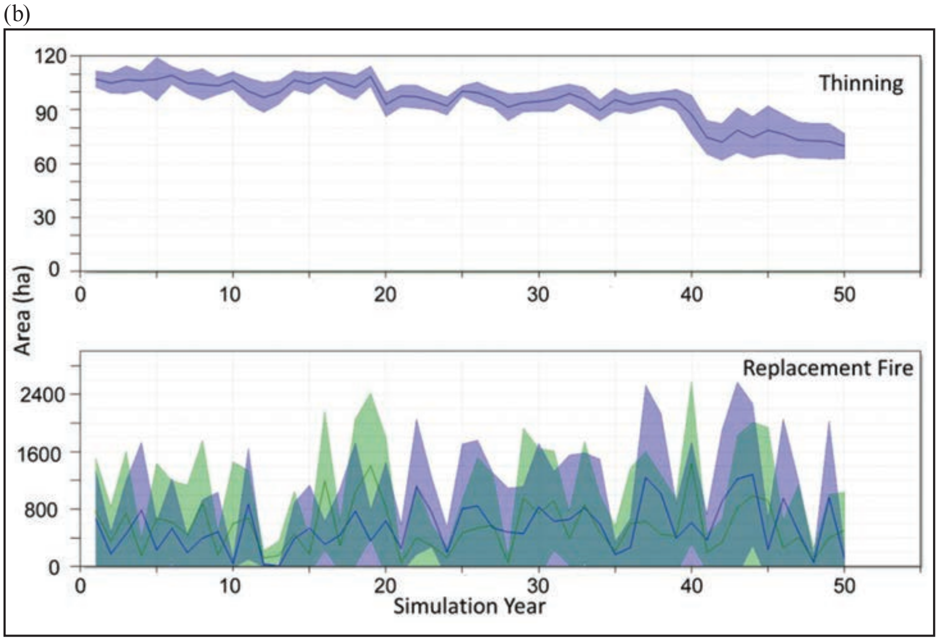

Figure 8—Sample STSM output for the Castle Creek landscape of fig. 3, comparing two simulation scenarios: No Management (green) and Current Management (blue). Shading shows 1.96 standard errors around the mean across 10 Monte Carlo iterations. (a) Area over time, across the entire landscape, in two of the model's state classes: Pine Juniper Closed and Pine Juniper Open. (b) Area over time, across the entire landscape, undergoing two of the model's transitions: Low Density Thinning and Replacement Fire.

Typically, the pathways and probabilities in STSMs are determined through analysis of historical data regarding rates of succession, disturbance and management; in situations where data is lacking, however, STSMs can be readily parameterized using literature and/or expert opinion (e.g., Czembor et al. 2011, Price et al. 2012), making the approach suitable for projects that contain at least some vegetation communities for which data and/or knowledge is limited. Finally with STSMs there are no hidden assumptions or relationships in the models, particularly when developed using software such as the Path Landscape Model, although, as with all models, it becomes increasingly challenging to keep track of all of the relationships as the complexity of the model increases. This makes the models well-suited for decision making in situations where stakeholder engagement and consensus are important.

STSM modeling projects usually begin by developing a whole-landscape or spatially-stratified STSM; avoiding a spatially-explicit model in the early stages of a modeling project makes it much simpler to parameterize and validate an initial model. Many, but not all, modeling projects eventually convert their STSMs to a spatially-explicit form. Factors that tend to limit the development of spatially-explicit STSMs include: (1) the objectives of the analysis not requiring spatially-explicit predictions; (2) the spatial extent of the landscape being too large to run in a spatially-explicit form; (3) a shortage of spatial data and/or resources required to develop and run the more complicated spatially-explicit models.

The first known use of STSMs was in support of landscape-level vegetation modeling for the Interior Columbia River Basin Ecosystem Management Project in the U.S. Pacific Northwest (Hann et al. 1997). In this project, a spatially-stratified STSM was developed using the VDDT software. The STSM approach provided two key benefits to this project: firstly, it provided a modelling platform that could handle both forests and rangelands across a single landscape; secondly, it allowed input from both stakeholders and experts to be incorporated into the models (Kurz et al. 1999).

Since this first project, STSMs have been applied to a wide range of management questions across a variety of ecological settings (table 1). The remainder of this section provides an overview of some of the questions for which the STSM approach has been used; additional examples can be found in Kerns et al. (2012b).

Forest Ecosystems

STSMs have been used extensively to inform the management and ecology of forest ecosystems. Klenner et al. (2000) were the first to develop a spatially-explicit STSM, using it to predict the combined effects of management actions and natural disturbances on old-growth habitat and patch size changes in British Columbia, Canada. Carlson and Kurz (2007) used a spatially-explicit STSM to explore the ability of alternative timber harvest strategies to approximate natural fire patterns in a boreal mixedwood forest of Alberta, Canada. Hemstrom et al. (2007) and Strand et al. (2009) used spatially-explicit STSMs to simulate the effects of alternative fire suppression and prescribed burning strategies on the structure and composition of forested systems in Idaho and Oregon. Klenner and Walton (2009) used a spatially-explicit STSM to predict the effects of alternative forest management treatments, such as partial cutting and fuel management treatments, on indicators of wildlife habitat, including understory productivity. Finally, a spatially-explicit STSM analysis was used to recreate the pre-European settlement structure and composition of the 12 million hectare Great Lakes St. Lawrence forest region of Ontario (Ontario Ministry of Natural Resources 2010).

Ecological Restoration

A second major application area for STSMs has been their use in supporting ecological restoration, particularly in rangeland and grassland vegetation communities. Forbis et al. (2006) developed a spatially-stratified STSM to explore the effects of alternative restoration scenarios across a 4.6 million hectare landscape of desert scrub (*Atriplex* spp. and others) and sagebrush (*Artemisia* spp.) communities in the Great Basin ecoregion of the western U.S. The model compared the effects of varying levels of fire suppression, livestock grazing, and restoration treatments on the predicted vegetation cover of native perennial understory and tree-invaded and weed-dominated states. Provencher et al. (2007) developed a spatially-explicit STSM to compare the cost and effectiveness of varying levels of managed fire, livestock grazing, and non-native species management in an effort to restore degraded vegetation types for a 140,000 hectare rangeland landscape in eastern Nevada. An STSM was developed to explore the effectiveness of alternative control strategies for the management of the invasive crested wheatgrass (*Agronpyron cristatum*) in Grasslands National Park of Canada (Frid and Wilmshurst 2009); this spatially-explicit STSM included an explicit representation of the inter-annual spread of invasive vegetation across a landscape.

The Nature Conservancy (TNC) has extended the application of STSMs for ecosystem restoration to help

Table 1—Examples of past STSM applications

Reference	Ecosystems	Application	Geographic area	Spatial extent (ha)	Spatially-explicit?
Provencher et al. 2008	Rangeland	Natural range of variability	Nevada USA	18,000	No
Strand et al. 2009	Forest	Fire and aspen management	Idaho USA	37,000	Yes
Frid and Wilmshurst 2009	Grassland	Grassland invasive management	Saskatchewan Canada	42,000	Yes
Miller 2007	Forest	Wildfire regimes	Idaho USA	50,000	Yes
Wondzell et al. 2007	Aquatic/Riparian	Disturbances and salmonid habitat	Oregon USA	54,000	No
Klenner et al. 2000	Forest	Management and wildlife habitat	British Columbia Canada	63,000	Yes
Low et al. 2010	Rangeland/Forest	Conservation planning	California USA	76,000	No
Klenner and Walton 2009	Forest	Management and wildlife habitat	British Columbia Canada	100,000	Yes
Provencher et al. 2007	Rangeland	Restoration and invasive management	Nevada USA	142,000	Yes
Hemstrom et al. 2007	Forest/Rangeland	Fuel and wildfire management	Oregon USA	178,000	No
Zweig and Kitchens 2009	Wetland	Wetland hydrology management	Florida USA	200,000	No
Czembor and Vesk 2009, Czembor et al. 2011	Forest	Restoration	Victoria Australia	250,000	No
Carlson and Kurz 2007	Forest	Harvest and landscape pattern	Alberta Canada	270,000	Yes
Price et al. 2012	Forest	Forest management and climate change	Michigan USA	272,000	Yes
Forbis et al. 2006	Rangeland	Restoration and invasive management	Great Basin USA	4.3 million	No
Ontario Ministry of Natural. Resources 2010	Forest	Pre-European settlement conditions	Ontario Canada	11.9 million	Yes
Rollins 2009, Swaty et al. 2011	Terrestrial	Assessment of ecological departure	Entire USA	~900 million	No
Kerns et al. 2012b	Various	12 papers covering a range of applications	Worldwide	Various	Yes and no

them determine how to get the best return on the restoration investment (Low et al. 2010). As part of their conservation planning process, TNC first uses an STSM approach to help predict how far departed a landscape is from “desired” condition; they then extend this reference STSM to explore how alternative future management actions will change the vegetation composition and structure of the landscape, and use this to predict which combination of management actions, on a per unit cost basis, will move the landscape most cost-effectively towards their desired condition.

Aquatic/Riparian Ecosystems

In addition to their long-standing application to upland vegetation communities, STSMs have also been developed for use in aquatic and riparian systems. A spatially-stratified STSM was developed to evaluate the effects of disturbances and land management practices, such as grazing, flooding, debris flows and wildfire disturbances, on riparian and aquatic vegetation for a river system in Oregon (Wondzell et al. 2007). In this STSM the states were defined by channel morphology and riparian vegetation, and model output was interpreted in terms of its suitability as habitat for anadromous salmonids. As a second example, Zweig and Kitchens (2009) developed a non-spatial STSM for predicting the effects of alternative hydrological regimes on wetland vegetation in the Florida Everglades.

Regional Land Management

Finally, because of their flexibility in handling a wide range of vegetation types and varying data availability, STSMs are well suited for use in regional and national land management initiatives. For example, the LANDFIRE program, a 5-year national, multi-agency project, developed over 1200 STSM models to predict the pre-European settlement conditions for all of the major ecosystems of the United States (Rollins 2009, Swaty et al. 2011). The Integrated Landscape Assessment Project (ILAP), a 3-year project to prioritize land management actions in the western U.S., developed over 200 STSMs, one for every potential vegetation type (PVT) found across all lands in Arizona, New Mexico, Oregon and Washington (Hemstrom et al. 2012).

Conclusions

STSMs provide a simple, flexible approach for predicting landscape-level vegetation change in response to both natural disturbances and management actions. They are stochastic, empirical simulation models, based on a Markov chain approach, that predict how vegetation will transition between states over time across a landscape. With STSMs a landscape is divided into a set of simulation cells (e.g. either raster pixels or polygons), each cell is assigned an initial vegetation state, and then the model predicts how the state of each cell changes over time. Spatial processes are typically represented in STSMs in one of two ways: by stratifying the landscape into zones that behave similarly with respect to vegetation change, and then representing each of these strata with its own non-spatial STSM; or alternatively, by creating a spatially-explicit STSM for the landscape in which the dynamics of every simulation cell is potentially dependent on its neighbours.

Over the past 20 years, the software available for developing STSMs has evolved considerably. Modellers are now able to represent an array of complex dynamics, including tracking and modifying the age-structure of the landscape, setting transitions to be contingent on past events, providing targets for transition areas (e.g. for forest harvest and management treatments), changing transition probabilities over time (e.g., due to climate change), and spreading disturbances across a landscape (e.g., for fire and invasive species).

As with all models, there are limitations to the use of STSMs for predicting vegetation change. Likely the most significant limitation is their purely empirical nature, whereby model relationships are typically fitted to existing knowledge and data. This makes it challenging to use the models to make predictions under novel conditions, as knowledge and data may not exist upon which to base the input parameters. To overcome this limitation users are increasingly turning to other, more specialized models to inform the model inputs for STSMs; for example Kerns et al. (2012a) have used the MC1 dynamic global vegetation model to incorporate climate change effects into STSMs, while others have used output from the Forest Vegetation

Simulator (FVS) to calibrate transition probabilities in STSMs (Shlisky and Vandendriesche 2012; Weisz and Vandendriesche 2012). This approach allows modellers to combine the integrative capabilities of STSMs with the mechanistic capabilities of other models. A second limitation with STSMs is that all relationships must be ultimately be expressed in terms of a Markov chain. This restriction can be limiting in certain circumstances: for example while it might be desirable to represent post-fire succession as a Markov chain, the fire spread component of the model might be better represented using a more mechanistic simulation approach. At present it is not possible to create such hybrid modeling approaches with STSMs. Efforts are underway, however, to extend the capabilities of STSM software (i.e. the Path Landscape Model) so that it can dynamically link to other models; this would provide the capability to generate hybrid modelling approaches, whereby external models could be called upon to provide specialized predictions within an STSM model run.

The STSM approach has been applied to a wide range of management questions and vegetation communities. Because STSMs can be developed for any vegetation community, they are well suited for landscapes that include a range of vegetation types. Examples of ecosystems in which STSMs have been developed include those with forest, rangeland and wetland components; they have also been used extensively to represent the dynamics of non-native (i.e., invasive) species. The spatial extent over which STSMs have been applied ranges from a few thousand to millions of hectares. Finally, STSMs are well suited for use in situations with imperfect knowledge or data, as the models can be parameterized, when necessary, using expert opinion, and a wide range of scenarios can be simulated to represent uncertainty in model inputs.

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References

Apex Resource Management Solutions Ltd. 2012. Path Landscape Model. http://www.apexrms.com/path (31 March 2012).

Baker, W.L. 1989. A review of models of landscape change. Landscape Ecology. 2(2): 111–133.

Bestlemeyer, B; Tugel, A.J.; Peacock, Jr., G.L.; Robinett, D.G.; Shaver, P.L.; Brown, J.R.; Herrick, J.E.; Sanchez, H.; Havstad, K.M. 2009. State-and-transition models for heterogeneous landscapes: a strategy for development and application. Rangeland Ecology and Management. 62: 1–15.

Carlson, M.; Kurz, W. 2007. Approximating natural landscape pattern using aggregated harvest. Canadian Journal of Forest Research. 37: 1846–1853.

Chiarucci, A.; Araújo, M.B.; Decocq, G.; Beierkuhnlein, C.; Fernández-Palacios, J.M. 2010. The concept of potential natural vegetation: an epitaph? Journal of Vegetation Science. 21(6): 1172–1178. Blackwell Publishing Ltd. http://doi.wiley.com/10.1111/j.1654-1103.2010.01218.x

Czembor, C.A.; Morris, W.K.; Wintle, B.A.; Vesk, P.A. 2011. Quantifying variance components in ecological models based on expert opinion. Journal of Applied Ecology. 48: 736–745 doi: 10.1111/j.1365-2664.2011.01971.x

Czembor, C.A.; Vesk, P.A. 2009. Incorporating between-expert uncertainty into state-and-transition simulation models for forest restoration. Forest Ecology and Management. 259(2): 165–175.

ESSA Technologies Ltd. 2007. Vegetation Dynamics Development Tool User Guide, Version 6.0. Prepared by ESSA Technologies Ltd., Vancouver, BC. 196 pp. http://essa.com/wp-content/uploads/2010/09/VDDT-60-User-Guide.pdf (31 March 2012).

ESSA Technologies Ltd. 2008. TELSA: Tool for Exploratory Landscape Scenario Analyses, Model Description, Version 3.6. Vancouver, BC. 64 pp. http://essa.com/wp-content/uploads/2010/09/ModelDescription.pdf (31 March 2012).

Forbis, T.A.; Provencher, L.; Frid, L.; Medlyn, G. 2006. Great Basin land management planning using ecological modeling. Environmental Management. 38(1): 62–83.

Frid, L.; Wilmshurst, J. 2009. Decision analysis to evaluate control strategies for Crested Wheatgrass in Grasslands National Park of Canada. Invasive Plant Science and Management. 2(4): 324–336.

Grant, C.D. 2006. State-and-transition successional model for bauxite mining rehabilitation in the Jarrah Forest of Western Australia. Restoration Ecology. 14: 28–37.

Hann, W.J.; Jones, J.L.; Karl, M.G.; Hessburg, P.F.; Keane, R.E.; Long, D.G.; Menakis, J.P; McNicoll, C.H.; Leonard, S.G.; Gravenmier, R.A.; Smith, B.G. 1997. Landscape dynamics for the basin. In T.M. Quigley and S.J. Arbelbide, tech. eds, An assessment of ecosystem components in the interior Columbia Basin and portions of the Klamath and Great Basins. Volume 2. Gen. Tech. Rep. PNW-GTR-405. Portland, OR: U.S. Department of Agriculture, Forest Service, Pacific Northwest Research Station: 337–1055

Hemstrom, M.A.; Merzenich, J.; Reger, A.; Wales, B. 2007. Integrated analysis of landscape management scenarios using state and transition models in the upper Grande Ronde River Subbasin, Oregon, USA. Landscape and Urban Planning. 80: 198–211.

Hemstrom, M.A.; Salwasser, J.; Halofsky, J.; Kagan, J.; Comfort, C. 2012. The Integrated Landscape Assessment Project. In: Kerns, B.K.; Shlisky, A.; Daniel, C.J., tech. eds. Proceedings of the First Landscape State-and-Transition Simulation Modeling Conference, June 14–16, 2011, Portland, Oregon. Gen. Tech. Rep. PNW-869. Portland, OR: U.S. Department of Agriculture, Forest Service, Pacific Northwest Forest and Range Experiment Station: 57–71.

Keane, R.E.; Cary, G.J.; Davies, I.D.; Flannigan, M.D., Gardner, R.H.; Lavorel, S.; Lenihan, J.M.; Li, C; Rupp, R.H. 2004. A classification of landscape fire succession models: spatial simulations of fire and vegetation dynamics. Ecological Modelling. 179: 3–27.

Kerns, B.K.; Hemstrom, M.A.; Conklin, D.; Yospin, G.; Johnson, B.; Bachelet, D.; Bridgham, S. 2012a. Approaches to incorporating climate change effects in state and transition simulation models of vegetation. In: Kerns, B.K.; Shlisky, A.J.; Daniel, C.J., tech. eds. Proceedings of the First Landscape State-and-Transition Simulation Modeling Conference, June 14–16, 2011, Portland, Oregon. Gen. Tech. Rep. PNW-GTR-869. Portland, OR: U.S. Department of Agriculture, Forest Service, Pacific Northwest Research Station: 161–171.

Kerns, B.K.; Shlisky A.; Daniel, C.J., tech. eds. 2012b. Proceedings of the First State-and-Transition Simulation Modeling Conference, June 14–16, 2011, Portland, Oregon. Gen. Tech. Rep. PNW-GTR-869. Portland, OR: U.S. Department of Agriculture, Forest Service, Pacific Northwest Forest and Range Experiment Station. 214 p.

Klenner, W.; Kurz, W.A.; Beukema, S.J. 2000. Habitat patterns in forested landscapes: management practices and the uncertainty associated with natural disturbances. Computers and Electronics in Agriculture. 27: 243–262.

Klenner, W.; Walton, R. 2009. Landscape-level habitat supply modelling to develop and evaluate management practices that maintain diverse forest values in a dry forest ecosystem in southern British Columbia. Forest Ecology and Management. 258: 146–157.

Knapp, C.N.; Fernandez-Gimenez, M.; Kachergis, E.; Rudeen, A. 2011. Using participatory workshops to integrate state-and-transition models created with local knowledge and ecological data. Rangeland Ecology and Management. 64(2): 158–170.

Kurz, W.A.; Beukema, S.J.; Klenner, W.; Greenough, J.A.; Robinson, D.C.E.; Sharpe, A.D.; Webb, T.M. 2000. TELSA: the Tool for Exploratory Landscape Scenario Analyses. Computers and Electronics in Agriculture. 27: 227–242.

Kurz, W.; Beukema, S.; Merzenich, J.; Arbaugh, M.; Schilling, S. 1999. Long-range modeling of stochastic disturbances and management treatments using VDDT and TELSA. Paper presented at Society of American Foresters National Convention, Sept. 14, 1999, Portland OR. http://essa.com/wp-content/uploads/2011/03/vddt_telsa.pdf. (31 March 2012).

Long, J L.; Miller, M.; Menakis, J.P.; Keane, R.E. 2006. Chapter 6—Developing the LANDFIRE Vegetation and Biophysical Settings Map Unit Classifications for the LANDFIRE Prototype Project. In: Rollins, Matthew G.; Frame, Christine K., tech. eds. 2006. The LANDFIRE Prototype Project: nationally consistent and locally relevant geospatial data for wildland fire management Gen. Tech. Rep. RMRS-GTR-175. Fort Collins, CO; U.S. Department of Agriculture, Forest Service, Rocky Mountain Research Station: 123–179.

Low, G.; Provencher, L.; Abele, S.L. 2010. Enhanced conservation action planning: Assessing landscape condition and predicting benefits of conservation strategies. Journal of Conservation Planning. 6: 36–60.

MacKenzie, W.H. 2012. Biogeoclimatic ecosystem classification of non-forested ecosystems in British Columbia. Prov. B.C., Victoria, B.C. Tech. Rep. 068. http://www.for.gov.bc.ca/hfd/pubs/Docs/Tr/Tr068.htm. (31 March 2012).

Miller, C. 2007. Simulation of the consequences of different fire regimes to support wildland fire use decisions. Fire Ecology. 3: 83–102.

Mladenoff, D.J. 2004. LANDIS and forest landscape models. Ecological Modelling. 180: 7–19.

Ontario Ministry of Natural Resources 2010. Forest Management Guide for Great-Lakes St. Lawrence Forests. Toronto: Queen's Printer for Ontario. 57 p. http://www.mnr.gov.on.ca/en/Business/Forests/Publication/272845.html (31 March 2012).

Price, J.; Silbernagel, J.; Miller, N.; Swaty, R.; White, M.; Nixon, K. 2012. Eliciting expert knowledge to inform landscape modeling of conservation scenarios. Ecological Modelling. 229: 76–87. doi: 10.1016/j.ecolmodel.2011.09.010.

Provencher, L.; Anderson, T. 2011. Climate Change Revisions to Nevada's Wildlife Action Plan: Vegetation Mapping and Modeling. Report prepared by The Nature Conservancy for the Nevada Department of Wildlife. http://www.ndow.org/wild/conservation/cwcs/index.shtm. (31 March 2012).

Provencher, L.; Forbis, T.; Frid, L.; Medlyn, G. 2007. Comparing alternative management strategies of fire, grazing, and weed control using spatial modeling. Ecological Modelling. 209: 249–263.

Provencher, L.; Campbell, J.; Nachlinger, J. 2008. Implementation of mid-scale fire regime condition class mapping. International Journal of Wildland Fire. 17: 390–406.

Rollins, M.G. 2009. LANDFIRE: a nationally consistent vegetation, wildland fire, and fuel assessment. International Journal of Wildland Fire. 18: 235–249.

Scheller, R.M.; Mladenoff, D.J. 2006. An ecological classification of forest landscape simulation models: tools and strategies for understanding broad-scale forested ecosystems. Landscape Ecology. 22: 491–505. doi:10.1007/s10980-006-9048-4.

Shlisky, A.; Vandendriesche, D. 2012. Use of state-and-transition simulation modeling in National Forest planning in the Pacific Northwest, U.S.A. In: Kerns, B.K.; Shlisky, A.J.; Daniel, C.J., tech eds. Proceedings of the First Landscape State-and-Transition Simulation Modeling Conference, June 14–16, 2011, Portland, Oregon. Gen. Tech. Rep. PNW-GTR-896, Portland, OR: U.S. Department of Agriculture, Forest Service, Pacific Northwest Research Station: 24–41.

Strand, E.K.; Vierling, L.A.; Bunting, S.C. 2009. A spatially explicit model to predict future landscape composition of aspen woodlands under various management scenarios. Ecological Modelling. 220(2): 175–191.

Swaty, R.; Blankenship, K.; Hagen, S.; Fargione, J.; Smith, J.; Patton, J. 2011. Accounting for ecosystem alteration doubles estimates of conservation risk in the conterminous United States. PLoS ONE. 6(8): e23002. doi:10.1371/journal.pone.0023002.

Weisz, R.; Vandendriesche, D. 2012. Use of the Forest Vegetation Simulator to quantify disturbance activities in state and transition models. In: Kerns, B.K.; Shlisky, A.J.; Daniel, C.J., tech. eds. Proceedings of the First Landscape State-and-Transition Simulation Modeling Conference, June 14–16, 2011, Portland, Oregon. Gen. Tech. Rep. PNW-GTR-869. Portland, OR: U.S. Department of Agriculture, Forest Service, Pacific Northwest Research Station: 143–160.

Westoby, M.; Walker, B.; Noy-Meir, I. 1989. Opportunistic management for rangelands not at equilibrium. Journal of Range Management. 42(4): 266–274.

Wondzell, S.M.; Hemstrom, M.A.; Bisson, P.A. 2007. Simulating riparian vegetation and aquatic habitat dynamics in response to natural and anthropogenic disturbance regimes in the Upper Grande Ronde River, Oregon, USA. Landscape and Urban Planning. 80(3): 249–267.

Xi, W.; Coulson, R.N.; Birt, A.G.; Shang, Z.B; Waldron, J.D.; Lafon, C.W.; Cairns, D.M.; Tchakerian, M.D.; Klepzig, K.D. 2009. Review of forest landscape models: Types, methods, development and applications. Acta Ecologica Sinica. 29: 69–78.

Zweig, C.L.; Kitchens, W.M. 2009. Multi-state succession in wetlands: a novel use of state and transition models. Ecology 90(7): 1900–1909.

Use of State-and-Transition Simulation Modeling in National Forest Planning in the Pacific Northwest, U.S.A.

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Abstract

Effective national forest planning depends on scientifically sound analyses of land management alternatives relative to desired future conditions and environmental effects. The USDA Forest Service Pacific Northwest Region is currently using state-and-transition simulation models (STMs) to simulate changes in forest composition and structure for the revisions of five forest plans in Oregon and Washington. We illustrate the use of STMs to examine current and desired forest conditions, develop forest plan environmental impact statement (EIS) alternatives, and evaluate environmental effects, with examples from the Okanogan-Wenatchee National Forest (Washington, U.S.A.). Model parameters include ecosystem states and natural and human-caused disturbances, which were derived from empirical studies, published literature, and expert opinion. Forest growth rates were calibrated using Forest Vegetation Simulator (FVS) modeling of national forest inventory plot data and FVS post-processors, such as the Preside program. Preside was used to classify forest inventory plots into STM states, estimate mean residence times (within a state) and transition probabilities (between states), and summarize the alternative pathways between states. In some cases for the Okanogan-Wenatchee National Forest, Preside showed longer residence times for dense multi-story stands than assumed in previously developed STMs for the forest plan area. STMs are being used to simulate the effects of alternative combinations of forest treatments such as forest thinning, regeneration harvest, and prescribed burning. The effects of treatments on a suite of indicators and decision criteria, including forest structural states, departure from reference conditions, woody biomass yield, wildlife habitat, and fire severity and frequency, will be estimated using the output of STMs. STMs used in forest plan revisions have proved useful in testing assumptions, developing alternative restoration scenarios, and documenting current knowledge.

Keywords: Land use planning, state-and-transition model, alternative development, model calibration.

National Forest Management Planning in the United States

Management of national forests in the United States is guided by the strategic goals of the U.S. Department of Agriculture and USDA Forest Service, as well as laws, regulations, and agency policies. The Forest Service is responsible for managing the lands and resources of the National Forest System (NFS), which includes approximately 193 million acres in 44 states, Puerto Rico, and the Virgin Islands. The NFS is composed of 155 national forests, 20 national grasslands, and one national tallgrass prairie. Goals for managing NFS lands include (USDA FS 2007):

- Restore, sustain, and enhance the nation's forests and grasslands,
- Provide and sustain benefits to the American people,
- Conserve open space,
- Sustain and enhance outdoor recreation opportunities,
- Maintain basic management capabilities of the Forest Service,
- Engage urban America with Forest Service programs, and
- Provide science-based applications and tools for sustainable natural resources management.

The focus of National Forest System land management is continually evolving. Since the 1990s, the primary focus of NFS land management has shifted from optimization of commodity output toward the restoration of ecosystem function and resiliency.

The framework for NFS land management plans (Forest plans) was established by Congress in the National Forest Management Act (NFMA) of 1976 (16 U.S.C. § 1604).

Under NFMA, Forest plans:

1. Establish forest multiple-use goals and objectives;
2. Establish forest-wide standards and guidelines;
3. Establish management areas and direction applying to future activities;
4. Designate lands suitable and unsuitable for timber production;
5. Evaluate potential wilderness areas; and
6. Establish monitoring and evaluation requirements.

NFMA is implemented under an agency planning rule, which establishes administrative procedures for developing, revising, and amending forest plans (36 Code of Federal Regulations Part 219). Current forest plans and plan revisions for all national forests follow guidance from the 1982 Planning Rule (47 Federal Register 43037, Sept. 30, 1982). However, the Final Programmatic Environmental Impact Statement (EIS) for a new land management planning rule was released in January 2012;[1] future forest plan revisions will take place under this 2012 rule until a new rule is adopted. Forest planning must also adhere to the requirements of the National Environmental Policy Act (NEPA) of 1969 (42 U.S.C. § 4321), which set up procedural requirements for all federal government agencies to prepare environmental assessments (EAs) and EISs, which disclose the environmental effects of proposed federal agency actions.

Forest plans establish direction so that all future decisions on the forest will consider physical, biological, economic, and other sciences, and assure coordination of multiple-uses and a sustainable yield of products and services. Coupled with laws and regulations, forest plans create a management system for future decisionmaking. Projects and activities are, proposed, analyzed and carried out within the framework of the plan.

Currently, Forest plans consider the roles and capabilities of NFS lands within a complex matrix of climate change, increased forest densities, extended drought, uncharacteristic insect epidemics, intense wildfires, expansion of residential development into forest lands, and rapidly changing socioeconomic settings. Given the mix of ecosystem processes, management objectives, and an uncertain future regarding NFS lands, plus the need to meet NEPA requirements (e.g., to disclose the effects of Forest Service management in EAs and EISs), National Forest planners depend on scientifically sound modeling tools to analyze the effects of alternative management scenarios.

Forest plans are developed by first drafting a proposed plan that attempts to address known challenges, public concerns, and new information. This proposed plan is released for public comment. After analyzing comments on the proposed plan, alternatives to the proposal are included in a draft environmental impact statement (DEIS). Generally, the DEIS: (1) compares alternative ways of managing national forest lands; and (2) outlines the physical, biological, social, and economic effects of each alternative. Alternatives include a "no change" (or "no action") proposal, which represents a continuation of current forest plan direction. After analysis of public comment on the DEIS, a final environmental impact statement (FEIS) and Record of Decision (ROD) are released along with a final forest plan which becomes management direction for the next 10 to 15 years.

Table 1 summarizes the key national forest planning considerations and their relationships to ecological state-and-transition modeling (STM). Given the 2012 Planning Rule, science must be taken into account, appropriately interpreted, and applied when planning models are developed. Incomplete or unavailable information, scientific uncertainty, and risk are evaluated and disclosed as a part of model and forest plan documentation. Published research,

[1] The Final Programmatic EIS for the 2012 Forest Service planning rule is available at: http://www.fs.usda.gov/detail/planningrule/home/?cid=stelprdb5349164.

Table 1—Key National Forest Planning considerations and their relationship to ecosystem modeling

Forest planning consideration	State-and-transition modeling task related to national forest planning
Desired conditions	Define ecosystem states and transitions that are relevant to current, future, and desired conditions.
	Simulate long-term reference conditions to serve as baselines.
	Define relationships of key issues to desired ecosystem states and/or transitions (e.g., wildlife habitat, smoke production, watershed health).
Objectives	Simulate management actions and outcomes that will move the forest toward desired conditions. This is done by management area and/or forest-wide.
Standards and Guidelines	Use transition targets or adjust transitions probabilities to simulate effects of standards and guidelines.
	Define states that are important to track relative to standards and guidelines (e.g., threatened and endangered species habitat).
Draft Proposed Action (PA) and alternatives to the PA	Build PA and alternative scenarios based on desired conditions, natural processes, temporal variability, and varying management strategies and extents.
Analysis of effects of the PA and alternatives	Simulate alternative outcomes for key indicators and decision criteria.
	Simulate cumulative effects of the PA and alternatives for ecosystem and social values across treatment types and land jurisdictions.
Monitor, adapt and amend plan	Model parameters serve to document assumptions.
	Adjust model probabilities, temporal and transition multipliers, initial conditions, and other parameters through time as conditions change.
Best available science	Appropriately use empirical data whenever possible.
	Implement quality control on all models.

empirical studies, expert opinion, or combinations of each are used to define model states, transition probabilities between states, and temporal cycles of disturbance.

Forest Plan Modeling in the Pacific Northwest

The Pacific Northwest (PNW) Region of the USDA Forest Service includes 16 national forests. Five national forests are currently revising their Forest Plans (fig. 1).

Two landscape level modeling applications are supported in the PNW Region for forest planning: the linear programming model Spectrum (USDA FS 1995) and the state-and-transition simulation model VDDT/Path (ApexRMS and ESSA Technologies Ltd. 2012; ESSA Technologies Ltd. 2007). The original planning efforts in the 1980s and early 1990s aimed to meet goals and objectives to optimize or maximize net public benefit from various forest outputs. The first round of forest plans were developed using FORPLAN (the predecessor to Spectrum; Johnson et al. 1986, Kent et al. 1991). FORPLAN was used to choose the best mix of management options to meet specified goals and objectives given resource connstraints.

Now, into a second round of planning, objectives for national forest management have changed along with social values and knowledge. These changes have also led to a shift in approach away from linear programming toward STM. National forest management goals currently emphasize restoring and maintaining ecosystem health, biodiversity, and resilience, while contributing to economic and social sustainability (USDA FS 2007). A linear programming approach generally has less applicability in addressing these types of goals. Further, forest planning staff now commonly use GIS to spatially analyze key issues, such as wildlife viability and habitat distribution, invasive species, and fire behavior. VDDT/Path is a state-and-transition modeling framework used in the PNW and elsewhere for examining the role of various disturbance agents and management actions in changing vegetation composition and structure. With STMs, users create and

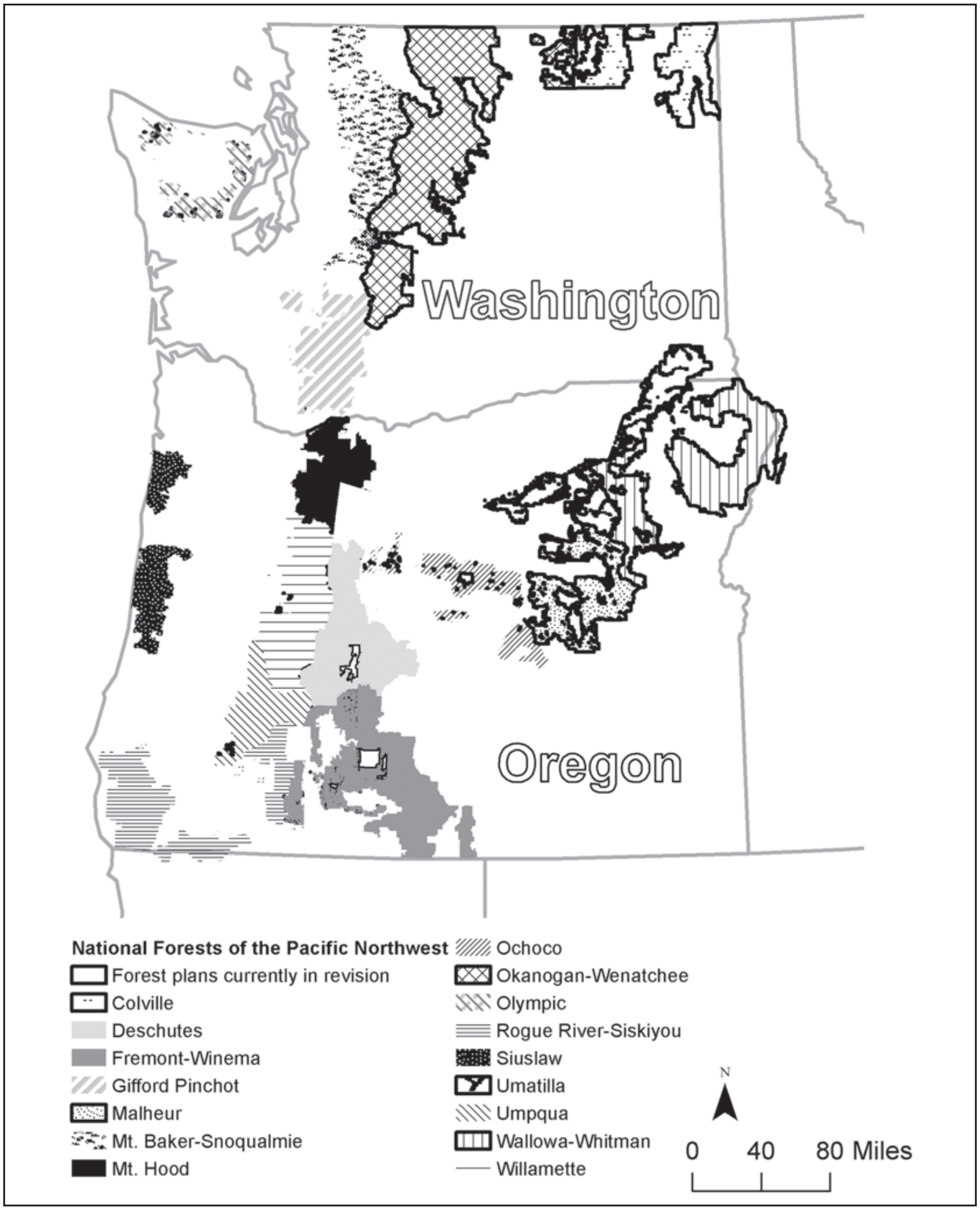

Figure 1—National Forests in the Pacific Northwest Region, USA, including five forests that are currently revising forest plans.

test quantitative assumptions about vegetation dynamics, and simulate their effects on ecosystems into the future at the landscape level. Unlike linear programming models, STMs can address the interaction of many complex natural and human factors (e.g., tree harvests, fire, insects, pathogens, mammals, weather, growth, competition) and their combined effects over long periods, but they do not directly optimize a solution given ecosystem functions and management objectives. Principally, the VDDT/Path STM is:

- Flexible
- In the public domain

- Relatively user friendly
- Able to model both deterministic and probabilistic processes
- A non-equilibrium model, which characterizes Ecosystems as constantly changing as a result of disturbance and other processes
- Compatible with the Tool for Exploratory Landscape Scenario Analysis (TELSA) spatial model (Kurz et al. 2000)
- Technically supported at the USFS PNW Regional and National levels

All five national forest plan revision efforts in the PNW Region are using STMs as their primary landscape level vegetation modeling framework in conjuntion with regiona/local spatial and non-spatial data. The plan revision team in northeast Washington for the Okanogan-Wenatchee and Colville National Forests is using STMs compiled, standardized, and enhanced by the Integrated Landscape Assessment Project (ILAP[2]). ILAP has produced VDDT/Path STMs and associated GIS data seamlessly for all broad vegetation types across the PNW Region (Oregon and Washington), regardless of land ownership (Hemstrom et al. 2012). Three national forests in the Blue Mountains are revising their plans using models that were precursors to ILAP, in combination with locally derived existing and potential vegetation spatial data.

Throughout this paper, examples of concepts will be presented from the Okanogan-Wenatchee National Forest Plan revision effort that is currently underway. The Okanogan-Wenatchee National Forest encompasses more than 4-million acres in Washington State and stretches from the Canadian border to about 180 miles south, and from the Cascade Crest east into the Okanogan highlands. The forest is very diverse—from high, glaciated alpine peaks along the Cascade Crest, through valleys of old growth forest, to dry shrub-steppe at its eastern edge. Annual precipitation varies widely from more than 70 inches along the crest to less than 10 inches at its eastern edge.

The Okanogan and Wenatchee National Forests first developed their forest plans in 1989 and 1990, respectively. The forests were administratively combined into one forest in 2000. The plan revision process began in 2003 but was delayed by the development of a new U.S. Forest Service Planning Rule and subsequent litigation action between 2005 and 2008. The Okanogan-Wenatchee National Forest faces at least the following challenges, which are being considered as they revise their forest plan for the next 10–15 years:

- Climate is already a significant stressor in the Columbia River Basin and eastern Cascade Range and if predictions are correct, the land area affected by wildfire could double by 2040. Vegetation communities will also change, likely in unpredictable ways (CIG 2009).
- Climate change, increasing pollution, spreading invasive plant and animal species, demand for natural resources, and human activities threaten to destabilize ecosystems.
- Fragmentation of wildlife habitat resulting from use patterns on lands adjacent to national forests, management activities, and increased demand of NFS lands is affecting the ability to manage for federally protected species, such as the northern spotted owl, Canada lynx, grizzly bear, and gray wolf.
- In the past ten or more years, there have been extensive outbreaks of, or increases in insects and disease (e.g., mountain pine beetle, western spruce budworm, balsam woolly adelgid, white pine blister rust), in some cases resulting in widespread tree mortality over large landscapes.

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[2] ILAP is a two-year project working on the watershed-level prioritization of land management actions based on fuel conditions, wildlife and aquatic habitats, economic values, and projected climate change across all lands in Arizona, New Mexico, Oregon, and Washington. The project creates a variety of analytical and graphical tools—including VDDT/Path models—that generate tables, graphs, and maps that land managers and planners can use to integrate and prioritize management activities. ILAP is a partnership between the U.S. Forest Service and Oregon State University. See http://oregonstate.edu/inr/ilap/ and Hemstrom et al. (2012) for more information.

- Uncharacteristic wildfires are inherently dangerous and difficult to suppress. Associated costs are rising.

The Forests released a draft proposed plan for public comment in June 2011. At this point in time, the plan revision team is developing themes for management alternatives that will eventually be analyzed in a draft environmental impact statement. Hence, applications of STM to-date have been limited to setting up model structures, transitions, and initial conditions, and developing a reference condition scenario. Future efforts will result in the development of more complex model scenarios and outputs than are described in this paper.

Ecological Models

Currently, ecological modeling for forest plan revisions in the PNW Region start with ILAP state-and-transition models and data, which are consistent with national and regional vegetation mapping and classification standards. Individual Forest Plan revision teams adjust these regionally-compiled models to meet their needs, such as adding state classes, adjusting potential vegetation type (PVT) boundaries, using local vegetation data in lieu of regional data sets, condensing the number of model classes into fewer states or PVTs, or other adjustments. Figure 2 provides a generalized overview of key relationships between STMs and spatial and non-spatial data within the context of national forest planning.

Potential vegetation types—
A potential vegetation type encompasses a group of plant associations that are characterized by a particular development pattern due to environment conditions and disturbance regimes (Henderson et al. 2011). Each VDDT/Path model represents one PVT within a specific ecoregion. For the Okanogan-Wenatchee forest plan revision, the forested landscape is stratified into five potential vegetation type groups[3] that are each depicted by one or more PVT models, including dry forest (dry pine, dry mixed conifer PVTs), mesic forest (moist mixed conifer, cool-moist forest), cold-moist forest (Pacific silver fir), cold-dry forest (mountain hemlock), and alpine forest including subalpine parkland).

The Integrated Landscape Assessment Project has created spatial GIS data for PVTs across the Pacific Northwest Region, including all land ownerships (fig. 3). These PVT maps display the spatial distribution of each state-and-transition model. They were created by either cross walking existing plant association group maps (created from 2004 to 2008 using a non-linear regression technique) to ILAP PVTs, or using a Random Forest Nearest Neighbor imputation process to map PVT distributions (completed in 2010). Potential vegetation type maps being used by the Okanogan-Wenatchee forest plan revision were created by cross walking a plant association group map created by nonlinear regression in 2004 to ILAP PVTs.

State-and-transition model structure—
Within each STM, states are defined by standardized combinations of cover type and structure (tree size, canopy density, and canopy layering). ILAP forested vegetation model state class specifications include at least the following:

- Dominant cover type (one, two, three, or mixed species dominance)
- Seven tree diameter classes (<1", 1–5", 5–10", 10–15", 15–20", 20–30", >30")
- Four tree density (canopy cover) classes (<10 percent, 10–40 percent, 40–60 percent, 60 + percent)
- Two tree layer classes (single storied, or multiple storied)

The Okanogan-Wenatchee National Forest Plan revision is largely using state-and-transition models created by ILAP but has added or deleted a few model states based on local observations and empirical data availability. In some cases, there are close to 50 vegetation states for some ILAP PVT models. For the Okanogan-Wenatchee plan revision, giant tree size classes (30" + diameter) were added to a number of ILAP models. Also natural successional pathways were adjusted in the mountain hemlock model to capture the

[3] Okanogan-Wenatchee revision efforts are currently focused on constructing forest and woodland VDDT/Path models. Non-forested ecosystems make up a relatively smaller portion of the National Forest.

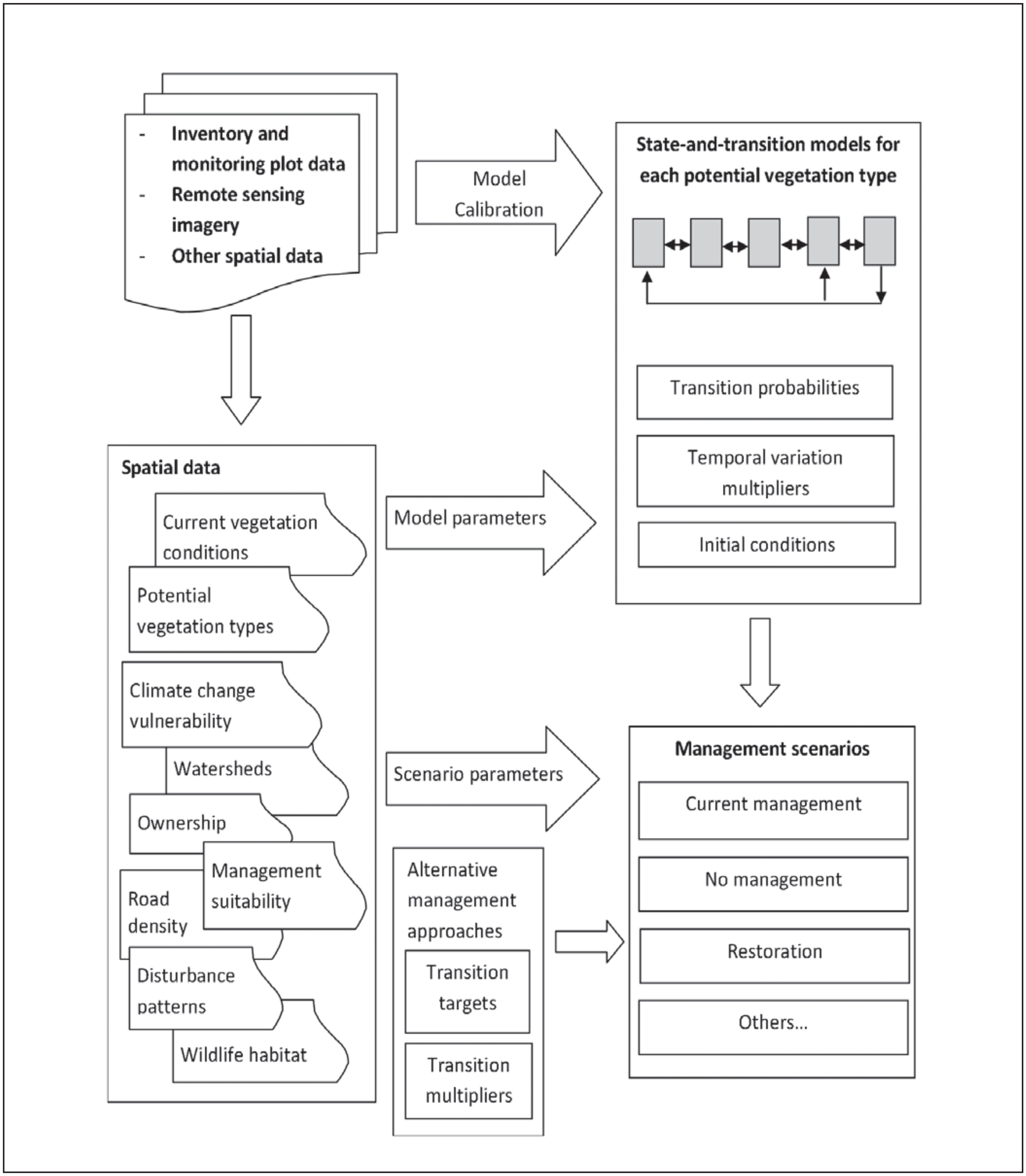


Figure 2—Overview of relationships between state-and-transition models, and spatial and non-spatial data within the context of national forest planning.

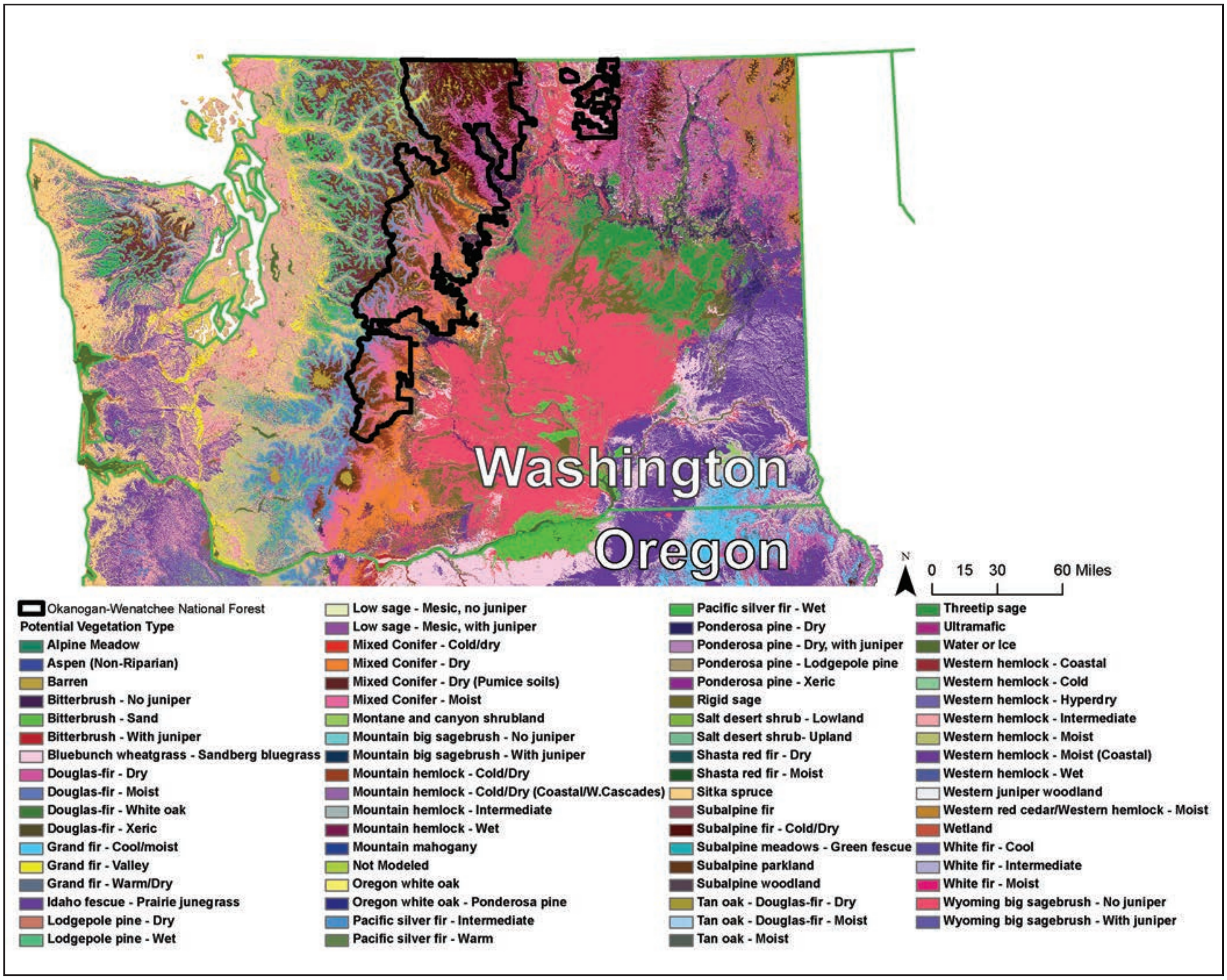
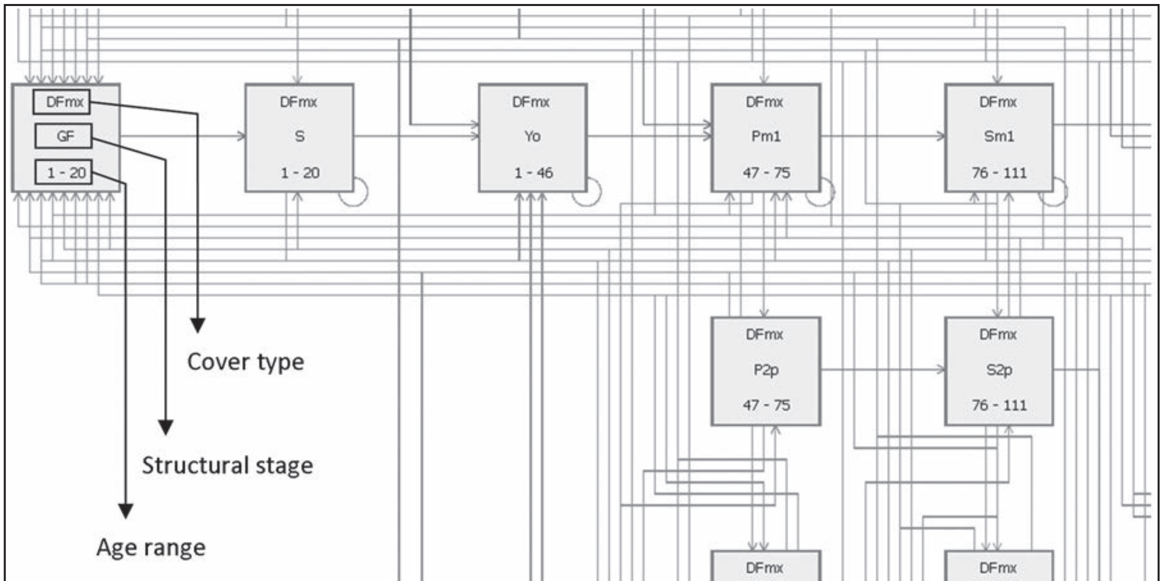


Figure 3—Potential vegetation types of Washington and the Okanogan-Wenatchee National Forest as mapped by the Integrated Landscape Assessment Project (Hemstrom et al. 2021).

observed dynamics between vegetation states dominated by lodgepole pine cover, and bark beetle/fire disturbances, which frequently prevent forests from developing toward later seral stages. Consequently, to simplify analysis and interpretation, and create compatibility with existing data on historical reference conditions, forested structural states were combined into seven structural groups for most analyses. These seven classes were derived from the Interior Columbia Basin Ecosystem Management Project (Hessburg et al. 1999), and include: (a) Stand Initiation, (b) Stem Exclusion Open Canopy, (c) Stem Exclusion Closed Canopy, (d) Understory Re-initiation, (e) Young Forest Multi-Strata, (f) Old Forest Single Strata, and (g) Old Forest Multi-Strata.

Figure 4 illustrates a small portion of the mesic (cool-moist) forest STM being used for the Okanogan-Wenatchee National Forest plan revision. This mesic forest model contains 34 vegetation states ranging from post-disturbance grass/forb conditions to closed canopy, multi-layer giant tree conditions. Transitions between states result from tree growth (increase in tree size and/or stand density) or disturbances (i.e., fire, insects, disease, severe weather, or management actions). Generally, growth transitions were



State class codes. *Cover type*: DFmx = Douglas-fir/mixed species cover type. *Structural stages*: GF = grass/forb; S = shrubs; Yo = Seedling/sapling open density; Pm1 = pole, medium density, single story; Sm1 = small tree, medium density, single story; P2p = pole, multi=story, post-disturbance; S2p = small tree, multi-story, post-disturbance.

Figure 4—A portion of the mesic forest STM model being used for the Okanogan-Wenatchee National Forest plan revision. The complete mesic (cool-moist) forest model includes 34 state classes and 207 probabilistic transitions. Tables 2a and 2b display examples of transitions between state classes.

calibrated by running forest inventory plot data through the Forest Vegetation Simulator (FVS) (Dixon 2002). This process is explained in more detail below in the section titled "Empirical calibration of vegetation succession rates." Fire transition probabilities were derived from Monitoring Trends in Burn Severity mapping data (Eidenshink et al. 2007).

Existing vegetation type map and initial model conditions—

The PNW Region has a spatial database of existing vegetation that was produced by the USFS PNW Research Station and Oregon State University, in close collaboration with the Western Wildland Environmental Threat Assessment Center (WWETAC), Interagency Mapping and Assessment Project (IMAP—a precursor to the ILAP project), Northwest Forest Plan Effectiveness Monitoring program, Remote Sensing Applications Center (RSAC), and the Forest Inventory and Analysis (FIA) program at the PNW Research Station. Existing vegetation mapping was integrated with ongoing sample-based forest inventories conducted by FIA at the PNW Research Station and Current Vegetation Survey (CVS) of the USFS PNW Region and Bureau of Land Management (BLM) lands in western Oregon. Gradient imputation (Gradient Nearest Neighbor, or GNN; Ohmann and Gregory 2002) was used to map vegetation composition and structure for areas of forest and woodland. GNN uses multivariate gradient modeling to incorporate data from FIA field plots with satellite imagery and mapped environmental data. A suite of fine-scale plot variables is imputed to each pixel in a digital map, and regional maps can be constructed for many of the same vegetation attributes available for FIA plots. Nonforest areas were mapped by ILAP scientists using a similar GNN imputation method. All GNN map products are grid-based at 30-m spatial resolution.

For the Okanogan-Wenatchee forest plan revision, these GNN existing vegetation maps were combined with the PVT layer, an ownership/land jurisdiction layer, and a management emphasis layer to determine initial forest conditions by PVT and management emphasis. Spatial land ownership data is used to delineate lands that fall under USDA Forest Service management jurisdiction and as such under Forest Plan revision direction. The spatial boundaries of non-NFS lands within the general planning area (e.g., other federal, state, private lands) are used to delineate areas that do not fall under forest plan direction, but may be important in analyses of cumulative effects of proposed management actions across all lands in the general planning area. The management emphasis layer for the Okanogan-Wenatchee plan revision divides NFS lands into "Wilderness," "Timber Suitable," and "Other" categories, each of which represent lands with similar forest management constraints and hence similar management transition probabilities. Initial conditions are the starting point for modeling alternative future scenarios representing forest plan revision EIS alternatives. The ownership/land jurisdiction classes used in the Okanogan-Wenatchee Plan revision are defined primarily by NFS versus other lands. The management emphasis layer is defined by the management emphasis being used for the forest plan revision, such as wilderness, timber suitable lands, and "other" management areas. It is expected that probabilities for management activities will generally differ between areas with differing management emphases.

Probabilistic transitions—
Natural and anthropogenic disturbances are characterized in STMs by deterministic and probabilistic transitions. A standardized suite of probabilistic transition types is used across the USFS PNW Region to maintain the ability to combine model parameters and ensure ease of model interpretation across geographical areas. Tables 2a and 2b show deterministic transition probabilities (primary successional pathways), and examples of probabilistic transition probabilities of the mesic (cool-moist) forest STM being used for the Okanogan-Wenatchee National Forest plan revision, respectively. Table 3 lists all probabilistic transition types currently being used in STMs for forest planning in the USFS PNW Region. Probabilistic transitions between states result from tree growth, natural disturbances, and anthropogenic management actions. In general, VDDT/Path STMs are attributed with average annual probabilities of transition. Transition multipliers are used to increase or decrease average transition probabilities by transition type across all states.

The Okanogan-Wenatchee National forest plan revision is using the probabilistic transition types created by ILAP, with corrections to some of the types of management treatments assigned to each state and adjustment of some natural disturbance probabilities (i.e. wildfire, insects and disease) to better reflect local situations. The forest plan revision effort is also using ILAP's Monte Carlo temporal multipliers, which were developed from existing data and expert knowledge about temporal patterns in the frequency and intensity of fire[4] and insect disturbances.

Compatibility with other models—
The National Forest Management Act (NFMA) requires that land management plans provide for diversity of plant and animal communities based on the suitability and capability of the land area while meeting overall multiple-use objectives. For terrestrial wildlife species, diversity is assessed through the use of a regional assessment procedure (Suring et al. 2011), which includes the identification of source habitat[5] and modeling of viability relative to reference conditions. The structure of STMs in the PNW Region are designed, to the extent possible, to be compatible with these terrestrial species viability assessment models, at least for upland forests. First, the classes of potential vegetation types used to stratify national forests for forest planning are amenable to analysis of potential wildlife habitat for terrestrial species viability assessments. Second, structural states (e.g., tree size class and density) within forest planning vegetation models are generally compatible with the definition of the structural components of wildlife species source habitat. Third, the reference point used to assess ecosystem health, design forest plan objectives, as well as assess terrestrial species viability is partly determined by the "historical or natural range of variability".

[4] ILAP uses data from the Monitoring Trends in Burn Severity (MTBS) database to develop temporal variation sequences and multipliers for wildfire disturbances.
[5] Source habitat includes vegetation states that contribute to stationary or positive wildlife population growth (Wisdom et al. 2000).

Table 2a—Deterministic transitions (primary successional pathways) of the mesic (cool-moist) forest STM being used for the Okanogan-Wenatchee National Forest plan revision. The mesic forest model contains 34 vegetation states ranging from postdisturbance grass/forb conditions to closed canopy, multi-layer, giant tree conditions

From class (all states have a Douglas-fir/mixed tree cover type)	Start age (years)	End age (years)	To class (all states have a Douglas-fir/mixed tree cover type)
Grass/forb	1	20	Shrubs
Grass/forb post disturbance	1	20	Shrubs post disturbance
Shrubs	1	20	Seedling/sapling open cover
Shrubs post disturbance	1	20	Seedling/sapling open cover
Seedling/sapling medium cover	1	46	Pole closed cover single story
Seedling/sapling open cover	1	46	Pole medium cover single story
Seedling/sapling post disturbance	1	46	Pole medium cover single story
Pole single story post disturbance	47	75	Small tree single story post disturbance
Pole multi-story post disturbance	47	75	Small tree multi-story post disturbance
Pole closed cover single story	47	75	Small tree closed cover single story
Pole closed cover multi-story	47	75	Small tree closed cover multi-story
Pole medium cover single story	47	75	Small tree medium cover single story
Pole medium cover multi-story	47	75	Small tree medium cover multi-story
Small tree single story post disturbance	76	111	Medium tree medium cover single story
Small tree multi-story post disturbance	76	111	Medium tree medium cover multi-story
Small tree closed cover single story	76	111	Medium tree closed cover single story
Small tree closed cover multi-story	76	111	Medium tree closed cover multi-story
Small tree medium cover single story	76	111	Medium tree medium cover single story
Small tree medium cover multi-story	76	111	Medium tree medium cover multi-story
Medium tree single story post disturbance	112	167	Medium tree medium cover single story
Medium tree closed cover single story	112	167	Large tree closed cover single story
Medium tree closed cover multi-story	112	167	Large tree closed cover multi-story
Medium tree medium cover single story	112	167	Large tree medium cover single story
Medium tree medium cover multi-story	112	167	Large tree medium cover multi-story
Large tree single story post disturbance	168	300	Large tree medium cover single story
Large tree closed cover single story	168	300	Giant tree closed cover single story
Large tree closed cover multi-story	168	300	Giant tree closed cover multi-story
Large tree medium cover single story	168	300	Giant tree medium cover single story
Large tree medium cover multi-story	168	300	Giant tree medium cover multi-story
Giant tree single story post disturbance	301	500	Giant tree single story post disturbance
Giant tree closed cover single story	301	500	Giant tree closed cover single story
Giant tree closed cover multi-story	301	500	Giant tree closed cover multi-story
Giant tree medium cover single story	301	500	Giant tree medium cover single story
Giant tree medium cover multi-story	301	500	Giant tree medium cover multi-story

For more information on the use and limitations of the use HRV in establishing desired conditions, see the section below titled “Modeling Alternative Management Scenarios and their Environmental Consequences.”

Empirical calibration of vegetation succession rates— Growth projections of forest inventory data using the Forest Vegetation Simulator (FVS) (Dixon 2002) provide empirically based information for validating model state age ranges and successional transition rates. Relatively large amounts of plot data from regional forest inventories are available for most modeled states in the PNW Region’s forested STMs. For the Okanogan-Wenatchee plan revision, a computer program called Preside (Vandendriesche 2009) was used to classify data from over 3,200 U.S.

Table 2b—A sample of probabilistic transitions of the mesic (cool-moist) forest STM being used for the Okanogan-Wenatchee National Forest plan revision. Probabilistic transitions between states result from tree growth, natural disturbances, and anthropogenic management actions

From state class[a]	To state class[a]	Transition type[b]	Annual probability	Age shift (years)[c]	Time since transitions (years)[d]
Giant tree single story post disturbance	Grass/forb	Stand-replacement wildfire	0.0040	0	0
Giant tree closed cover single story	Giant tree closed cover single story	Non lethal wildfire	0.0008	5	0
Giant tree closed cover single story	Large tree single story post disturbance	Spruce budworm outbreak	0.0100	0	0
Grass/forb	Shrubs	Stand-replacement wildfire	0.0027	0	0
Grass/forb	Seedling/sapling open cover	Natural regeneration, mid-seral species	0.0075	0	0
Giant tree medium cover single story	Giant tree medium cover multi-story	Understory development	0.0400	1	20
Large tree single story post disturbance	Grass/forb	Stand replacement wildfire	0.0012	0	0
Large tree single story post disturbance	Large tree single story post disturbance	Non lethal wildfire	0.0015	5	0
Large tree single story post disturbance	Large tree medium cover multi-story	Understory development	0.0400	1	20
Large tree closed cover single story	Large tree single story post disturbance	Spruce budworm outbreak	0.0100	0	0
Medium tree single story post disturbance	Grass/forb	Stand-replacement wildfire	0.0012	0	0
Medium tree single story post disturbance	Medium tree single story post disturbance	Non lethal wildfire	0.0015	5	0
Medium tree closed cover multi-story	Medium tree closed cover single story	Non lethal wildfire	0.0011	5	0
Medium tree medium cover single story	Grass/forb post disturbance	Stand-replacement wildfire	0.0012	0	0
Pole single story post disturbance	Pole multi-story post disturbance	Understory development	0.0400	1	20
Pole multi-story post disturbance	Grass/forb	Stand-replacement wildfire	0.0015	0	0
Pole multi-story post disturbance	Pole single story post disturbance	Non lethal wildfire	0.0012	5	0
Shrubs	Seedling/sapling open cover	Natural regeneration, mid-seral species	0.0038	0	0
Shrubs post disturbance	Shrubs post disturbance	Non lethal wildfire	0.0015	5	0
Small tree multi-story post disturbance	Shrubs post disturbance	Non lethal wildfire	0.0012	5	0
Small tree multi-story post disturbance	Small tree closed cover multi-story	Understory development	0.0100	0	20
Small tree closed cover single story	Shrubs post disturbance	Spruce budworm outbreak	0.0100	0	0
Small tree single story post disturbance	Seedling/sapling post disturbance	Natural regeneration, mid-seral species	0.0038	0	0

Table 2b—A sample of probabilistic transitions of the mesic (cool-moist) forest STM being used for the Okanogan-Wenatchee National Forest plan revision. Probabilistic transitions between states result from tree growth, natural disturbances, and anthropogenic management actions (continued)

From State Class[a]	To State Class[a]	Transition Type[b]	Annual Probabilistic	Age Shift (years)[c]	Time Since Transitions (years)[d]
Seedling/Sapling medium cover	Grass/Forb	Stand replacement wildfire	0.0020	0	0
Seedling/Sapling post disturbance	Seedling/Sapling post disturbance	Non-lethal wildfire	0.0015	1	0

[a] All states of this mesic forest model have a Douglas-fir/mixed tree cover type.
[b] The mesic forest model contains 207 transitions between 34 vegetation states. Only a sample of natural disturbance probabilities is displayed here. Anthropogenic management actions not shown here include various intensities of harvest, thinning, salvage, tree planting, and prescribed fire (see table 3).
[c] For transitions that accelerate forest development, "Age shift" refers to the number of years added to the age of the state class as a result of the transition.
[d] For transitions that are dependent on a certain number of years without prior disturbance, "Time since transition" refers to the number of years that a state must be "disturbance free" for the transition to occur.

Table 3—Natural and anthropogenic types of probabilistic transitions used in Forest Planning in the USDA Forest Service Pacific Northwest Region

Natural transition types	Anthropogenic transition types
Wildland fire (non-lethal, mixed severity, and stand replacement)	Prescribed fire (non lethal, mixed severity, and stand replacement)
Dwarf mistletoe	Partial harvest salvage
Canopy growth	Pre-commercial thin
Western pine beetle	Selection harvest
Mountain pine beetle	Group selection harvest
Spruce budworm	Regeneration harvest
Douglas-fir beetle	Salvage
High severity wind	Planting
Understory development	Livestock grazing
Natural regeneration	Partial harvest (15 types, depending on current tree size and density)
Alternative successional pathway	
Root disease	

Forest Service forest inventory and monitoring plots[6] into vegetation classes (i.e., cover type, size class, canopy cover, canopy layers) for subsequent FVS projection of the plot data into the future. The Preside program calculates the average time plots from a particular vegetation state stay in that state and the probability of movement to other model states.

The general sequence of steps being used to integrate FVS projections into STMs is:

1. Prepare the inventory data for projection by FVS.
2. Adjust FVS default parameters for growth, mortality, and regeneration for each PVT model.
3. Develop natural growth projections to estimate rates of forest succession.
4. Process FVS output through the Preside program and accumulate the results into a matrix summarizing mean residence times within states and transition probabilities between states.
5. Compare empirically derived transition rates (from FVS) to STM parameters and adjust the STM where necessary.

[6] More information on forest inventory and analysis data for the USFS PNW region is available at: http://fia.fs.fed.us/library/fact-sheets/default.asp.

Table 4—Inventory plot distribution by potential vegetation type for FVS calibration of STM models used in the Okanogan-Wenatchee forest plan revision

		Number of inventory plots at start of FVS simulation by tree size class						
Potentia vegetation type	Total number of inventory plots used in FVS simulations	Grass/ forb (<1 in.)	Young (1-5 in.)	Pole (5-10 in.)	Small (10-15 in.)	Medium (15-20 in.)	Large (20-30 in.)	Giant (>30 in.)
Dry forest	1,467	169	5	105	548	504	132	4
Mesic forest	911	24	13	92	368	287	120	7
Cold-moist forest	207	4	4	29	69	54	37	10
Cold-dry forest	622	45	8	144	309	80	36	0
Alpine forest	71	9	0	14	26	22	0	0
Total plots	3,278	251	30	384	1,320	947	325	21

Table 4 summarizes the number of inventory plots used by tree size class to calibrate STMs for the Okanogan-Wenatchee forest plan revision. Preside analyses showed only a few discrepancies in growth transition rates for some vegetation states between FVS and the ILAP models. For example, in some cases Preside showed longer residence times for dense multi-story stands than related parameters in the ILAP models so growth rates for these states were consequently adjusted.

While not yet accomplished for the Okanogan-Wenatchee forest plan revision, FVS can also be used to validate or estimate transition pathways and/or probabilities for management, insects and disease, fire, or other natural or human-caused disturbances. It can also be used to report model attributes such as woody biomass volume or smoke emissions from fire. Additionally, climatic effects have recently been integrated into FVS to produce a new model called Climate–FVS, which provides a tool to allow climate change impacts to be incorporated in forest and project plans (Crookston et al. 2010).

Modeling Alternative Management Scenarios and their Environmental Consequences

Forest plan alternatives are developed in-part by evaluating and current ecosystem and socioeconomic conditions, and designing alternative suites of management actions to achieve desired conditions. State-and-transition models are the primary framework used in the PNW Region for designing and testing Forest planning alternatives and analyzing some of their environmental effects.

For the Okanogan-Wenatchee National Forest plan revision, desired conditions are derived primarily from information on historical ranges of variation in vegetation structure (Hessburg et al. 1999; Landres et al. 1999). Hessburg et al. (1999) used sample-based aerial photointerpretation on the eastern slope of the Cascade Mountains in Washington State to build spatially continuous historical (1938–1956) vegetation maps for 48 randomly selected sub-watersheds. These data were used to build desired conditions used in the forest plan revision. While attempts to strictly recreate conditions of the past are often not desirable or feasible (e.g., due to climate change, non-native species invasion, soil erosion, social intolerance of fire frequencies at levels representative of Native American burning (Kay 2007), and future climatic changes may in-time reduce the relevancy of historical references, the HRV remains an objective reference for at least the short-term management of natural resources (Keane et al. 2009). The Okanogan-Wenatchee Forest Plan revision team is currently comparing likely climate change projections and “future ranges of variation” in ecosystem structure and function against HRV to improve forest plan development and implementation.

Objectives for vegetation management in the draft Okanogan-Wenatchee proposed plan were developed by comparing current and desired vegetation and fire regime conditions, and testing alternative suites of management practices (e.g., tree harvest, forest thinning, prescribed fire) to support a variety of resource objectives ranging from creating diverse wildlife habitats to maintaining scenic values while providing forest products. For example, relative to desired conditions, dry mixed conifer forests are generally deficit in stand initiation, stem exclusion/closed canopy, and old forest single story structures (Hessburg et al., 1999). Overall, proposed plan direction is intended to make ecosystems more resilient to disturbance driven by climate change, reduce impacts of insects and diseases, and produce quality forest commodities.

The acceptability of these alternative management scenarios are also based on their feasibility within current budget assumptions and ability to ensure public safety. Development of these alternative suites of management practices—scenarios—begins with a STM that is parameterized with current annual disease, fire, and weather transition probabilities. Alternative sets of transition targets and transition multipliers are then iteratively used to test options—which integrate natural processes and management activities—for moving the forested landscape from current (initial) conditions toward desired goals. During scenario modeling, 10 to 30 Monte Carlo simulations are run to capture stochastic variability.

Comparisons of the environmental consequences of forest plan alternatives help support the selection of the final forest plan. For example, figure 5 illustrates the outcomes of two possible management scenarios for dry mixed conifer forest lands suitable for timber production on the Okanogan-Wenatchee National Forest. The "No Management" scenario excludes all management actions except fire suppression and allows all other natural disturbances to operate freely. The "Restoration" scenario represents one possible suite of regenerating stand densities harvest, thinning, and prescribed fire actions designed to move dry forest toward desired conditions. Forest harvest and thinning at a rate of 1-2 percent of timber suitable lands represents a level of activity consistent with historical budgets. As figure 5 demonstrates, over the 10-15 year planning horizon this restoration scenario moves dry mixed-conifer forests toward desired conditions (e.g., greater amounts of old forest single story; less dense, multistory conditions).

Table 5—Common monitoring indicators calculated from STM model output that are used in forest planning in the USDA Forest Service Pacific Northwest Region

Stand replacement fire hazard	Wildlife habitat quality
Fire regime condition class	Timber volume
Wildlife habitat abundance	Biomass
Vegetation density class	Revenue
Successional stage	Cost
Single versus multi-layered	Smoke production
Forest structural groups	

However, this scenario also reveals that changes in management approach across large landscapes may take decades to effectively change ecosystem structure and composition due to the length of time needed for forest development, and constraints that limit the extent or intensity of management actions (e.g., budget, air quality regulations). Some of the environmental consequences of alternatives can also be estimated by calculating indicators that are a function of the model's predicted area over time for state classes and transitions. Table 5 displays some common indicators used in forest planning in the PNW Region. Attributes such as forest structural groups, suitable wildlife habitat, or biomass volume can be assigned to one or more state classes, or calculated from model outputs.

Conclusions

The USFS PNW Region has extensive experience applying STMs and has an expanding model library. These models represent the integration of the best available science, albeit from a variety of sources including published research, peer-reviewed literature, unpublished papers, and expert judgment. Scenario planning in general, and STMs specifically often incorporate a variety of quantitative and qualitative information and consideration of this diverse information in a systemic way frequently leads to better decisions (Peterson et al. 2003). On the other hand, no matter what the underlying data source, models carry with them a certain degree of error and uncertainty. Sources of error in STMs include, but are not limited to:

- data or knowledge gaps
- omitting states not currently present but which could occur on the landscape

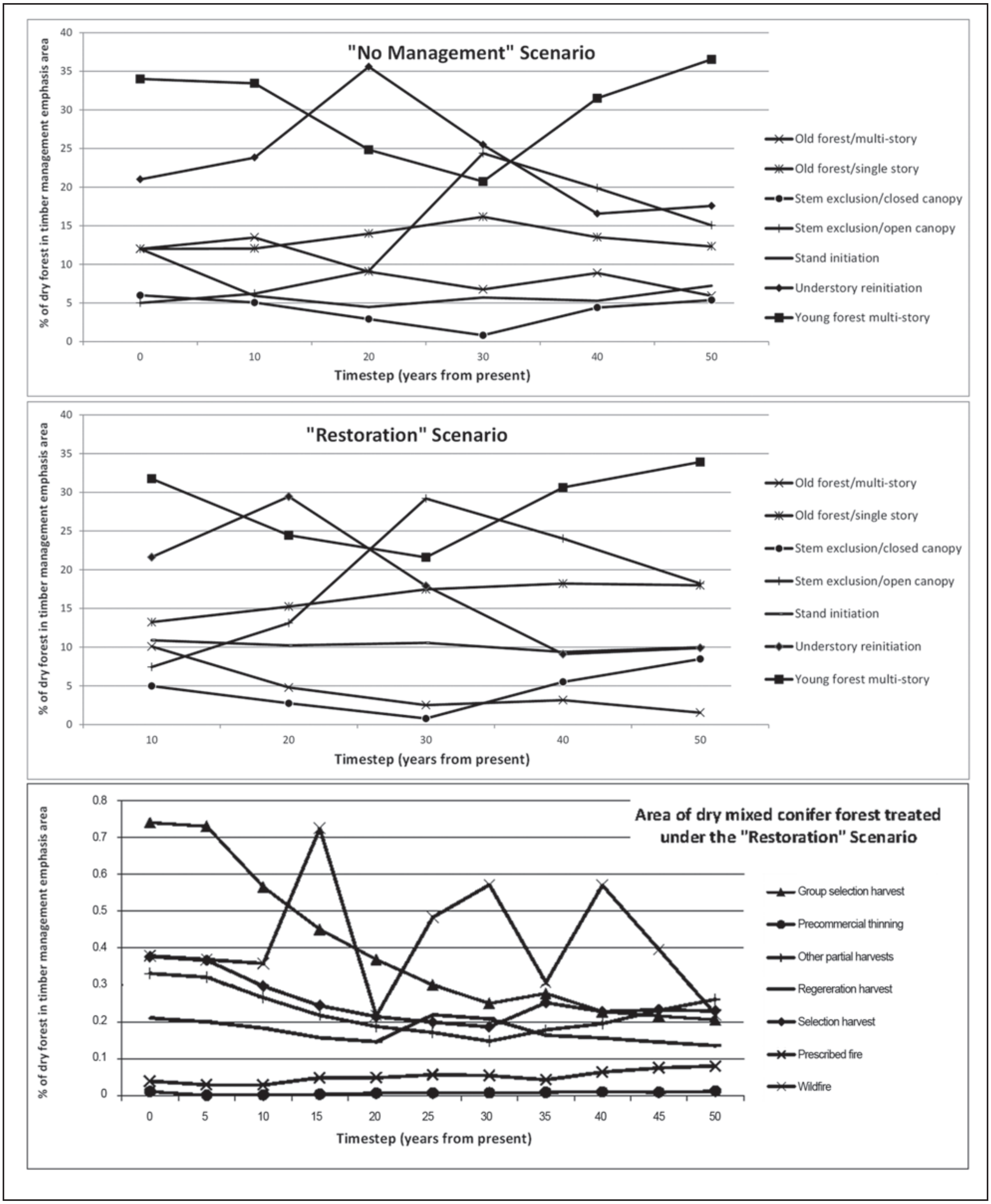


Figure 5—Outcomes of 50-year STM simulations of two potential management scenarios for dry mixed conifer forest on lands suitable for timber production on the Okanogan-Wenatchee National Forest. Also shown is a graph of treatments used in the "restoration" scenario.

- application of transition probabilities from similar, but not identical processes or systems
- inappropriately broad application of data from localized sources
- using mean probability values for processes with wide or bimodal variability
- failure to incorporate temporal variation in disturbance processes
- failure to capture spatially controlled processes
- failure to capture the effects of past land use on ecosystem structure and function

While the use of expert knowledge can introduce additional sources of model uncertainty, published literature alone rarely provides the detailed, site-specific information necessary to fully parameterize STMs. When empirical data is available, it is usually limited in extent and rarely includes an analysis of appropriate application scales or spatial heterogeneity (Bestelmeyer et al. 2011). Experts and practitioners are often the only source of information about ecosystem structure, function, and dynamics, especially at local scales (Drescher et al., 2008).

Local resource management experts, including forest, fire, and insect and pathogen ecologists, have qualitatively reviewed STM parameters and modeled outcomes for the Okanogan-Wenatchee forest plan revision. These experts and managers generally concur that the models are fair representations of the landscape-level forest structure, composition and function for the purposes of broad forest planning. However, non-spatial STMs generally simplify spatial heterogeneity, particularly at resolutions finer than the analysis area (Bestelmeyer et al. 2011), and these limitations should be considered during the planning process and subsequent implementation. For example, riparian systems are generally under-represented in the suite of STMs currently available for national forest planning, and as such are often assumed to behave similarly to adjacent uplands during broad landscape analyses. Development of models for unique ecosystems, spatial modeling of systems and processes dependent on spatial constraints, and continual integration of lessons learned from forest plan and climate monitoring can be used to improve the accuracy of STM parameters through time.

State-and-transition models are being used successfully to integrate resources for developing forest plan alternatives and analyzing the relative effects of those alternatives. These models can also be used to test assumptions within the complex matrix of climate change, increased forest densities, extended drought, uncharacteristic insect epidemics and fires, expansion of residential development into wild lands, and uncertainty in our understanding of ecosystem structure and function. The USFS PNW region's library of STMs provide an opportunity to link landscape and forest-level analyses to broadscale analyses, while also establishing a framework for adaptive feedback between levels of analysis. This capacity is particularly important relative to the analysis of regional issues, such as conservation of old forest dependent species, effects of road density on habitat integrity, commodity production, and effectiveness of alternative ecosystem restoration strategies. Broader applications of STMs by other landowners, such as other federal agencies and state departments of natural resources, and development of consistent models across large geographic extents (such as ILAP's suite of STMs for the PNW and Southwestern regions) are creating opportunities for integration of science and planning across larger geographic extents. Greater collaboration can help integrate multiple sources of scientific information, and improve our collective ability to effectively manage natural resources.

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References

Apex Resource Management Solutions Ltd. and ESSA Technologies Ltd. 2012. Path Landscape Model version 4.0.10. [Documentation and software download are available at http://www.apexrms.com/path].

Bestelmeyer, B.; Goolsby, D.P.; Archer, S.R. 2011. Spatial perspectives in state-and-transition models: a missing link to land management? Journal of Applied Ecology. 48(3): 746–757.

Climate Impacts Group (CIG). 2009. The Washington Climate Change Impacts Assessment, M. McGuire et al. Center for Science in the Earth System, Joint Institute for the Study of the Atmosphere and Oceans, University of Washington, Seattle, Washington. http://www.cses.washington.edu/db/pdf/wacciareport681.pdf. (December 12, 2011).

Crookston, N.L.; Rehfeldt, G.E.; Dixon, G.E.; Weiskittel, A.R. 2010. Addressing climate change in the forest vegetation simulator to assess impacts on landscape forest dynamics. Forest Ecology and Management. 260(2010): 1198–1211.

Dixon, G.E., comp. 2002. Essential FVS: A user's guide to the Forest Vegetation Simulator. Internal report. (Revised: February 3, 2010) Fort Collins, CO: U.S. Department of Agriculture, Forest Service, Forest Management Service Center. 240 p.

Drescher, M.; Perera, A.; Buse, L.; Ride, K.; Vasiliauskas, S. 2008. Uncertainty in expert knowledge of forest succession: a case study from boreal Ontario. The Forestry Chronicle. 84: 194–209.

Eidenshink, J.; Schwind, B.; Brewer, K.; Zhu, Z.; Quayle, B.; Howard, S. 2007. A project for monitoring trends in burn severity. Fire Ecology. Special Issue Vol. 3(1): 3–21.

ESSA Technologies Ltd. 2007. Vegetation Dynamics Development Tool User Guide, Version 6.0. Prepared by ESSA Technologies Ltd., Vancouver, BC. 196 p.

Hemstrom, M.A.; Salwasser, J.; Halofsky, J.; Kagan, J.; Comfort, C. 2012. The Integrated Landscape Assessment Project. In: Kerns, B.K.; Shlisky, A.J.; Daniel, C.J., tech. eds. Proceedings of the First Landscape State-and-Transition Simulation Modeling Conference, June 14–16, 2011, Portland, Oregon. Gen. Tech. Rep. PNW-GTR-869. Portland, OR: U.S. Department of Agriculture, Forest Service, Pacific Northwest Research Station: 57–70.

Henderson, J.A.; Lesher, R.D.; Peter, D.H.; Ringo, V. 2011. A landscape model for predicting potential natural vegetation of the Olympic Peninsula USA using boundary equations and newly developed environmental variables. Portland OR: U.S. Department of Agriculture, Forest Service, Pacific Nortwest Research Station Gen. Tech. Rep. PNW-GTR-841. 41 p.

Hessburg, P.F.; Smith, B.G.; Kreiter, S.D.; Miller, C.A.; Salter, R.B.; McNicoll, C.H.; Hann, W.J. 1999. Historical and current forest and range landscapesin the interior Columbia River Basin and portions of the Klamath and Great Basins. Part I: Linking vegetation patterns and landscape vulnerability to potential insect and pathogen disturbances. Gen.Tech. Rep. PNW-GTR-458. Portland, OR: U.S. Department of Agriculture, Forest Service, Pacific Northwest Research Station. 357 p. (Quigley, Thomas, M., ed., Interior Columbia Basin EcosystemManagement Project: scientific assessment).

Johnson, K.N.; Stuart, T.W.; Crim, S.A. 1986. FORPLAN Version 2: an overview. Washington DC: USDA, Forest Service, Land Management Planning Systems Section.

Kay, C.E. 2007. Are lightning fires unnatural? A comparison of aboriginal and lightning ignition rates in the United States, Masters, R.E.; Galley, K.E.M., eds. Proceedings of the 23rd Tall Timbers Fire Ecology Conference: fire in the grassland and shrubland ecosystems. Tallahassee, FL: Tall Timbers Research Station: 16–28.

Keane, R.E.; Hessburgb, P.F.; Landres, P.B.; Swanson, F.J. 2009. The use of historical range and variability (HRV) in landscape management. Forest Ecology and Management. Vol 258: 1025–1037.

Kent, B.; Bare, B.B.; Field, R.C.; Bradley, G.A. 1991. Natural Resource Land Management Planning using Large-Scale Linear Programs: the USDA Forest Service experience with FORPLAN. Operations Research. 39: 13–27.

Kurz, W.A.; Beukema, S.J.; Klenner, W.; Greenough, J.A.; Robinson, D.C.E.; Sharper, A.D.; Webb, T.M. 2000. TELSA: the Tool for Exploratory Landscape Scenario Analyses. Computers and Electronics in Agriculture. 27: 227–242.

Landres, P.B.; Morgan, P.; Swanson, F.J. 1999. Overview and use of natural variability concepts in managing ecological systems. Ecological Applications Vol 9: 1179–1188.

Ohmann, J.L.; Gregory, M.J. 2002. Predictive mapping of forest composition and structure with direct gradient analysis and nearest neighbor imputation in coastal Oregon, U.S.A. Canadian Journal of Forest Research. 32(4): 725–741.

Peterson, G.; Cumming, G., et al., 2003. Scenario planning: a tool for conservation in an uncertain world. Conservation Biology 17, 358–366.

Suring, L.H., Gaines, W.L.; Wales, B.C.; Mellen-McLean, K.; Begley, J.S.; Mohoric, S. 2011. Maintaining populations of terrestrial wildlife through land management planning: a case study. Journal of Wildlife Management. 75(4): 945–958.

U.S. Department of Agriculture, Forest Service [USDA FS]. 1995. Spectrum User Guide, Version 1.0. Prepared by Washington Office Ecosystem Management Analysis Center, Fort Collins, Colorado. 324 p.

U.S. Department of Agriculture, Forest Service [USDA FS]. 2007. USDA Forest Service Strategic Plan FY 2007–2012. U.S. Department of Agriculture, Forest Service. FS-880. July 2007. 38 p. http://www.fs.fed.us/publications/strategic/fs-sp-fy07-12.pdf. (March 20, 2012).

USDA Office of the General Counsel (OGC). 2002. Overview of forest planning and project level decision-making. USDA OGC Natural Resources Division. June 2002. 40pp. [Accessed online on July 7, 2011: http://www.fs.fed.us/emc/nfma/includes/overview.pdf].

Vandendriesche, D.A. 2009. Advanced FVS Topics for Forest Planning. Preside: Process Residence Times. Interoffice publication. Fort Collins, CO: U.S. Department of Agriculture, Forest Service, Forest Management Service Center. 129 p.

Wisdom, M.J.; Holthausen, R.S.; Wales, B.C.; Hargis, C.D.; Saab, V.A.; Lee, D.C.; Hann, W.J.; Rich, T.D.; Rowland, M.M.; Murphy, W.J.; Eames, M.R. 2000. Source habitats for terrestrial vertebrates of focus in the interior Columbia basin: broad-scale trends and management implications. Gen. Tech. Rep. PNW-GTR-485. Portland, OR: U.S. Department of Agriculture, Forest Service, Pacific Northwest Research Station.

Modeling on the Grand Scale: LANDFIRE Lessons Learned

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Abstract

Between 2004 and 2009, the LANDFIRE project facilitated the creation of approximately 1,200 unique state-and-transition models (STMs) for all major ecosystems in the United States. The primary goal of the modeling effort was to create a consistent and comprehensive set of STMs describing reference conditions and to inform the mapping of a subset of LANDFIRE's spatial products. STMs were created by more than 700 experts through a series of modeling workshops, individual meetings and web conferences hosted around the country. While model- building speed, efficiency and consistency may have been enhanced by using a small group of project employees to develop STMs, our participatory approach to model development encouraged early engagement in the LANDFIRE project as a whole, helped to incorporate a broad spectrum of knowledge into the STMs and built modeling capacity. The depth and breadth of the LANDFIRE modeling effort provides an opportunity to learn about expert-based modeling efforts. In this paper we reflect on that effort and, based on our collective experience facilitating the development of the LANDFIRE STMs, we offer 10 lessons learned: (1) create a flexible modeling process, (2) incorporate a learn-by-doing method, but know that it takes work, (3) engage a broad spectrum of experts from the start, (4) agree on what is being modeled, (5) implement procedures to maintain quality control, (6) if possible, build from existing models, (7) thoroughly document results, (8) never forget the modeling purpose, (9) set realistic modeling goals, and (10) model to document known ecological information and identify gaps in understanding. In this paper, we discuss these lessons in detail and offer observations and examples from our experience to help others efficiently build more useful models for land management and planning efforts in the future.

Keywords: pre-settlement, vegetation ecology, vegetation dynamics, state-and-transition model, LANDFIRE, experts, VDDT, Vegetation Condition Class.

Introduction and Background

Between 2004 and 2009, the Landscape Fire and Resource Management Planning Tools Project (LANDFIRE; http://www.landfire.gov) developed state-and-transition models (STMs) for all major Ecological Systems (Comer et al. 2003) in the United States through an expert-based model development process (Rollins 2009). LANDFIRE (now the LANDFIRE Program) is a shared program between the U.S. Department of Agriculture Forest Service and the U.S. Department of the Interior that is chartered to develop a suite of more than 20 vegetation, fire and fuel related products (table 1) that support fire and land management activities at regional and national levels. The datasets were created using consistent methods and cover all lands, public and private, in the United States (Rollins 2009).

LANDFIRE developed STMs to estimate pre-settlement reference conditions and to inform the mapping of a subset of its spatial products (table 1). Pre-settlement reference conditions as applied in LANDFIRE refer to the estimated percent of the landscape within given seral stages for

Table 1—LANDFIRE created and is continually updating its suite of more than 20 related fuel, vegetation and fire regime products. The STMs (called Vegetation Dynamics Models) are used directly and indirectly to create a subset of the spatial products

Fuel products	Vegetation products	Fire regime products
13 fire behavior fuel models[a]	Existing vegetation type[a]	Fire regime groups[b]
40 fire behavior fuel models[a]	Existing vegetation cover	Mean fire return interval[b]
Canadian forest fire danger rating system	Existing vegetation height	Percent low-severity fire[b]
Fuel characteristic classification system fuelbeds	Biophysical settings[a]	Percent mixed-severity fire[b]
Fuel loading models	Vegetation dynamics models	Percent replacement-severity fire[b]
Forest canopy cover	Environmental site potential	Vegetation condition class[b,c]
Forest canopy height		Vegetation departure[b,d]
Forest canopy bulk density		Succession classes[e]
Forest canopy base height		

[a] Products that were developed using STMs and associated description documents as an ancillary data source.
[b] Products that were generated directly by STMs in all versions of LANDFIRE except LANDFIRE National where they were used as inputs to the LANDSUM model (Keane et al. 2006) which generated these products.
[c] Vegetation Condition Class was formerly called Fire Regime Condition Class.
[d] Vegetation Departure was formerly called Fire Regime Condition Class Departure Index.
[e] Product that was generated using rule sets in the STM description document.

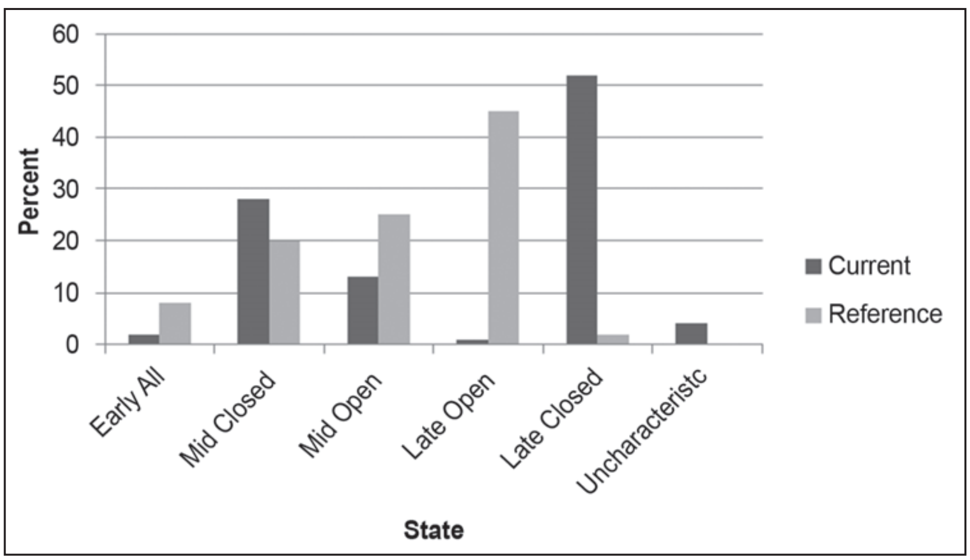


Figure 1—The primary use of STMs by LANDFIRE was to compare reference and current conditions to calculate Vegetation Condition Class. This example compares reference conditions estimated from the Ozark-Ouachita Dry Oak Woodland STM (LANDFIRE 2012a) to current conditions (LANDFIRE 2012b) in Map Zone 44—Ozark and Ouachita Mountains.

an ecosystem that would have occurred prior to European settlement. The reference period included both the influence of Native Americans (e.g., use of fire) throughout much of the continental U.S. and the influence of Polynesian settlers (e.g., agriculture) in the Hawaiian Islands. The primary use of the STM generated reference conditions by LANDFIRE was to calculate Vegetation Condition Class (formerly referred to as Fire Regime Condition Class or FRCC; Barrett et al. 2010), a metric which quantifies the difference in vegetation cover, height and type between reference and current conditions (fig. 1). The model documentation and model outputs for the reference scenario were also used directly to provide mapping rule sets for developing the Fire Regime Group, Succession Class, Fire Frequency and Fire Severity spatial layers and as an ancillary data source for mapping Biophysical Settings, Existing Vegetation Type and Fire Behavior Fuel Models (Rollins 2009; table 1).

The primary objective of the LANDFIRE modeling effort was to create a consistent and comprehensive set of STMs describing reference conditions for every ecosystem mapped by LANDFIRE. In addition, we wanted to:

- create a STM library as a foundation for future modeling efforts,
- develop a sense of buy-in and ownership by the community of potential LANDFIRE data users,
- train participants in the concepts and applications of STMs, and
- provide a forum for scientific and land management networking.

These objectives guided the model development process and modeling rules. The process we implemented served to market the LANDFIRE project as a whole, taking it from a top, own effort that delivered maps and models built by a small team of project employees to a participatory effort where user input was incorporated directly to build a subset of the products. While the former approach would probably have led to greater consistency in the STMs, it would have likely compromised training, outreach and networking objectives.

Each LANDFIRE STM represents a single ecosystem called a Biophysical Setting (BpS). A BpS is a vegetation concept mapped by LANDFIRE, based on the Ecological Systems classification (Comer et al. 2003), which represents the potential vegetation community that could exist on the landscape given the current biophysical environment (e.g. soils and precipitation) and an approximation of the historical disturbance regime (e.g. fire return interval and flooding frequency). A LANDFIRE STM consists of two related parts (fig. 2):

1. a quantitative state-and-transition model developed with the Vegetation Dynamics Development Tool (VDDT; ESSA Technologies Ltd. 2007) and
2. a description document developed in the Model Tracker Database (MTDB).

VDDT was used to attribute each state within a BpS with an age range and probabilities for deterministic (i.e. succession) and probabilistic (i.e. disturbance) transitions. VDDT was then run for 1,000 years to estimate reference conditions (i.e. the percent of the landscape in each state) and the frequency of fire and other disturbances. VDDT was chosen as the modeling platform by LANDFIRE because it is in the public domain, relatively user-friendly, compatible with related spatial models and capable of running multiple iterations quickly. VDDT is also supported by some federal and state agencies as a land management planning tool.

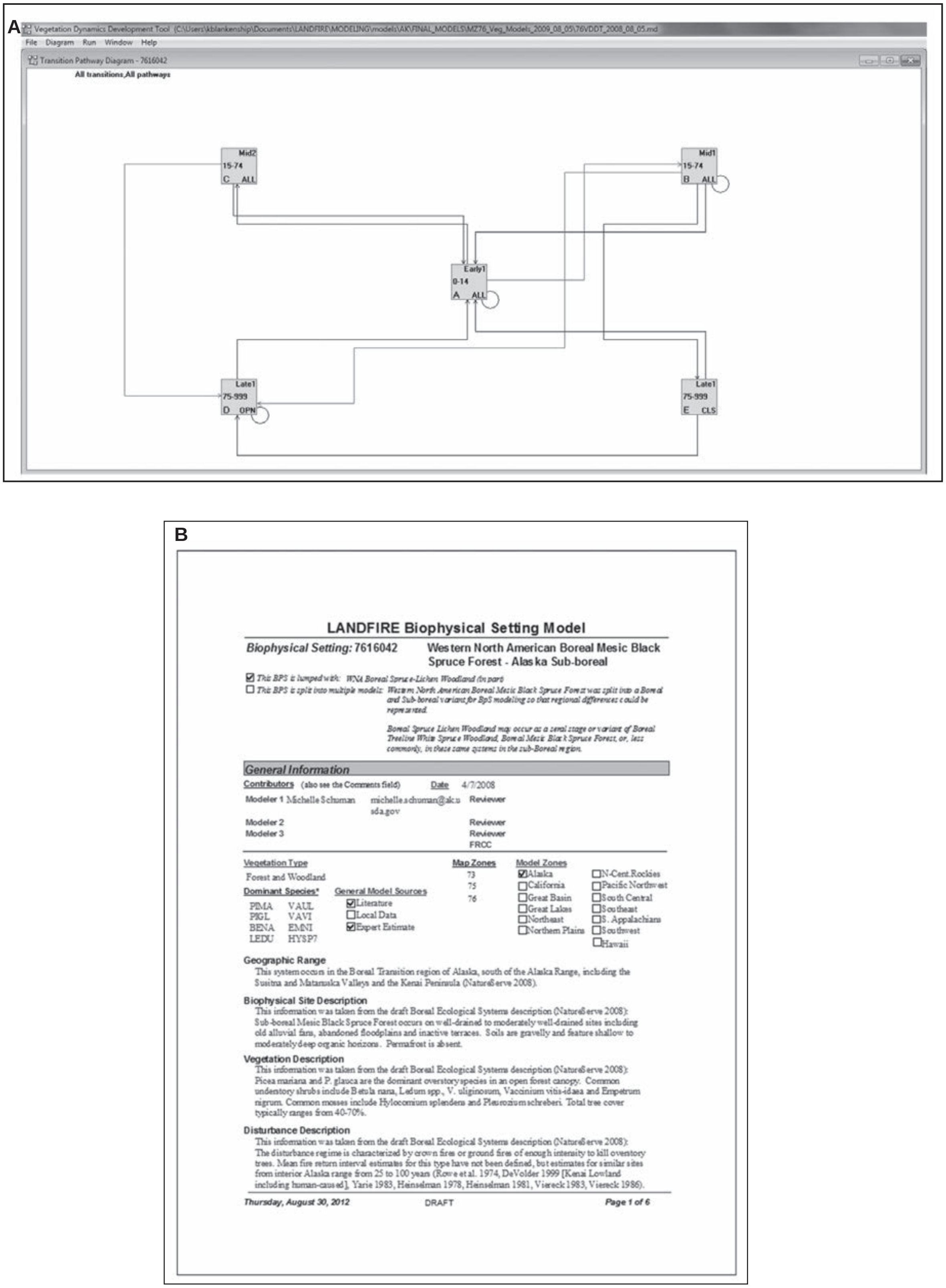
MTDB is a Microsoft Access database designed by LANDFIRE to document model development (fig. 3). The database was used by modelers to record:

- a description of the modeled ecosystem including geographic range, biophysical setting, disturbance regime, vegetation characteristics and dominant species,
- mapping rules for each state (called s-class or succession class by LANDFIRE) in the STM,
- STM results including the estimated reference condition (i.e. the percent of ecosystem in the various states) and the fire frequency and severity, and
- relevant literature, model contributors, model reviewers and modeling assumptions.

A report was generated from the MTDB which became the description document (metadata) that accompanies each STM.

Modeling rules were established to ensure that the models would be consistent and comparable across the country and could be used to develop map products. For example, LANDFIRE models are consistent in resolution (they have five or fewer states), capture the main successional pathway without gaps or overlap in age and use a pre-defined subset of VDDT functions including:

- a standardized set of definitions for cover types, structural stages, transition types and transition groups,



LANDFIRE Biophysical Setting Model

Biophysical Setting: 7616042 — Western North American Boreal Mesic Black Spruce Forest - Alaska Sub-boreal

☑ *This BPS is lumped with:* *WNA Boreal Spruce-Lichen Woodland (in part)*

☐ *This BPS is split into multiple models:* *Western North American Boreal Mesic Black Spruce Forest was split into a Boreal and Sub-boreal variant for BpS modeling so that regional differences could be represented.*

Boreal Spruce Lichen Woodland may occur as a seral stage or variant of Boreal Treeline White Spruce Woodland, Boreal Mesic Black Spruce Forest, or, less commonly, in these same systems in the sub-Boreal region.

General Information

Contributors (also see the Comments field) — Date 4/7/2008

Modeler 1 Michelle Schuman michelle.schuman@ak.usda.gov — Reviewer

Modeler 2 — Reviewer

Modeler 3 — Reviewer

FRCC

Vegetation Type: Forest and Woodland

Dominant Species*: PIMA, VAUL, PIGL, VAVI, BENA, EMNI, LEDU, HYSP7

General Model Sources: ☑ Literature ☐ Local Data ☑ Expert Estimate

Map Zones: 73, 75, 76

Model Zones: ☑ Alaska ☐ California ☐ Great Basin ☐ Great Lakes ☐ Northeast ☐ Northern Plains ☐ N-Cent.Rockies ☐ Pacific Northwest ☐ South Central ☐ Southeast ☐ S. Appalachians ☐ Southwest ☐ Hawaii

Geographic Range

This system occurs in the Boreal Transition region of Alaska, south of the Alaska Range, including the Susitna and Matanuska Valleys and the Kenai Peninsula (NatureServe 2008).

Biophysical Site Description

This information was taken from the draft Boreal Ecological Systems description (NatureServe 2008): Sub-boreal Mesic Black Spruce Forest occurs on well-drained to moderately well-drained sites including old alluvial fans, abandoned floodplains and inactive terraces. Soils are gravelly and feature shallow to moderately deep organic horizons. Permafrost is absent.

Vegetation Description

This information was taken from the draft Boreal Ecological Systems description (NatureServe 2008): Picea mariana and P. glauca are the dominant overstory species in an open forest canopy. Common understory shrubs include Betula nana, Ledum spp., V. uliginosum, Vaccinium vitis-idaea and Empetrum nigrum. Common mosses include Hylocomium splendens and Pleurozium schreberi. Total tree cover typically ranges from 40-70%.

Disturbance Description

This information was taken from the draft Boreal Ecological Systems description (NatureServe 2008): The disturbance regime is characterized by crown fires or ground fires of enough intensity to kill overstory trees. Mean fire return interval estimates for this type have not been defined, but estimates for similar sites from interior Alaska range from 25 to 100 years (Rowe et al. 1974, DeVolder 1999 [Kenai Lowland including human-caused], Yarie 1983, Heinselman 1978, Heinselman 1981, Viereck 1983, Viereck 1986).

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Figure 2—A LANDFIRE model consists of a STM developed in VDDT (A) and a description document developed in the MTDB (B).

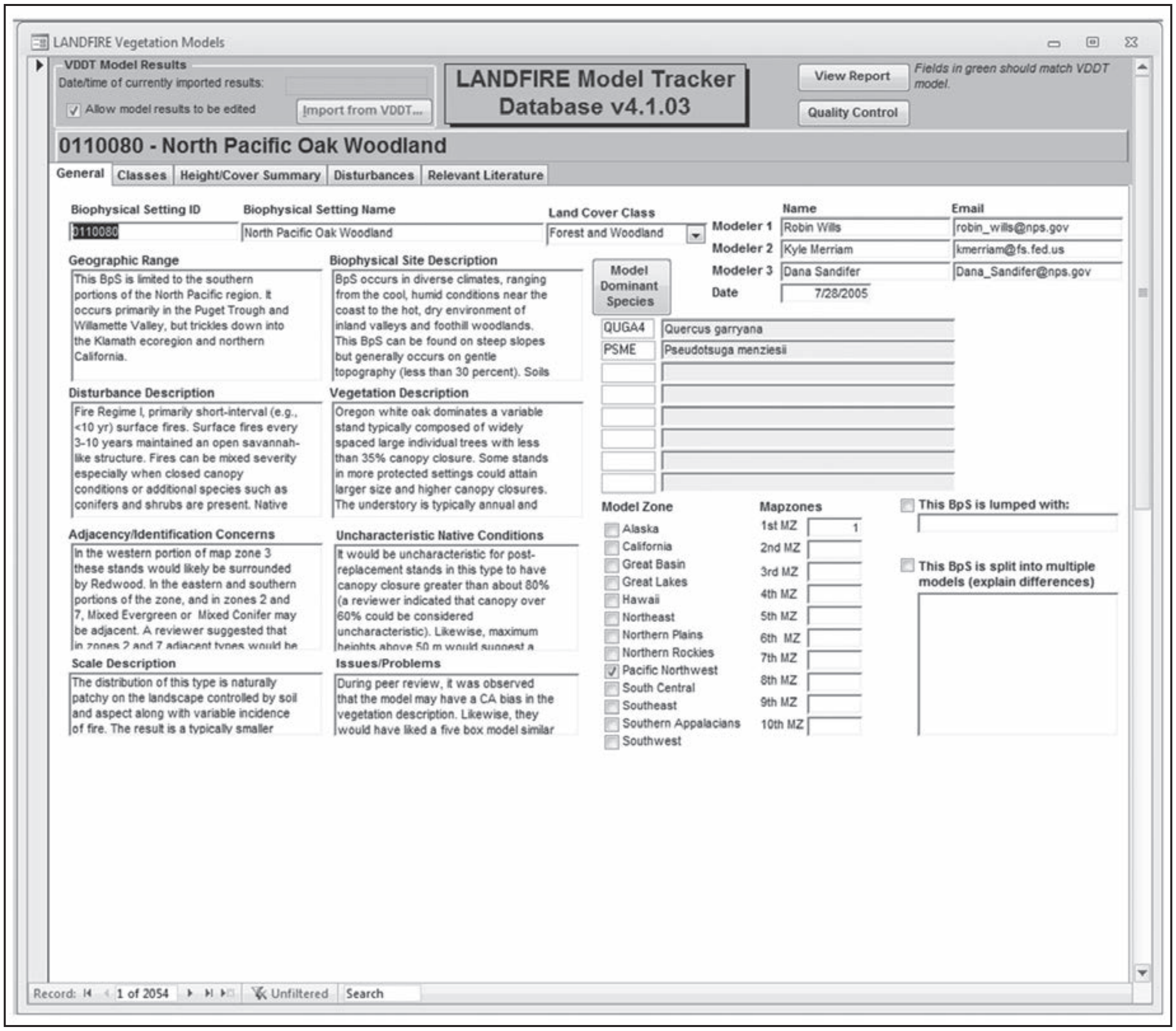

Figure 3—MTDB is Microsoft Access tool developed by LANDFIRE to document the modeling process.

- use of Time Since Disturbance (TSD) only with alternate succession pathways, and
- use of relative Age (RelAge) limited to replacement severity disturbances occurring in the initial state (normally state A in LANDFIRE).

These modeling rules created a number of weaknesses, including:

- the need to develop crosswalks between differing vegetation classifications,
- the potential loss of information available at resolutions finer than the project objectives, and
- constraint of expert modelers to a select suite of model functions, sometimes below their skill level.

Another limitation was imposed by the project schedule which limited the time available for STM development and review. During the National phase of the project, LANDFIRE developed, reviewed and revised seven unique STMs each week on average for 183 weeks.

The STMs were developed through a series of more than 40 expert workshops held around the country, some 35 web conferences and many more individual meetings. A modeling leader was designated for each of 13 geographic regions and provided funding for STM development activities in their area. LANDFIRE modeling leaders cast a wide net for experts in the fields of vegetation, fire and landscape ecology as well as land managers and stewards—in short, anyone with the training and/or experience necessary to

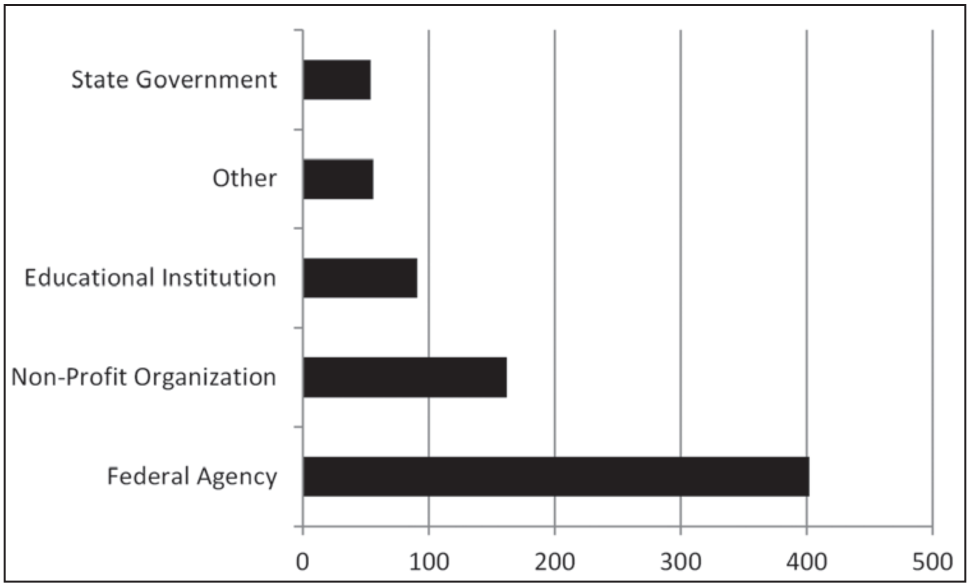

Figure 4—More than 700 individuals from the government, non-profit and private sectors contributed to the development of LANDFIRE STMs. The category "Other" refers to individuals who did not affiliate with any group or organization, such as independent contractors and consultants.

Table 2—LANDFIRE created two sets of STMs: 1) Rapid Assessment models which were coarser scale and covered the conterminous U.S. and 2) National models which were finer scale and covered the entire United State including Alaska and Hawaii. The relationship between types mapped and types modeled is one-to-many because in some cases multiple STMs were needed to represent the geographic variation in widespread vegetation types. Of the 2,426 total LANDFIRE STMs, about 1,200 are unique because in some cases one STM was used to represent the same BpS in different Map Zones

Project phase	Vegetation types mapped	Vegetation types modeled
LANDFIRE Rapid Assessment	242	262
LANDFIRE National	541	2,164
TOTAL	783	2,426

populate the ecological information in a STM, and who had supervisory support and funding for engagement in the project. Workshops were open to all interested individuals, and a modest amount of funds were available to support travel for a portion of participants. Model parameters were developed based on literature, local data and professional judgment. When disagreements occurred regarding model inputs, LANDFIRE modeling leaders consulted additional experts, reviewed the literature, performed sensitivity analysis and ultimately made the final decision about the parameters. Both the decision-making process and all the opinions were thoroughly documented in MTDB to ensure transparency. Written and verbal evaluations were solicited at each workshop to fuel to facilitate adaptive workshop planning.

During the Rapid Assessment phase of the project, 262 coarse-scale STMs representing 242 BpS units mapped in the conterminous U.S. were developed. These STMs were then refined during the LANDFIRE National phase of the project to create 2,164 STMs representing 541 mid-scale (ranging from 10–1000's of hectares in size) BpS units mapped in the U.S., including Alaska and Hawaii. In total, LANDFIRE engaged more than 700 experts from various sectors (fig. 4) to create 2,426 STMs (approximately 1,200 of which were unique) representing 783 vegetation units in the U.S. (table 2). Not all of the STMs were unique because in some cases, based one expert feedback, one STM was used to represent the same BpS in different Map Zones (National Landcover Database Map Zones). However, the associated documentation in MTDB may have been adjusted to better represent the geographic variation in plant species or environmental gradients so a unique record was maintained for each Map Zone even when quantitative information in the STM was not changed. In some cases, the opposite situation occurred—a given BpS had multiple STMs associated with it to represent the geographic variation across Map Zones with quantitative changes in the model. For example, the Inter-Mountain Basins Big Sagebrush Shrubland BpS occured in 20 Map Zones and had at least three distinct STMs associated with it to represent the variability in successional rates and disturbance probabilities throughout its extensive range.

In this paper we offer 10 lessons learned based on our collective experience facilitating the development of LANDFIRE's STM library. We discuss lessons related to all aspects of model building including developing a modeling process, eliciting expert input, defining modeling units, checking for model errors, documenting model results and setting appropriate expectations. These lessons may help others build more useful models for land management and planning in the future.

Lesson 1: Create a Flexible Modeling Process

Imagine embarking on a project to develop thousands of structurally-consistent, scientifically-sound vegetation STMs while also engaging hundreds of people with diverse knowledge, skills and personalities. Together these objectives necessitate a relatively large degree of flexibility in approach (e.g., to address a diversity of learning styles), while there may also exist many constraints on the modeling mechanics (e.g., to ensure each model is built at the appropriate scale of resolution). Being flexible in the modeling approach does not necessarily mean scientific quality will suffer; scientific quality may in fact be enhanced when the approach allows a greater diversity of experts to contribute their knowledge and skills. Flexibility in approach provides the wiggle-room necessary to work with individuals or organizations that have different styles or processes.

Goals for developing STMs for LANDFIRE included engagement of a large diversity of experts for the purposes of compiling the best available science on ecosystem structure and function, and capacity-building for the future application of completed models. Some modeling participants were interested in learning how to build and apply the STMs, as well as providing and/or compiling the best available science for translation into a STM format. Other participants were primarily focused on compiling the best available information and were not interested in being able to use the STMs themselves. Workshop participants also differed in learning styles. The "experiential learners" needed to run the models in a hands-on manner to understand how they worked; while others were "abstract learners" and could understand enough about the modeling process through lectures to meet the project objectives.

While LANDFIRE, in part, aimed to build applied modeling capacity in each modeling participant, the diversity of learning styles and participant motivations necessitated a flexible approach. For example, we learned that if a small group of experts was expected to build a STM, it must include at least one person willing to listen openly and patiently to others and run the model software while also incorporating their own expert knowledge in an unbiased manner. If a group of experts lacked the skills and desire to build a STM in VDDT, they had to be provided more abstract methods to document the best available science on model parameters (e.g., flipcharts or forms where experts could fill-in tables of transition probabilities, or draw box-and-arrow diagrams) so that the STM could be built later. If experiential learners were willing to build STMs, but no other experts were available to assist, they had to be comfortable working individually, or be provided one-on-one support throughout the model-building process. In general, based on written workshop evaluations by participants, most LANDFIRE modelers appreciated the in-person, facilitated workshop approach and the opportunity to interact with other experts. However, where time, travel budgets, modeling skills and/or a desire for increased modeling capacity was lacking, first iteration "straw man" models were built by LANDFIRE staff which could be reviewed individually by experts on their own time.

We suggest holding in-person workshops and one-on-one meetings whenever possible, using techniques that the modeling leaders and modelers are comfortable with. We used multiple techniques to build models often within the same workshop, including:

- facilitated modeling using flip charts and/or VDDT software,
- modeling individually or in small groups,
- modeling by pairing an expert with a VDDT "driver" who could run the software but did not necessarily understand the ecology, and
- LANDFIRE staff creating straw man models which were later critiqued by experts during or outside a workshop event.

Online web conferences can be effective when in-person meetings are not feasible.

Essentially, the need to build structurally consistent STMs using the best available science (see Lesson 5) does not preclude taking a flexible modeling approach. A flexible modeling approach facilitates engaging the broadest suite of learning styles and personal motivations possible.

Lesson 2: Incorporate a Learn-By-Doing Method, but Know That it Takes Work

Building STMs requires a general understanding of modeling concepts and specific knowledge of modeling tools such as VDDT. While these concepts and tools can be taught through a didactic approach, we found that an experiential learning approach, where users learned directly by doing, facilitated two of our project objectives: building many STMs in a short amount of time and building modeling capacity within our expert community. While this constructivist-guided approach is well documented (two publications by Jean Piaget, attributed as "father" of constructivism, have over 7,000 citations in Google Scholar), it requires (1) teamwork, (2) motivation, and (3) preparation on the part of the facilitator.

Team learning , such as building STMs in small groups as was done at most LANDFIRE workshops, has proven to be valuable in virtually every educational setting (Daniels and Walker 2001). Team situations provide opportunities for reflective observation (i.e., asking "why?"), and further, learning is often motivated by conflict (Kolb 1993). The teams in the LANDFIRE modeling effort were selected for expertise, not necessarily for agreement in learning styles, age or type of experience. In one example from a LANDFIRE modeling workshop in Michigan, a young college professor was paired with an older ecologist from The Nature Conservancy. The professor was very comfortable with both the modeling software and the ecosystem from literature review, whereas The Nature Conservancy ecologist was relatively uncomfortable with the software but had decades of field experience. The two experts often questioned each other—the ecologist questioning how the professor ran the model; the professor questioning the ecologist's field-based observations. The tension forced both modelers to alternate between the four modes of experimental learning: reflection, action, feeling and thinking (Daniels and Walker 2001). We did not test the experts, but both stayed engaged with LANDFIRE, built a nuanced and complete model and most importantly, commented that they "learned a lot" from the experience.

Addressing the built-in tensions between people, the challenges of quantifying ecosystem processes with substantial levels of uncertainty, high expectations and simply "being away from the office" required motivation. Motivation was both internal to the experts and created on site during workshops and meetings. The simple fact that experts prioritized their work to be involved often indicated that there was motivation and that the topic at hand had immediate relevance. However, some experts may have been directed to attend by a supervisor, for example. It is important that leaders do not assume adequate motivation among participants. In the LANDFIRE modeling process, motivation was developed through several means: (1) immediate engagement (e.g. minimizing lectures and moving quickly to hands-on modeling), (2) creation of a "safe" environment where risks of questioning and being questioned were kept to a minimum, (3) accountability based on STM review and (4) fun (see below).

Working with many people of varied backgrounds requires structure and preparation. Corroborating many of Vella's 12 fundamental principles of adult learning (Vella 1994), we found that for the processes to be effective there had to be physical comfort (plenty of food, quiet location, etc.), clear expectations and clear, but evolving roles. In LANDFIRE workshops, the facilitator took on the leadership role in establishing the aforementioned "motivational setting," but as the process matured the facilitator would often be replaced as the leader by experts. It was apparent to us that peer-to-peer learning and collaboration increased the value of the workshop approach over STMs being developed by individuals working independently.

Finally, it was helpful to throw in some fun whenever possible. For example, we used an acronym contest, where we learned that TNC, i.e. The Nature Conservancy, also can mean "Totally Non-Confrontational." Administering an "Are you a lumper or a splitter?" quiz to workshop participants not only brought laughter, but helped participants recognize their potential modeling strengths and weaknesses.

Lesson 3: Engage a Broad Spectrum of Experts from the Start

We believe that engaging a broad spectrum of experts in the development and review processes results in more robust and useful SMTs. Consider inviting individuals who will be critical to building future support for the use of STMs. Research has shown that experts are the greatest source of variation in the modeling process (Czembor 2011) but for many ecosystems expert knowledge is virtually the only information source available. If variation is inevitable, modeling leaders need to increase the sample size, that is, identify and engage as many experts as possible within time and resource constraints. LANDFIRE modelers included scientists, managers and resource specialists from all the major U.S. land management agencies (e.g., Forest Service and National Park Service), teachers and students from academic institutions and foresters, ecologists, botanists, managers and others from a variety of non-governmental organizations (e.g., The Nature Conservancy and NatureServe; fig. 4).

Once experts are involved, managing their input in constructive ways is the key to successful engagement. Through experience we developed several techniques for responding appropriately to issues encountered when working with a diverse group of experts (table 3). When modeling is complete, it is important to follow up with modeling participants to communicate project results and the importance of their efforts to the success of the project.

Lesson 4: Agree on What is Being Modeled

Defining *what* is being modeled and communicating that explicitly are essential to the modeling process. This includes both the vegetation concept (e.g., BpS) and the individual vegetation units to be modeled (e.g., Alaska Arctic Wet Sedge-Sphagnum Peatland). When explaining the vegetation concept, we found it helpful to discuss it within the context of various other potential vegetation classifications familiar to our experts such as Potential Natural Vegetation Type (e.g., Schmidt et al. 2002), Habitat Type (e.g., Daubenmire 1968, Pfister et al. 1977), Land Type Association (e.g., ECOMAP 1993) and others.

After the modeling concept is defined and understood, the individual modeling units themselves must be examined and modelers must come to agreement on the distinction between related and sometimes overlapping ecosystems. For example, LANDFIRE created STMs for seven California chaparral ecosystems: California Maritime Chaparral, California Mesic Chaparral, California Montane Woodland and Chaparral, California Xeric Serpentine Chaparral, Mediterranean California Mesic Serpentine Woodland and Chaparral, Northern and Central California Dry-Mesic Chaparral and Southern California Dry-Mesic Chaparral. While time consuming, examining similar ecosystems like those listed above before initiating modeling is essential to preventing confusion during model development and later model use.

Lesson 5: Implement Procedures to Maintain Model Quality

Maintaining model quality through standards, rules and error checking are critical when STMs are to be comparable across ecosystems and/or if they are to be used in other software programs (e.g., LANDSUM) or applications (e.g., mapping state classes). LANDFIRE developed modeling

Table 3—Successfully engaging experts and eliciting the information required to build STMs involves developing ways to constructively mange expert input and work with experts with diverse backgrounds and skills

Issue	Potential solution	LANDFIRE example
Working with "lumpers" and "splitters" (i.e., individuals who tend to focus on similarities and define fewer ecosystems vs. those who tend to focus on differences and define more ecosystems)	Illuminate individual tendencies to prevent either over-simplification or wasted time tracking down unnecessary information.	We gave a light-hearted "Are you a lumper or a splitter?" quiz to identify individual tendencies early on in the model development process.
Managing experts with agendas	Respectfully mange their input so as to separate agendas from science.	MTDB provided a place where all opinions could be documented and robust model review helped ensure that the best available science was incorporated into each STM.
Building modeling confidence and capacity	Make sure all participants feel valued and acknowledge that only some experts will be willing to learn modeling tools such as VDDT.	We paired experts who were comfortable modeling with those who were not, and/or provided other devices to record expert input such as flip charts or cheat sheets. We started by eliciting information within the expert's area of interest and worked towards less familiar information.
Limited budget for compensating experts	Payment may increase motivation and timeliness. Use web conferences instead of in-person meetings when budgets are limited.	We used limited financial support to engage key experts and ensure completion of all STMs, especially for rare ecosystems and ecosystems for which little research existed.

standards and rules to maintain the quality and consistency of its STMs and to ensure their compatibility with mapped products. Standards, such as modeling forested ecosystems with a standardized set of state classes including one early, two open and two closed states, create a sense of unity among the LANDFIRE models. While this standard was followed most, but not all, of the time, LANDFIRE had a set of rules that were applied to the entire model set such as using five or fewer states in a STM to create consistency in resolution and prohibition of the use of Monte Carlo multiplier files to capture temporal variation in disturbances. The VDDT software includes functionality for setting the temporal variation in disturbances, but it was not incorporated into the project design because of large gaps in data or knowledge about temporal variation of natural disturbances geographically and for particular disturbance types (see lesson eight).

An automated and a manual set of quality control checks were developed to ensure that LANDFIRE modeling rules were followed and errors were minimized. Keeping models as simple as possible (see lesson nine), makes finding and fixing errors easier. Modeling efforts like LANDFIRE should clearly communicate the benefits of standards or rules such that modelers and reviewers can understand how the rules may work in their favor in the long term.

Lesson 6: If Possible, Build from Existing Models

Starting with an existing STM and modifying it as needed to represent a new ecosystem can promote modeling efficiency and may help build modeling confidence among experts with little modeling experience. Working with a variety of experts we found that most preferred to modify an existing STM rather than start from scratch. This seemed to be particularly helpful for individuals who had no previous modeling experience or who had not used VDDT before—the case for most LANDFIRE modelers.

In addition to helping modelers get started, working from a similar, existing STM aided efficiency by allowing users to focus on the quantitative differences between ecosystems (see lesson seven). For ecosystems with little research from which to develop STM parameters, starting with a related ecosystem's STM and focusing on relative differences in those numbers was an effective strategy. The downside of starting with an existing STM is that, to reveal hidden bias, it becomes extremely important to question the existing model's assumptions within the context of the new ecosystem.

Lesson 7: Thoroughly Document Results

Documenting model results promotes later evaluation, understanding and application of the model set by identifying information sources, stating assumptions and identifying knowledge gaps. The use of LANDFIRE's MTDB as a place to document this information promoted transparency in the modeling process, which supported scientific confidence in the STMs. This was particularly important in cases where there was disagreement between experts on model parameters and/or where there was little research from which to glean succession and disturbance rates. In addition to its use internally, documentation makes models more transferable and readily useable by others. When a modeler starts with an existing model (see lesson 6), and understands its assumptions, documentation of new model parameters can be facilitated by merely editing existing documentation.

Lesson 8: Never Forget the Modeling Purpose

Modeling is generally undertaken to achieve a specific objective and this objective should help guide modeling decisions. The LANDFIRE project used its STMs primarily to estimate reference conditions and to assist with mapping vegetation and fuel spatial products. Keeping these goals in mind allowed us to focus on the required outputs and minimize issues that did not impact the results the project needed. For example, we found it was often difficult to quantify infrequent disturbances (with return intervals of 1,000 years or more such as severe insect outbreaks or weather events) without the use of Monte Carlo multipliers, a VDDT function not used in LANDFIRE STMs (see lesson five). Without the use of multipliers, disturbances with long return intervals occur in the model more frequently (because at every time step there is a probability of their occurrence) but at a lower intensity (i.e., the disturbance affects fewer pixels or landscape area) at any given time than would be expected by the real world event. For short duration simulations, the loss of variability could have a significant effect on the results but by running the STMs for a long time period (1,000 years) and, taking the average of the outputs for that period, the impact on the results needed by LANDFIRE was minimal so we could document our assumptions and move on without delay. However, this example illustrates the trade-offs between modeling rules set in place by the project for consistency and the ability to model some complex ecological phenomena. The modeling purpose impacts the modeling rules; it can help determine what to include and what to leave out of a model.

Lesson 9: Set Realistic Modeling Goals

As a matter of practicality and philosophy, we recommend keeping STMs as simple as they can be while still meeting the project goals. Philosophically, modelers must remember that every model is an intentional simplification of reality, and that it is the modeler's responsibility to decide how much simplification is appropriate to meet his or her objectives. Practically, modelers should remember that STMs must be parameterized and understood to be useful, and the more complex the STM, the more difficult both these tasks are. There are at least two levels of simplification that we recommend, what to model and how to model.

To decide *what* to model, identify those things that are important to your project. For instance, in LANDFIRE, the significance of fire and fire regime was paramount, although not exclusive. When the list of BpS to be modeled was defined by the experts involved, we asked them to "lump" and "split" modeling units intelligently based upon LANDFIRE's needs. If two vegetation systems were very similar ecologically and had very similar fire regimes, such as riparian types, the distinction between them was not critical to LANDFIRE, so we asked them to "lump" the two BpS into a single STM. If a BpS occurred in two variants that had significantly different fire regimes, such as Douglas-fir at different elevations or on different aspects, we asked the experts to "split" the system into two distinct STMs. Within the context of the modeling objectives, the simpler the model, the easier it is to maintain model quality.

There are many reasons to keep the content and structure of each STM as simple as possible, such as parameter specification, model over-specification and model exploration. A vegetation STM is composed of states, transitions and transition parameters (frequency and destination at minimum). A STM with five states and five transitions has at least 50 potential parameters to specify, and each transition has five possible destinations. Imagine a second STM with 10 states and 10 transitions. This STM has 100 or more potential parameters to specify, and each transition has 10 possible destinations that must be sorted out. Often the reference information or experience that is needed to specify all these model parameters is not substantial, and is spread very thin indeed for more complex models. It is also possible to over-specify a STM. Consider a system with two types of flooding disturbances: one has a return interval of 50 years and the other's is 500 years. It is highly likely that the second type of flooding disturbance may not add useful information to a STM being used in a 100-year planning process. Finally, it is much easier to understand and explore a simpler STM. By minimizing the number of states and transitions in a STM, errors are found and diagnosed more quickly, and the interpretation process is more thorough and efficient.

Lesson 10: Model to Document Known Ecological Information and Identify Gaps in Understanding

The process of quantitatively modeling every mid-scale ecosystem in the United States helped us identify the many gaps in our collective ecological understanding. We found that ecosystems with commercial value such as ponderosa pine and longleaf pine forests tend to have more research associated with them, allowing for more robust estimates of succession and disturbance rates. In contrast, noncommercial and/or rare ecosystems such as California chaparral or Great Lakes alvar (limestone plains with sparse vegetation) have comparatively less information from which to build quantitative models. Our efforts highlighted research needs in many ecosystems.

The modeling process was often as beneficial as the STM results. For example, the process allowed us to:

- document what is known about ecosystems,
- identify areas where information is lacking about ecosystems,
- test assumptions about ecosystem function,
- look at relative differences between ecosystems,
- create a shared understanding about ecosystem function, and
- stimulate collaborative learning.

These were valuable outcomes of the modeling process above and beyond the creation of STMs.

The creation of a comprehensive, national STM library led by LANDFIRE in collaboration with hundreds of experts across the country represents a significant contribution to the understanding and synthesis of information related to pre-settlement ecosystems across the entire U.S. In addition to their use in understanding and setting reference conditions, the models can be adapted to represent current or desired conditions, to predict future conditions and/or test land management strategies (sensu Low et al. 2010, Pohl et al. 2001, Shlisky et al. 2005, Shlisky and Vandendriesche 2012, Weisz et al. 2009). The LANDFIRE STMs combined with these lessons learned can serve as a solid foundation for future model development efforts related to land management and planning in the United States.

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References

Barrett, S.; Havlina, D.; Jones, J. [and others]. 2010. Interagency Fire Regime Condition Class Guidebook. Version 3.0 [Homepage of the Interagency Fire Regime Condition Class website, U.S. Department of Agriculture, Forest Service, U.S. Department of the Interior, and The Nature Conservancy]. [Online], http://www.frcc.gov. 126 p.

Comer, P.; Faber-Langendoen, D.; Evans, R. [and others]. 2003. Ecological Systems of the United States: A Working Classification of U.S. Terrestrial Systems. Arlington, VA: NatureServe. 75 p.

Czembor, C.A.; Morris, W.K.; Wintle, B.A.; Vesk, P.A. 2011. Quantifying variance components in ecological models based on expert opinion. Journal of Applied Ecology. 48(3): 736–745.

Daniels, S.E.; Walker, G.B. 2001. Working Through Environmental Conflict: The collaborative learning approach. Westport, CT: Praeger. 328 p.

Daubenmire, R. 1968. Plant Communities: A Textbook of Plant Synecology. New York, NY: Harper and Row Publ. 300 p.

ECOMAP. 1993. National Hierarchial Framework of Ecological Units. Washington D.C.: U.S. Department of Agriculture, Forest Service. 20 p.

ESSA Technologies Ltd. 2007. Vegetation Dynamics Development Tool User Guide, Version 6.0. Prepared by ESSA Technologies Ltd., Vancouver, BC. 196 p.

Keane, R.E.; Holsinger, L.M.; Pratt, S.D. 2006. Simulating historical landscape dynamics using the landscape fire succession model LANDSUM version 4.0. Gen. Tech. Rep. RMRS-GTR-171CD. Fort Collins, CO: U.S. Department of Agriculture, Forest Service, Rocky Mountain Research Station. 73 p.

Kolb, D. 1993. The process of experiential learning. In: M. Thorp; Edwards R.; Hanson A., eds. Culture and process of adult learning: A reader. London and New York: Routledge: 138–156. in association with: the Open University.

LANDFIRE. 2012a. LANDFIRE 1.1.0 Vegetation Dynamics Models. [Homepage of the LANDFIRE Project, U.S. Department of Agriculture, Forest Service; U.S. Department of Interior], [Online]. http://www.landfire.gov/index.php. (2012, March 15).

LANDFIRE. 2012b. LANDFIRE 1.1.0 Succession Class layer. U.S. Department of Interior, Geological Survey. [Online]. http://landfire.cr.usgs.gov/viewer/. (2011, March 18).

Low, G.; Provencher, L.; Abele, S. 2010. Enhanced conservation action planning: Assessing landscape condition and predicting benefits of conservation strategies. Journal of Conservation Planning. 6: 36–60.

Pfister, R.D.; Kovalchik, B.L.; Arno, S.F.; Presby, R.C. 1977. Forest habitat-types of Montana. Gen. Tech. Rep. GTR-INT-34. Ogden, UT: U.S. Department of Agriculture, Forest Service, Intermountain Forest and Range Experiment Station. 174 p.

Pohl, K.; Schulz, T.; Strenge, E. [and others]. 2005. Dynamic vegetation modeling: driving management changes today by looking into the past and future. In: Proceedings, Society of American Foresters National Convention. Fort Worth, TX. October 19–23, 2005. 10 p.

Rollins, M.G. 2009. LANDFIRE: a nationally consistent vegetation, wildland fire, and fuel assessment. International Journal of Wildland Fire. 18: 235–249.

Schmidt, K.M.; Menakis, J.P; Hardy, C.C. [and others]. 2002. Development of coarse-scale spatial data for wildland fire and fuel management. Gen. Tech. Rep. RMRS-GTR-87. Fort Collins, CO: U.S. Department of Agriculture, Forest Service, Rocky Mountain Research Station. 41 p.

Shlisky, A.J.; Guyette, R.P.; Ryan, K.C. 2005. Modeling reference conditions to restore altered fire regimes in oak-hickory-pine forests: validating coarse models with local fire history data. In: Proceedings, EastFire conference. George Mason University, Fairfax, VA. May 12–13, 2005.

Shlisky, A.; Vandendriesche, D. 2012. Use of state-and-transition simulation modeling in National Forest planning in the Pacific Northwest, U.S.A. In: Kerns, B.K.; Shlisky, A.J.; Daniel, C.J. tech. eds. Proceedings of the First Landscape State-and-Transition Simulation Modeling Conference, June 14–16, 2011, Portland, Oregon. Gen. Tech. Rep. PNW-GTR-869. Portland, OR: U.S. Department of Agriculture, Forest Service, Pacific Northwest Research Station: 23–41.

Weisz, R.; Tripeke, J.; Truman, R. 2009. Evaluating the ecological sustainability of a ponderosa pine ecosystem on the Kaibab Plateau in Northern Arizona. Fire Ecology. 5: 100–114.

The Integrated Landscape Assessment Project

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Abstract

The Integrated Landscape Assessment Project (ILAP) is a three-year effort that produces information, models, data, and tools to help land managers, policymakers, and others examine mid- to broad-scale (e.g., watersheds to states and larger areas) prioritization of land management actions, perform landscape assessments, and estimate potential effects of management actions for planning and other purposes. ILAP provides wall-to-wall, cross-ownership geospatial data and maps on existing, potential and future vegetation conditions, land ownership and management allocation classes, and other landscape attributes. State and transition models integrate vegetation development, management actions, natural disturbances, and climate change to allow users to examine the mid- and long-term effects of alternative management, disturbance, and climate scenarios. State-and-transition model (STM) outputs are used to produce information on many landscape characteristics, including vegetation conditions, disturbance regimes, fuel conditions, wildlife habitats, and economic values of natural resource-related products in Arizona, New Mexico, Oregon, and Washington. The project consists of science delivery (e.g., state and transition models, spatial data) and knowledge discovery (e.g., new linkages to wildlife habitat relations, fuel treatment economics, aboveground carbon pools, biomass, water supplies, and trends in wildfire and fuel conditions) that are integrated through decision support systems. The spatial data, state and transition models, model outputs, and interpretations cover all major upland vegetation types, including forests, woodlands, shrublands, grasslands, and deserts. To date, more than 50 GIS layers and 250 unique state and transition models have been produced across the 4-state area (over 117 million hectares). ILAP data, models, and tools will be accessible through a Western Landscapes Explorer portal to be publicly launched in 2012 (INR and OSU Libraries 2012). Products from ILAP can be used by land managers, program managers, analysts, planners, and policymakers to evaluate management strategies that reduce wildfire risk, improve habitat, generate revenues, benefit rural communities, and inform restoration investment decisions. Because it allows for integration of many natural resource management objectives, ILAP facilitates collaborative landscape planning over very large areas. ILAP methods should be widely applicable for all lands.

Keywords: landscape assessment, science delivery, knowledge discovery, vegetation models, decision support.

Introduction

Fire suppression, vegetation management activities, grazing, climate change, and other factors produce constantly changing vegetation, fuel, 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 natural resource management are tight. Techniques are needed to prioritize where natural resource management activities, such as fuel treatments, could be most effective and most likely to result in desirable conditions. Solutions driven by single resource concerns have proven problematic in most cases, since ecological and human systems are necessarily intertwined. More than $5.5 million of funding from the American Recovery and Reinvestment Act in 2009 provided the opportunity to hire a team of more than 50 technical experts to provide an integrated approach to assess landscape conditions and forecast potential future effects of alternative natural resource management strategies in Arizona, New Mexico, Oregon, and Washington. This paper summarizes the approach and methods used in the ILAP. Examples of applications can be found elsewhere (Creutzburg et al. 2012, Morzillo et al. 2012, Shlisky et al. 2012, Zhou and Hemstrom 2012).

The Integrated Landscape Assessment Project (ILAP) produces databases, reports, maps, analyses, and other information showing mid- to broad-scale (thousands to millions of hectares) vegetation conditions and trends, key wildlife habitat conditions and trends, 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 (fig. 1).

ILAP integrates these landscape attributes into databases, reports, and maps that show a continuum of integrated priority areas (from high to low priority) considering a combination of vegetation trends, treatment costs, and likely effects of treatments and climate change impacts on key wildlife habitat, fuel conditions, and other landscape characteristics. ILAP gathers and consolidates key information and filled in data gaps across the 4-state area. The project packages and delivers knowledge in usable ways and allows for the development of new knowledge. In addition, ILAP is modular and allows updates or exchange of vegetation data sets, including incorporation of new plot data, resource interpretations, and other elements as knowledge improves.

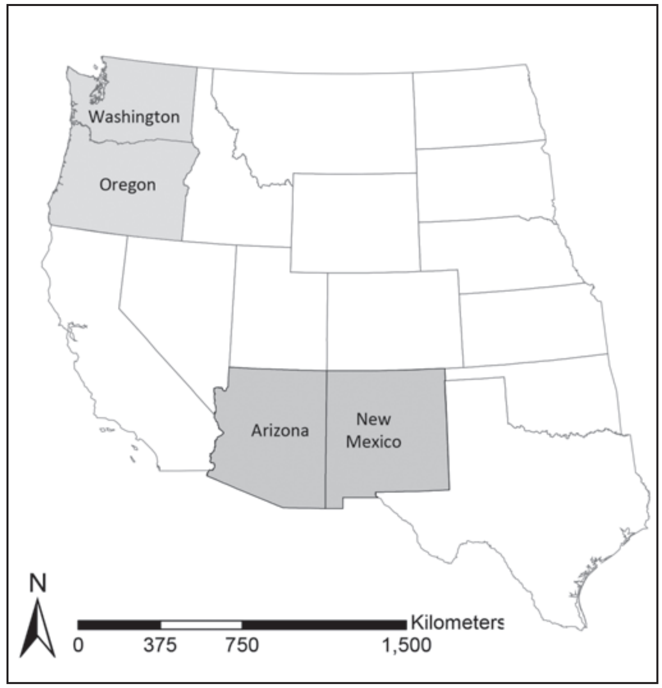

Figure 1—Integrated Landscape Assessment study area.

ILAP relies on regional advisory groups—with representation from state and federal land management agencies, conservation organizations, and industry groups—for definition of goals and priorities. With their input, a set of resource management questions have been defined relating to all major upland ecological systems in the 4-state area:

1. What are the existing vegetation conditions across forests, woodlands, shrublands, grasslands, deserts, and other ecological systems?
2. What are the implications of vegetation 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 and in the face of climate change?

While ILAP models and data cover all major upland vegetation types in the 4-state area, the project also works with collaborative groups involved with restoration decisionmaking in focus landscape areas to demonstrate utility and refine the landscape analysis process. At present, focus areas include the Tapash Sustainable Forest Collaborative in

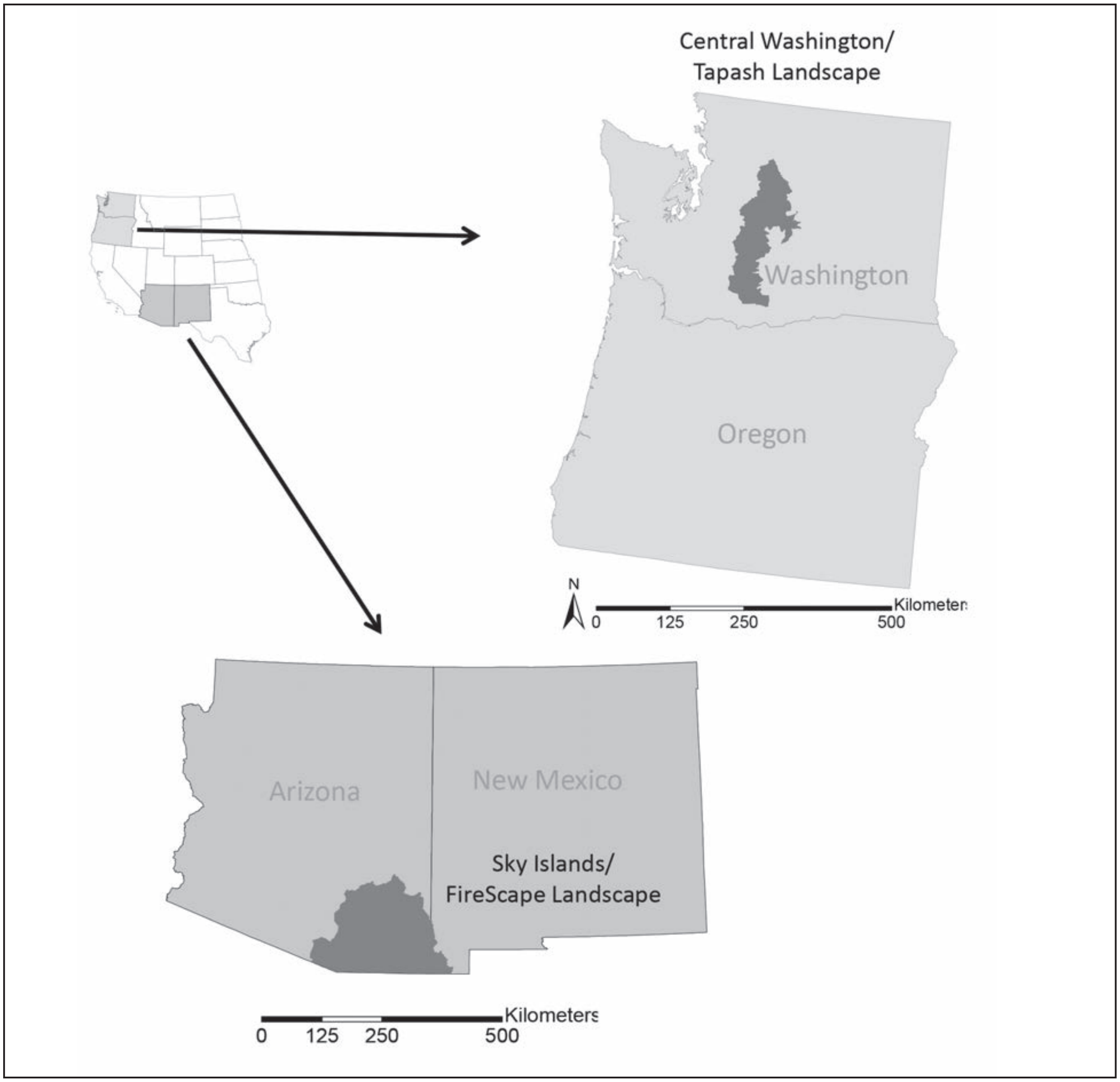

Figure 2—The central Washington and Sky Islands landscape focus areas within the Integrated Landscape Assessment Project.

central Washington and the Sky Islands/FireScape group in southeastern Arizona (fig. 2). While management questions are specific to each focus area, example questions include:

1. What general kinds of treatments might produce the desirable combinations of fuel reductions and wildlife habitat conditions? Is it possible to slow or stop the upward spiral of fire suppression costs through fuel treatments? What are the economic costs of such an approach?
2. How likely are fuel treatments to generate valuable economic products? Can the treatments pay for themselves?
3. What areas and management regimes might be most likely to produce high combined potential to reduce critical fuels, improve or not degrade key wildlife habitat, and generate positive economic value?

4. How will projected climate scenarios affect future vegetation, habitat, and fuel conditions over the long term (100 years)?

Because ILAP is funded by the American Recovery and Reinvestment Act, the project will be completed in a relatively short time (about 3 years). Both the project timeframe and the size of the four state project area (>117 million hectares) necessitate reliance on existing information (vegetation data, state and transition models, etc.) rather than development of extensive new information. The 4-state project area was selected because of the state and transition models and collaborations that were already in place between Region 3 of the USDA Forest Service (based in Albuquerque, New Mexico), Region 6 of the Forest Service, and the Forest Service PNW Research Station (based in Portland, Oregon). Because ILAP aims to be an "all-lands" approach, new data sets and models are developed to fill data gaps or where models do exist, as is the case for much of the arid land. Existing data and models are refined as new insights into ecological interactions, natural resource conditions and trends, potential climate change effects, economic and social interactions, and other topics become apparent.

Organization

The project is a collaborative effort and incorporates expertise from several institutions and disciplines (fig. 3). An oversight team, composed of representatives from the funding agency and major collaborators (Institute for Natural Resources, Oregon State University College of Forestry, USDA-FS PNW Research Station, and USDA-FS Region 3) provides overall direction at monthly meetings. Two groups of project advisors, one from Oregon and Washington and one from Arizona and New Mexico, connect the project goals, objectives, and products to state agencies, federal

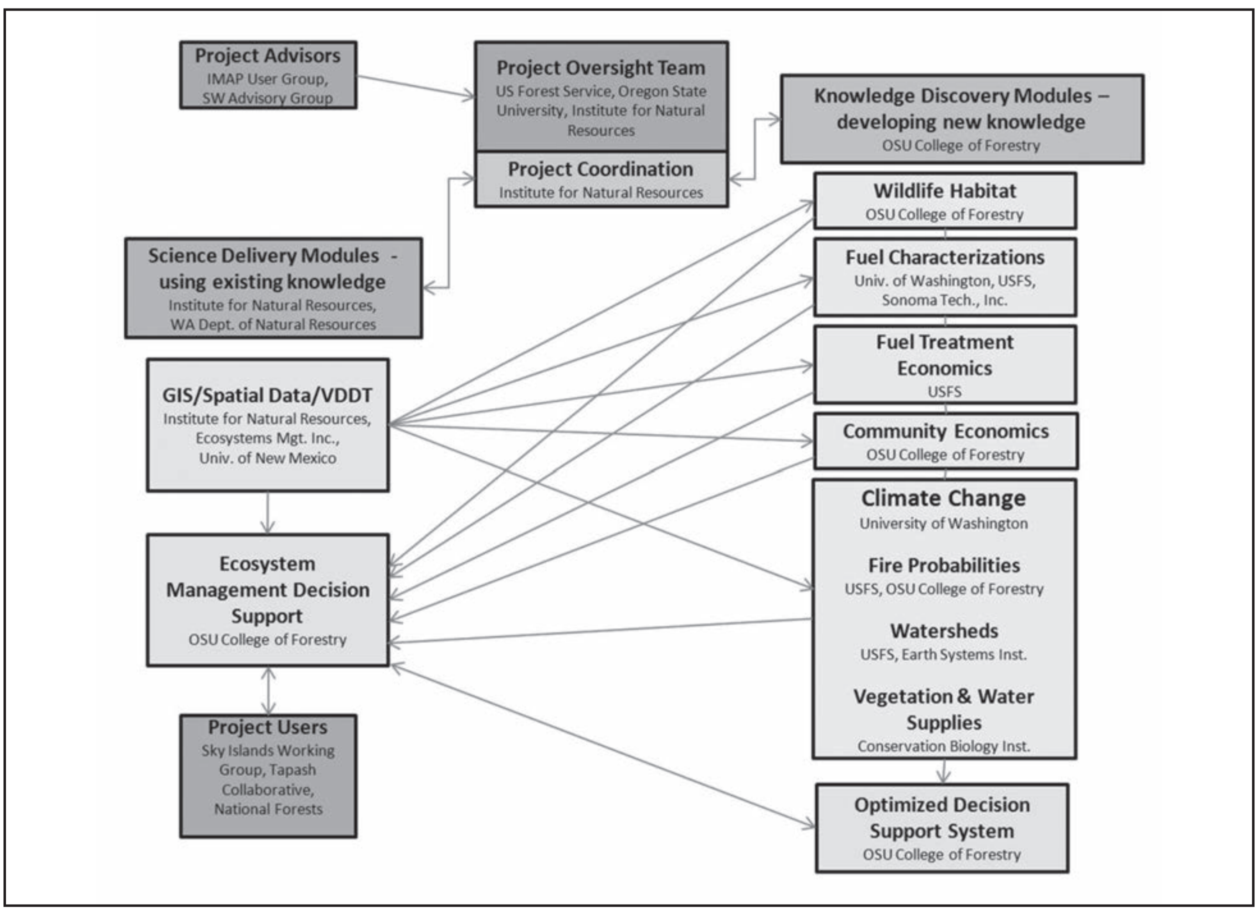

Figure 3—Organization of science delivery and knowledge discovery partners in the Integrated Landscape Assessment Project.

agencies, non-profit organizations, private contractors and industries, universities, and the interested public by providing comment, feedback, and review at twice-yearly working sessions. The project lead scientist and project coordinator oversee the technical and outreach aspects of project work. Science delivery, as a whole, is jointly led by scientists from the Institute for Natural Resources and the Washington Department of Natural Resources. Each science delivery module has a lead investigator and production team, as necessary. Knowledge discovery modules are led by several universities and nonprofit organizations and each module has a lead scientist and, as appropriate, a production team. User involvement is critical to establishing project priorities and developing useful products.

Science Delivery and Knowledge Discovery

The project is separated into two major components: science delivery and knowledge discovery. Science delivery teams generally work with *existing* methodologies to develop landscape-level information (primarily relating to vegetation conditions) while the knowledge discovery teams develop and apply *new* methodologies to develop, project, and integrate associated landscape-level information on wildlife habitat, fuel conditions, treatment economics, community impacts, and climate change impacts using STM simulation outputs. From a timing perspective, the science delivery teams produce the foundational data and model outputs that are then used by the knowledge discovery teams as inputs to their models and tools to enable a multidimensional approach to assess landscapes and inform management priorities for restoration over very large areas. The ILAP team is organized into 12 module teams: GIS, vegetation modeling, wildlife habitat, fire and fuels, fuel treatment economics, community economics, climate change and vegetation, climate change and watersheds, climate change and fire, EMDS decision support, optimized decision support, and data portal.

Science Delivery Modules

ILAP's science delivery modules include the Geographic Information System (GIS), State and Transition Modeling (STM), and Ecosystem Management Decision Support (EMDS) modules.

Geographic Information System Module

The GIS module provides spatial data to the other ILAP modules. These data include current and potential vegetation conditions, watershed boundaries, ownership categories, management activities, and others. The GIS team gathers data from various public and private sources, merges and appends it into seamless datasets, combines attribute data into consistent formats, creates detailed documentation, and provides data to the broader ILAP team and partners. Much effort focuses on standardizing datasets across administrative units and between modeling regions. Data is delivered in raster/grid and polygon formats. The GIS team uses a long-term data management process to facilitate the incorporation of any data updates or use of new and improved datasets, as well as maintenance of original datasets. GIS data on current and potential natural vegetation are developed by using imputation methods and geo-referenced plot data from various sources. Much of the plot data, especially for forested vegetation types, are from permanent inventory plots on federal lands, such as the Forest Inventory and Analysis (FIA) program (USDA FS 2012a). A significant plot gathering and database compilation effort in the Southwest is under sub-contract with Ecosystem Management, Inc. (EMI) and Natural Heritage New Mexico at the University of New Mexico. Local field offices of many federal agencies (USFS, NRCS, BLM, etc.) are visited to collect plot data. Often times, these plot data are not in a digital form or georeferenced, so efforts are made to select the highest priority data to digitize and compile. Existing or current vegetation and potential vegetation types (PVTs) are mapped using gradient nearest neighbor imputation (GNN; Ohmann and Gregory 2002) for forested vegetation and a combination of GNN and random forest nearest neighbor imputation (RFNN; Breiman et al. 2006) for arid lands, both of 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 existing vegetation and an assignment of potential vegetation types (PVT) in Oregon and Washington and potential natural vegetation types (PNVT; PVT for simplicity) in Arizona and New Mexico. The vegetation data cover all major vegetation types across all wildlands (forests, woodlands, shrublands, grasslands, and desert). Riparian areas, minor upland types, urban, agricultural and other developed areas are excluded.

State and Transition Modeling Module

The state and transition modeling (STM) module collects, assembles, and builds models for forest and arid land vegetation types. Using input datasets from the GIS module (existing vegetation cover and structure, potential vegetation, ownership and management data layers, and watershed boundaries), the STM module projects future landscape conditions for all major upland vegetation types using a "no management" scenario (no management other than continued wildfire suppression on all lands and continued grazing in arid lands) and for the landscape focus areas according to a few example alternative management scenarios.

ILAP builds on STMs currently used by various organizations for federal land management planning, restoration planning, and ecoregional assessments in the 4 state area and elsewhere (e.g., Forbis et al. 2006, Hann et al. 1997, Hemstrom et al. 2007, Holsinger et al. 2006, Merzenich and Frid 2005, Weisz et al. 2009). The STM approach treats vegetation as states, with each state defined as a combination of cover type and structural stage within potential vegetation types. Transitions among states represent natural disturbances, management actions, and vegetation growth and development. At present, STMs are implemented in the Vegetation Development Dynamics Tool (VDDT) (ESSA Technologies Ltd. 2012) and run using the Path Landscape Modeling Framework (Apex Resource Management Solutions Ltd. 2012). STMs are adapted from existing models available from the USDA Forest Service, Pacific Northwest and Southwest Regions, The Nature Conservancy, and LANDFIRE (LANDFIRE 2012). In some cases, STMs consistent with project methods are not available and new models are constructed using similar existing models as templates. STMs are developed for each PVT within each modeling region (fig. 4), resulting in 124 STMs for Oregon and Washington and 90 STMs for Arizona and New Mexico. Transitions are developed from a combination of expert opinion, available literature, and empirical data analysis (e.g., the Monitoring Trends in Burn Severity data—for wildfire probabilities; MTBS 2011). Transitions include all major natural disturbances, including wildfire (low, mixed, and high severity), insect outbreaks, wind disturbance, drought mortality, and others as appropriate to the ecological system being modeled, as well as a variety of management activities.

The 4-state area is stratified by combinations of land ownership, land allocation classes, and potential vegetation types within modeling regions. Fifth-code (Hydrologic Unit Code; HUC; USGS 2012) watersheds were used to further stratify results to improve spatial resolution. HUC boundaries within a modeling region do not affect ecological relationships in the models but allow modelers to better target management treatments to relatively small areas (e.g. 1000s of hectares). STM are run on each modeling region, PVT, land ownership/allocation, and HUC stratum. Alternative land management scenarios are generated by changing assumptions about vegetation management treatments and rates by modeling region, PVT, land ownership, land use allocation and HUC. Resulting forecasts of vegetation conditions, management activities, and natural disturbances are linked to wildlife habitat characteristics, economic values, and other important conditions (Barbour et al. 2007, Hemstrom et al. 2007, Reeves et al. 2006, Wales et al. 2007). In this fashion, model simulation results forecast potential future amounts and distributions of important landscape characteristics at the scale of modeled strata without implying pixel or stand-level accuracy.

Ecosystem Management Decision Support (EMDS) Module

The Ecosystem Management Decisions Support (EMDS) module integrates the separate factors of vegetation, fuels, wildlife habitat, and economic conditions into a combined, flexible assessment and prioritization process. Likely future trends are included in the prioritization process along with important ancillary data (e.g. wildland-urban interface boundaries, roads, key watershed delineation, etc.). The EMDS tool helps managers and others explore and set

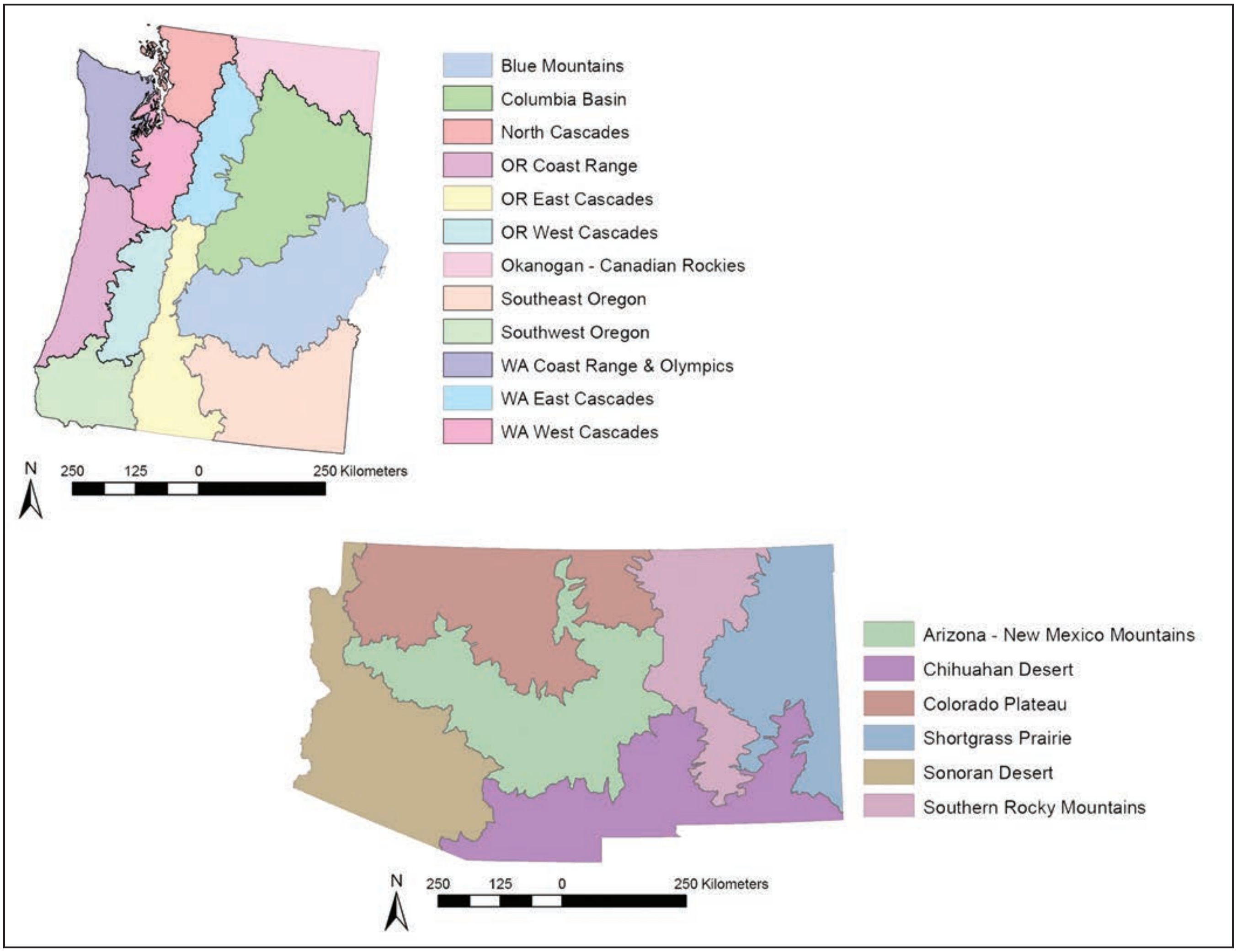

Figure 4—Modeling regions used in the Integrated Landscape Assessment Project.

priorities using maps, tables, and reports based on different combinations of characteristics that best reflect their values.

Knowledge Discovery Modules

ILAP's knowledge discovery modules include fire and fuel characterization, wildlife habitat, fuel treatment economics, community economics, climate change, and an optimized decision support system.

Fire and Fuel Characterization Module

The fire and fuel characterization module evaluates potential future fire hazard, focusing on how land management and natural disturbances might affect fuels and answering question such as: How do fuel characteristics vary across the four states? How might different forest management scenarios affect fuel conditions and fire hazard across landscapes? To what extent can fuel treatment programs reduce fire hazards over the long term? Fuel beds (descriptions of burnable biomass extending from the forest floor to the canopy) have been built for each vegetation state in forested STM using inventory and other plots classified into each state class. Resulting fuel beds allow users to assess current conditions and trends in fuels and potential fire behavior. Deliverable module products include: (1) characterized fuel properties from inventory plots, describing a range of conditions in each STM state class and (2) characterization of fire hazard for each land ownership and PVT stratum by watershed, including both qualitative characterization (fire potential on a scale of 0 to 9) and quantitative characterization (output such as simulated fire flame length, rate of spread, and crown fire potential). One application of this module's outputs is to assess the likelihood of crown fire given different management approaches.

Fuel Treatment Economics Module

The fuel treatment economics module assesses the financial feasibility of proposed forest vegetation management treatments. This module estimates potential supplies of timber and biomass (by diameter classes and tree species groups) and aboveground, tree-based carbon pools by STM state class for forested lands in Oregon and Washington. In addition, methods and data from this module allow users to conduct financial analyses that compare alternative vegetation treatment scenarios. Methods are tested by comparing a base "no management" scenario to a hypothetical restoration scenario in the central Washington landscape area. This example uses STM simulation outputs of removed products from proposed treatments over time to develop cost-benefit analyses. It considers harvesting cost associated with each treatment using a fuel reduction cost simulator (FRCS) (USDA FS 2012b), transportation cost to the desired mills, products prices, and other economic factors. Deliverable products include: (1) data and methods for examining available biomass and timber across all forested lands in Oregon and Washington, (2) methods and data for examining potential timber and biomass removals associated with a wide variety of alternative management scenarios for forested lands in Oregon and Washington, and (3) an example analysis of economic attribute trends and variability for a no management and an alternative restoration management scenario in the central Washington landscape area, and (4) documentation of modeling methods and results. The outputs of this module can help land managers and others evaluate prospective areas for timber and forest product extraction and assess watersheds where forest management treatments may have the largest economic potential in terms of revenue and jobs for communities or where those products may help offset the costs of management treatments.

Wildlife Habitats Module

The wildlife habitat module generates look-up tables for STM state classes to estimate potential habitat area for more than 50 species and habitats in Arizona, New Mexico, Oregon, and Washington. The module develops databases that allow users to derive aggregate area of vegetation composition for each modeling region, PVT, ownership-land allocation, and watershed stratum and match this information to species-habitat relationships to determine potential aggregate habitat area within each stratum. Deliverable products include: (1) a list of focal species-habitat relationships (Oregon and Washington) or habitats of interest (Arizona and New Mexico), (2) example generation of wildlife habitat analyses based on STM outputs for a no-management scenario and a restoration management scenario in the central Washington landscape area, and (3) written documentation of methods, results, and findings. The outputs of this module provide land managers and planners with an ability to evaluate how specific habitats may be impacted by various land management decisions and proposed policies across modeled lands in the 4-state area.

Community Economics Module

The community economics module addresses the question of to what degree can large-scale forest vegetation treatment programs support economic activity and contribute to well-being in communities that have been negatively impacted by recent federal forest policy changes. Essentially, this module asks: How would priorities for fuel treatment areas be affected by including community well-being as a criterion for treatment prioritization, along with fire hazard reduction and wildlife habitat quality? Communities are assigned an "Impact Score" based on their level of socio-economic distress, their ability to utilize harvested forest materials, and whether they have been impacted by changes in federal forest policy, such as the Northwest Forest Plan. One application of this module's output is to help describe the potential for fuel treatments to produce economic benefits to nearby communities for the forested landscapes in Arizona, New Mexico, Oregon, and Washington.

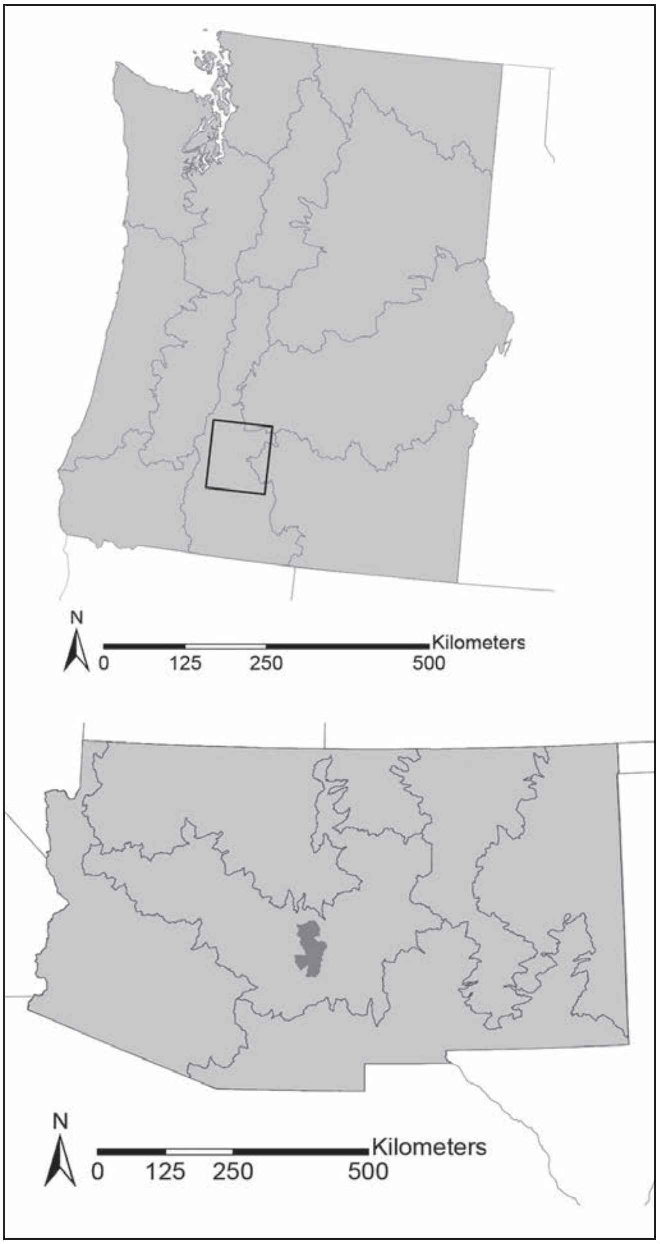

Figure 5—The central Oregon and eastern Arizona climate change prototype area within the Integrated Landscape Assessment Project.

Climate Change

The three ILAP climate change modules address potential climate change impacts on vegetation, watershed conditions, and fire probabilities. Together, these modules evaluate potential future effects that climate change might have on: (1) mid- to broad-scale vegetation conditions and wildfire regimes, (2) hydrology at the watershed scale, and (3) local, stand-scale vegetation and wildfire interactions.

Climate change and vegetation module—
The climate change and vegetation module provides estimates of potential future climate change on major vegetation types and wildfire conditions in study areas in central Oregon and eastern Arizona (fig. 5). In normal usage, the areal extent of potential vegetation types and associated STM remain constant over time (Kerns et al. 2012). In the future, however, climate change is expected to alter the mix and distribution of PVTs. This module builds “mega-models” in which many individual STMs are combined. Landscape area can move among PVTs over time as a function of changes in climatic conditions and wildfire. The module gathers vegetation change and wildfire trend data from simulations of three global climate model and emissions scenarios (MIROC-A2, CSIRO-A2, and Hadley-A2) run with the MC1 dynamic vegetation model (Bachelet et al. 2001). That information is used to build new transitions that cross PVT boundaries in response to vegetation and wildfire trends from MC1 (fig. 6). Potential future wildfire trends under different climate change scenarios are also included under the assumption that wildfire will be a major contributor to climate change effects. The resulting models will allow users to answer questions such as: How might the forests and arid lands in the study areas change in the future given the three different climate scenarios? What kinds of management activities might exacerbate climate change effects on vegetation conditions, natural disturbances, and associated resource values? Conversely, are there suites of management activities that might foster relatively resilient vegetation communities? Module products will include a set of “climatized” STMs for the two study areas, simulations of the three climate scenarios at 4 km grid scale for all of the modeling regions in the four state area, GIS tools to extract hydrography and other data from dynamic global vegetation model output, and methods that can be used to construct similar models in other areas.

Climate change and watersheds module—
The climate change and watersheds module applies and enhances the NetMap system (Earth Systems Institute

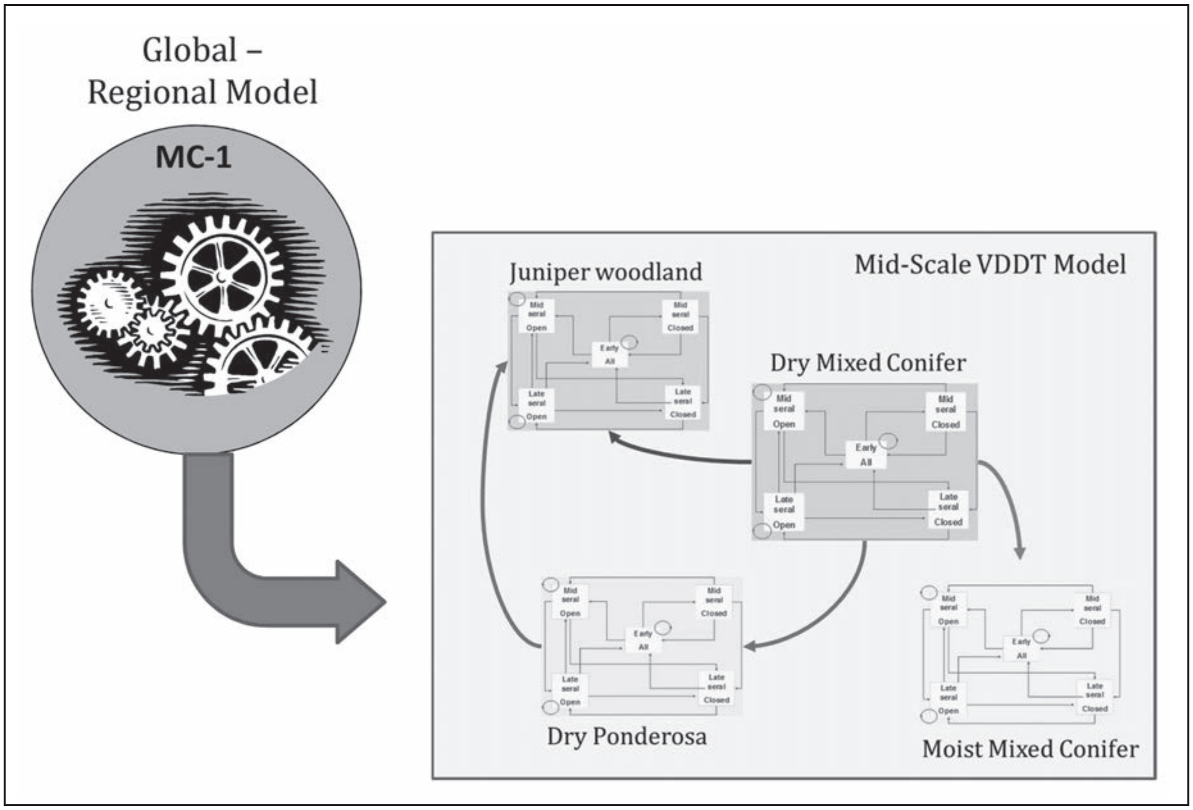

Figure 6—Conceptual process for connecting a Dynamic Global Vegetation Model (MC1) to STM.

2012) to generate estimates of erosion hazards, in-channel habitat conditions, and other important watershed characteristics for all 5th code hydrologic unit (U.S. Geological Survey 2012) watersheds that contain national forest lands in Oregon and Washington. Based on a computer model of stream systems, different inputs such as bank slope and sediment erosion potential are used to create maps of priority restoration areas. It also provides 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 from alternative climate change scenarios. NetMap is free downloadable software that can be overlaid on Google Earth to help managers identify areas where changes are most likely, and where restoration activities may be most effective.

Climate change and fire probabilities module— The climate change and fire probabilities module provides refined insight at the stand scale into the variation of wildfire probabilities and vegetation dynamics with climate change in a prototype area of the upper Deschutes sub-basin in Oregon. This module uses the FireBGCv2 computer model (USDA FS 2012c) to generate data and maps of fire ignition, spread, frequency and severity and the resulting shifts in vegetation arrangement and distribution. In the future, results from the FireBGCv2 and similar models will help to inform STMs by characterizing vegetation change and altered fire regimes under a range of potential climate change conditions.

Optimized Decision Support Module

The optimized decision support module integrates fuels, wildlife habitat, and economic conditions into a spatially-based analytical decision support process. Input criteria, such as preserving habitat for a particular wildlife species, generating revenue from forest products, or encouraging desirable future forest conditions will be variously weighted in the optimization process. The analysis procedure is

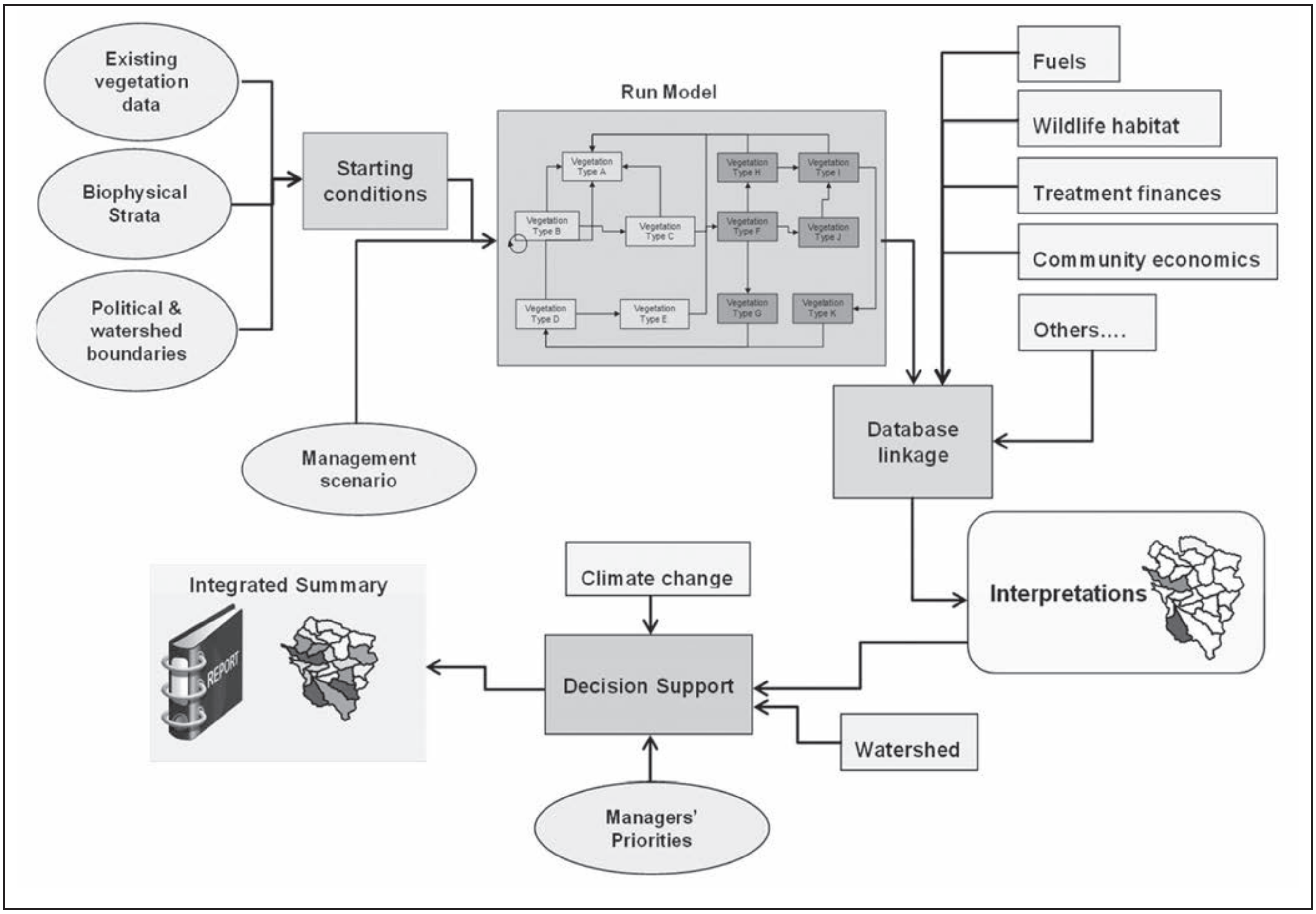

Figure 7—Process flow for landscape analysis and prioritization in the Integrated Landscape Assessment Project.

computationally intensive and often limits the size of the landscapes that can be analyzed using optimization methods. This module will explore the geographic limits of optimization methods and provide a spatial representation of the integrated landscape components.

Forecasting Future Conditions Process

While many users are satisfied w ith information on current landscape conditions contained in various ILAP data sets, ILAP STM can be used for examining likely future outcomes of alternative management strategies. For example, land management planners may use STM to see if a proposed set of management activities are likely to produce desired conditions. Alternatively, resource managers can use a gaming approach to generate management approaches that provide an acceptable mix of natural resource conditions and values. This scenario forecasting process requires several steps (fig. 7).

- Assemble base data needed to run state and transition models. This involves aggregating and cross-walking existing vegetation, potential vegetation types, watershed boundary, and land ownership/allocation data into the strata used for modeling. Cross-walking and aggregation are not trivial tasks given the potential variety of data that may be useful. Aggregation of finer-scale grid or polygon data to coarser modeling strata reduces spatial detail, but often improves data compatibility. Spatial detail is critically dependent on the scale and accuracy of input data. It is important to understand the limitations of input data.

- Generate management scenarios. Users are consulted about the issues with which they are most concerned. Most often, this process results in a series of natural resource management and socioeconomic questions that relate to one another. These questions must be translated into assumptions about vegetation management treatment types and rates, differences among the vegetation and management objectives of different land owners and on different land allocations, and the kinds of resulting landscape conditions of interest. The combination of assumptions and desired products is a management scenario. Developing a management scenario is not a trivial step and often takes several in-depth discussions with users to clarify their intent and interests.
- Run STM simulations for many decades or longer with management scenarios of interest. ILAP typically runs models for 50 to 100 years or more. Simulations usually include 30 Monte Carlo sequences of randomly drawn wildfire years and insect outbreaks as well as the prescribed management activities.
- Analyze model output and link to interpretations of fuel conditions, wildlife habitat, economic values, and other forecast landscape characteristics. The modeling process generates output by year, state class, and transition. In essence, every state class and every transition is output for every year. Output data are linked by look-up tables to the combinations of state classes (area by year by model stratum) and transitions (area by year by model stratum) that are interpretations of key wildlife habitats, fuel conditions, economic values, and other landscape characteristics.
- Integrate and prioritize. Separate analyses of fuels, wildlife habitats, and economics provide useful information for examining mid- and broad-scale trends, but the real power of the project comes through integrating these separate factors into a combined, flexible prioritization process. ILAP uses decisions support systems to help managers and others interact with combinations of characteristics that best answer their particular questions. Outputs are maps, tables, databases, and reports.
- Product delivery. ILAP products range from relatively simple geographic and other data sets to integrated landscape analyses, to white papers and science journal articles. ILAP products are accessible from the Western Landscapes Explorer portal (INR and OSU Libraries 2012). Most basic geographic data and many landscape analysis reports will be freely available. Some data sets and analyses may be available for only limited distribution if they contain data that users deem sensitive.

Conclusion

ILAP creates a variety of analytical and graphical tools that will help land managers and planners integrate and prioritize management activities. The project's publications, models, maps, data, and tools will be available online and archived so that scientists and managers in years to come will be able to use and build on the project's products. The project will also create a web-enabled decision support system if time and resources permit. Land managers, planners, analysts, scientists, policymakers, and large-area landowners can use the project's tools and information for many applications including, but not limited to:

- Watershed restoration strategies
- Land management planning
- Statewide assessments and bioregional plans
- National forest plan revisions

ILAP data and models are currently in use by the Okanogan-Wenatchee and Colville National Forests to support current forest plan revisions. Forest planning analysts are running the ILAP models with help from the ILAP team and the Forest Service regional planning analyst. Other national forests in the Pacific Northwest (Deschutes, Ochoco, and Fremont-Winema) are using the ILAP data and models to support wildlife species viability analyses. National forests in the southwest are using local versions of ILAP models for forest plan revisions. Several other USDA Forest Service regions use STMs for land management planning and assessments and those regions could be easily included in ILAP. Remaining Forest Service regions may not have STMs except for LANDFIRE models, but more detailed STMs suitable for planning and assessments could be developed rather quickly. In addition, the USDA Natural Resources Conservation Service and USDI Bureau of Land Management often use STMs in ecological site descriptions. The ILAP process could be applied to natural resource management issues in any ecological system as long as that system can be described in state and transition terms.

As the first phase of ILAP concludes, a strong foundation of landscape-level data, STMs, tools and expertise has been built that can be efficiently applied to other landscapes in the West. The goal is to have complete Western coverage of ILAP data and STMs to support regional and statewide issues of importance by groups such as the Western Governors' Association, Western Forestry Leadership Coalition, and Landscape Conservation Cooperatives.

Acknowledgments

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References

Apex Resource Management Solutions Ltd. 2012. Path Landscape Model. http://www.apexrms.com/path. (14 March 2012)

Bachelet, D., Lenihan,J.M.; Daly, C.; Neilson, R.P.; Ojima, D.S.; Parton, W.J. 2001. MC, a Dynamic Vegetation Model for estimating the distribution of vegetation and associated carbon and nutrient fluxes, Technical Documentation Version 1.0. Gen. Tech. Rep. PNW-GTR-508. Portland, OR: U.S. Department of Agriculture, Forest Service, Pacific Northwest Station. 95 p.

Barbour, R.J., Singleton, R.; et al. 2007. Evaluating forest product potential as part of planning ecological restoration treatments on forested landscapes. Landscape and Urban Planning. 80(3): 237–248.

Breiman, L.; Cutler, A.; Liaw, A.; Wiener, M. 2006. Breiman and Cutler's random forests for classification and regression. http://CRAN.R-project.org/ [R package version 4.5–16].

Creutzburg, M. K.; Halofsky, J.S.; Hemstrom, M.A. 2012. Using state-and-transition models to project cheatgrass and juniper invasion in southeastern Oregon sagebrush steppe. In: Kerns, B.K.; Shlisky, A.J.; Daniel, C.J. tech. eds. Proceedings of the First Landscape State-and-Transition Simulation Modeling Conference, June 14–16, 2011, Portland, Oregon. Gen. Tech. Rep. PNW-GTR-869. Portland, OR: U.S. Department of Agriculture, Forest Service, Pacific Northwest Research Station: 73–84.

Earth Systems Institute. 2012. NetMap. http://www.netmaptools.org/coverage. (12 March 2012).

ESSA Technologies. 2007. VDDT. http://essa.com/tools/vddt/. (14 March 2012).

Forbis, T.A., Provencher, L.; et al. 2006. Great Basin land management planning using ecological modeling. Environmental Management. 38(1): 62–83.

Hemstrom, M.A., Merzenich, J. [et al.]. 2007. Integrated analysis of landscape management scenarios using state and transition models in the upper Grande Ronde River Subbasin, Oregon, USA. Landscape and Urban Planning. 80(3): 198–211.

Hann, W.J., Jones, J.L. [et al.]. 1997. Landscape dynamics of the basin. An assessment of ecosystem components in the Interior Columbia Basin and portions of the Klamath and Great Basins. T.M. Quigley and S.J. Arbelbide. Portland, OR, U.S. Department of Agriculture, Forest Service, Pacific Northwest Research Station: 337–1055.

Holsinger, L., Keane, R.E.; [et al.]. (2006). Using historical simulations of vegetation to assess departure of current vegetation conditions across large landscapes. The LANDFIRE Prototype Project: nationally consistent and locally relevant geospatial data for wildland fire management. Gen. Tech. Rep. RMRS-GTR-175. M.G. Rollins and C.K. Frame. Fort Collins, CO, U.S. Department of Agriculture, Forest Service, Rocky Mountain Research Station: 315–366.

Institute for Natural Resources and Oregon State University Libraries. 2012. Western Landscapes Explorer. http://www.westernlandscapesexplorer.info/. (14 March 2012).

Kerns, B.K.; Hemstrom M.A.; Conklin D.; Yospin, G.; Johnson, B.; Bachelet, D.; Bridgham, S. 2012. Approaches to incorporating climate change effects in state and transition simulation models of vegetation. In: Kerns, B.K.; Shlisky, A.J.; Daniel, C.J. tech. eds. Proceedings of the First Landscape State-and-Transition Simulation Modeling Conference, June 14–16, 2011, Portland, Oregon. Gen. Tech. Rep. PNW-GTR-869. Portland, OR: U.S. Department of Agriculture, Forest Service, Pacific Northwest Research Station: 161–171.

LANDFIRE. 2012. LANDFIRE. http://www.landfire.gov/. (14 March 2012).

Merzenich, J.; Frid, L. 2005. Projecting landscape conditions in southern Utah using VDDT. Systems Analysis in Forest Resources: Proceedings of the 2003 Symposium Portland, OR: U.S. Department of Agriculture, Forest Service, Pacific Northwest Research Station. 157–163.

Morzillo, A.T.; Halofsky, J.S.; DiMiceli, J.; Hemstrom, M.A. 2012. Balancing feasibility and precision of wildlife habitat analysis in planning for natural resources. In: Kerns, B.K.; Shlisky, A.J.; Daniel, C.J. tech. eds. Proceedings of the First Landscape State-and-Transition Simulation Modeling Conference, June 14–16, 2011, Portland, Oregon. Gen. Tech. Rep. PNW-GTR-869, Portland, OR: U.S. Department of Agriculture, Forest Service, Pacific Northwest Research Station: 103–114.

MTBS. 2011. Monitoring Trends in Burn Severity. http://www.mtbs.gov/. (14 March 2012)

Ohmann, J.; Gregory, M.J. 2002. Predictive mapping of forest composition and structure with direct gradient analysis and nearest neighbor imputation in coastal Oregon, U.S.A. Canadian Journal of Forestry. 32: 725–741.

Reeves, M.C., Kost, J.R.; [et al.]. 2006. Fuels products of the LANDFIRE project. Fuels management—how to measure success: Conference Proceedings. 28–30 March, 2006, Portland, OR. Proceedings RMRS-P-41. P.L. Andrews and B.W. Butler. Fort Collins, CO: U.S. Department of Agriculture, Forest Service, Rocky Mountain Research Station: 239–252.

Shlisky, A.; Vandendriesche, D. 2012. Use of state-and-transition simulation modeling in national forest planning in the Pacific Northwest, U.S.A. In: Kerns, B.K., Shlisky, A.J.; Daniel, C.J. tech. eds. Proceedings of the First Landscape State-and-Transition Simulation Modeling Conference, June 14–16, 2011, Portland, Oregon. Gen. Tech. Rep. PNW-GTR-869. Portland, OR: U.S. Department of Agriculture, Forest Service, Pacific Northwest Research Station: 23–41.

U.S. Department of Agriculture, Forest Service [USDA FS] 2012. Forest Inventory and Analysis National Program. http://fia.fs.fed.us. (12 March 2012).

U.S. Department of Agriculture, Forest Service [USDA FS] 2012. Fuel Reduction Cost Simulator. Portland, OR: Pacific Northwest Research Station. http://www.fs.fed.us/pnw/data/frcs/frcs.shtml. (14 March 2012).

U.S. Department of Agriculture, Forest Service [USDA FS] 2012. Fire, Fuel, and Smoke Science Program. Missoula, MT: Rocky Mountain Research Station. http://www.firelab.org/research-projects/fire-ecology/139-firebgc. (14 March 2012).

U.S. Geological Survey. 2012. Hydrologic Unit Maps. http://water.usgs.gov/GIS/huc.html. (12 March 2012).

Wales, B.C.; Suring, L.H. et al. 2007. Modeling potential outcomes of fire and fuel management scenarios on the structure of forested habitats in northeast Oregon, USA. Landscape and Urban Planning. 80(3): 223–236.

Weisz, R., Triepke, J. et al. 2009. Evaluating the ecological sustainability of a ponderosa pine ecosystem on the Kaibab Plateau in Northern Arizona, Fire Ecology. 5(1): 100–115.

Using State-and-Transition Models to Project Cheatgrass and Juniper Invasion in Southeastern Oregon Sagebrush Steppe

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Abstract

Many threats are jeopardizing the sagebrush steppe of the Columbia Basin, including the spread of invasive species such as cheatgrass (*Bromus tectorum* L.) and the expansion of western juniper (*Juniperus occidentalis* Hook.) into historic shrub steppe. Native sagebrush steppe provides productive grazing lands and important habitat for many wildlife species, and managers are in need of landscape-scale tools to assess shrub steppe conversion risk and management options to maintain native shrub steppe. We used a state-and-transition modeling approach to project changes in sagebrush steppe vegetation across the landscape of southeastern Oregon. Models were constructed using both empirical data, including empirically derived fire probabilities, and expert opinion for processes that are still poorly documented, such as livestock grazing effects. With unrestricted grazing and no restoration treatments, future invasion by exotic annual grasses in warm, dry sagebrush steppe and juniper expansion into cool, moist sagebrush steppe are likely to accelerate in the next 50 years under current climatic conditions. Invasions are also likely to be spatially heterogeneous, depending on the mix of sagebrush steppe environments, current rangeland condition, disturbances, and management activities across the landscape. We conclude that state-and-transition models provide a useful framework for conceptualizing vegetation dynamics of sagebrush steppe systems, identifying gaps in knowledge, projecting future vegetation conditions, and identifying potential areas for restoration at landscape scales.

Keywords: *Bromus tectorum*, cheatgrass, invasive species, western juniper, *Juniperus occidentalis*, rangeland, sagebrush steppe, state-and-transition modeling.

Introduction

Sagebrush steppe ecosystems across much of the West have experienced significant declines over the last few decades (Connelly et al. 2004, Hemstrom et al. 2002). Among the major threats, intensive livestock grazing, species invasions, altered fire regimes, development, and climate change are all thought to contribute to the decline of shrub steppe (DiTomaso 2000, Jones 2000, Mack 1981, Miller et al. 2005). Rangelands provide an important source of forage for livestock, and degradation of shrub steppe may reduce the ability of rangelands to support livestock (Belsky 1996, Young and Clements 2009). Conversion of sagebrush steppe also occurs against a backdrop of increasing concern about loss of habitat for sagebrush-obligate species such as greater sage-grouse (*Centrocercus urophasianus*) (Connelly et al. 2004). Restoration of degraded shrub steppe can be exceedingly difficult due to the complex and often unpredictable interaction of site potential, fire, climate, invasive species, and management practices such as grazing (Di-Tomaso 2000, McIver and Starr 2001).

Exotic species invasions and native juniper expansion in particular have dramatically changed the landscape in eastern Oregon over the last century. Exotic annual grasses, such as cheatgrass (*Bromus tectorum* L.), ventenata (*Ventenata* Koeler spp.), and medusahead (*Taeniatherum caput-medusae* (L.) Nevski), have invaded many warm-dry sites, and have changed the vegetation structure and fire regime by forming dense, dry grass stands and promoting frequent fire (Pellant 1996, Whisenant 1990). Another contemporary threat to shrub steppe ecosystems comes from expansion of western juniper beyond its historic range. Western juniper (*Juniperus occidentalis* Hook.) is native to eastern Oregon but has expanded rapidly in the past 130 years due to fire suppression, reduction of fuels from livestock grazing, changes in precipitation patterns, and other factors (Burkhardt and Tisdale 1976, Miller et al. 2005). Juniper trees can deplete soil water, alter species composition and biodiversity of shrub steppe, increase soil erosion, reduce stream flows, and reduce forage production for livestock (Miller et al. 2000, 2005). Because of these complex threats and the vast extent of shrub steppe ecosystems, there is a need for a broad, multi-ownership perspective to examine landscape-scale trends in vegetation and effects of rangeland management.

One approach to examine vegetation dynamics, natural disturbances, and management across large areas is through the use of state-and-transition models (STMs). STMs are widely used in land management across both forest and rangeland landscapes (Forbis et al. 2006, Hemstrom et al. 2004, Holmes et al. 2010, Weisz et al. 2010). The models provide a conceptual framework for understanding ecological dynamics (Bestelmeyer et al. 2009, Briske et al. 2006, Stringham et al. 2003, Westoby et al. 1989), and challenge ecologists to define their assumptions in terms of vegetation composition (states and phases) and processes that cause vegetation change (transitions). In the process of building STMs, existing literature and data can be explored and gaps in our knowledge and data are revealed as areas for future study. STMs allow the user to easily test alternative hypotheses about vegetation dynamics and change by evaluating different models, and allow managers to compare alternative management strategies in terms of desired outcomes. In this study, we construct a suite of detailed STMs designed to capture the contemporary dynamics of southeastern Oregon shrub steppe ecosystems, and use them to project vegetation change 50 years into the future.

We focus on two major sagebrush steppe ecosystems of southeastern Oregon. The most common sagebrush sites in southeast Oregon are warm, dry lowland sites (called warm-dry sites) primarily occupied by Wyoming big sagebrush (*Artemisia tridentata* Nutt. ssp. *wyomingensis* Beetle and Young), basin big sagebrush (*Artemisia tridentata* Nutt. ssp. *tridentata*), bluebunch wheatgrass (*Pseudoregeneria spicata* (Pursh) A. Löve), Thurber needlegrass (*Achnatherum thurberianum* (Piper) Barkworth), and needle-and-thread (*Hesperostipa comata* (Trin. and Rupr.) Barkworth). In these areas, exotic annual grasses have invaded many sites and partially or wholly converted the shrub steppe to exotic grass. The second major sagebrush system is characterized by cool, moist upland sites (called cool-moist sites) primarily occupied by mountain big sagebrush (*Artemisia tridentata* Nutt. ssp. *vaseyana* (Rydb.) Beetle), low sagebrush (*Artemisia arbuscula* Nutt.), Idaho fescue (*Festuca idahoensis* Elmer), and bluebunch wheatgrass. Cool-moist sagebrush sites are more productive and less susceptible to invasion by exotic grasses, but many are rapidly converting into woodlands as western juniper expands its range. As part of the Integrated Landscape Assessment Project (ILAP), we project changes in sagebrush steppe vegetation under unrestricted livestock grazing and no restoration treatments, and focus on cheatgrass and juniper invasion as indicators of contemporary landscape change.

Methods

Study Area

We modeled sagebrush steppe vegetation types across a 5.3 million hectare (13.2 million acre) portion of southeastern Oregon, bounded by the Blue Mountains to the north and the foothills of the Cascade Mountains to the west (fig. 1). This area roughly corresponds to the Malheur High Plateau, Humboldt Area, and Owyhee High Plateau Major Land Resource Areas (MLRA) that are contained within the state of Oregon. The study area was comprised primarily of warm-dry sagebrush sites (59.8 percent) and cool-moist sagebrush sites (18.8 percent), with salt desert shrub, woodlands, playas, and other minor systems comprising

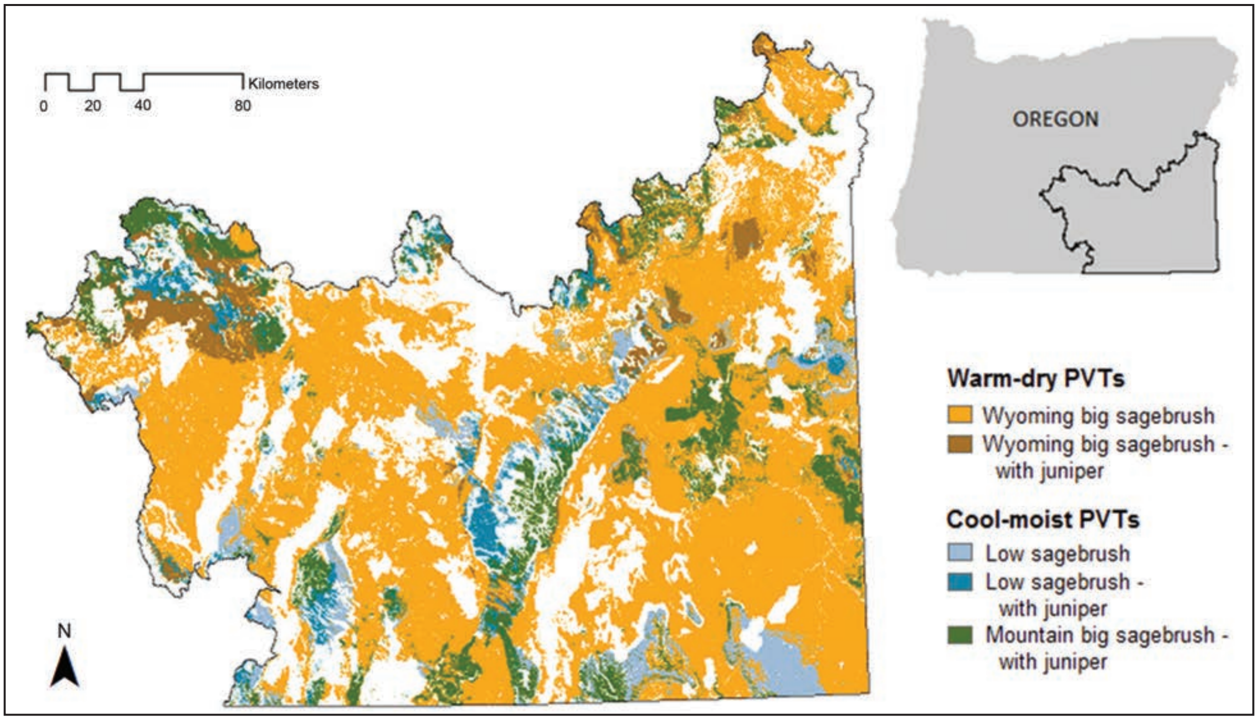

Figure 1—The sagebrush steppe in southeastern Oregon is divided into two major site types and five potential vegetation types (PVTs). Blank (white) areas represent other PVTs not modeled for this study, barren areas, urban areas, or agriculture.

the remaining undeveloped landscape. The Bureau of Land Management (BLM) was the primary land steward in the region, covering 72.7 percent, followed by private land and state agencies.

Spatial Layers

Three spatial layers were used to define our modeling units: potential vegetation type (PVT), ownership/allocation, and watershed. PVT described the vegetation potential of a site based on soils, climate, and disturbance regime. PVTs defined the spatial extent of each vegetation type as a modeling unit, and each STM simulated vegetation dynamics of alternative vegetation communities within a single PVT. We modeled five major PVTs, including two warm-dry types (Wyoming big sagebrush with and without juniper encroachment potential), and three cool-moist PVTs (mountain big sagebrush with juniper encroachment potential and low sagebrush with and without juniper encroachment potential) (fig. 1). Where plot data did not differentiate subspecies of big sagebrush, an elevation cutoff of 1200 m was applied, assuming that the mountain subspecies would occur above this elevation and the Wyoming subspecies would occur below this elevation. Potential vegetation was modeled using a random forest nearest neighbor imputation (RFNN) method (Crookston and Finley 2008, Ohmann and Gregory 2002), which related plant association data to grids of climate (PRISM climate group, Oregon State University) and topographic (National Elevation Dataset) environmental variables. Because projections of future vegetation condition must allow for expansion of juniper beyond its historic range, Wyoming big sagebrush and low sagebrush PVTs were divided into areas with and without potential for juniper encroachment in the PVT map using RFNN predictions of juniper cover. Mountain big sagebrush was considered susceptible to juniper invasion across its entire extent. All PVTs in this study were considered as sagebrush steppe potential, and thus any juniper presence was considered to be expansion beyond its historic range. The second spatial layer used to define modeling units was ownership/allocation, which categorized the landowner or land steward (BLM, Forest Service, private, tribal, state, and other) and management intent, ranked into five categories based on the intensity of intended use. For this study, the same STMs were run across all ownership/allocation levels, but the ownership/allocation layer will be used to inform varying grazing and restoration treatment levels for future studies. Third, we used 5th-field (10 digit) Hydrologic Unit Codes (HUCs) to define watersheds, downloaded from the United States Geological Survey (http://water.usgs.gov/GIS/huc.html). The combination of PVT, ownership/allocation, and watershed thus provided the spatial basis of our modeling, allowing us to stratify our model output in terms of site characteristics, management intent, and hydrologic unit location.

Additionally, we initialized our STMs with spatial maps of current vegetation. Current vegetation was modeled using a RFNN method. The mapping method was similar to PVT mapping, but in this case the RFNN method imputed field plot data to pixels using the association between field data (species composition and cover), grids of environmental data, and LANDSAT TM (thematic mapper) imagery from 2000. Vegetation communities in the current vegetation map were linked to states and phases in the STMs using a series of rule sets that allocated every pixel in the landscape into a state or phase.

State-and-Transition Models (STMs)

We constructed STMs using the Vegetation Dynamics Development Tool (VDDT) (ESSA Technologies 2007) to characterize vegetation dynamics of the major sagebrush ecosystems in southeastern Oregon and project future vegetation change. VDDT allows users to divide the landscape into distinct combinations of vegetation cover and structure (states and phases), linked together by processes (transitions) such as succession, disturbance, and management activities. Users define a pathway for each transition and its annual probability of occurring, and VDDT uses Monte Carlo simulations to project landscape change over time. VDDT is a non-spatial model, and tracks each simulation cell independently of neighboring cells. Simulations were run in the Path Landscape Model (Apex RMS and ESSA Technologies), which uses VDDT as a simulation engine but allows the user to run multiple STMs and scenarios (such as alternative management options) in a single landscape analysis.

One STM was constructed for each PVT, describing alternative vegetation states and phases within each potential vegetation unit. STMs developed by the BLM for the Malheur High Plateau MLRA in southeastern Oregon (Evers 2010) and STMs built by the USDA Forest Service for the Blue Mountains of northeastern Oregon[1] were used heavily to aid in constructing and parameterizing models. Conceptually, our sagebrush STMs can be divided into a few broad states (large boxes, fig. 2), with community phases (smaller boxes within states, fig. 2) describing varying combinations of cover and structure (Bestelmeyer et al. 2009). Major states include shrub steppe, exotic grass, juniper woodland, juniper with exotic grass, and seeded grass. Semi-degraded phases represent disturbance-impacted vegetation that is recoverable to native conditions (dashed lines, fig. 2) but is at-risk of crossing a threshold to an alternative state. Note that each STM varies; not all states and phases in figure 2 are present in each model, and transition probabilities vary substantially among STMs, particularly between PVTs on warm-dry and cool-moist sites.

Specific criteria were used to define the vegetation composition and structure of each state and phase within each STM. Herbaceous composition was used as an indicator of native, semi-degraded, or exotic-dominated condition. Exotic grass states were defined by a minimum absolute and relative cover of exotic annual grasses, primarily cheatgrass and other invasive bromes, ventenata, medusahead, vulpia (*Vulpia bromoides* (L.) Gray), and others. Native states were defined by a minimum absolute and relative cover of 8 grass species sensitive to disturbance (decreasers such as bluebunch wheatgrass, Idaho fescue, needle- and-thread, some *Achnatherum* and *Elymus* species, and others). Cover

[1] Personal communication: Dave Swanson, former area ecologist, USDA Forest Service, Baker City, OR.

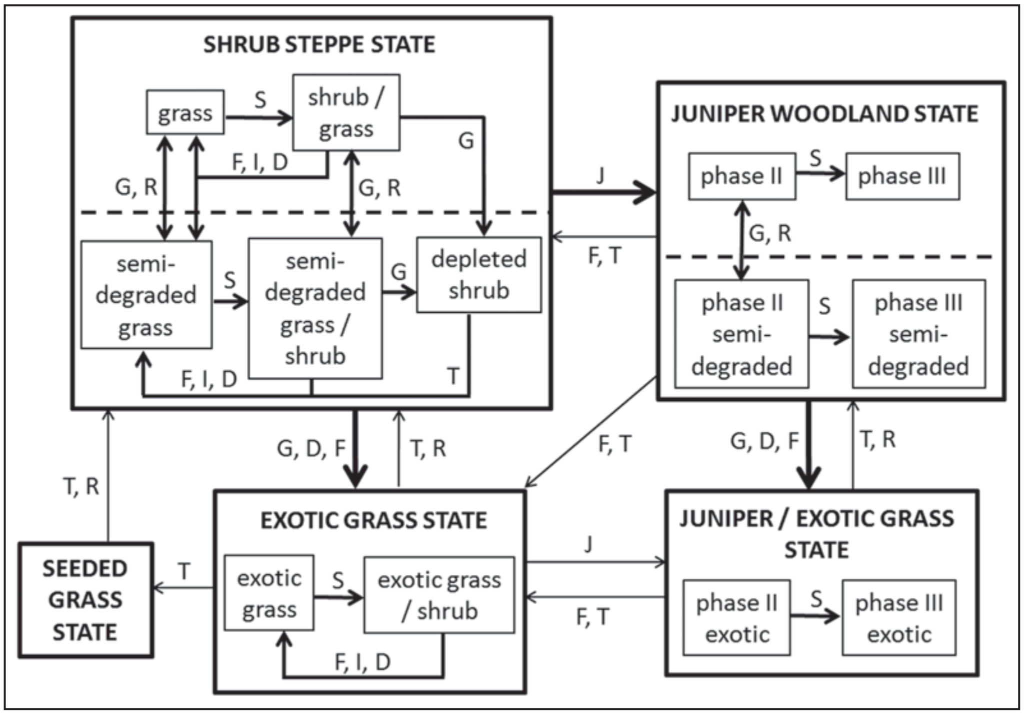

Figure 2—A conceptual state-and-transition model (STM) diagram of vegetation dynamics in the Wyoming big sagebrush with juniper potential vegetation type (PVT), showing major model states (large boxes), representative phases within states (small boxes), and transitions that link states and phases (arrows). Transitions include succession (S), fire (F), juniper establishment (J), grazing (G), insect outbreaks (I), drought (D), recovery (R), and management treatments (T). Dashed lines separate native from semi-degraded condition, and represent reversible thresholds where recovery to native condition is more likely. Transitions between states, however, are often irreversible without management intervention or major disturbance, and low probability recovery transitions are shown as thin arrows. Juniper phases II and III represent increasing juniper dominance (Miller et al. 2005), and phase I juniper woodlands are omitted for simplicity. Each individual STM varies and this figure represents a simplified model for a single PVT.

thresholds are variable among models. In PVTs where seeding of nonnative species occurs, seeded states were defined by *Agropyron* species and others that are commonly seeded in rangelands. Anything that did not meet the minimum threshold for these indicator species was considered to be in a semi-degraded state. Depleted shrub was defined as high shrub cover (≥25 percent) and low grass cover (<5 percent) and is only present in warm-dry sagebrush models. Juniper woodlands were divided into phases I, II and III based on Miller (2005), where phase I represented shrub steppe with scattered juniper, phase II represented codominant juniper and shrubs/grasses, and phase III indicated mature woodlands where juniper was dominant. Juniper cover classes of 2–10 percent, 11–20 percent and >20 percent were used to distinguish phases I, II and III, respectively, based on feedback from expert reviewers.

Disturbance dynamics, including succession, natural disturbance (e.g., wildfire, drought, and insects) and management transitions (e.g., seeding, cutting, and prescribed fire) were modeled by specifying transition pathways between boxes and defining annual probabilities of growth or disturbance events occurring (table 1). Models were constructed so the primary mechanism for degradation (exotic grass invasion) into sagebrush steppe was through the interaction of grazing with fire or drought disturbance (Curtin 2002, Evers 2010, Loeser et al. 2007). In warm-dry sites, recovery of native species in exotic grass states was slow and required rest from grazing disturbance before it

Table 1—Transitions used to model sagebrush steppe vegetation dynamics in southeastern Oregon

Transition	Description
Replacement fire	Wildfire that results in return to early-successional phases. Replacement fire is modeled in all states and phases.
Mosaic Fire	Patchy fire that thins shrubs or trees. Mosaic fire occurs in most phases except those dominated by exotics, closed or depleted shrub, and phase III juniper.
Surface fire	Surface fire that burns the woodland understory (phase III juniper with exotic grass only).
Maintenance grazing	Low-impact grazing that does not affect plant community composition or structure.
Moderate grazing	Grazing that causes successional change by increasing shrub cover but is not severe enough to promote exotic grass invasion.
Graze degrade	Heavy grazing that causes degradation from native to semi-degraded to exotic grass-dominated shrub steppe. The transition probability is low, as we assume that the interaction of grazing with disturbance (Post-disturbance graze degrade) is more likely to lead to degradation.
Post-disturbance Graze degrade	Heavy grazing after major disturbance, leading to semi-degraded condition or exotic grass-dominated states. This transition can only occur within two years following a fire or drought, and the transition probability is 10-fold higher than Graze degrade.
Drought	Moderate multi-year drought that does not cause vegetation change.
severe drought	Drought severe enough to kill shrubs, causing a transition to early-successional shrub steppe. This transition occurs only once every 100-200 years.
Natural regeneration	Recovery of native herbaceous vegetation in a degraded site by natural regeneration. This transition usually requires rest from grazing to occur.
Juniper establishment	Juniper establishment that converts shrub steppe to phase I juniper.
Insect Outbreaks	Cyclical outbreaks of sagebrush-defoliating insects, occurring once occur every 20-30 years.

would begin to occur. In cool-moist sites, recovery from exotic grass states was modeled to occur automatically unless it was heavily grazed, reflecting the higher competitive ability of native bunchgrasses in mesic sites. Fire probabilities varied among states and phases based on the cover of exotic grass species (see “Fire Probabilities”). Insect outbreaks and severe drought affected vegetation by thinning shrub cover, and juniper establishment events (where applicable) occurred from late-successional shrub steppe into phase I woodlands (Evers 2010). Management transitions were built into the models but were deactivated for this study to evaluate future landscape condition without active management.

Fire Probabilities

We used the Monitoring Trends in Burn Severity (MTBS) data to derive fire probabilities and interannual variability in fire year (Eidenshink et al. 2007, www.mtbs.gov). The MTBS data set is a publicly-available, 25-year record of all fires >405 hectares (>1,000 acres) in size across the United States from 1984 to 2008. It includes fire perimeters and burn severity ratings for each fire occurring in the study area from 1984 to 2008. For this study, we used fire perimeters for ~250 fires to infer the proportion of the landscape in each PVT group burned annually. We overlaid the yearly maps of fire perimeters with a map of PVT groups (warm-dry and cool-moist) and exotic grass cover groups in a GIS. Exotic grass cover was derived from our current vegetation map, and divided into three groups: 0-10 percent, representing places with little to no invasion; 10-25 percent, representing areas that are semi-degraded; and >25 percent, where exotic grasses dominate the herbaceous layer. We extracted the landscape proportion burned and calculated annual fire probabilities for each combination of PVT group and exotic grass group (table 2), and assigned these probabilities to wildfire transitions in the STMs. Fire return intervals (FRIs) were calculated as the inverse of fire probability.

Running Simulations

Simulations were run in the Path model to project vegetation change 50 years into the future. One model was run for

Table 2—Annual fire probabilities and corresponding fire return intervals derived from Monitoring Trends in Burn Severity (MTBS) data for the two major sagebrush steppe site types and three levels of exotic grass cover. Numbers reflect fire return intervals under current levels of fire suppression

Site Type	Exotic grass cover	Annual probability	Fire return interval
Cool-moist sagebrush steppe	0-10 percent	0.0068	148
	10-25 percent	0.0089	112
	>25 percent	0.0173	58
Warm-dry sagebrush steppe	0-10 percent	0.0063	160
	10-25 percent	0.0114	88
	>25 percent	0.0179	56

each modeling unit (combination of PVT, watershed, and ownership/allocation). Where modeling units were <405 hectares (<1,000 acres) in size, a rule set was applied to combine small units within a watershed with other similar vegetation or management types. Where small units did not meet the criteria to combine with others, they were dropped from the analysis. The study area consisted of 889 modeling units in the major sagebrush steppe PVTs that were large enough to be retained for analysis, with <5 percent of the landscape not modeled due to small modeling unit size. Each STM was run for 30 Monte Carlo simulations with random draws of fire severity year and insect outbreak occurrence, and we reported average trends.

Model Output

To simplify results for graphical purposes, we combined states and phases into seven groups, including native, semi-degraded, and exotic shrub steppe, exotic grass, and phase I, II, and III juniper. Seeded states were not included for simplicity. Although VDDT is a non-spatial model, current and future projections of exotic grass and juniper woodlands can be summarized and mapped back to our spatial modeling units. We summarized the percent of pixels within each modeling unit that contained exotic grass or juniper woodland states, and displayed a single value scaled between 0 (low invasion level) and 1 (high invasion). Exotic grass maps displayed all exotic grass phases, and juniper woodland maps depicted woodlands in phases II and III only, since phase I juniper is similar to sagebrush steppe.

Results

Current and Projected Future Conditions

Our imputed current vegetation conditions (2000) for warm-dry sites across the extent of the study area indicate that much of the sagebrush steppe (~70 percent) was semi-degraded, with exotic grass encompassing ~15 percent of the landscape. In cool-moist sites, current vegetation maps show that half of the landscape was semi-degraded shrub steppe, one-third was native shrub steppe, and juniper encroachment affected <15 percent of the cool-moist shrub steppe. Where juniper had encroached it was still largely in phase I, with shrubs and grasses still dominant.

STM projections to year 2050 indicate a decline in rangeland condition in both warm-dry and cool-moist sites, assuming unrestricted grazing and no restoration treatments (fig. 3). Much of the current semi-degraded sagebrush steppe is projected to convert to exotic grass, increasing to nearly half of the landscape in warm-dry sites. In cool-moist sites, model projections indicate an increase in juniper woodlands to more than half of the extent of cool-moist PVTs, with rapid expansion of phase I in the first 25 years and conversion to phase II in the second half of the simulation. In both site types, native and semi-degraded shrub steppe decline as they are converted to exotic grass or juniper woodlands.

Invasion Maps

STM projections suggest that much of the landscape is likely to convert to either exotic grass or juniper woodland,

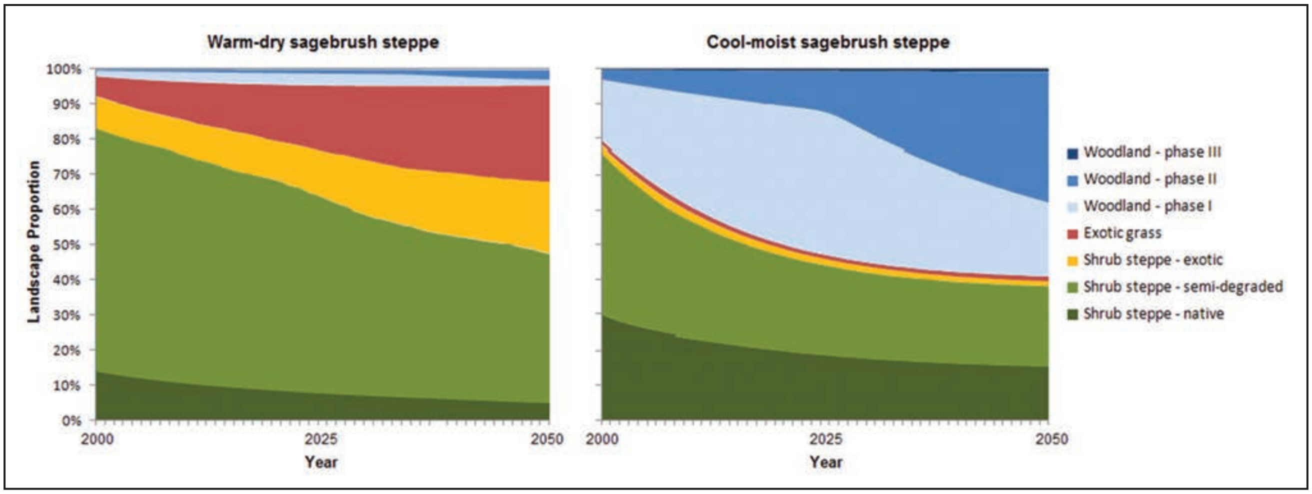

Figure 3—Projected vegetation change from 2000-2050 for warm-dry sagebrush steppe (left) and cool- moist sagebrush steppe (right). Graphs show average modeled landscape proportion across 30 Monte Carlo simulations for southeastern Oregon.

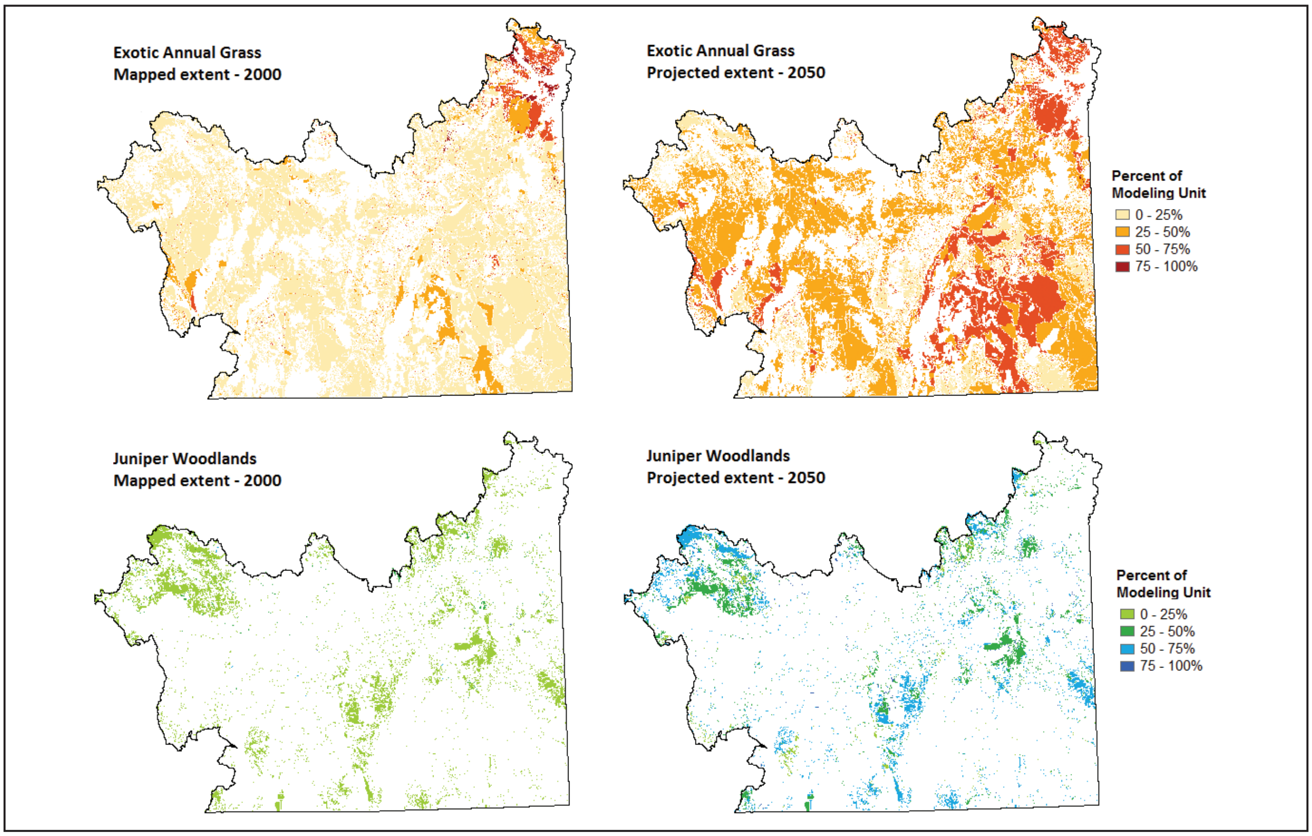

Figure 4—Exotic annual grass (top) and juniper woodlands (bottom) as mapped in 2000 (left) and projected for year 2050 using state-and-transition models (right). Colors depict the percent of each mapped modeling unit in an invaded state. Blank (white) areas were not modeled or represent PVTs where invasion cannot occur.

but that the severity of invasion is highly variable across the landscape (fig. 4). We report results at the modeling unit level because the STMs are non-spatial, and thus pixel-by-pixel projections are not possible.

Discussion

Model projections indicate that much of the sagebrush steppe landscape of southeastern Oregon is likely to experience invasion by exotic grasses or juniper under a scenario of unrestricted grazing and no restoration treatments. Exotic grass in particular is projected to expand across a large extent of southeastern Oregon, whereas juniper is more limited by site potential in drier (warm-dry) sites. However, these projections represent a worst case scenario, and managing grazing (particularly by limiting grazing during and after major disturbances such as fire and drought) or implementing restoration treatments could result in much improved landscape condition. Invasion risk maps show that the projected level of exotic grass and juniper invasion is highly heterogeneous across the landscape of southeastern Oregon. This heterogeneity stems from varying susceptibility of each PVT to invasion and variation in current vegetation condition at the initialization of the model runs (year 2000 conditions). Although the maps provide coarser-scale projections summarized at the modeling unit level instead of individual pixel-by-pixel predictions, they can nonetheless aid in large landscape-level assessment of rangeland condition and invasion risk, and provide a broader context for management decisions and prioritization across the study area.

The trends in exotic grasses largely reflect grazing effects and the interaction between grazing and major disturbance (fire and drought). We assumed that heavy grazing in warm-dry sites leads to degradation by reducing the presence of native grasses while providing a competitive advantage to nonnative species, which is exacerbated under conditions of abiotic stress. Under heavy grazing, a feedback loop is created whereby grazing leads to more exotic grasses, which in turn leads to more frequent wildfires, and leads to an even greater exotic grass presence. Furthermore, once range condition has deteriorated to semi-degraded conditions, some disturbances even in the absence of grazing can lead to dominance by exotic grasses. Grazing also removes grasses in sites that are susceptible to juniper invasion, which provides greater opportunity for juniper establishment under existing shrubs and reduces fuel that would historically cause establishing juniper woodlands to periodically burn (Burkhardt and Tisdale 1976, Miller and Wigand 1994). Much of the landscape that was historically shrub steppe is now considered to be vulnerable to future juniper expansion (fig. 1), and our projections suggest rapid juniper expansion, as has been documented on many of these sites (Miller et al. 2005).

STMs have been adopted by many land management agencies because of their useful characteristics for organizing ecological knowledge and informing management. They provide a relatively simple and intuitive modeling framework that managers can use as a mid or broad-scale land management tool. STMs can be used as conceptual models as well as predictive models, and they force ecologists to formalize their assumptions about landscape dynamics. They are easily incorporated into sensitivity analysis to test the importance of different processes under a certain set of assumptions, and can challenge and expand ideas about rangeland ecosystem dynamics and management. Constructing models can also be valuable for highlighting areas where little empirical data exists. We used a variety of data sources to construct our STMs, including empirical data to construct fire probabilities (Eidenshink et al. 2007) and drought frequencies (Knapp et al. 2004), published (Evers 2010) and unpublished STMs, and several experts to construct our models. The STM framework can readily accommodate new data and information as it becomes available to test our assumptions and understanding of sagebrush steppe ecosystems.

A novel aspect of our study was the inclusion of MTBS fire perimeter data to quantitatively derive fire return intervals for each site type and varying levels of exotic annual grass invasion (table 2). It was particularly important to capture the effects of exotic grass in our analysis, since exotic grasses can dramatically increase fire frequency and severity (Pellant 1996, Whisenant 1990) and fire probabilities are expected to vary among states and phases in the STMs. Our analysis assumes a similar level of exotic grass cover over the 25-year record, but is likely to be more robust to interannual variability in grass productivity and cover since we group exotic grass cover into three broad categories. Consistent with previous studies, we detected an increase in fire with increasing exotic grass cover, although fire return intervals are not as frequent as some previous studies suggest (Evers 2010, Pellant 1996, Whisenant 1990). Because the MTBS record has captured fires under a fire suppression policy, our model projections assume a fire suppression policy and effectiveness similar to that of recent decades. The MTBS data provides the most detailed spatial record of wildfire occurrences we are aware of, but it is likely that wildfires are underreported to some degree, particularly on nonfederal lands and earlier in the recorded history (1980s). Although the MTBS data set has several limitations, we maintain that the benefits of using over two decades of spatial, quantitative data outweigh the limitations of the data set.

Although STMs have proven useful to many land managers and rangeland scientists, various drawbacks to the approach limit the interpretations we can make with STMs. Non-spatial STMs by nature cannot model spatial processes explicitly or incorporate fine-scale site variation, resulting in generalized predictions that can only be applied at mid to broad spatial scales. STMs are also not mechanistic, and rely upon the modeler to determine the effects of disturbance and management processes and how they cause state and phase change. Most STMs, including those presented here, rely at least in part on expert judgment to determine transition pathways and probabilities, and therefore each expert will likely build a slightly different model. Even where some data are available, it is generally not available across large landscapes, adding uncertainty about the effects of environmental heterogeneity on transition probabilities. Given these limitations, we frame STMs as working hypotheses that describe the state of the knowledge about each ecological system given various assumptions. They are designed to conceptualize and project vegetation dynamics across broad spatial scales, and should be coupled with field studies to refine local vegetation dynamics where possible. Lastly, our models do not address climate change effects, as our projections are relatively short-term (to year 2050), but methodology is being developed as part of ILAP to incorporate climate change effects in our STMs (Kerns et al. 2012).

In this study, we demonstrate the utility of STMs for evaluating the risk of sagebrush steppe conversion to exotic grass or juniper across the landscape of southeastern Oregon. Although we projected large increases in both exotic grasses and juniper, we only ran a worst-case scenario of no restoration treatments and unrestricted grazing. With the models and data available we can now begin to incorporate alternative management scenarios to address a range of questions such as: given a limited budget, what combination of fuel treatments, seeding, grazing levels, and/or juniper control could maintain or improve current levels of good condition sagebrush? Where should we prioritize such treatments? How do our projections relate to habitat for species such as sage-grouse? How might our answers differ under a changing climate? The resulting projections and maps of model output can be useful to public and private land managers in answering important management questions and providing a broader context for landscape treatments and restoration.

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References

Belsky, A.J. 1996. Viewpoint: Western juniper expansion: Is it a threat to arid Northwestern ecosystems? Journal of Range Management. 49: 53–59.

Bestelmeyer, B.T.; Tugel, A.J.; Peacock, Jr., G.L. [and others]. 2009. State-and-transition models for heterogeneous landscapes: a strategy for development and application. Rangeland Ecology and Management. 62: 1–15.

Briske, D.D.; Bestelmeyer, B.T.; Stringham, T.K. [and others]. 2006. Recommendations for development of resilience-based state-and-transition models. Rangeland Ecology and Management. 61: 359–367.

Burkhardt, J.W.; Tisdale, E.W. 1976. Causes of juniper invasion in southwestern Idaho. Ecology. 57(3): 472–484.

Connelly, J.W.; Knick, S.T.; Schroeder, M.A. [and others]. 2004. Conservation assessment of Greater sage-grouse and sagebrush habitats. Unpublished report. Cheyenne, WY: Western Association of Fish and Wildlife Agencies. 610 p.

Crookston, N.L.; Finley, A.O. 2008. yaImpute: An R package for kNN imputation. Journal of Statistical Software. 23: 1–16.

Curtin, C.G. 2002. Livestock grazing, rest, and restoration in arid landscapes. Conservation Biology. 16(3): 840–842.

DiTomaso, J.M. 2000. Invasive weeds in rangelands: Species, impacts, and management. Weed Science. 48(2): 255–265.

Eidenshink, J.; Schwind, B.; Brewer, K. [and others]. 2007. A project for monitoring trends in burn severity. Fire Ecology. 3: 3–21.

ESSA Technologies, Ltd. 2007. Vegetation Dynamics Development Tool, User's Guide, Version 6.0. ESSA Technologies Ltd., Vancouver, BC.

Evers, L. 2010. Modeling sage-grouse habitat using a state-and-transition model. Corvallis, OR: Oregon State University. Ph.D. dissertation. 180 p.

Forbis, T.A.; Provencher, L.; Frid, L. [and others]. 2006. Great Basin land management planning using ecological modeling. Environmental Management. 38: 62–83.

Hemstrom, M.A.; Ager, A.A.; Vavra, M. [and others]. 2004. A state and transition approach for integrating landscape models. In: Hayes, J.L.; Ager, A.A.; Barbour, R.J., eds. Methods for integrated modeling of landscape change: Interior Northwest Landscape Analysis System. Gen. Tech. Rep. PNW-GTR-610. Portland, OR: U.S. Department of Agriculture, Forest Service, Pacific Northwest Research Station: 17–32.

Hemstrom, M.A.; Wisdom, M.J.; Hann, W.J. [and others]. 2002. Sagebrush-steppe vegetation dynamics and restoration potential in the interior Columbia Basin, U.S.A. Conservation Biology. 16: 1243–1255.

Holmes, A.L.; Miller, R.F. 2010. State-and-transition models for assessing grasshopper sparrow habitat use. Journal of Wildlife Management 74: 1834–1840.

Jones, A. 2000. Effects of cattle grazing on North American ecosystems: a quantitative review. Western North American Naturalist. 60: 155–164.

Kerns, B.K.; Hemstrom, M.A.; Conklin, D.; Yospin, G.; Johnson, B.; Bachelet, D.; Bridgham, S. 2012. Approaches to incorporating climate change effects in state and transition simulation models of vegetation. In: Kerns, B.K.; Shlisky, A.J.; Daniel, C.J., tech. eds. Proceedings of the First Landscape State-and-Transition Simulation Modeling Conference, June 14–16, 2011, Portland, Oregon. Gen. Tech. Rep. PNW-GTR-869. Portland, OR: U.S. Department of Agriculture, Forest Service, Pacific Northwest Research Station: 161–171.

Knapp, P.A.; Soule, P.T.; Grissino-Mayor, H.D. 2004. Occurrence of sustained droughts in the interior Pacific Northwest (A.D. 1733–1980) inferred from tree-ring data. Journal of Climate. 17(1): 140–150.

Loeser, M.R.R.; Sisk, T.D.; Crews, T.E. 2007. Impact of grazing intensity during drought in an Arizona grassland. Conservation Biology. 21: 87–97.

Mack, R.N. 1981. Invasion of B*romus tectorum* L. into western North America: an ecological chronicle. Agro-Ecosystems. 7: 145–165.

McIver, J. and Starr, L. 2001. Restoration of degraded lands in the interior Columbia River basin: passive vs. active approaches. Forest Ecology and Management. 153: 15–28.

Miller, R.F.; Bates, J.D.; Svejcar, T.J. [and others]. 2005. Biology, ecology, and management of western juniper. Tech Bull 152. Corvallis, OR: Oregon State University, Agricultural Experiment Station. 82 p.

Miller, R.F.; Svejcar, T.J.; Rose, J.A. 2000. Impacts of western juniper on plant community composition and structure. Journal of Range Management. 53: 574–585.

Miller, R.F.; Wigand, P.E. 1994. Holocene changes in semiarid pinyon-juniper woodlands. BioScience. 44: 465–474.

Ohmann, J.L.; Gregory, M.J. 2002. Predictive mapping of forest composition and structure with direct gradient analysis and nearestneighbor imputation in coastal Oregon, U.S.A. Canadian Journal of 6 Forest Research. 32: 725–741.

Pellant, M. 1996. Cheatgrass: the invader that won the West. Unpublished Report. Interior Columbia Basin Ecosystem Management Project. Bureau of Land Management, Idaho State Office. 23 p.

Stringham, T.K.; Kreuger, W.C.; Shaver, P.L. 2003. State and transition modeling: An ecological process approach. Journal of Range Management. 56: 106–113.

Weisz, R.; Triepke, J.; Vandendriesche, D. [and others]. 2010. Evaluating the ecological sustainability of a pinyon-juniper grassland ecosystem in northern Arizona. In: Jain, T.B.; Graham, R.T.; Sandquist, J., eds. Integrated management of carbon sequestration and biomass utilization opportunities in a changing climate: Proceedings of the 2009 National Silviculture Workshop. RMRS-P-61. Boise, ID: U.S. Department of Agriculture, Forest Service: 321–336.

Westoby, M.; Walker, B.; Noy-Mier, I. 1989. Opportunistic management for rangelands not at equilibrium. Jounal of Range Management. 42(4): 266–274.

Whisenant, S.G. 1990. Changing fire frequencies on Idaho's Snake River Plains: ecological and management implications. In: Proceedings: symposium on cheatgrass invasion, shrub die-off, and other aspects of shrub biology and management. Gen. Tech. Rep. INT-276. Ogden, UT: U.S. Department of Agriculture, Forest Service, Intermountain Research Station: 4–10.

Young, J.A.; Clements, C.D. 2009. Cheatgrass: fire and forage on the range. Reno, NV: University of Nevada Press. 348 p.

Landscape Development and Mule Deer Habitat in Central Oregon

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Abstract

This research explored the ecological consequences of rural residential development and different management regimes on a tract of former industrial timberland in central Oregon known as the Bull Springs. Forage quality and habitat suitability models for mule deer (*Odocoileus hemionus*) winter range were joined to the outputs of a spatially explicit vegetation dynamics model under two management scenarios. In one scenario, the tract was managed as a working forest excluding development, and in the other, development was allowed to occur at historical rates. Landscape pattern analysis was used to measure differences between the outcomes of the two scenarios. Our efforts showed that allowing development on the tract could potentially lead to greater isolation, smaller habitat patches, and decreased extensiveness of patches used for foraging across mule deer winter range. Patches providing multiple habitat functions also became more isolated and less numerous in our simulations. Although neither scenario prevented habitat degradation, restricting development on Bull Springs had slightly more favorable simulated outcomes for forage and multifunctional habitat conditions. Management of this tract as a working forest in a region under pressure for more residential development could reduce the negative effects of development on an iconic species in the region. This research provides insight into how the land use change trajectory of a small portion of the landscape can influence the larger ecological conditions of a region undergoing rapid rural residential development.

Keywords: Mule deer; landscape ecology; rural residential development; alternative development scenarios; habitat suitability.

Introduction

Low-density housing has expanded into rural lands and the wildland-urban interface (WUI) across the United States (Theobald and Romme 2007) and represents an accelerating phenomenon in the West (Brown et al. 2005) capable of altering social and ecological landscapes. The relative permanency of development distinguishes it from extractive land uses such as logging and grazing (Hansen et al. 2005), and its impacts extend beyond the walls of individual homes. Disturbance regimes, biodiversity, and myriad other ecosystem services have all demonstrated sensitivity to the extent and nature of residential development on rural lands (Dale et al. 2005, Hansen et al. 2005, Rindfuss et al. 2004). For example, individual residences create localized disturbance zones for wildlife (Theobald et al. 1997), and developments propagate disturbances, especially fire and invasive species, into adjacent undeveloped lands (Hansen and DeFries 2007). The cumulative effects of individual land use change decisions can lead to substantial ecological impacts (Theobald et al. 2005), and uncoordinated residential development over time can have disproportionate effects on potential wildlife habitat (Spies et al. 2007) and disrupt migration corridors (Hansen and Defries 2007).

Central Oregon has exemplified the western American trend of residential development taking place on forested land previously managed for timber and other non-residential uses. When plans emerged to develop housing on a

13 000-ha private tract of former timberland just west of Bend, the region's largest city, questions arose about how different management actions on the tract would affect the region's natural resources. Although the tract, known as the Bull Springs, was zoned for forest use and a minimum lot size of 97 ha at the time of the proposal, there was a precedent of converting land zoned for forest use to residential use in other parts of the state (Lettman 2002, 2004). Furthermore, the tract overlaps a substantial portion of the observed winter range for mule deer (*Odocoileus hemionus*). One potential outcome of land use, ownership, and management changes in Central Oregon is a shift in the amount, quality, and spatial pattern of habitat for wildlife such as mule deer. Mule deer use higher elevation woodlands in the summer, however, their movement to lower elevation valleys and sagebrush in the winter could bring them into contact with the proposed development. As mule deer require large home ranges (up to 500 ha for solitary bucks) and long dispersal distances (often exceeding 1 km), the proposed development could fragment important habitat patches or disrupt migration corridors. Development in the winter range also has potential to impact mule deer foraging spaces as well as valuable hiding habitats that provide camouflage from predators and thermal cover habitats that protect them from wind and sun (Csuti et al. 1997, Johnson and O'Neil 2001, Oregon Department of Fish and Wildlife (ODFW) 2003). An iconic species in the region and an important source of game (ODFW 2003), mule deer and their winter range requirements have often been at the center of the debate over the possible policy options for guiding land use change on the Bull Springs tract following its ownership change, and so we chose mule deer habitat as the indicator of the ecological outcomes of alternative policies for this study.

To better understand the impacts of shifts in the landscape patterns resulting from changes in land ownership and land use on mule deer winter range habitat, we analyzed differences in the landscape patterns between two management options by examining potential habitat configurations simulated under each option. We present methods that quantify the landscape consequences of differing policy options and, by providing a means to weigh alternative policy scenarios, may be useful to decision makers. Two alternative development scenarios for the Bull Springs tract were considered plausible in the future:

(1) The tract is managed as a working forest and not developed (WF).

(2) The tract becomes developed for residential use at historical rates (DEV).

Development was allowed to occur outside of the Bulls Springs regardless of the scenario.

To understand how different policy options might influence the spatial arrangement of mule deer habitat, we used spatial pattern metrics to quantify change under both scenarios for a mule deer forage quality model and a habitat suitability model. These types of analyses can help identify the effects of working forest management on mule deer habitat in the region, and show the role the Bull Springs tract could play in the larger suite of developments forecast to occur within the region.

Data and Methods

Vegetation dynamics in the region surrounding the Bull Springs tract and mule deer winter range were simulated under two scenarios for 60 years. The two scenarios were based on the proposed options for the tract arising in the public policy debate in Oregon. The Working Forest (WF) scenario excluded development from the Bull Springs tract and managed it for restoration goals. The Development (DEV) scenario allowed development to occur at historical rates on the tract. We classified the initial (2000) and final (2060) vegetation conditions into forage and habitat suitability categories using wildlife-habitat relationship models (described below). These categorical maps were then analyzed using spatial pattern metrics to quantify the changes in mule deer habitat and compare the ecological outcomes of the scenarios. Analyses were conducted at three spatial extents: the full study area, mule deer winter range, and Bull Springs boundary (fig. 1).

Study Area

Centered on the Bull Springs tract, the study area is 430 000 ha of the dry, eastern slopes of the Cascade Range and the western edge of Oregon's high desert and ranges in

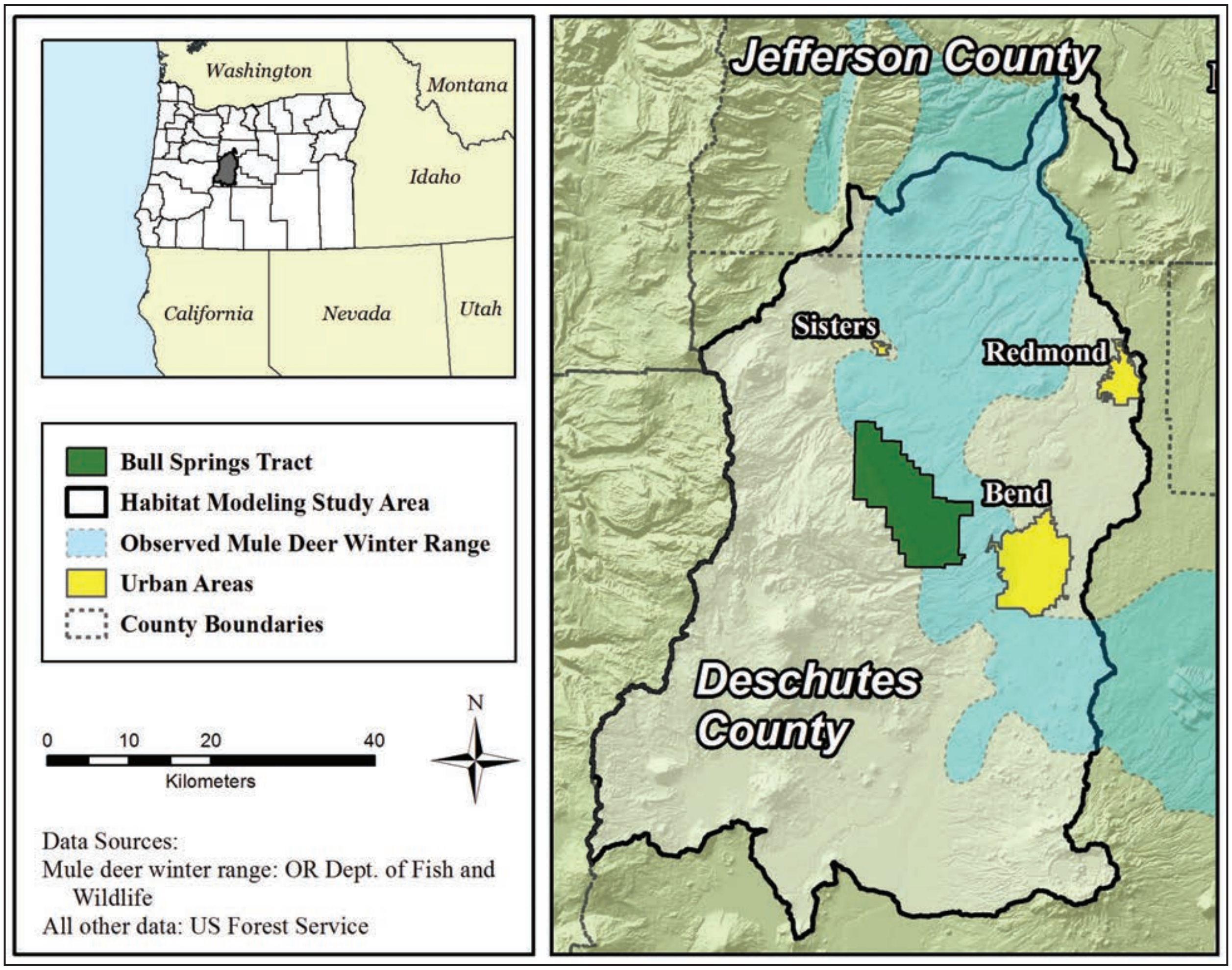

Figure 1—The study area and its ownership context. The solid black line shows the boundary of the study area used in the STM; the dashed line shows the observed extent of the mule deer winter range. The Bull Springs tract is shown in the center of the modeling area.

elevation from 590 m to 3150 m (fig. 1). High elevation forests are dominated by spruces (*Picea* spp.) and firs (*Abies* spp.) whereas lower regions of the mountain slopes consist primarily of ponderosa pine (*Pinus ponderosa* C. Lawson) and lodgepole pine (*Pinus contorta* Douglas ex Louden). East of the foothills, the landscape is dominated by juniper (*Juniperus occidentalis* Hook.) and sagebrush (*Artemisia* spp.) plant communities. With precipitation ranging from 25.4 cm to 239.7 cm annually with large seasonal variation (Thorson et al. 2003) and soils with low water retention capacity, available soil water is often the limiting factor for plant growth (Simpson 2007). The ownership landscape is a mixture of federal and private ownerships. The USDA Forest Service (FS) and the U.S. Bureau of Land Management (BLM) administer much of the land in the study area, with 29.4 percent in private industrial and nonindustrial ownership. The mapped mule deer winter range[1] consists of 133 100 ha (30.7 percent) of the total study area. The Bull Springs tract covers 3.0 percent of the total study area and 9.8 percent of the observed mule deer winter range (fig. 1).

Vegetation Dynamics

Spatially-explicit state and transition models (STMs) were used to simulate residential development and vegetation

[1] Personal communication with Glenn Ardt, Biologist, Oregon Department of Fish and Wildlife (ODFW).

dynamics. STMs have been applied to various ecological systems in which multiple stable states are possible (Westoby et al. 1989, Hemstrom et al. 2007, Vavra et al. 2007, Petersen et al. 2009). Vegetation was simulated for 60 years (year 2000 to 2060) using a suite of STMs representing individual development stages, or *states*, within potential vegetation types (PVTs) for climax communities. PVTs are vegetation assemblages that take into account physical settings and species communities (Hall 1998). States are defined by a *cover type*, typically the dominant species or vegetation type, and a *structural stage* that describes the vegetation size class, canopy cover, and vertical layering. Transitions, such as wildfire, management activities, and development, defined possible pathways between states. Base transition rates control the speed at which vegetation assemblages change from one state to another given a particular transition, however, rates may be modified using multipliers to increase or decrease base rates to represent, for example, different intensities of the same management type among ownership types. The order, occurrence, and location of transitions are determined stochastically for each annual time step. Other transition-inducing mechanisms modeled were defoliators (e.g., western pine beetle) and parasites (e.g., mistletoe). We used the Tool for Exploratory Landscape Scenario Analyses (Version 3.06) to model landscape dynamics and the Vegetation Dynamics Development Tool (Version 6.0.25) to design, build, and calibrate the STMs (ESSA 2007, ESSA 2008). Models were designed by local ecologists or derived from previous work (Hemstrom et al. 2007) and ongoing planning activities. Fourteen potential vegetation types were modeled, three shrub steppe types and 11 forest types. PVTs modeled were: mountain shrub/meadow, Wyoming big sage/juniper, mountain big sage/juniper, ponderosa pine dry (pumice soils), ponderosa pine dry (hot dry; residual soils), ponderosa pine/lodgepole pine, lodgepole pine dry (pumice soils), lodgepole pine wet, mixed conifer dry (pumice soils), mixed conifer dry (other soils), mixed conifer moist, Shasta red fir (dry), upper montane (cold), and subalpine parklands.

Initial conditions were created by intersecting spatial data sets that represented vegetation stand boundaries, PVT boundaries, ownership/allocation boundaries, and development zones. The vegetation stand boundaries were derived using segmentation in eCognition 5 (Baatz et al. 2004) over a multi-image stack, where image segments represented homogeneous patches of vegetation, such as stands. The multi-image stack was composed of individual Landsat 5 and NAIP bands and image texture. Vegetation cover and structure attributes assigned to vegetation stands came from a vegetation layer developed using a gradient nearest neighbor (GNN) analysis technique where imputation is used to assign plot-level vegetation data to pixels (Ohmann and Gregory 2002; LEMMA 2011). PVT boundaries were determined from plant association maps developed by the USDA FS. Plant associations were grouped to form PVTs and represented the entire geographic extent over which PVTs could occur rather than the mean. Ownership-allocation boundaries were developed by the Oregon Department of Forestry. Residential development was restricted to development zones derived from projections of future building densities based on development rates from the 1970s to 2000 and environmental factors such as slope, elevation, distance to roads, zoning, and distance to other buildings (see Kline 2005 and Kline et al. 2010). For this study, we assumed PVT distribution, ownership-allocation, and development zone boundaries remained static over time. We used the TELSA Voronoi tessellation algorithm (Kurz et al. 2000) to subdivide our landscape into simulation polygons with an average size of 1 ha.

Development and Land Management

To model development, we determined the rates of change within five development density classes based on initial and ending development patterns determined by Kline et al. (2010), and applied these rates to our models as the annual probabilities for development. In other words, for each development density class, we defined development transitions and a probability for a development event to occur; the probabilities were determined from work by Kline et al. (2010). Development was assumed to occur linearly so that a patch must be developed at the lowest density class before being developed to a higher density and must pass through all density classes, in ascending order, before reaching the highest density class possible given the initial development zone. Exact locations of development were assigned stochastically within the development zones. Development density classes were defined, going from lowest to highest density, as more than 194 ha per structure (NotDev), 97 to 194 ha per structure (D2), 32 to 97 ha per structure (D3), 4 to 32 ha per structure (D4) and less than 4 ha per structure (D5). Most of the landscape began in the lowest development class (NotDev). When housing densities exceeded one house per 97 ha, active forest management ceased, as sustainable forest management has been shown to decrease with parcelization (Germain et al. 2007). Development analyses are presented for the whole study area below, but model outcomes were examined at all three spatial extents.

Modeled management activities varied by ownership (e.g., private industrial), federal land allocations (e.g., wilderness), and vegetation types and were applied to the landscape based on ownership class and development density. Federal lands were managed primarily for restoration and to reduce fuel loads, with lower rates of treatments than private lands. Management activities modeled for federal lands included pre-commercial thinning, commercial thinning, prescribed fire, and other harvest types; the frequencies at which management occurred varied by ownership, vegetation type, cover type, and structure. Overall, private land without residential development was assumed to be managed using similar methods to federal lands, but at higher intensities. For example, salvage activities were assumed to be 50 times more likely to occur on private ownership types than on USDA Forest Service land. Under the working forest scenario, the Bull Springs tract was managed for restoration of open ponderosa pine stands typical of the region under historical conditions prior to Euro-American settlement (Youngblood et al. 2004).

Quantifying Changes in Mule Deer Habitat

To represent the landscape in terms of mule deer habitat, the states in the initial and final vegetation maps were classified into habitat categories using a wildlife-habitat relationship (WHR) model (Johnson and O'Neil 2001). WHR models map the habitat of a particular wildlife species to the landscape based on vegetation type and structure, and have been compiled for many species in Oregon (Johnson and O'Neil 2001). With these models, the spatial structure of the landscape is linked to ecologically relevant life history traits such as home range size and dispersal distances, allowing species-specific responses to changes in landscape structure over time to be inferred (Johnson and O'Neil 2001).

The wildlife-habitat model for mule deer for the central Oregon region was developed in consultation between the USDA FS and wildlife biologists from ODFW. Vegetation type and structure were classified in three dimensions: forage quality (poor/none, low, moderate, high), thermal cover (yes, no), and hiding cover (yes, no). In addition, these dimensions were combined into a single rating model called the habitat suitability index (HSI). Classifications drew mainly on natural history information and sources reviewed above, but some modifications were made based on the expert knowledge of area biologists. We limited our pattern analysis of habitat area and patches to the observed mule deer winter range. All outputs generated by the vegetation dynamics model were converted to raster data sets with a 30-m cell size to coincide with the nominal grain size of the GNN vegetation data and analyzed in FRAGSTATS 3.3 (McGarigal et al. 2002). Habitat patches were defined in FRAGSTATS as adjacent cells sharing a cell boundary.

Forage Quality

Mule deer forage quality is related to a combination of forest structure (corresponding to structural stages in the STMs) and dominant tree species (cover types in the STMs). High-quality forage patches were typically open, with grass/forb, closed shrub, and seedling/sapling conditions in areas that supported most conifer tree species or were older stands of very large trees with multistory, open canopies. The exceptions were mesic stands of high elevation mixed conifer species, ponderosa pine, white fir (*Abies concolor* (Gord. and Glend.) Lindl. ex Hildebr.), and lodgepole pine, which were considered to be of low-quality due to snow accumulation in the winter. Moderate-quality forage consisted of younger stands of mostly conifer species and open canopies, with the exceptions listed above. Development densities less than one structure per 97 ha were considered moderate-quality forage provided the cover type was a tree

Table 1—Vegetation conditions defining each forage quality rating level and the habitat suitability index scores obtained when these levels are combined with hiding and thermal cover classifications (see text for definitions of hiding and thermal cover conditions). Certain combinations were not possible in these models due to the structural requirements for habitat to be called hiding or thermal cover, and are indicated with '--' in the table

		Habitat suitability index scores when combined with hiding and thermal cover			
Forage quality	Description of forage quality ratings	Hiding and thermal cover	Hiding cover	Thermal cover	Neither hiding nor thermal cover
High	*Structure:* open, grass/forb, closed shrub, seedling/sapling or older very large trees with multistory, open canopies *Composition:* all species except mesic, high elevation stands of mixed conifer species (ponderosa pine, white fir, and lodgepole pine), where snow accumulates in winter	--	6	--	3
Moderate	*Structure:* younger and open canopies *Composition:* mostly conifer species	--	5	--	2
Low	*Structure:* closed canopy stands *Composition:* any species when canopy is closed, and any occurrence of mesic, high elevation stands of mixed conifer species (e.g., ponderosa pine, white fir, and lodgepole pine)	7	--	4	--
Poor/none	*Structure and composition*: sagebrush, juniper, grassland or development densities higher than 97 ha per structure	6	3	3	0

species and the site was not mesic and high in elevation. All closed-canopy stands were considered to be low-quality forage. Forage quality in states with development higher than one structure per 97 ha or in sagebrush, juniper, or grassland trajectories were classified as poor/none, following the concept of disturbance zones (Theobald et al. 1997). These classifications are summarized in table 1.

Habitat Suitability Index

To capture change in modeled states that provide multiple habitat benefits or functions, we defined a habitat suitability index. The index combined forage quality habitat classes with thermal and hiding cover classes to provide a means to consider all habitat types together. Hiding cover was based on vegetation structure. Seedling/sapling stands, denser stands of poles, and all stands with a multilayered canopy and trees greater than 25 cm in diameter provided hiding cover. Closed shrub conditions in aspen types and dry ponderosa types were also considered hiding cover. All other modeled states, including all development states, provided no hiding cover. Canopy closure was the primary determinant of thermal cover, as it provides both shade in the summer and reduced wind exposure in the winter. All areas with coniferous tree species larger than 25 cm in diameter and canopy closure exceeding 40 percent provided adequate thermal cover. Numerical equivalents were given to each level of habitat classification: forage quality received scores of 0 through 3 corresponding to each level defined above from poor/none to high, while hiding and thermal cover were each given a score of 0 when absent and 3 when present. These numerical equivalents were then summed to generate the HSI (table 1). This formulation of habitat suitability distinguished between vegetation states that

Table 2—The landscape metrics used, a description of what they measure, and their units

Level	Metric	Definition	Ecological interpretation	Units
Class	Mean patch area	Average area in hectares of all patches in the same class	Home range size needed by an individual for mating, breeding, and foraging	Hectares
	Mean patch radius of gyration	Average distance between each cell in a patch and the patch centroid, averaged over all patches	Connectivity and corridor characteristics of the landscape	Meters
	Mean Euclidean nearest neighbor	Average distance from the edge distance of a patch to the edge of the nearest neighbor of the same type for all patches in that class	Overall isolation of patches in each class	Meters
	Number of patches	Total number of patches in each class	Fragmentation or consolidation of a class	Count
Landscape	Shannon's diversity index	richness and evenness of the distribution of patch types	Sensitive measure of rare patch types	None
	Largest patch index	percentage of the landscape occupied by the single largest patch	Homogeneity and the dominance of patches within the	Percent
	Contagion	The likelihood that adjacent cells will be of the same class type	General aggregation of the patches in the landscape and connectedness	Percent

provided just one habitat function (single-function classes) from those that provided 2 or 3 habitat functions (multi-functional classes). A rating of 2, for example, corresponded to only moderate-quality forage being present, whereas a rating of 7 indicated that both hiding and thermal cover were present in combination with low forage quality. HSI values of 2 or 3 only reflected forage conditions (except for very large multistory stands of open-canopy white fir, which provided only hiding cover), while HSI values of 4, 5, 6, and 7 represented multifunctional classes. Values of 0 signified no habitat provision and values of 8 and 9 were not possible, as all states that are moderate- or high-quality forage are inherently poor thermal and hiding cover; similarly, HSI = 1 does not occur because low-quality forage conditions provide hiding or thermal cover.

Landscape Pattern Analysis

Landscape pattern analysis can quantify the effects of human activities on the spatial arrangement of the landscape, and many metrics have been developed for measuring these arrangements (McGarigal et al. 2002, Turner 2005, Turner et al. 2001). Various studies in the field of landscape ecology have used landscape pattern metrics to study the relationship between ownership, land use change, and ecological processes (McComb et al. 2007, Stanfield et al. 2002, Swenson and Franklin 2000). Landscape structure can be measured within habitat types (class level) or across all habitat types (landscape level), and depends on the spatial extent considered and the grain size of the smallest unit of area. Metrics were calculated at the class and landscape levels, where habitat quality and HSI values represented classes, using FRAGSTATS 3.3 and were chosen to minimize redundancy while capturing the broadest set of landscape characteristics possible. Within the observed mule deer winter range, patches were defined as adjacent pixels of the same forage and HSI classes. At the class level, we calculated mean patch size, total class area, mean Euclidean nearest neighbor, mean radius of gyration and the number of patches (table 2). At the landscape level, we calculated the Shannon's Diversity Index (SHDI), contagion, and LPI (table 2). These metrics were calculated for each habitat model component for the initial and final conditions under

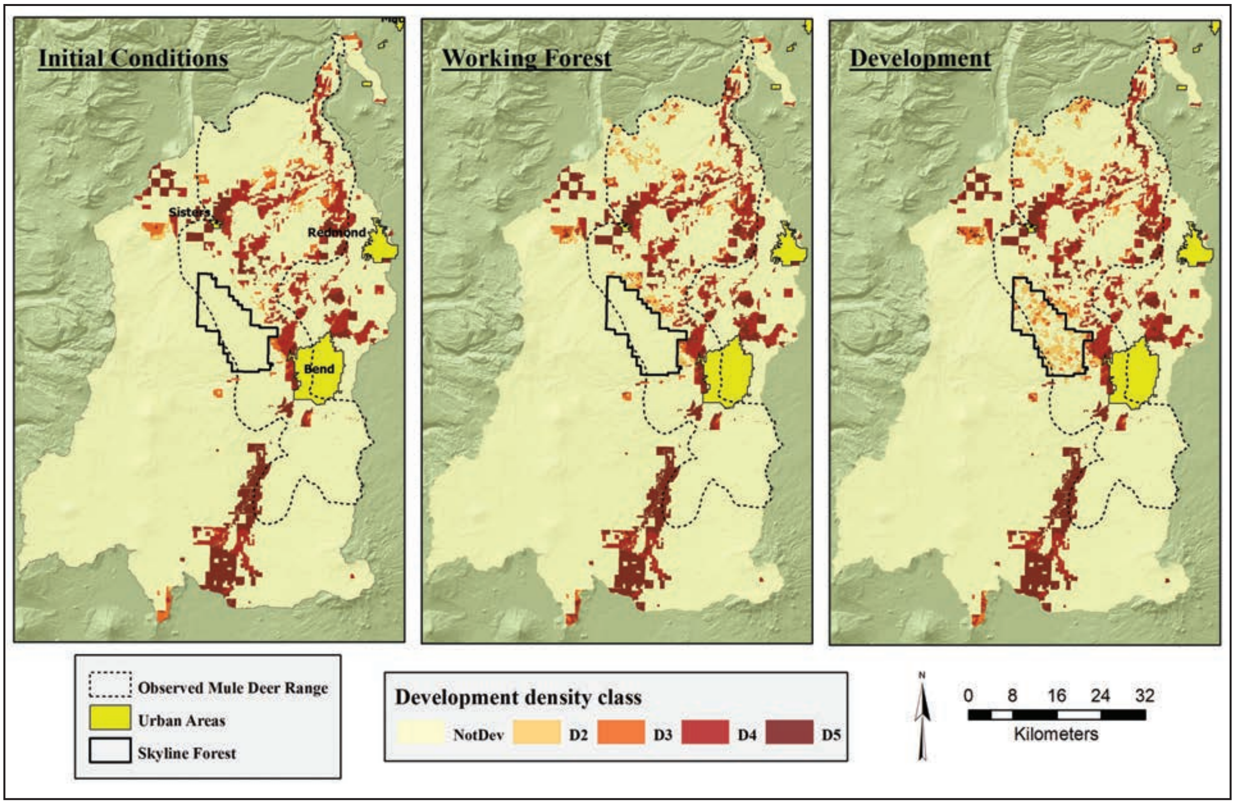

Figure 2—Spatial distribution of development classes for initial conditions and working forest (WF) and development (DEV) scenarios at 60 years. NotDev = greater than 194 ha/structure; D2 = 97 to 194 ha/structure; D3 = 32 to 97 ha/structure; D4 = 4 to 32 ha/structure; D5 = less than 4 ha/structure.

each scenario within the mule deer winter range. Percentage changes and absolute changes from present to future were calculated. In addition, change in developed land and vegetation states were calculated to provide context for interpreting the results of the pattern analysis. Our approach did not include a weight for neighborhood or proximity to high quality habitat. For example, moderate quality patches located beside high quality patches did not receive a higher habitat value than those far from high quality patches.

Results

The two scenarios generated different spatial patterns of development (fig. 2) and affected differences in vegetation structure and cover combinations after 60 years of simulation. Overall, the DEV scenario showed greater amounts of land conversion from lower to higher densities of development than the WF scenario. Urban densities increased in both scenarios (1.0 percent in WF and 0.7 percent in DEV), and undeveloped land area decreased by 1.2 percent and 2.5 percent in the WF and DEV scenarios respectively. In the observed mule deer winter range 5.8 percent of the land moved from undeveloped into the lowest developed density (D2, 97 to 142 ha per structure) in DEV as compared to only 3.0 percent in WF (fig. 3). In contrast, 42 percent of the Bull Springs tract converted from undeveloped to developed land (D2) under the DEV scenario, with 3670 ha of undeveloped land converted to D2 and 1890 ha converted to development densities of 32 to 97 ha per structure (D3) at the end of 60 years. On the Bull Springs tract alone, nearly twice as much land entered the lowest density class in the DEV scenario as compared to the WF scenario.

Forage Quality

Mapping the initial and future forage quality conditions illustrated differences in the locations and nature of land use conversions between scenarios (fig. 4A) and the resulting distributions of forage quality levels (fig. 4B).

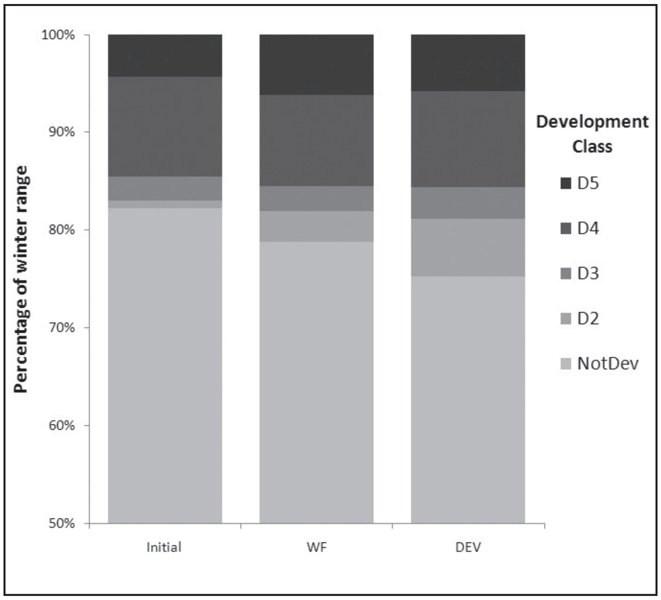

Figure 3—Proportions of development in five classes within mule deer winter range for the development (DEV) scenario and the working forest (WF) scenario. Winter range comprises 9800 ha of the Bull Springs tract and 133 100 ha of the entire study area. NotDev = greater than 194 ha/structure; D2 = 97 to 194 ha/structure; D3 = 32 to 97 ha/structure; D4 = 4 to 32 ha/structure; D5 = less than 4 ha/structure. As NotDev comprises most of the winter range, the vertical axis begins at 50 percent to better illustrate the differences in the other development classes.

Landscape level metrics did not change dramatically from initial conditions for either scenario or show substantial differences between scenarios (table 3). Overall, low-quality forage experienced the greatest negative change from the initial conditions for both scenarios in all landscape metrics (fig. 5). The largest total increases in patch area, abundance, and extensiveness occurred in high- and moderate-quality classes, with slightly larger increases in total area, largest patch index, patch area, and extensiveness in the WF scenario. Nonforage patches were less numerous, less extensive, and closer together under the WF scenario compared to DEV; under the DEV scenario nonforage patches became smaller and constituted a larger percentage of the landscape. Moderate-quality forage patches became larger and more extensive under WF compared to DEV, but they also were more isolated, according to nearest neighbor distances. High-quality forage patches became smaller on average in both scenarios. The decrease in mean high-quality patch area was less pronounced in the WF scenario, but fragmentation through the creation of more patches was more pronounced. High-quality patches also became less isolated and more extensive, with somewhat greater gains under the WF scenario compared to DEV. The largest differences between the scenarios were an increase in nearest neighbor distances of low-quality forage patches and a reduction in the patch abundance under DEV compared to a moderate increase in patch number under WF.

Habitat Suitability Index

Similar to outcomes for forage quality, the spatial mapping of the habitat suitability index showed subtle differences between the initial conditions and the two scenarios in terms of the location and nature of habitat quality changes (fig. 6A), as well as the overall distribution of habitat suitability levels in the two scenarios (fig. 6B). Landscape-level metrics were fairly similar between scenarios (table 3), but at the class level, differences were more apparent (fig. 7). For single-function habitat types (HSI = 2 or 3), the mean patch size increased in both scenarios although the increase was slightly smaller in DEV. Other metrics showed little difference between the two scenarios. In contrast, multifunctional habitat types (HSI = 4–7) displayed substantial differences between scenarios. For all multifunctional habitat types, mean radius of gyration was slightly lower under DEV than under WF. The patch isolation (mean

Table 3—Landscape level metrics calculated for the forage and habitat suitability index landscapes

	Forage			HSI		
Metric	**Initial**	**Development**	**Working Forest**	**Initial**	**Development**	**Working Forest**
Largest patch index	19.60	19.68	19.69	19.61	19.68	19.69
Contagion	53.22	61.95	63.33	52.22	63.33	61.95
Shannon's Diversity Index	1.57	1.32	1.27	1.57	1.32	1.27

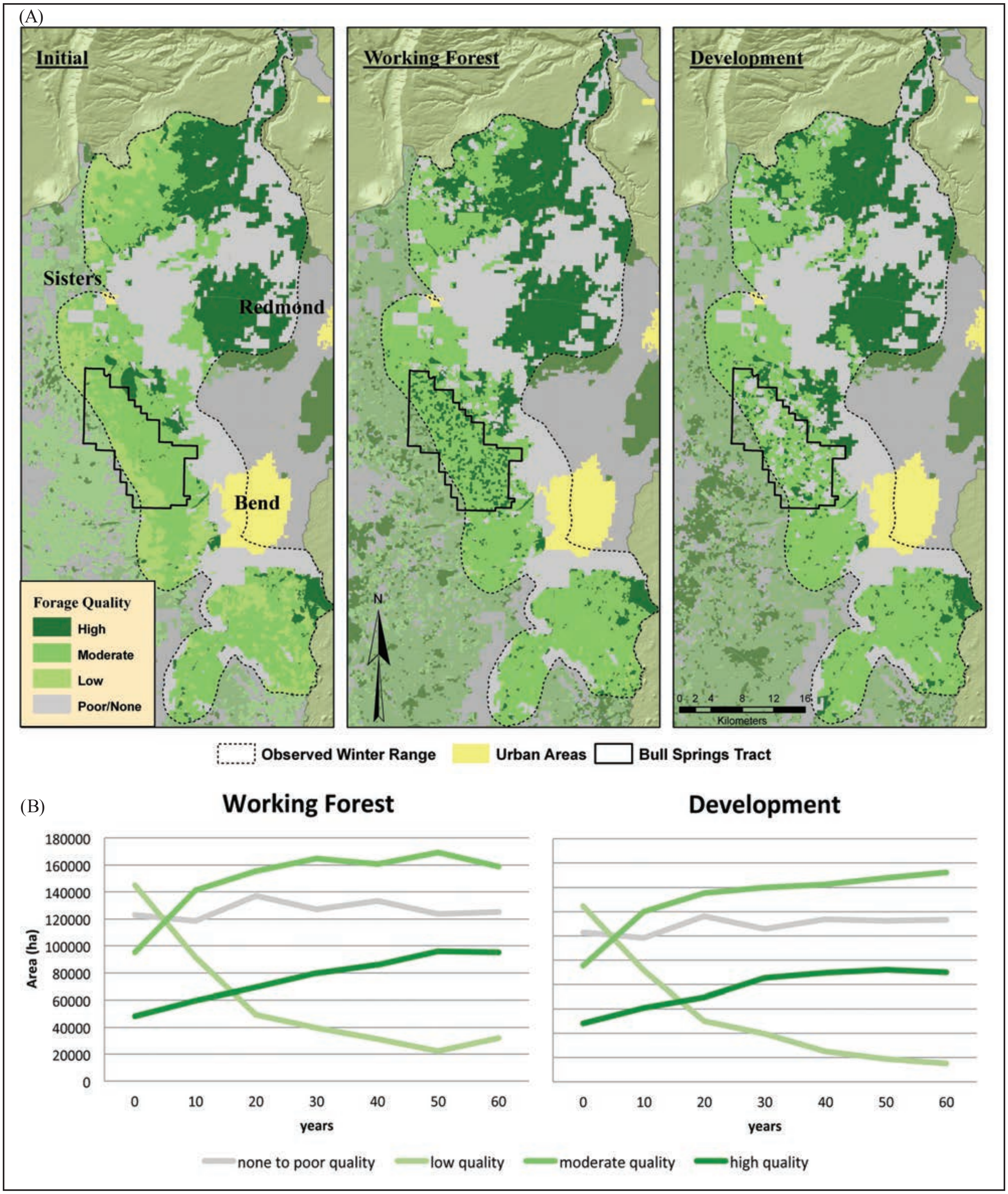

Figure 4—(A) Initial forage quality conditions (left) as well as the simulated forage quality conditions for the working forest scenario (middle) and the development scenario (right). The conditions within the mule deer winter range are highlighted and the location of the Bull Springs tract is shown in each map, as well as the major urban areas in the landscape. (B) Trajectories of the total amount of land in each forage quality class over the simulation.

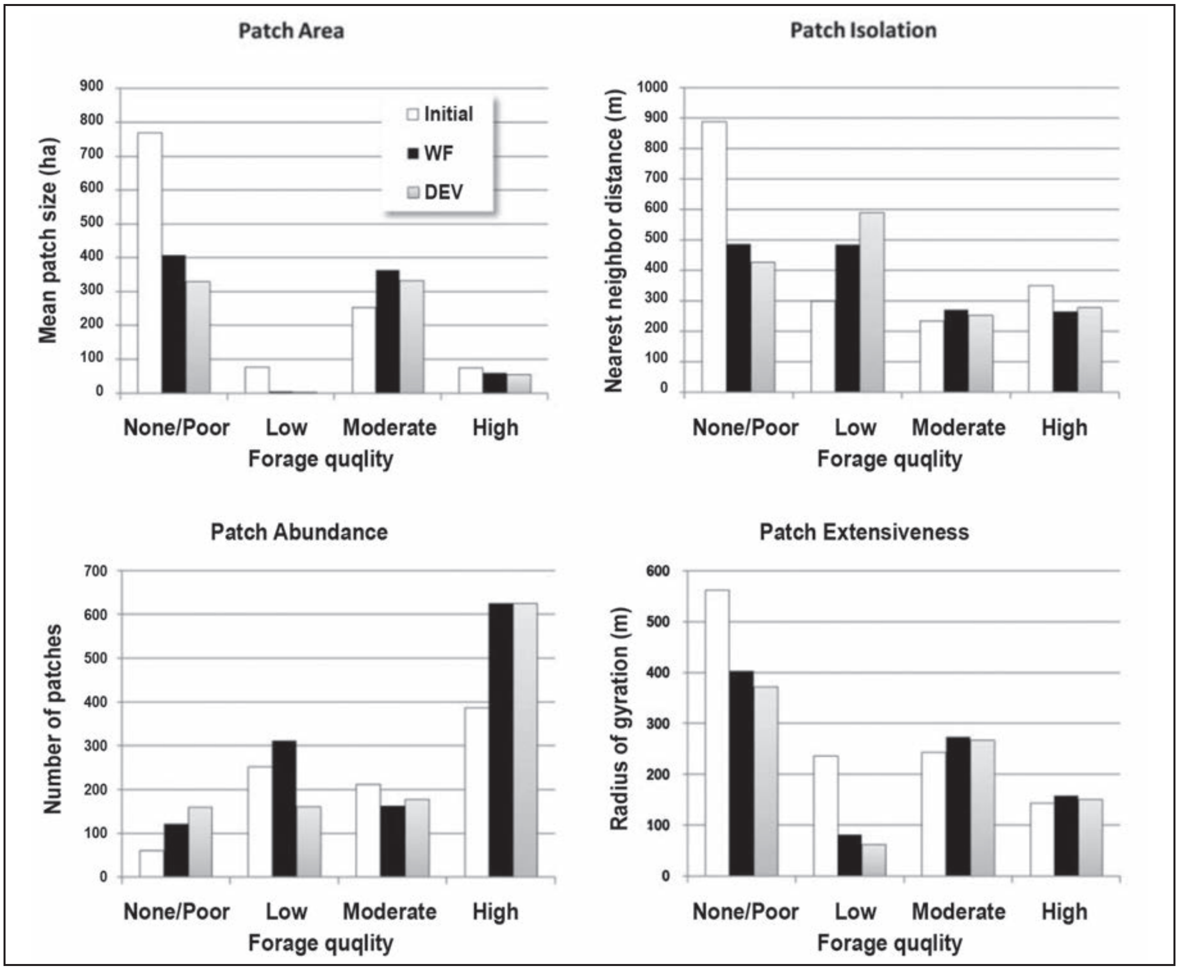

Figure 5—Differences in patch metrics between the initial conditions and modeled scenarios indicate changes in the spatial patterns of mule deer forage quality classes. The calculated values of the landscape metrics are given for the mean patch area (top left), the number of patches (bottom left), the mean Euclidean nearest neighbor (top right), and the mean radius of gyration (bottom right).

nearest neighbor distances) under DEV nearly doubled for HSI values of 4, 5 and 7. These habitat types provide low- or moderate-quality forage, in combination with either hiding or thermal cover. Our results suggest that vegetation conditions supplying more than one habitat function on the landscape would become more isolated, less extensive, and smaller in the future, with these effects amplified under DEV in most cases.

Interestingly, the outcomes for HSI values of 6 differed from those of the other multifunctional classes. Total class area remained within 10 percent of initial conditions, but both scenarios resulted in very large increases in the number of patches and little change in isolation and extensiveness. Coupled with the overall decrease in patch size, this indicates greater fragmentation of this class compared to the other multifunctional classes. There were also marked differences between scenarios in terms of nearest neighbor distance, with WF resulting in increased isolation compared to decreased isolation under DEV for this value.

Discussion

Our spatial models demonstrated that rural residential development and forest management have the potential to alter the Bull Springs tract and the surrounding landscape regardless of the development fate of the Bull Springs

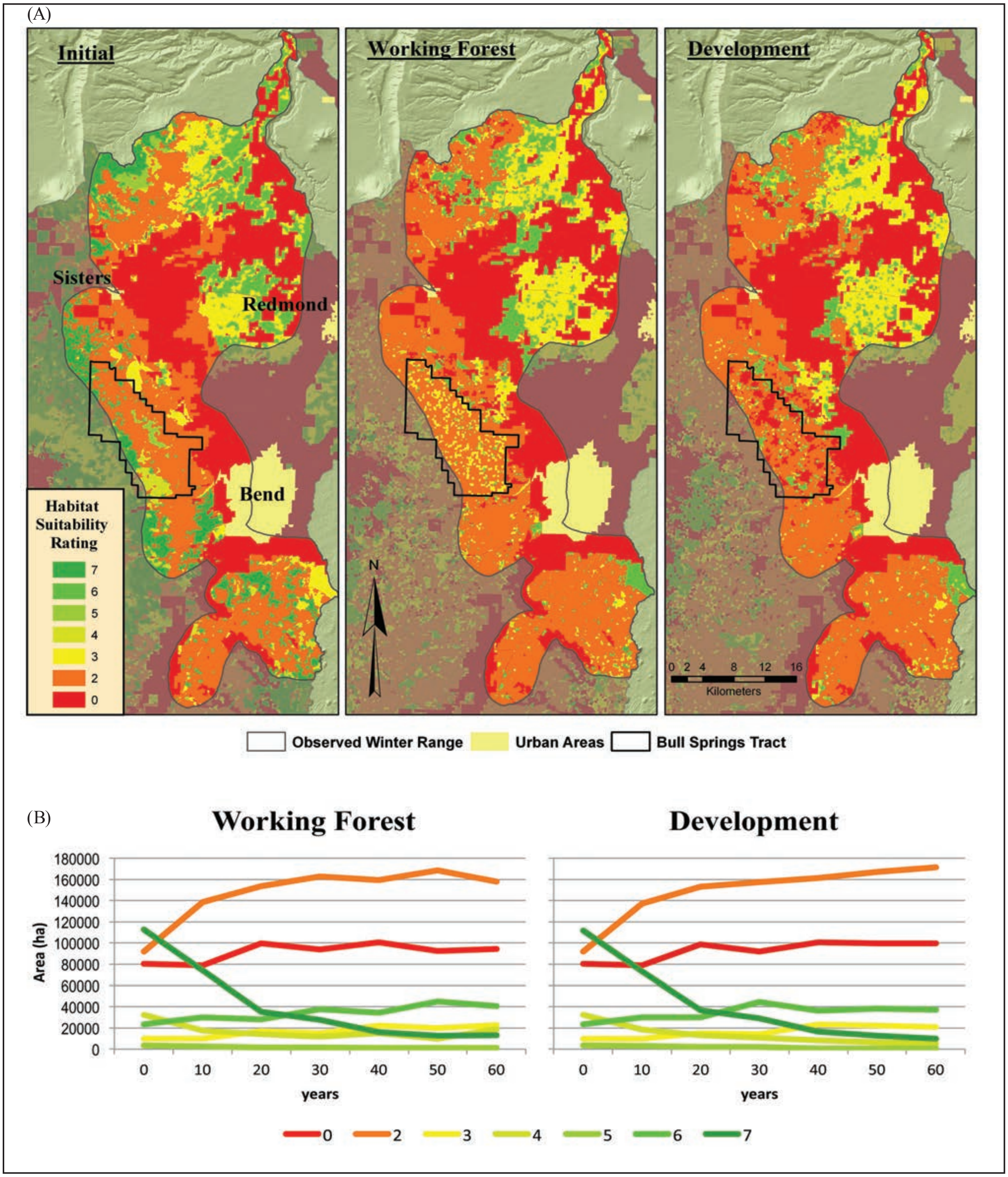

Figure 6—Maps of the habitat suitability conditions initially (left) and after 60 years under the working forest (WF, center) and development (DEV) scenarios (right). The conditions within the mule deer winter range are highlighted and the location of the Bull Springs tract is shown in each map, as well as the major urban areas in the landscape.

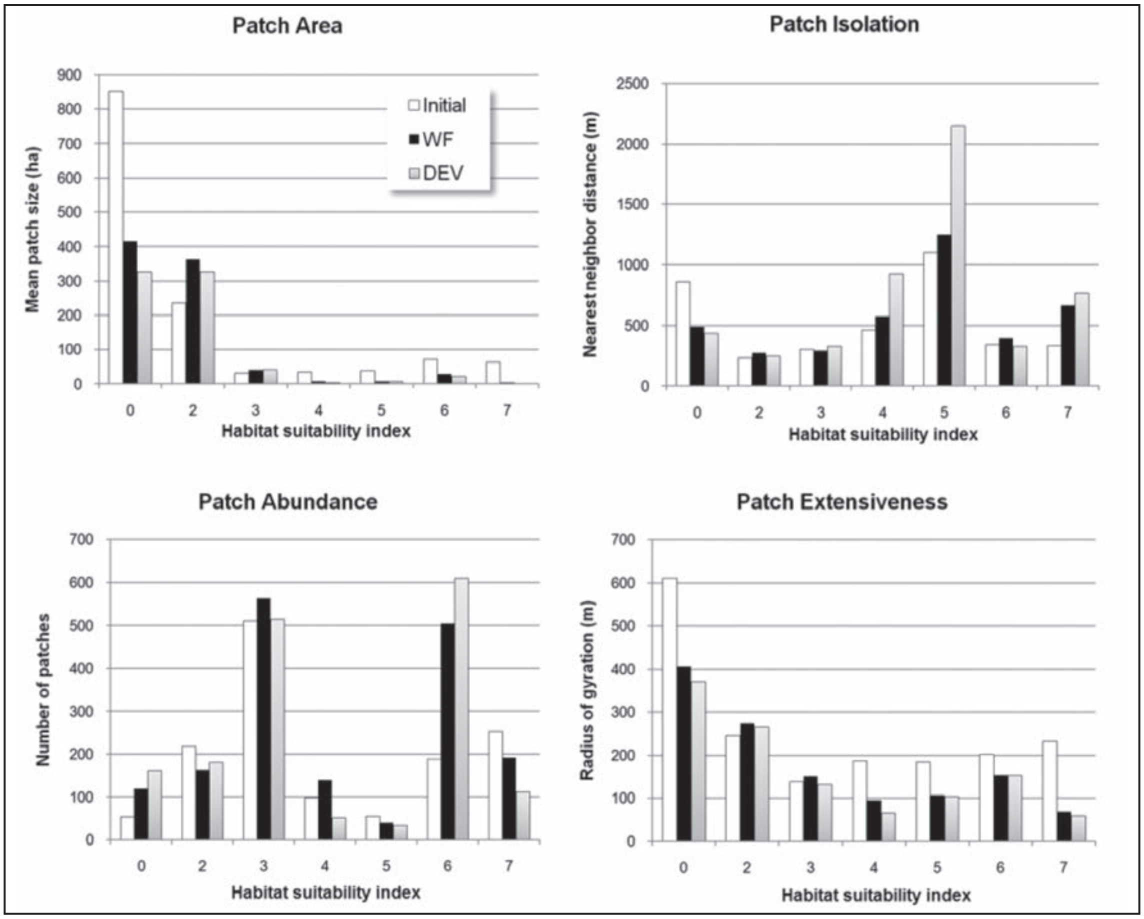

Figure 7—Class-level measures of the spatial pattern of the mule deer habitat suitability index. HSI = 0 has no habitat functions, HSI = 2 or 3 represent single-function habitat classes, HSI > 3 represent multifunction habitat classes. The calculated values of the landscape metrics are given for the mean patch area (top left), the number of patches (bottom left), the mean Euclidean nearest neighbor (top right), and the mean radius of gyration (bottom right).

tract. Results from both modeled scenarios suggest that landscape-wide changes projected to occur in the next several decades have potential to affect mule deer habitat, largely because development is expected to continue, but the differences may be subtle at the landscape scale. However, due to the Bull Springs' landscape position, large proportion of mule deer winter range, and large size relative to other private single-ownership tracts, preventing development on Bull Springs may offset landscape-level habitat degradation that could result from nearby rural development. Moreover, the modeled interaction of management and changes in forest structure and composition due to forest maturation suggests improved forage and multifunctional habitat conditions for mule deer over the next several decades if the land is managed as a working forest. This expectation was exemplified by increased patch abundance and area of high-quality forage patches in the working forest scenario when compared to the development scenario. Though more total area of high quality forage was added under the working forest scenario, both scenarios created the same number of high-quality forage habitat patches and resulted in decreased isolation of these patches relative to the initial conditions. This relationship suggests that mule deer may find more high-quality forage patches in the future and have to travel shorter distances between patches once they find the high-quality forage patches. The overall shift from multifunctional to single-function HSI ratings in both scenarios signals a possible stratification of the landscape and could lead to higher energetic costs if mule deer must travel farther distances between patches of different habitat types to obtain all needed resources. In contrast, if the single-function patches are closely interspersed and all habitat requirements are met within short travel distances, energetic costs may decline.

Mule deer is an iconic species in eastern Oregon, and maintenance of the herd for aesthetic and hunting purposes is a stated goal; our results suggest habitat supply and arrangement will be impacted by Bull Springs tract management decisions. Based on the limits of the observed winter range, there is evidence from the simulation that lower rates of development on the tract could enhance mule deer habitat conditions in the future and offset or ameliorate the impacts of development elsewhere in the region. Promotion of mule deer persistence in their winter range will likely require attention to the location of residential development outside the Bull Springs tract. Attention must also be paid to the potential for isolating high-quality forage and multi-functional habitat conditions and converting these to less suitable types because restricting development within the Bull Springs may increase development pressures elsewhere in the landscape.

Conversion of forested land to rural residences is only one process affecting vegetation dynamics and landscape change. Decision makers are often expected to respond to landscape-scale processes, but may not have access to landscape-scale information to ensure the broader policy governing land use change balances wide-ranging land use conversion with gains made through more localized conservation actions. Landscape simulation modeling provides one approach for investigating the numerous effects future policy choices may have in a region by defining the ecological significance of a particular piece of land in a given landscape as well as the limits of conservation on a single tract when considering the larger suite of changes taking place elsewhere in the landscape. By coupling spatial vegetation dynamics modeling that incorporates both human- and disturbance-driven modifications of the landscape with landscape pattern analysis, we can inform the broader debates on planning for land use changes in the future. Moreover, landscape simulation modeling results such as these can be extended to provide supporting data to a decision support system designed to prioritize scenario outcomes. Another extension of the work presented here that could benefit decision support would be to relate the landscape metrics for forage quality and HSI classes to mule deer carrying capacity. Incorporating the role that proximity to various habitat types plays in carrying capacity on the landscape would be another powerful means to understand the complex interactions of human activity and wildlife requirements. Landscape models and analyses such as the ones presented here provide a means for increasing knowledge of dynamic landscapes that are important for many management objectives.

Limitations

A major limitation of this study is that the allocation of development and simulation of final vegetation states were done using a single simulation, rather than multiple simulations, which limits the conclusions that can be drawn from the predictions. While we performed limited uncertainty and sensitivity analyses with the nonspatial and spatial vegetation dynamics models, we are limited in our ability to distinguish between the impacts of development and artifacts arising from the spatial simulation method; our results must be tempered by this uncertainty. In addition, the pattern metrics used in this study represent a set of assumptions about what makes "good" mule deer habitat, both in defining patches and in deciding what spatial aspects are more important than others. We also used a limited set of independent and complementary spatial pattern metrics for simplicity, but these are dependent on the scale and grain of measurement (Li and Wu 2004) and can often be highly correlated with cumulative changes in habitat area (Fahrig 2003). While these metrics were chosen to relate to life history requirements of mule deer, they are just one set of measurements of landscape structure in the region, and could not capture all aspects of the spatial structure of habitat, such as inter-class spatial relationships. Other limitations arise from not including other forms of human-wildlife interactions such as subsidized food supplies, suppression of predators, and climate change in the models.

With regards to the input data, the base GNN vegetation data layer was generated from a statistical model. When combined with the spatial vegetation dynamics model there is the potential for nontrivial classification errors (Ohmann and Gregory 2002). As such, spatial results from these products should be considered at the regional level, not the site level. These tools could not predict landowner behavior and changes in management approaches or ownership boundaries, and are limited to our historical understanding of and assumptions about these phenomena. Human and ecological processes and interactions were simplified in our modeling, but while the results must be viewed as only a small subset of many possible outcomes, they provide a point of departure for understanding the different outcomes that could be experienced under alternative policy and management regimes.

Conclusion

Interactions between land use change, shifts in management priorities, and natural disturbance processes can drive landscape dynamics and resulting patterns. The habitat of mule deer and other wildlife are determined by these dynamics and patterns. Using wildlife-habitat relationships, spatial analysis, and spatial vegetation dynamics modeling, it is possible to provide concise measurements of landscape change and facilitate ecological interpretations of differences between alternative land use policy futures. Our analysis of a large private tract in central Oregon and surrounding landscape showed that alternative management regimes might affect mule deer habitat, but more information is needed to understand the effect on carrying capacity. Notably, working forest management on the Bull Springs could result in somewhat better mule deer habitat outcomes, particularly with respect to higher quality forage and multifunctional habitat types. That said, simulated habitat was degraded to some extent under both scenarios, but marginal gains due to conservation over dispersed rural development could arise through working forest management, according to our simulation outcomes. Managing this tract as working forest could reduce the negative ecological effects of residential development and other land use changes within the region. Future management decision-making should pay continued attention to where residential development is expected to occur outside the Bull Springs tract and consider the isolating effect development might have on important habitat types. Managing the use and conditions of a single portion of the landscape can only do so much when that portion is embedded in a larger context of change, but this research provides critical insight into how land use policies on a specific tract can influence the ecological trajectory of change in the broader landscape.

Acknowledgments

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Literature Cited

Baatz, M.; Benz, U.; Dehghani, S.; Heynen, M. 2004. eCognition User Guide, Definiens Imaging GmbH, Munchen, Germany.

Brown, D.G.; Johnson, K.M.; Loveland, T.R.; Theobald, D.M. 2005. Rural land-use trends in the conterminous United States, 1950–2000. Ecological Applications. 15: 1851–1863.

Csuti, B.; Kimerling, A.J.; O'Neill, T.A.; Shaughnessy, M.M.; Gaines, E.P.; Huso, M.M.P. 1997. Atlas of Oregon wildlife: distribution, habitat and natural history. Corvallis, OR: Oregon State University Press. 492 p.

Dale, V.; Archer, S.; Chang, M.; Ojima, D. 2005. Ecological impacts and mitigation strategies for rural land management. Ecological Applications. 15: 1879–1892.

ESSA Technologies Ltd. 2007. Vegetation Dynamics Development Tool User Guide, Version 6.0. Prepared by ESSA Technologies Ltd., Vancouver, BC. 196 p.

ESSA Technologies Ltd. 2008. TELSA: Tool for Exploratory Landscape Scenario Analyses, Model Description, Version 3.6. Prepared by ESSA Technologies Ltd., Vancouver, BC. 235 p.

Fahrig, L. 2003. Effects of habitat fragmentation on biodiversity. Annual Review of Ecology, Evolution and Systematics. 34: 487–15.

Germain, R.H.; Anderson, N.; Bevilacqua, E. 2007. The effects of forest land parcelization and ownership transfers on nonindustrial private forest land forest stocking in New York. Journal of Forestry. 105: 403–408.

Hall, F.C. Pacific Northwest ecoclass codes for seral and potential natural plant communities. 1998. Gen. Tech. Rep. PNW-GTR-418. Portland, Oregon: U.S. Department of Agriculture, Forest Service, Pacific Northwest Research Station.

Hansen, A.J.; Knight, R.L.; Marzluff, J.M.; Powell, S.; Brown, K.; Gude, P.H.; Jones, K. 2005. Effects of exurban development on biodiversity: patterns, mechanisms, and research needs. Ecological Applications. 15: 1893–1905.

Hansen, A.J.; DeFries, R. 2007. Ecological mechanisms linking protected areas to surrounding lands. Ecological Applications. 17: 974–988.

Hemstrom, M.A.; Merzenich, J.; Reger, A.; Wales, B. 2007. Integrated analysis of landscape management scenarios using state and transition models in the upper Grande Ronde River subbasin, Oregon, USA. Landscape and Urban Planning. 80: 198–211.

Johnson, D.H.; O'Neil, T.A., eds. 2001. Wildlife-habitat relationships in Oregon and Washington. Corvallis, OR: Oregon State University Press. 736 p.

Kline, J.D. 2005. Forest and farmland conservation effects of Oregon's (USA) land use planning program. Environmental Management. 35: 368–380.

Kline, J.D.; Moses, A.; Burcsu, T. 2010. Anticipating forest and range land development in central Oregon (USA) for landscape analysis, with an example application involving mule deer. Environmental Management. 45: 974–984.

Kurz, W.A.; Beukema, S.J.; Klenner, W.; Greenough, J.A.; Robinson, D.C.E.; Sharpe, A.D.; Webb, T.M. 2000. TELSA: The Tool for Exploratory Landscape Scenario Analyses. Computers and Electronics in Agriculture 27: 227–242.

LEMMA. 2011. GNN Mapping of Existing Vegetation, Pacific Coast States. Corvallis, OR. http://www.fsl.orst.edu/lemma/main.php?project=imampersandid=home (1 November 2011).

Lettman, G.J. 2002. Land use change on non-federal land in western Oregon 1973–2000. Salem, OR: Oregon Department of Forestry.

Lettman, G.J. 2004. Land use change on non-federal land in eastern Oregon 1975–2001. Salem, OR: Oregon Department of Forestry.

Li, H.; Wu, J. 2004. Use and misuse of landscape indices. Landscape Ecology. 19: 389–399.

McComb, B.C.; Spies, T.A.; Olsen, K.A. 2007. Sustaining biodiversity in the Oregon Coast Range: potential effects of forest policies in a multi-ownership province. Ecology and Society. 12: 29.

McGarigal, K.; Cushman, S.A.; Neel, M.C.; Ene, E. 2002. FRAGSTATS: Spatial pattern analysis program for categorical maps. Computer software program produced by the authors at the University of Massachusetts, Amherst.

Ohmann, J.L.; Gregory, M.J. 2002. Predictive mapping of forest composition and structure with direct gradient analysis and nearest neighbor imputation in coastal Oregon, U.S.A. Canadian Journal of Forest Research. 32: 725–741.

Oregon Department of Fish and Wildlife (ODFW), 2003. Oregon's Mule Deer Management Plan. Portland, OR: Oregon Department of Fish and Wildlife. http://www.dfw.state.or.us/ODFWildlife/management_plans/docs/MuleDeerPlanFinal.PDF. (October, 4, 2012).

Petersen, S.L.; Stringham, T.K.; Roundy, B.A. 2009. A Process-Based Application of State-and-Transition Models: A Case Study of Western Juniper (*Juniperus occidentalis*) Encroachment. Rangeland Ecology and Management 62: 186–192.

Provencher, L.; Forbis, T.A.; Frid, L.; Medlyn, G. 2007. Comparing alternative management strategies of fire, grazing, and weed control using spatial modeling. Ecological Modelling. 209: 249–263.

Rindfuss, R.R.; Walsh, S.J.; Turner II, B.L.; Moran, E.F.; Entwisle, B. 2004. Linking pixels to people. In: Gutman, G.; Janetos, A.C.; Justice, C.O.; Moran, E.F.; Mustard, J.F.; Rindfuss, R.R.; Skole, D.; Turner II, B.L.; Cochrane, M.A., eds. Land change science: observing, monitoring and understanding trajectories of change on the Earth's surface. Boston, MA: Kluwer Academic Publishers: 379–394.

Simpson, M. 2007. Forested plant associations of the Oregon East Cascades. In: U.S. Department of Agriculture Forest Service. Pacific Northwest Region Technical Paper R6-NR-ECOL-TP-03-2007, July.

Stanfield, B.J.; Bliss, J.C.; Spies, T.A. 2002. Land ownership and landscape structure: a spatial analysis of sixty-six Oregon (USA) Coast Range watersheds. Landscape Ecology. 17: 685–697.

Swenson, J.J.; Franklin, J. 2000. The effects of future urban development on habitat fragmentation in the Santa Monica Mountains. Landscape Ecology. 15: 713–730.

Theobald, D.M.; Miller, J.R.; Hobbs, N.T. 1997. Estimating cumulative effects of development on wildlife habitat. Landscape and Urban Planning. 39: 25–36.

Theobald, D.M.; Romme, W.H. 2007. Expansion of the US wildland-urban interface. Landscape and Urban Planning. 83: 340–354.

Theobald, D.M., Spies, T.; Kline, J.; Maxwell, B.; Hobbs, N.T.; Dale, V.H. 2005. Ecological support for rural land-use planning. Ecological Applications. 15: 1906–1914.

Thorson, T.D.; Bryce, S.A.; Lammers, D.A.; Woods, A.J.; Omernik, J.M.; Kagan, J.; Pater, D.E.; Comstock, J.A. 2003. Ecoregions of Oregon (color poster with map, descriptive text, summary tables, and photographs). Reston, VA: U.S. Geological Survey.

Turner, M.G. 2005. Landscape ecology: what is the state of the science? Annual Review of Ecology, Evolution and Systematics. 36: 319–344.

Turner, M.G.; Gardner, R.H.; O'Neill, R.V. 2001. Landscape ecology in theory and practice: pattern and process. New York, NY: Springer Verlag.

Vavra, M.; Hemstrom, M.A.; Wisdom, M.J. 2007. Modeling the effects of herbivores on the abundance of forest overstory states using a simple state-transition approach. Landscape and Urban Planning. 80: 212–222.

Westoby, M.; Walker, B.; Noy-Meir, I. 1989. Opportunistic management for rangelands not at equilibrium. Journal of Range Management. 42: 266–274.

Youngblood, A.; Max, T.; Coe, K. 2004. Stand structure in eastside old-growth ponderosa pine forests of Oregon and northern California. Forest Ecology and Management. 199: 191–217.

Yung, L.; Blesky, J.M. 2007. Private property rights and community goods: negotiating landowner cooperation amid changing ownership on the Rocky Mountain front. Society and Natural Resources. 20: 689–703.

Balancing Feasibility and Precision of Wildlife Habitat Analysis in Planning For Natural Resources

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Abstract

Wildlife conservation often is a central focus in planning for natural resource management. Evaluation of wildlife habitat involves balancing the desire for information about detailed habitat characteristics and the feasibility of completing analyses across large areas. Our objective is to describe tradeoffs made in assessments of wildlife habitat within a multiple-objective vegetation-based state-and-transition model (STM) framework. Species-habitat relationships derived from STM vegetation characteristics require careful interpretation for several reasons. First, the observational unit for wildlife analysis is habitat, which does not provide information about actual species occurrence or distribution. Second, subjectivity exists in researcher interpretation of species-habitat relationships derived from past literature, particularly qualitative descriptions. For quantified species-habitat relationships that exist, only information that matches output criteria directly may be used for analysis. Third, visual interpretation of results may vary based on scale of analysis used in STMs. When preparing wildlife habitat information from STM output and its application to natural resource planning, there is a need to focus on consistent and defensible information and emphasize the limitations of knowledge derived from data analysis.

Keywords: fuels management, habitat, land use planning, state-and-transition model, wildlife management.

Introduction

Policymakers and planners often consider tradeoffs between resources available and the information gained through resource investment when making decisions about natural resources management. State-and-transition models (STMs) can play an important role in assisting natural resource managers and policymakers with such decisions. STMs allow users to create and test descriptions of vegetation dynamics, management regimes, and natural disturbances by simulating them simultaneously across a landscape of interest through time. Vegetation-based STMs have been used in many applications, including wildlife habitat analysis (e.g., Evers et al. 2011, Wisdom et al. 2002).

Because wildlife habitat is not the observational unit of focus for vegetation STMs, but rather a representation of habitat based on pre-determined vegetation characteristics, mismatches between data and inferences made from those data can have drastic consequences for species and habitat, as well as for land use managers who rely on planning and policy decisions for the sustainable management of natural resources. For example, fine-scale details such as relative habitat quality and spatial distribution of habitat may not be available from coarse-level aggregated state class characteristics of STMs. Therefore, our objective is to highlight a number of considerations that are necessary for accurate interpretation of wildlife habitat analyses from STMs. First, we discuss two reasons why wildlife habitat analyses are important in policy and planning decisions that may involve use of STM tools, and how resulting policies may vary across jurisdictional levels. Then, we evaluate four factors that affect the interpretation of habitat information as derived from STMs: (1) the observational unit of analysis, (2) available versus desired species-habitat information, (3) interpretation of data by different researchers, and (4) scale of analysis. We conclude by highlighting factors to consider when balancing feasibility and precision of wildlife habitat analysis with interpretation of results for policy decisions, and provide some basic guidelines for making inferences from wildlife habitat analyses.

Why Apply STMs to Wildlife Habitat Analysis?

Wildlife habitat analysis involves conditional inferences and, when placed within a broader natural resource policy, can conflict with other natural resource management objectives. Despite potential conflicts, we highlight two motivations for including wildlife in policy analyses and linkages with STMs.

Wildlife Is an Indicator of Ecosystem Conditions

Each wildlife species depends on a range of environmental and habitat features for activities such as foraging, roosting, nesting, denning, and hiding from predators (Bolen and Robinson 1999). Presence or absence of a species in a location that contains characteristics linked to that species may provide clues about both habitat condition and the integrity of ecological processes (Grimm 1995).

Habitat characteristics for species span both non-living and living features within ecosystems. Non-living features may include ground moisture, particular soil conditions, water, leaf litter or detritus, and down and dead wood. For example, snags provide roosting locations for many bat species as well as sources of foraging for woodpeckers (Saab et al. 2004). Like non-living features, living features such as shrubs and other understory features provide shelter, nest locations, cover from predators, as well as foraging opportunities for wildlife. For example, a variety of forest ecosystems can provide seasonal food needs for black bears (*Ursus americanus*), such as berries and mast (Baldwin and Bender 2009, Hébert et al. 2008). Species such as marten (*Martes americana*), fisher (*Martes pennanti*; Powell et al. 2003), and flying squirrels (*Glaucomys sabrinus*; Mahan et al. 2010) rely on complex canopy structures for life activities. Presence of smaller species, such as flying squirrels, is a necessary condition for related predators, such as northern spotted owl (*Strix occidentalis caurina*; e.g., Forsman et al. 2005). Therefore, management for particular features can result in habitat benefits for many species.

Wildlife species also are indicators of ecosystem processes. For example, tree regeneration and seed dispersal can be highly dependent on mammals and birds (e.g., Auger et al. 2002, Gilgert and Zack 2010, Schupp 1993, Siepielski and Benkman 2007, Stiles 2000). Dam building by beavers (*Castor canadensis*) enhances soil nutrients and species richness of wetlands (Wright et al. 2002) and provides habitat for fish and waterfowl (McCall et al. 1996, Pollock et al. 2004). Therefore, evidence of the processes that species help maintain provides clues about the overall ecosystem function.

Wildlife Has Social Value

Wildlife holds social value in both consumptive and non-consumptive contexts. Utilitarian value (Decker et al. 2001) often is associated with fees generated for hunting and fishing opportunities, and in some instances can total tens of thousands of dollars for trophy species (Booth 2009). Naturalistic value (Decker et al. 2001) is associated with activities such as wildlife viewing. Several US national parks, including Yellowstone, Olympic, and the Great Smoky Mountains, cite the participation in wildlife viewing activities as an important reason for park visitation (Manni et al. 2007, Papadogiannaki et al. 2009, Van Ormer et al. 2001). Bird watching is a global industry that attracts participants with diverse backgrounds and motivations (Curtain 2010, Green and Jones 2010, Lindsey et al. 2007). Thus, similar to wildlife as indicators of ecosystem conditions, there is opportunity to relate STMs to social wildlife indicators, although we are unaware of any current applications of STMs in this area.

The social value of wildlife also is illustrated by the wealth of non-profit groups focusing on wildlife. For example, the World Wildlife Fund (http://www.worldwildlife.org) and National Audubon Society (http://www.audubon.org) rely on donations for operating revenue. In such cases, donors may seek to promote existence of particular species regardless of ever having the personal chance to see them (e.g., World Wildlife Fund conservation program for giant pandas; *Ailuropoda melanoleuca*), suggesting anthropomorphic or moral values (Decker et al. 2001).

Factors Affecting Interpretation of Wildlife Habitat Derived from STMs

Ecological and social values related to wildlife influence policy at federal, regional, and state jurisdictional levels. At the federal level, the Endangered Species Act (USFWS 2011) and National Environmental Policy Act (NOAA 1969) provide guidance for potential impacts to wildlife and wildlife habitat. Regional and state policies may complement or supplement federal guidelines for wildlife, or focus on regional priorities. For example, the U.S. Forest Service Northwest Forest Plan (NWFP) for the Pacific Northwest provided goals for sustainable forest management (NWFP; USDA FS 1997) through a vision of fish and wildlife habitat conservation, such as recovery of the northern spotted owl and its habitat (USDA FS 1997). However, mismatches between geopolitical boundaries and geographic ranges of wildlife species can result in inconsistent designations of wildlife. For example, western gray squirrel (*Sciurus griseus*) populations are stable in Oregon and California, but have declined in Washington, where three genetically isolated populations now exist. The Washington Department of Natural Resources has instituted a recovery plan for the species, the objectives of which include restoring and protecting habitat in appropriate areas of the state (Linders and Stinson 2006). In those cases, STMs may not only assist species management at the state level, but also interpretation of available knowledge may be necessary at multiple scales of analysis for decision-making across multiple geopolitical boundaries across the geographic range of the species.

STMs allow managers to evaluate tradeoffs and risks associated with land management decisions that may impact ecological and social wildlife values across multiple jurisdictional scales. However, regardless of scale and tools used, all wildlife models are the result of inferences made from evaluations of data about organisms and their habitat. In other words, wildlife habitat models are representations of species-habitat relationships that are limited to the scope of the information used to construct them. Therefore, time sensitive needs and variation in spatial scale of the policy context result in a need to balance feasibility of analyses necessary for decision-making with precision of the data used to make those decisions. Here we highlight four factors that affect habitat evaluation and the resulting inferences made from those evaluations.

Observational Unit

The observational unit (or unit of observation) is the unit used for analysis. For wildlife management, the unit of analysis can span a range of observations: an individual, a population, multiple populations, or a species. There often is a mismatch between the observational unit of wildlife and STM output, which can limit inferences that can be drawn from STMs. Analyses based on STMs may consider mere habitat presence as defined by particular characteristics, but may not be able to provide a precise assessment of habitat quality. For example, open canopy and recent disturbance may be used to identify presence of habitat or potential habitat for the black-backed woodpecker (*Picoides arcticus*). Ability to incorporate snag density into STMs would allow for a finer assessment of habitat quality because a greater density of snags is directly related to higher habitat quality (Saab et al. 2009). Similarly, habitat area is another variable that may be considered, but scale of analysis (discussed below) may limit interpretation of this variable. For example, assessment of habitat area at the watershed scale may provide information about the aggregated number of acres of black-backed woodpecker habitat available within a watershed. However, because those habitat data are aggregated at the watershed scale, there is limited ability to evaluate the spatial distribution of that habitat within each individual watershed. The lack of information on habitat arrangement would then affect the ability to assess home ranges, connectivity, and other landscape assessments (e.g., see analyses in Dudley and Saab 2007). In general, the selection of the observational unit will not only affect the analysis that can be completed, but also the interpretation and knowledge derived from the analysis.

a

Size Class (inches)					Canopy Closure (%)				Canopy Layers (#)	
Pole	Smtree	Mdtree	Lgtree	Gttree	GrassForb	Open	Medium	Closed	Single	Multi
(5-10)	(10-15)	(15-20)	(20-30)	(>30 in)	(<10)	(10-40)	(40-60)	(>60)	1	>1

b

Habitat variable	**Units of measurement**
Volume of snags	m^3/ha
Volume of stumps	m^3/ha
Volume of exposed root masses	m^3/ha
Volume of downed logs	m^3/ha
Basal area of live deciduous	m^3/ha
Basal area of live coniferous trees	m^3/ha
Density of live trees	number/ha
Tree height	m
Overhead cover	**Percent**
Litter depth	cm
Understory deciduous stem density	number/ha
Understory coniferous stem density	number/ha
Foliage density <0.5 m	percent
Foliage density 0.5-2.0 m	percent

Figure 1—Evaluation of wildlife habitat within a framework of vegetation dynamics creates difficulty in matching habitat information to vegetation attributes. In this example, a) Vegetation characteristics categories used for development of STMs were limited to tree size (dbh), canopy closure (percent) and canopy structure (layers). b) When evaluating habitat for the American marten from past research, a large number of variables cannot be used because of inconsistencies between important vegetation and wildlife habitat variables. For example, if the output from STMs such as that illustrated here are used to evaluate American marten habitat, only overhead cover (percent; in bold text) can be used from Payer and Harrison (2003).

Matching "What We Know" to "What We Need"

Habitat models are simplified representations of complex ecological relationships. Building habitat models from empirical wildlife data allow researchers to control selection of variables they perceive as important to a species based on existing knowledge. For example, habitat suitability indices (HSIs) are models that use an inductive process based on data from observed species-habitat relationships. However, habitat conservation may only be one consideration within a broader management objective and, therefore, would be evaluated using a framework and context derived from a perspective other than wildlife. For instance, a STM built with the primary purpose of evaluating vegetation growth and succession likely will contain parameters that align well with variables important to vegetation dynamics. However, those important vegetation variables may or may not overlap with variables important for a particular wildlife species and potentially result in a less-than-optimal relationship between model output and wildlife habitat. Ultimately, such inconsistencies, particularly at the fine scale, are important to interpret and convey clearly and precisely.

If modelers are interested in using existing literature to link species-habitat relationships to STMs, how state classes are defined in STMs will affect the ability to derive habitat information from published knowledge. We found that although numerous studies may exist for particular wildlife species, information often was not recorded or reported in ways that matched directly with the definitions of STM state classes (e.g., fig. 1). Either the observational unit of the study did not match the STM observational unit, or the variables incorporated into STMs were not recorded in the wildlife habitat information. The outcome resulted in limited data for even for the "most studied" species.

Consensus Among Researchers

Wildlife habitat modeling efforts often include a number of individuals working as a team, which can lead to differences in interpretation of habitat information. At the most basic level, ability of an individual researcher to define and match those relationships consistently may be influenced by life history traits of a species. Habitat specialists are restricted to a narrow range of habitat characteristics. For example, red tree voles (*Arborimus longicaudus*) reside a

Cover	Structure	A	B	C	D
Ponderosa Pine	Large tree, low density, single story	0	0	0	0
Douglas-fir/White Fir	Large tree, low density, single story	0	0	0	0
Grand Fir/Englishman Spruce	Large tree, low density, single story	0	0	0	0
White Fir	Large tree, low density, single story	0	0	0	0
Ponderosa Pine	Large tree, medium density, single story	1	1	1	1
Douglas-fir/White Fir	Large tree, medium density, single story	1	1	1	1
Grand Fir/Englishman Spruce	Large tree, medium density, single story	1	1	1	1
White Fir	Large tree, medium density, single story	1	1	1	1

b

Cover	Structure	A	B	C	D
Ponderosa Pine	Large tree, low density, single story	0	0	1	1
Douglas-fir/White Fir	Large tree, low density, single story	0	0	0	0
Grand Fir/Englishman Spruce	Large tree, low density, single story	1	0	1	1
White Fir	Large tree, low density, single story	0	0	0	0
Ponderosa Pine	Large tree, medium density, single story	0	1	0	1
Douglas-fir/White Fir	Large tree, medium density, single story	1	0	0	0
Grand Fir/Englishman Spruce	Large tree, medium density, single story	0	0	1	1
White Fir	Large tree, medium density, single story	0	1	0	0

Figure 2—Individual interpretation of information can lead to inconsistent definition of species-habitat relationships, such as illustrated for four hypothetical members of a research team "A," "B," "C," and "D." Cover and structure states identified as habitat for a given species are designated with a "1", whereas those considered not habitat are designated with a "0". a) The first species (top) is an old-growth habitat specialist with a narrow range of habitat characteristics, which results in relatively easy consensus among researchers. b) The second species is a forest habitat generalist with a broad range of possible habitat characteristics, which results a need to discuss inconsistencies in information interpretation among researchers.

among the canopy of mature Douglas-fir forests south of the Columbia River, and rely on conifer needles for both forage and water (e.g., Forsman et al. 2009). Those narrow habitat traits are relatively easy to identify consistently from STM output. Conversely, both white-tailed (*Odocoileus virginianus*) and mule deer (*O. hemionus*) are habitat generalists, and forage among a wide variety of ecosystems and forest types. Those complexities in habitat use may result in differences among researchers if each researcher focuses on a different habitat component (e.g., forage versus cover information).

Multiple researchers also can interpret the same information differently, which can confound efforts to match species-habitat relationships with STM state classes. For example, the northern spotted owl is associated with relatively large trees and closed and complex canopy structure (Forsman et al. 2005). Those habitat characteristics are relatively straightforward to identify among vegetation STM state classes such as (tree) size class (diameter at breast height; dbh), canopy cover (or percent closure), and canopy structure (single versus multiple layers; fig. 2). Therefore, it is expected that a research team would have a relatively easy time reaching consensus on state classes to be defined as habitat, particularly if quantitative measures are provided within references. However, qualitative descriptions such as "large," "closed," and "complex" are more difficult to interpret if metrics are not also defined quantitatively. For example, the pileated woodpecker (*Dryocopus pileatus*) inhabits forests with a variety of size classes and diverse canopy structures (Aubry and Raley 2002). In this case, there are many possible definitions of habitat, which may be difficult to evaluate if not defined with quantitative metrics and can lead to inconsistencies among a research team. Therefore, developing decision rules to account for potential inconsistencies among a team can play an important role in decreasing probability of error when describing habitat characteristics, as well as defining guidelines for inferences made from those data.

Scale

Scale is the spatial or temporal dimension of a pattern or process. The scale of analysis can be coarse (e.g., continent or geologic period) or very fine (individual grains of and or milliseconds). Regardless of size, scale used for STM analysis affects interpretation of data and potential comparisons of output to other wildlife habitat models. For example, the range of the black-backed woodpecker includes the east side of the Cascade Mountains as well as upper-elevation portions of west-side slopes (Dixon and Saab 2000); presentation of that information at the 30-meter scale illustrates this succinctly (fig. 3a). If the same results are presented at the watershed scale as devised from STMs (fig. 3b), the species range distribution includes the extent of all watersheds with boundaries along the Cascade crest. Although the actual geographic range of the species did not change for the two analyses, the visual interpretation of the species-habitat relationships data in the STM output is distorted, such that it appears to include a larger total area than the finer scale map. Without fully appreciating the role of scale or general ecological information about the species, STM data users may misinterpret the output and overestimate the habitat needs of and/or available habitat for black-backed woodpeckers and inaccurately emphasize the importance of ecosystems outside of this species' geographic range. Thus, without proper understanding, mapped output could lead to policy decisions that overly conserve, or do not sufficiently conserve, habitat.

Scale of analysis also affects comparison of STM data to results of other habitat mapping approaches. For example, the initial Gap Analysis Program (GAP), an effort by the US Geological Survey, involved use of wildlife-habitat relationship models (WHR) to assess underrepresented species and habitats for conservation planning (Scott et al. 1993). Initial GAP efforts used 30-meter resolution land cover data to map WHRs by estimating habitat presence from known species distribution and data derived from individual pixels (fig. 4a). Using land cover and habitat relationship data, the range and distribution of individual species were mapped and modeled to determine relative amount of habitat on and off protected lands.

A second and more recent approach, the Gradient Nearest Neighbor method (GNN; Ohmann and Gregory 2002), is an imputation designed to map vegetation characteristics at the 30-meter scale based data derived from forest inventory plots. Characteristics of the data collected for each plot, along with other biophysical attributes and satellite imagery, are used to interpolate vegetation data across the entire landscape where plot information is absent (fig. 4b). Although GNN is a powerful tool for evaluating vegetation characteristics, it does not consider fine-scale landscape attributes and geographic features important for wildlife species considered by the GAP approach. As a result, some aggregation to coarser scales (e.g., 1-hectare) is necessary in order to account for imputation bias.

Current efforts combining GAP and GNN data layers provides the ability to integrate the complex vegetation characteristics of GNN with the fine-scale presence of geographical features and land cover of GAP. The result is a more-precise process for ranking habitat for conservation priority based on known species distributions, and is currently being used in analyses such as for the Northwest Regional Gap Analysis Project (Re-GAP; http://gap.uidaho.edu/index.php/gap-home/Northwest-GAP; fig. 4c).

Confusion can result if decision-makers then attempt to compare results from STMs to GAP, GNN, Re-GAP or other moderate and coarse-scale assessments. For example, the American marten is a mature-forest species with a distribution across much of northern North America (Powell et al. 2003). In Oregon, GAP results predict habitat in occupied watersheds within mountain ranges (fig. 5a). More specifically, the observational unit (habitat) is predicted at the 30-meter scale only for watersheds that include known occurrence of martens. What if STM analysis evaluates distribution of marten habitat across Oregon, with habitat as the observational unit, but evaluation includes all watersheds regardless of marten occurrence? The discrete difference in how watersheds were selected (marten occurrence versus complete enumeration based on habitat characteristics) greatly affects interpretation of results (fig. 5b). Specifically, the evaluation based on complete enumeration appears to resemble historic range of marten as identified by

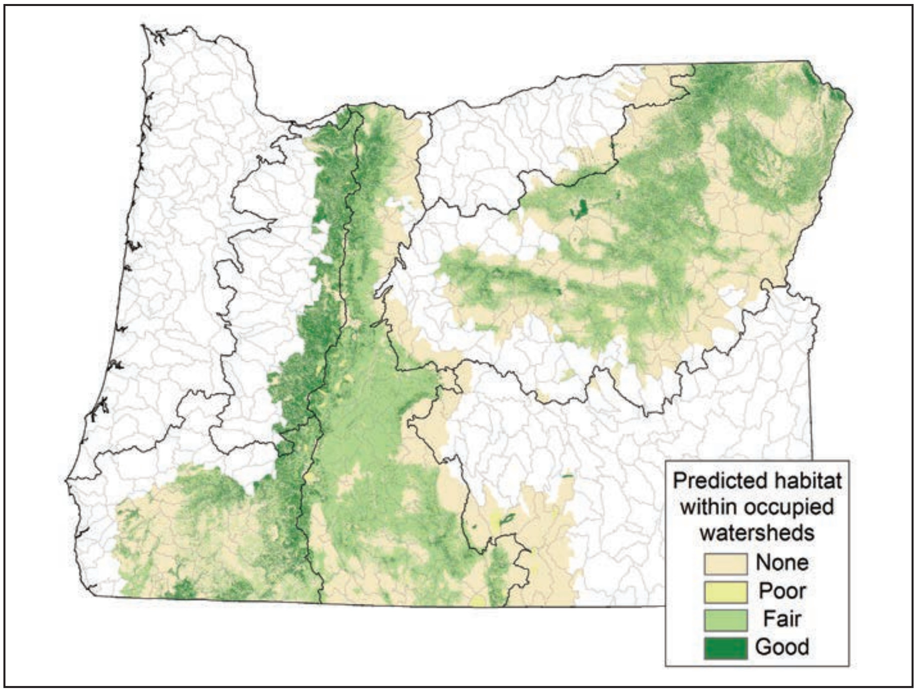


Figure3a—Black-backed woodpecker range presented at the 30-meter scale illustrates mainly dry forest habitat relationships for this species in Washington and Oregon, with some extension to the west side of the Cascade Mountains (INR 2011a).

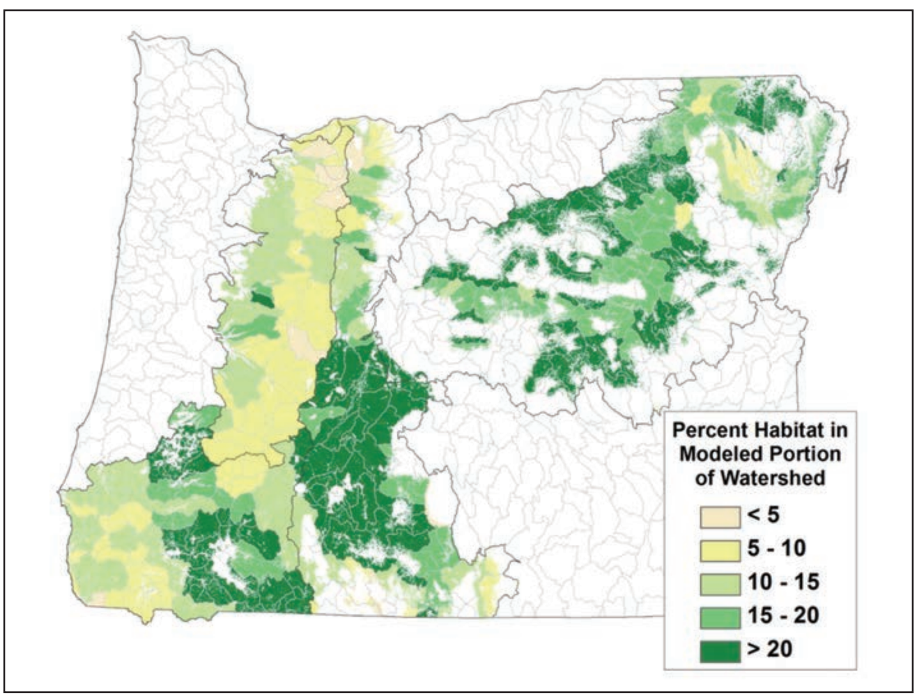
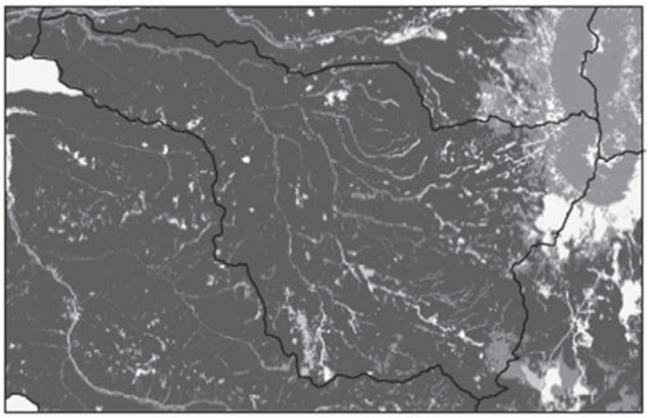


Figure 3b—Visualization of results at the HUC5 scale suggest a larger total range area than when visualized at the 30-meter scale because of aggregations of habitat information presented in relation to extent of associated watershed boundaries. Appreciation of both species ecology and scale of inference are needed for accurate assessment of habitat information.



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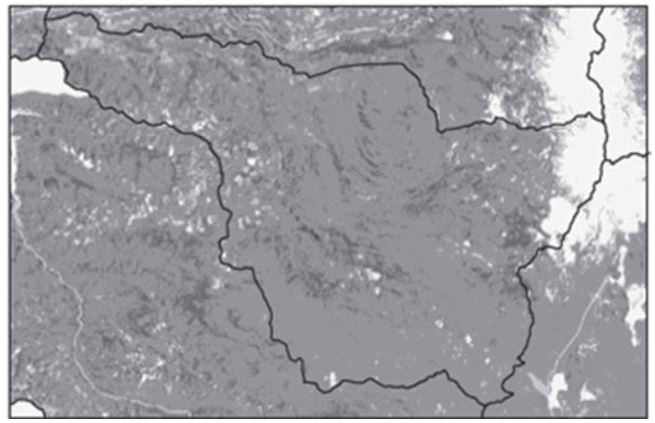

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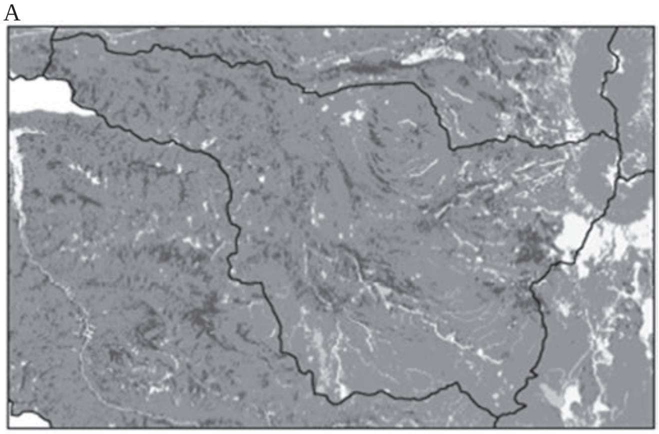


C

Figure 4—Research objectives, data used, and scale of analysis affect interpretation of data. Illustrated here are HUC5 watersheds in proximity to the Three Sisters and Mt. Washington Wilderness areas of the Cascade Mountains in Oregon. a) GAP used 30-meter resolution land cover data to extrapolate habitat based on known species distribution and data derived from individual pixels. Geographic features are preserved from land cover data. b) The GNN approach imputes vegetation characteristics based on focal and neighboring FIA plot data. Geographic features are not preserved in the imputation process c) Combined GAP and GNN provides the ability to integrate vegetation complexity and preservation of geographic features derived from raw land cover data. Darker gray colors indicate relatively “better” habitat than white and lighter gray colors. (Source: Oregon Natural Heritage Information Center 2005 updated ODFW Conservation Straegy GAP Analysis).

GAP analysis (fig. 5c) rather than current known distribution. The GAP and STM analyses both focused on the same observational unit and incorporated GNN data, but the slightly different objectives, specific selection process of observational units, and scales of analysis results in two different presentations of results. As a result, clarity of objectives and acknowledgement of differences of scale allows for understanding of why results may vary across potentially seemingly similar evaluations.

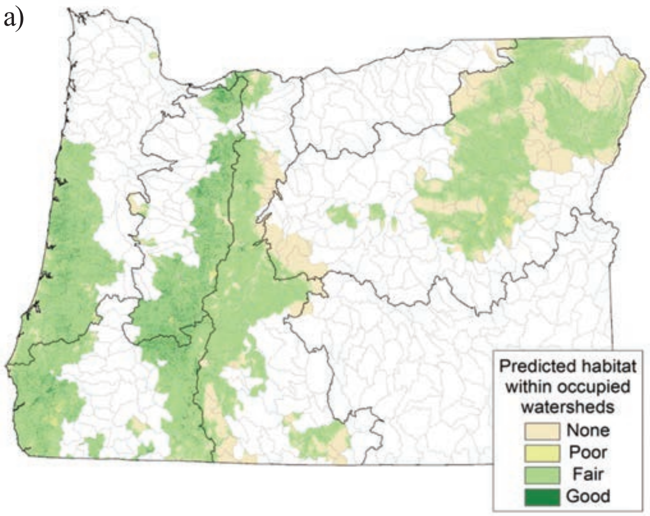
Conclusion

The objectives of this paper were to establish the importance of wildlife habitat analysis in policy and planning decisions and highlight some factors to be considered when balancing feasibility and precision of wildlife habitat analysis using STM data. Wildlife is important to policymaking for both ecological and social reasons. Evaluation of wildlife habitat is a complex process, and analysis can vary based on the observational unit, the ability to match species-habitat relationships with data related to other planning objectives, individual evaluation of information used to construct habitat models, and scale used for analysis. Even the most-studied species provide researchers limited information because of the complexities inherent in ecological systems. Therefore, results must be interpreted carefully with appropriate scrutiny based on the scope of the data.

Acknowledgments

The authors thank Joseph Bernert, Stephanie Graham, Maggie Massie, Emilie Henderson, three anonymous reviewers, and the technical editors of this general technical report. This research is part of the Integrated Landscape Assessment Project funded by the USDA Forest Service, PNW Research Station, under the American Recovery and Reinvestment Act.

a)



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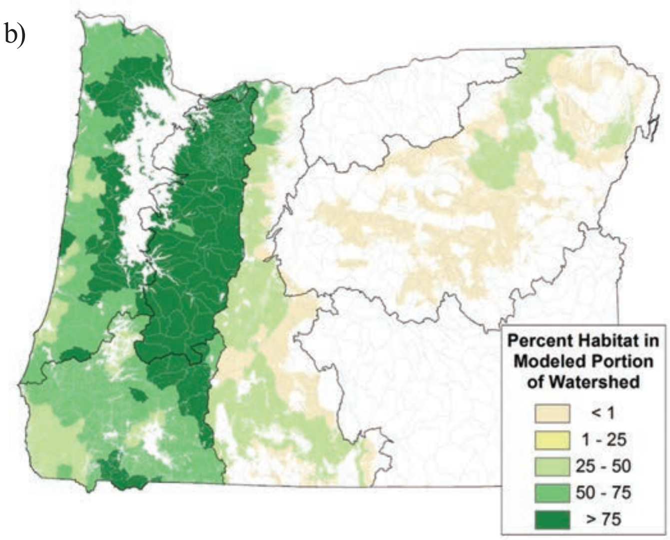

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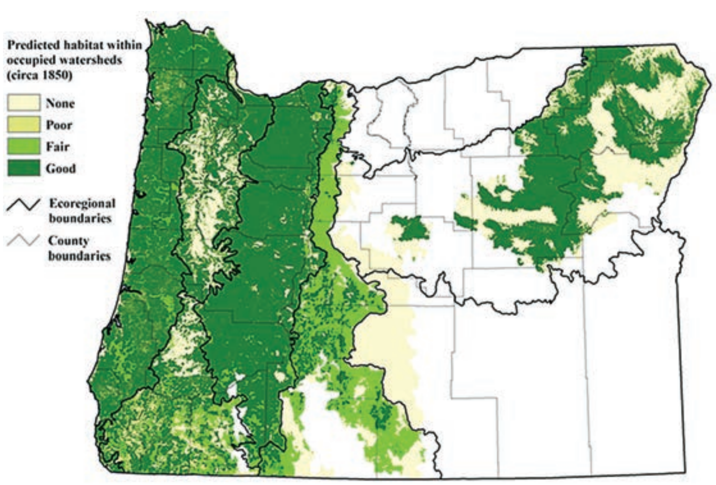

Figure 5—Comparing results from different habitat modeling efforts can have mixed results if research objectives, data used, and scale of analyses are not consistent between analyses. a) In Oregon, GAP predicts American marten habitat among mountain ranges of both states (INR 2011b). In a second analysis, results visually represent broader distribution because of varying objectives, data used, and scale of analysis from those of GAP. c) In fact, projected distribution of marten by within the second analysis "b" suggests closer relationship to historical distribution of the species than present occurrence (INR 2011b).

References

Aubry, K.B.; Raley, C.M. 2002. The pileated woodpecker as a keystone habitat modifier in the Pacific Northwest. In Laudenslayer, W.F. Jr.; Shea. P.J.; Valentine, B.E. [et al.], tech. coords. Proceedings of the symposium on the Ecology and Management of Dead Wood in Western Forests. Gen. Tech. Rep. PSW-GTR-181. Albany, CA: U.S. Department of Agriculture, Forest Service, Pacific Southwest Research Station: 257–274.

Auger, J.; Meyer, S.E.; Black, H.L. 2002. Are American black bears (*Ursus americanus*) legitimate seed dispersers for fleshy-fruited shrubs? American Midland Naturalist. 147: 352–367.

Baldwin, R.A.; Bender, L.C. 2009. Foods and nutritional components of diets of black bear in Rocky Mountain National Park, Colorado. Canadian Journal of Zoology. 87: 1000–1008.

Bolen, E.G.; Robinson, W.L. 1999. Wildlife ecology and management, fourth edition. Upper Saddle River, NJ: Prentice Hall. 605 p.

Booth, V.R. 2009. A comparison of the prices of hunting tourism in southern and eastern Africa. Joint publication of Food and Agriculture Organization of the United Nations and International Council for Game and Wildlife Conservation. Budapest, Hungary. 40 p.

Curtin, S. 2010. The self-preservation and self-development of serious wildlife tourists. International Journal of Tourism Research. 12: 17–33.

Decker, D.J.; Brown, T.L.; Siemer, W.F. 2001. Human Dimensions of Wildlife Management in North America. Bethesda: The Wildlife Society. 447 p.

Dixon, R.D.; Saab, V.A. 2000. Black-backed Woodpecker (*Picoides arcticus*), The Birds of North America Online (A. Poole, Ed.). Ithaca: Cornell Lab of Ornithology; Retrieved from the Birds of North America Online: http://bna.birds.cornell.edu/bna/species/509. (6 April 2012)

Dudley, J.G.; Saab, V.A. 2007. Home range size of black-backed woodpeckers in burned forests of southwestern Idaho. Western North American Naturalist. 67: 593–600.

Evers, L.; Miller, R.F.; Hemstrom, M. [et al.]. 2011. Estimating historical sage-grouse habitat abundance using a state-and-transition model. Natural Resources and Environmental Issues. 17: 1–13.

Forsman, E.D.; Kaminski, T.J.; Lewis, J.C. [et al.]. 2005. Home range and habitat use of northern spotted owls on the Olympic Peninsula, Washington. Journal of Raptor Research. 39: 365–377.

Forsman, E.D.; Swingle, J.K.; Hatch, N.R. 2009. Behavior of red tree voles (*Arborimus longicaudus*) based on continuous video monitoring of nests. Northwest Science. 83: 262–272.

Gilgert, W.; Zack, S. 2010. Integrating multiple ecosystem services into ecological site descriptions. Rangelands. 32: 49–54.

Green, R.J.; Jones, D.N. 2010. Practices, needs, and attitudes of bird-watching tourists in Australia. Sustainable Tourism Cooperative Research Center, Gold Coast, Queensland, Australia. 70 p.

Grimm, N.B. 1995. Why link species and ecosystems? A perspective from ecosystem ecology. In C.G. Jones and J.H. Lawton, eds. Linking species and ecosystems. New York: Chapman and Hall: 5–15.

Hébert, R.; Samson C.; Hout, J. 2008. Factors influencing the abundance of berry plants for black bears, *Ursus americanus*, in Quebec. The Canadian Field-Naturalist. 122: 212–219.

Institute for Natural Resources. 2011a. Black-backed woodpecker. Oregon Wildlife Explorer, Oregon Explorer, Institute for Natural Resources and Oregon State University. Retrieved (11/15/2011), from http://oe.oregonexplorer.info/Wildlife/ExternalContent/SpeciesDistributionMaps/ABNYF07090.pdf. (15 November 2011).

Institute for Natural Resources. 2011b. American marten. Oregon Wildlife Explorer, Oregon Explorer, Institute for Natural Resources and Oregon State University. http://oe.oregonexplorer.info/Wildlife/ExternalContent/SpeciesDistributionMaps/ABNYF07090.pdf. (15 November 2011).

Linders, M.J.; Stinson, D.W. 2006. Washington state recovery plan for the western gray squirrel. Olympia, WA: Washington Department of Fish and Wildlife. 91 p.

Lindsey, P.A.; Alexander, R.; Mills, M.G.L. 2007. Wildlife viewing preferences of visitors to protected areas in South Africa: implications for the role of ecotourism in conservation. Journal of Ecotourism. 6: 19–33.

Mahan, C.G.; Bishop, J.A.; Steele, M.A. [et al.]. 2010. Habitat characteristics and revised gap landscape analysis for the northern flying squirrel (*Glaucomys sabrinus*), a state endangered species in Pennsylvania. The American Midland Naturalist. 164: 283–295.

Manni M.F.; Littlejohn, M.; Evans, J. [et al.]. 2007. Yellowstone National Park Visitor Study, Summer 2006. University of Idaho Park Studies Unit, Visitor Services Project, Report 178. 107 p.

McCall, T.C. Hodgman, T.P.; Diefenbach, D.R. [et al.]. 1996. Beaver populations and their relation to wetland habitat and breeding waterfowl in Maine. Wetlands. 16: 163–172.

National Oceanic and Atmospheric Administration (NOAA). 1969. NEPA/EIS Fact Sheet. http://www.nmfs.noaa.gov/pr/pdfs/permits/nepa.pdf. (2 November 2011)

Ohmann, J.L.; Gregory, M.J. 2002. Predictive mapping of forest composition and structure with direct gradient analysis and nearest-neighbor imputation in coastal Oregon, USA. Canadian Journal of Forest Research. 32: 725–741.

Papadogiannaki, E.; Eury, D.; Hollenhorst, S.J. 2009. Great Smoky Mountains National Park Visitor Study, Summer 2008. University of Idaho Park Studies Unit, Visitor Services Project, Report 205. 102 p.

Payer, D.C.; Harrison, D.J. 2003. Influence of forest structure on habitat use by American marten in an industrial forest. Forest Ecology and Management. 179: 145–156.

Pollock, M.M.; Pess, G.R.; Beechie, T.J. [et al.]. 2004. The importance of beaver ponds to coho salmon production in the Stillaguamish River Basin, Washington, USA. North American Journal of Fisheries Management. 23: 749–760.

Powell, R.A.; Buskirk, S.W.; Zielinski, W.J. 2003. Fisher and marten. In: G.A. Feldhamer, B.C. Thompson, and J.A. Chapman (eds.). Wild mammals of North America: biology, management, and conservation, second edition. Baltimore, MD: John Hopkins University Press: 635–649.

Saab, V.A.; Dudley, J.; Thompson, W.L. 2004. Factors influencing occupancy of nest cavities in recently burned forests. The Condor. 106: 20–36.

Saab, V.A.; Russell, R.E; Dudley, J.G. 2009. Nest-site slection by cavity-nesting birds in relation to postfire salvage logging. Forest Ecology and Management. 257: 151–159.

Schupp, E.W. 1993. Quantity, quality and the effectiveness of seed dispersal by animals. Vegetatio. 107–108: 15–29.

Scott, J.M.; Davis, F.; Csuti, B. [et al.]. 1993. GAP analysis: a geographic approach to protection of biological diversity. Wildlife Monographs. 123: 3–41.

Siepielski, A.M.; Benkman, C.W. 2007. Convergent patterns in the selection mosaic for two North American bird-dispersed pines. Ecological Monographs. 77: 203–220.

Stiles, E.W. 2000. Animals as seed dispersers. In: Fenner, M., ed. Seeds: the ecology of regeneration in plant communities, second edition. Wellingford, United Kingdom: CABI Publishing: 111–124.

U.S. Department of Agriculture, Forest Service [USDA FS]. 1997. 1997 Northwest forest plan: an ecosystem management approach. Washington, DC: U.S. Department of Agriculture. http://www.fs.fed.us/pnw/pfp.pdf. (11 October 2011).

U.S. Fish and Wildlife Service [USFWS]. 2011. ESA basics: more than 30 years of conserving endangered species. Arlington, VA: U.S. Fish and Wildlife Service, http://www.fws.gov/endangered. (11 October 2011).

Van Ormer, C.; Littlejohn, M.; Gramann, J.H. 2001. Olympic National Park Visitor Study, Summer 2000. University of Idaho Park Studies Unit, Visitor Services Project, Report 121. 127 p.

Wisdom, M.J.; Rowland, M.M.; Wales, B.C. [et al.]. 2002. Modeled effects of sagebrush-steppe restoration on greater sage-grouse in the interior Columbia Basin, U.S.A. Conservation Biology. 16: 1223–1231.

Wright, J.P.; Jones, C.G.; Flecker, A.S. 2002. An ecosystem engineer, the beaver, increases species richness at the landscape scale. Oecologia. 132: 96–101.

Forecasting Timber, Biomass, and Tree Carbon Pools with the Output of State and Transition Models

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Abstract

The Integrated Landscape Assessment Project (ILAP) uses spatial vegetation data and state and transition models (STM) to forecast future vegetation conditions and the interacting effects of natural disturbances and management activities. Results from ILAP will help land managers, planners, and policymakers evaluate management strategies that reduce fire risk, improve wildlife habitat, and benefit rural communities. This case study illustrates the methodology for modeling timber volume, biomass estimates, and the forest carbon over time with the output of STM simulations. It presents how the Forest Inventory and Analysis data were applied to assist the interpretation of STM simulation results and derives useful information to the public. The method was applied in the central Washington study area to project the timber production, biomass supply potential, and aboveground carbon stock for two alternative management scenarios.

Keywords: State and transition model, biomass estimation, timber volume, aboveground tree carbon.

Introduction

The Integrated Landscape Assessment Project (ILAP) forecasts potential future vegetation conditions under long-term landscape management scenarios using Path (http://www.apexrms.com) and Vegetation Dynamics Development Tool (VDDT) (ESSA 2007), a state and transition modeling (STM) platform. Future vegetation conditions and the disturbance information from STM simulations are summarized in terms of acres by management allocation (ownership and management types), watershed, and vegetation state classes. The simulation output does not carry the key variables for estimating volume and biomass over time during the simulation such as tree species, diameter, and height. The vegetation conditions are summarized by management allocation for each watershed (table 1a), and the disturbance effects are also displayed by acres with detailed information about the state class changes associated with disturbance types (table 1b). The challenge for people who are interested in information about timber products and biomass is how area-only-related outputs can be translated to the desired attributes. The approach developed for this study uses Forest Inventory and Analysis (FIA) plot information to build look-up tables with volume and biomass attributes linked to each state class for each region. This paper will describe procedures to develop look-up tables for calculating timber volume and biomass as well as tree carbon from VDDT output. The method is demonstrated for central Washington to get the estimates of timber volume, biomass, and aboveground live tree carbon for both management and non-management scenarios. The method can be applied to other landscapes in the ILAP study regions.

Study Area

The study area is in central Washington and encompasses 2.65 million acres of mostly forested land in 25 5th hydrologic unit code (HUC) level watersheds and 16 combinations of ownership and management types (fig. 1). The terrain is mountainous with the highest peaks at over 7,874 feet (2400 m) elevation and the lowest valley bottoms at less than 1,640 feet (500 m). Forested vegetation ranges from very dry environments that support ponderosa pine (*Pinus ponderosa* Lawson and C. Lawson) stands to upper elevation forests of subalpine fir (*Abies lasiocarpa* (Hook.) Nutt.) and mountain hemlock (*Tsuga mertensiana* (Bong.) Carrière). Much of the forest consists of mixed-conifer stands

Table 1a—VDDT inventory output example—dry mixed conifer PVT for general management in the Washington East Cascades ecoregion

Time	Watershed code	Ownership	State class code	Acres
1	104	Forest Service	22210	21.91
1	104	Private	322705	30.11
1	105	Washington state	182705	43.00
1	131	Yakama Nation	332705	8.91
…	…			
5	104	Forest Service	372705	20.50
5	104	Washington state	462235	15.00
5	105	Forest Service	452235	50.90
5	106	Private	192210	18.71
5	131	Yakama Nation	462235	110.21

Table 1b—VDDT Output example for disturbances—dry mixed conifer PVT model for general management on Forest Service lands in the Washington East Cascades ecoregion

Time	Watershed code	Disturbance	From state class code	To state class code	Treated acres
1	104	Partial harvest, small—medium size and medium density	242210	182210	40.5
1	105	Partial harvest, small—medium size and medium density	312210	262210	31.9
1	106	Partial harvest, small—medium size and high density	292705	282705	11.1
2	105	Partial harvest, small –medium size and high density	212705	182705	20.4
3	105	Partial harvest, large—medium size and high density	362705	352705	7.8
3	106	Partial harvest, small—medium size and medium density	282210	262210	16.0

dominated by Douglas-fir (*Pseudotsuga menziesii* (Mirb.) Franco), grand fir (*Abies grandis* (Douglas ex D. Don) Lindl.), and ponderosa pine.

Data and Methodology

The existing vegetation conditions in ILAP came from Gradient Nearest Neighbor (GNN) imputation of inventory plots to 98-feet (30-m) pixels (Ohmann and Gregory 2002) http://www.fsl.orst.edu/lemma/method/methods.php) and other geographic information system (GIS) layers. Each 30-m pixel was associated with an FIA inventory plot. To develop the look-up table with the attributes of volume and biomass, the plots initially assigned to the GNN pixels for the modeling region were retrieved from the inventory database with the detailed individual sample tree information including species, diameter, and height. The following steps were taken to build the volume/biomass look-up table and estimate the volume and biomass:

- Step 1: Collect the plot data (tree list) from the FIA periodic or annual data used in GNN.
- Step 2: Assign a state class to each plot using predefined rules.
- Step 3: Calculate the volume and biomass for each plot associated with the state class using the equations documented in Zhou and Hemstrom (2010).
- Step 4: Summarize volume and biomass attributes for each plot and by product groups or for the total.
- Step 5: Compute average volume and biomass attributes for each state class and build a look-up table.

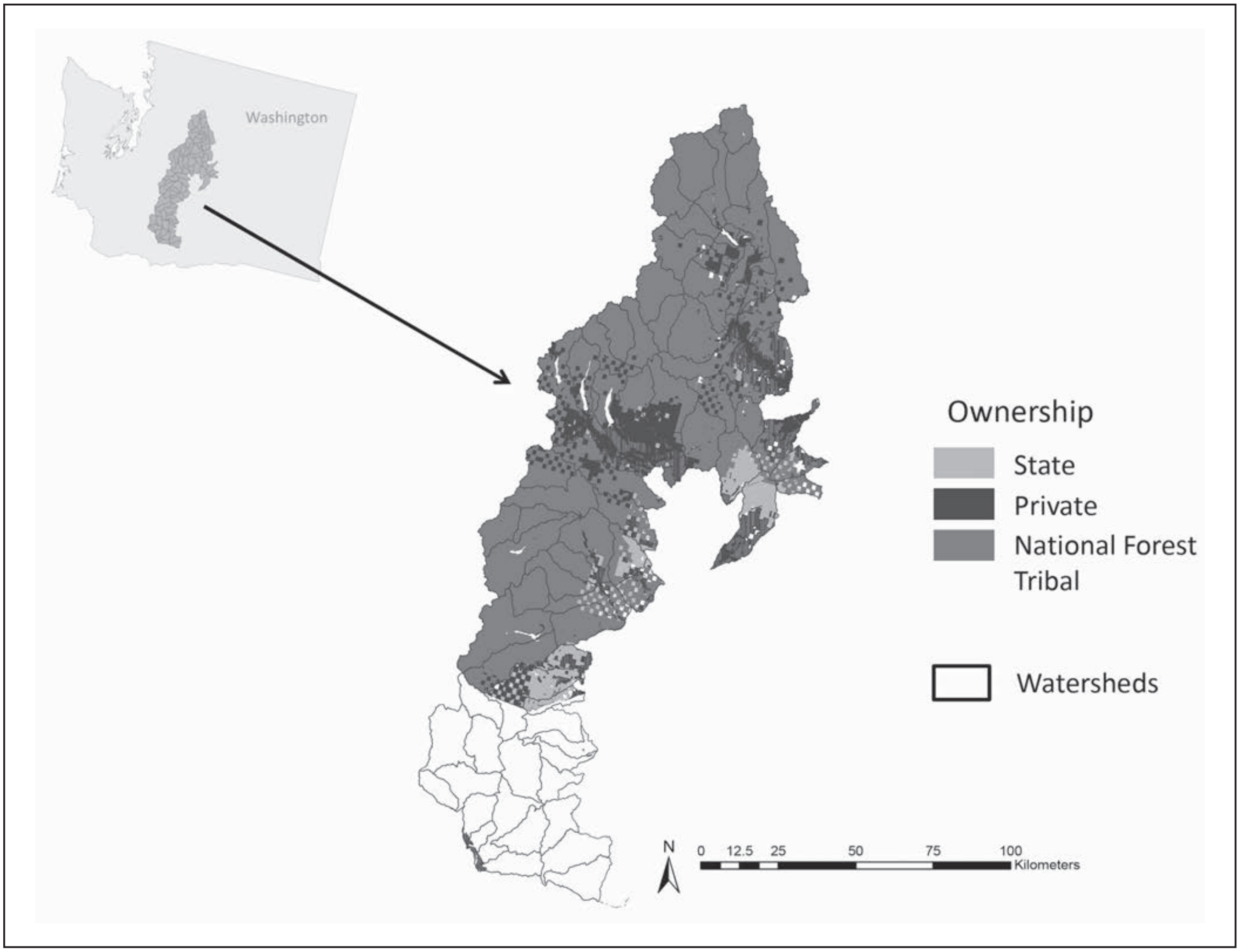

Figure 1—Central Washington landscape study area.

- Step 6: Fill data holes for state classes with inadequate plot representation using a hierarchical rule set for state class similarity.
- Step 7: Calculate volume and biomass by matching the state class from the VDDT output to the look-up table to generate estimates of volume and biomass by landscape stratum and year.
- Step 8: Calculate volume and biomass removals based on the difference in each state class before and after treatment assuming that the treatment happens middle of the year and there is no growth during the treatment year.

Look-up tables are built for each different landscape eco-region. Table 2a shows an example of total volume and biomass for each state class and potential vegetation type (PVT) for the Washington East Cascades (WEC). Volume attributes include total volume (for trees with a diameter at breast height (dbh $\geq$1 in), merchantable tree volume (dbh $\geq$5 in) and sawtimber volume (dbh $\geq$9 in for softwood and dbh $\geq$11 in 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. Table 2b is an example of a look-up table with detailed product classes for the WEC ecological region. Five products are defined in this study based on tree diameters: (1) small tree—dbh <5 in; (2) chip tree—5 in ≤dbh <7 in;

Table 2a—Volume (cubic feet/acre) and biomass (pounds/acre) look-up table example (total) in the Washington East Cascades ecoregion

Potential vegetation type	State class code	Total volume	Merchantable volume	…	Stem biomass	Branch biomass	Total biomass
		cf/ac	*cf/ac*		*lb/ac*	*lb/ac*	*lb/ac*
Dry mixed conifer	332705	1,031	977		24,886	3,778	40,597
Dry mixed conifer	342705	3,835	3,642		97,080	15,004	147,451
Dry mixed conifer	352705	2,732	2,582		67,771	10,765	105,275
Dry mixed conifer	362705	8,584	8,242		237,485	36,817	341,420
Dry mixed conifer	372705	1,866	1,767	…	43,816	6,260	66,699
Pacific silver fir	342710	7,286	6,960		178,732	27,411	274,912
Pacific silver fir	352710	1,203	1,144		31,421	6,303	58,604
Pacific silver fir	392710	10,733	10,186		262,566	38,511	400,021
Pacific silver fir	422710	4,492	4,324		106,910	17,916	159,528

Table 2b—Volume (cubic feet/acre) and biomass (pounds/acre) look-up table example by product groups in the Washington East Cascades ecoregion

Potential vegetation type	State class code	Product group	Total volume	Merchantable volume	…	Stem biomass	Branch biomass	Total biomass
			cf/ac	*cf/ac*		*lb/ac*	*lb/ac*	*lb/ac*
Dry mixed conifer	332705	Small tree	27	0		326	1,942	3,988
Dry mixed conifer	332705	Chip tree	29	9		278	140	577
Dry mixed conifer	332705	Pole tree	42	11		312	93	502
Dry mixed conifer	332705	Small saw	489	375		10,303	2,176	14,826
Dry mixed conifer	332705	Large saw	602	582	…	15,024	2,821	20,703
Pacific silver fir	392710	Small tree	131	0		2,325	2,625	6,651
Pacific silver fir	392710	Chip tree	150	92		3,378	1,027	5,396
Pacific silver fir	392710	Pole tree	248	204		6,011	1,543	9,012
Pacific silver fir	392710	Small saw	3,630	3,470		93,675	21,125	132,599
Pacific silver fir	392710	Large saw	6,642	6,420	…	171,187	39,810	246,362

(3) pole tree—7 in ≤dbh <9 in for softwood and 7 in ≤dbh < 11 in for hardwood; (4) small sawtimber—9 in ≤dbh <20in for softwood and 11 in ≤dbh <20 in for hardwood; and (5) large sawtimber—dbh ≥20 in.

Two management scenarios were simulated for 300 years using STMs in ILAP. The no-management scenario assumed no-management treatments except continued current levels of wildfire suppression; unsuppressed wildfires, and other natural disturbances were allowed to run their course. The current management scenario was based on estimated forest management treatment rates currently in place by land ownership and management types. Management type differed widely among ownerships and land allocations from no treatment (except wildfire suppression) in wilderness and similar reserved areas to commercial timber harvest on private timberlands. We worked with local collaborative groups to gather estimated treatment rates by ownership and land allocation. Treatments included regeneration harvests, commercial thinning, precommercial thinning, tree planting, prescribed fire, and mechanical fuel treatments. Wildfire probabilities were computed by PVT (dry, moist, and cold forest) using data from the Monitoring Trends in Burn Severity study (http://www.mtbs.gov, accessed on 11/29/2011) and reflect wildfire occurrence in the Washington East Cascades ecological region for the 1984 to 2008 time period.

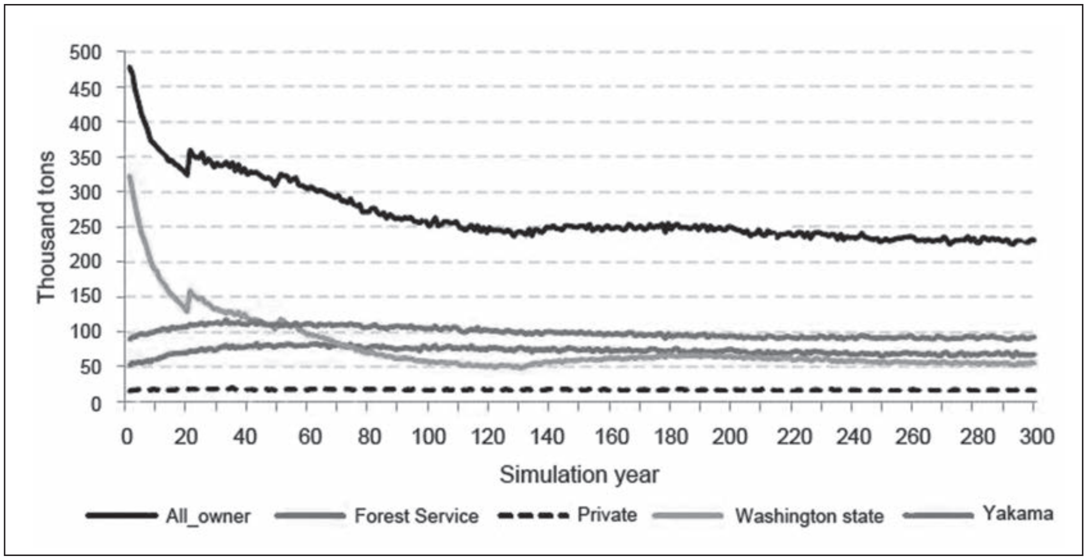

Figure 2—Total tree biomass removals by human disturbances (fuel treatments).

Results

The Biomass Removals From Fuel Reduction Treatment

The materials removed by fuel reduction treatment include merchantable timber products, and small trees or other non-merchantable parts of the trees such as branches, bark, and tops, which might be used as biomass for energy production or other purposes. Of course, depending on how the biomass is defined, potential biomass removals will be different. For simplicity, we considered biomass removals to include everything but foliage; a definition that can be easily changed since biomass components are computed separately. Simulated removals from Forest Service lands and Yakama Tribe lands increase gradually over the first two to three decades and then stabilize. The removals from private lands are relatively low during the entire simulation period because of the low acreage of privately owned lands in this study region (fig. 2). The removals from Washington state forests initially are about 67 percent of the total removals from all ownerships, it decreases dramatically for the first three decades to less than 40 percent of the total, ultimately stabilizing at 20 to 25 percent of the total biomass removals. Conversely, biomass removals from Forest Service and tribal lands increase steadily. The removals from Forest Service lands average 11 percent at the beginning and reach 25 percent at the end of 50 years, and removals from tribal lands during the same period increase from 19 to 35 percent. The removals from private forest land are projected to increase from 3.2 to 5.3 percent of the total biomass removals for the first 50 simulation years.

The Merchantable Tree Volumes

The total standing merchantable tree inventory increased rapidly under both management and no-management scenarios during the first three decades, stabilized for 10 to 20 years, and then decreased (fig. 3). The inventory under the current management scenario peeks at 10.9 billion cubic feet around year 33, about 13.5 percent higher than the initial inventory, and then declines. The inventory under the no-management scenario reaches its maximum, 11.5 billion cubic feet, around year 40, 20 percent more than the beginning inventory; remains steady for about a decade, then declines. In both scenarios, the inventories at the end of simulation period are above original levels although the merchantable tree inventory under the no-management scenario is projected to be 9.7 percent higher than the inventory under the management scenario.

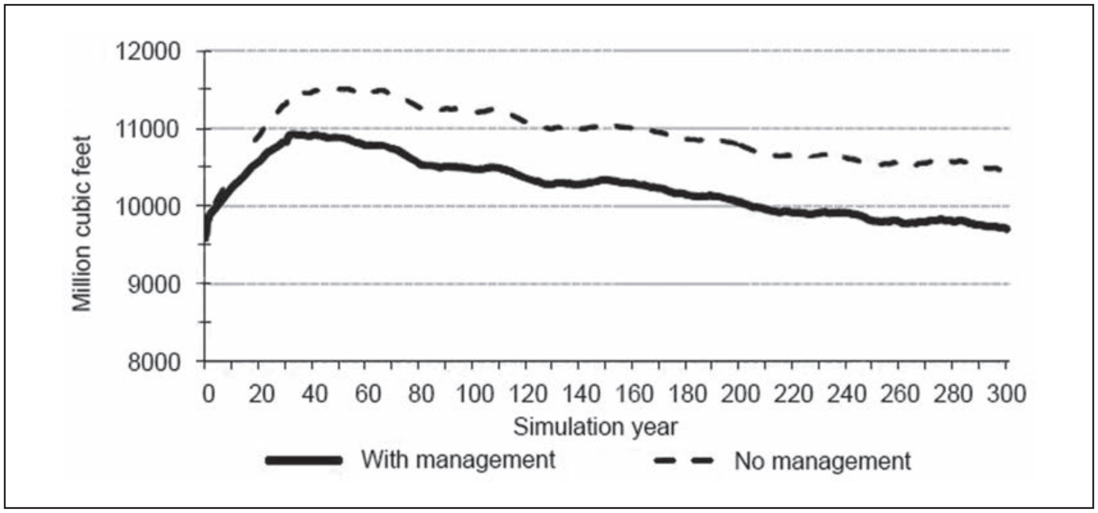

Figure 3—Total merchantable tree inventory in central Washington.

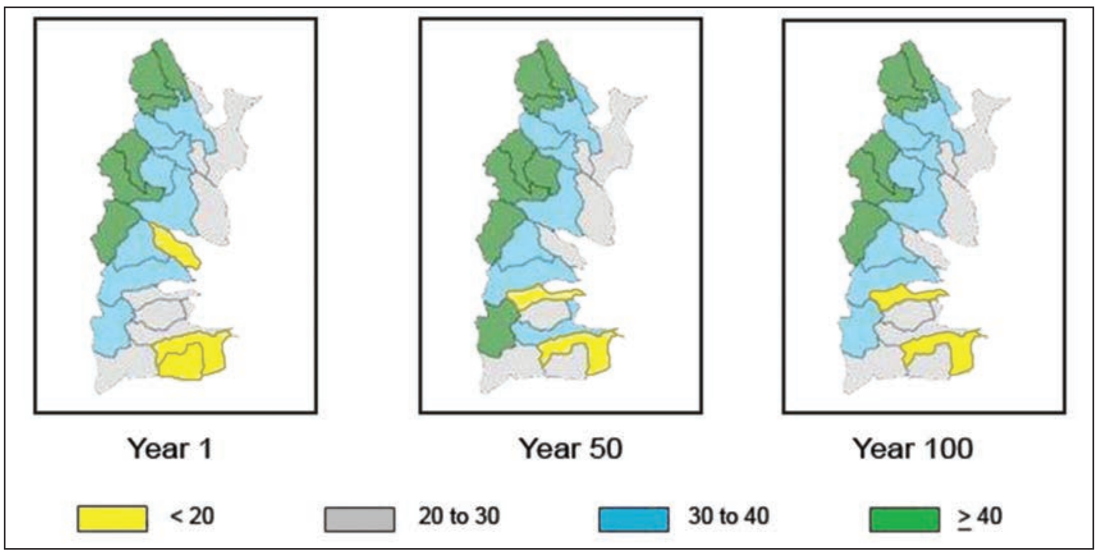

Figure 4—Aboveground carbon stock over time (tons/acre).

Aboveground Tree Carbon Stock Over Simulation Period

We estimate the tree carbon pool by multiplying whole-tree biomass by 0.5 (Heath 2007). Although other carbon pools, such as those in dead trees and soil, are equally important on forest lands, we calculate only the tree-based, aboveground carbon pool. The aboveground tree-based carbon stock varies over time under different management scenarios. The average total aboveground, tree-based carbon stock increased for 21 of the 25 watersheds during the first 50 years (fig. 4). Carbon pools decline slowly in 11 watersheds during the second fifty year and remain relatively constant in the rest. The aboveground live tree carbon stock was relatively stable after the first century for almost all the watersheds.

Discussion and Conclusion

State and transition models can project vegetation structures and landscape condition over time for different management assumptions across a variety of vegetation types, land ownerships, and land management allocations. Forest inventory data provide plot and tree level information to build look-up tables with the attributes of tree volume and biomass that can be attached to STM output. This process provides useful information to forest managers and other decision makers for forest management planning. Our methods could be applied to generate similar look-up tables for soil carbon, dead tree biomass, and other landscape attribute estimates. These estimates can be used to examine the potential long-term effects of various management approaches on forest vegetation conditions (including wildlife habitats, fuels, and others) and general amounts, types, and production locations of economically valuable forest products.

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References

ESSA Technologies Ltd. 2007. Vegetation Dynamics Development Tool user guide, Version 6.0. Vancouver, BC. 196 p.

Heath, L.S. 2007. Section 3.1—Methodology for estimating net carbon stock changes in forest lands remaining forest lands. In: Inventory of U.S. greenhouse gas emissions and sinks: 1990–2005. Unpublished document. On file with: Northern FIA Program, Durham, New Hampshire.

Ohmann, J.; Gregory, M.J. 2002. Predictive mapping of forest composition and structure with direct gradient analysis and nearest neighbor imputation in coastal Oregon, U.S.A. Canadian Journal of Forestry. 32: 725–741.

Zhou, X.; Hemstrom, M.A. 2010. Timber volume and aboveground live tree biomass estimations for landscape analyses in Pacific Northwest. Gen. Tech. Rep. PNW-GTR-819. Portland, OR: U.S. Department of Agriculture, Forest Service, Pacific Northwest Research Station. 31 p. http://www.fs.fed.us/pnw/pubs/pnw_gtr819.pdf.

Through a Glass, Darkly—Comparing VDDT and FVS

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Abstract

Land managers commonly use FVS and VDDT as planning aids. Although complementary, the models differ in their approach to projection, spatial and temporal resolution, simulation units and required input. When both are used, comparison of the model projections helps to identify differences in the assumptions of the two models and hopefully will result in more consistent results across models.

We used the transition probability matrix as the basis for comparing the two models, using side-by-side simulations with FVS and VDDT to project 250 mixed conifer stands. We designed and carried out a simulation experiment with managed and unmanaged scenarios, to explore the consequences of different approaches to filtering FVS outputs by removing censored and rare observations, as well as smoothing out side-effects such as jitter which result from creating categories from continuous data.

Our analysis includes verification of the Preside system, comparison of matrix behavior, as well as the behavior of VDDT with FVS runs imported into VDDT. Three useful conclusions are that: (1) including rare transitions from FVS is important to getting reasonable temporal dynamics; (2) initialization and censoring issues can be ignored; and (3) smoothing and jitter can also be ignored.

One surprising conclusion is that very different assumptions about regeneration cause VDDT and FVS results to be profoundly different for species, size and canopy structure. One nagging question is "how can we tell which model is right?" Field observations would help, and iterative model revision of both FVS and VDDT models is also helpful to a point. Our best advice is to use each model to challenge and improve the assumptions of the other, using each model to illuminate the "blind spots" of the other.

Keywords: Forest Vegetation Simulator, Vegetation Dynamics Display Tool, FVS, VDDT, scale, simulation.

Introduction

Land managers use simulation tools to help them analyze and understand how different management scenarios or disturbance regimes will affect future landscape conditions. Two commonly used tools are the Forest Vegetation Simulator (FVS; Stage 1973, Crookston and Dixon 2005) and the Vegetation Dynamics Development Tool (VDDT[1]; ESSA Technologies Ltd. 2007). The foundations of FVS lie in four decades of empirical growth and yield modeling and forest management, while the roots of VDDT are found in the lineage of probabilistic ecosystem succession models. Although both may be used to simulate the same landscape and vegetation, the models differ in their approaches to projecting vegetation change, in the spatial and temporal resolution of projections and in the fundamental simulation units, processes and required input data.

The decision to use FVS or VDDT can depend upon circumstances such as data availability, disturbance dynamics, forest management strategies, the presence of non-forested conditions, and the scale and type of issues that are being addressed. When there is an opportunity to use both models to project the same landscape, it raises the possibility of comparing the projections. Such comparisons can draw attention to differences in the projections, which can result in a more thorough understanding of the two modeling systems. Going deeper, they also create the opportunity to compare specific common parameters; to consider the reasons for any difference and to decide whether there are

[1] VDDT simulations can also be carried out using the Path Landscape model (http://pathmodel.com), in which the VDDT state transition model is embedded.

compelling reasons to transfer a parameter estimate made by one model, for use in the other. If good reasons exist for modifying previously-accepted VDDT parameters based on FVS, then this could be termed “calibrating” VDDT with FVS.

At the finest resolution, FVS projects the change in density, diameter- and height-growth of a collection of individual trees (the stand) and simulates the development of a non-spatial inventory over time, and often with management activities. When needed, attributes of the whole stand (e.g., percent canopy cover, quadratic mean diameter, basal area) can be easily derived from the finer scale attributes of individual trees. FVS simulations frequently use a 5- or 10-year timestep. In its most basic use, VDDT uses state-transition probabilities to simulate temporal changes in vegetation classes on an annual timestep. These probabilities are often developed in collaborative workshops at which ecologists synthesize published literature and their experience with a particular landscape or ecosystem, to define the vegetation classes found on the landscape, and the transitions between the classes. VDDT models can range from very simple models with a few classes, to very complex models with dozens of vegetation states, along with the disturbance and succession probabilities that control how vegetation classes change. The fundamental unit of resolution of VDDT is a physical abstraction called the pixel[2]. Pixels can represent almost any vegetation state (e.g., “open single-story ponderosa pine dominant” or “mature tall-grass prairie”), but do not explicitly represent an element or cell of any particular size or location in a spatial landscape map. Rather, each pixel represents an instance of one of the possible vegetation states defined for the landscape. Frequently, VDDT users use 1,000 pixels, but the actual number is arbitrary and is chosen to provide enough detail for the management questions under study. Over time, pixels change between different vegetation states in response to succession or disturbance using user-defined deterministic or probabilistic rules.

VDDT models usually define vegetation classes and transition probabilities based on a mixture of literature and expert judgment. Because of this subjective element, some land managers believe that VDDT models could be put on a stronger footing by incorporating parameters estimated from FVS simulations, thus providing more defensible land use decisions. In this way, the empirical basis of FVS might serve as an additional source of guidance for VDDT.

The key step in bridging the two models is to translate FVS results so that the two models have a common basis. This is achieved by post-processing FVS landscape simulations to create vegetation classes that are identical to those used by VDDT, then computing transition probabilities based on the class changes that take place during the FVS stand simulation. A software program called Preside (Moeur and Vandendriesche 2009)[3] has been created for this translation step, using the FVSSTAND keyword and the FVSStand (Vandendriesche 1997) post-processer. Tabular results from FVSStand are processed by Preside to create a classification of stand structure over time, classes that are user-defined and compatible with the class definitions used to create VDDT models for the same landscape. Once the classification step is complete, Preside creates a matrix of the occurrence and frequency of transitions between different states, similar to the transition matrix created by VDDT.

Transitions defined through FVS are the result of its internal dynamics combined with post-simulation classification, while those found in VDDT are defined by a model developer. In the case of VDDT, the model developer defines all possible transitions in advance, while transition pathways and probabilities predicted by FVS are derived from the detailed growth, mortality and regeneration of individual trees in stands simulated over time.

However they were created, the results of the VDDT and FVS simulations use a common set of vegetation classes, and the transitions between those classes should be comparable. That said, the differences between the two models—the continuously varying world of FVS and the

[2] Pixel is technically a misnomer and is unrelated to “picture elements” or remotely sensed data or Geographic Information Systems. This label for VDDT simulation units was coined many years ago and has come into common parlance.

[3] Available at: http://www.fs.fed.us/fmsc/ftp/fvs/software/pre_post/Preside_z.exe.

Table 1—High level summary of the simulation scenarios

Management Scenario	VDDT	FVS
Unmanaged	Succession + Natural Disturbance	Unmanaged
Managed	Succession + Natural Disturbance +	Restoration management
Data Simplification Scenarios		
All	Use all observed transitions	
Sub	Use a subset, removing transitions that occur fewer than 5 times	
Base	Include all timesteps	
Drop Zero (DZ)	Remove transitions in the first timestep	

class- and transition-based world of VDDT—are not simple to overcome. While class models are usually much simpler in structure, both models have subtle assumptions that can produce large effects, and the introduction of class boundaries to FVS simulations can create significant artifacts. As this paper shows, removing these artifacts is not trivial, and can introduce changes to the "signal" of FVS-based transition parameters.

Materials and Methods

As key factors in this study, we created simple managed and unmanaged scenarios in both FVS and VDDT and went on to analyze those simulations in different ways. Facing the challenges of comparing two models with markedly different approaches, we tested a variety of approaches for presenting the data from each model in ways that fostered comparisons. As there are inherent issues with trying to classify a continuous landscape, we performed extra analyses using the unmanaged FVS scenario to explore the sensitivity of our classification boundaries.

Scenario Design

To compare the landscapes predicted by the native-VDDT and FVS outputs converted to VDDT transitions, we made 300 year simulations of Unmanaged and Managed scenarios (see table 1) using both VDDT and FVS. Other scenarios were explored as well, as described below.

Management scenarios—
Two FVS scenarios were compared with matching VDDT simulations. After the results of the FVS simulations were processed with FVSStand, some analyses used a custom Excel spreadsheet to implement different smoothing methods (described below), which were then exported and formatted to conform to the Preside input file format. VDDT outputs were similarly treated, so that both modeling systems were processed identically as much as possible. Preside was configured to use the classification rules shown in table 2, and the information about the state of each stand at each timestep was used to create VDDT-like transition probability matrices. These matrices contained between 42 and 58 different state classes, depending on the scenario and model.

Data simplification scenarios—
One of the outputs of the Preside model is an estimate of the annual probability of changing from one vegetation class to another, based on FVS outputs. It is an attempt to classify FVS in the language of VDDT. Using options in Preside, we examined two different approaches to calculating the probability of different transitions. The first approach we used—*All*—accepts all observed transitions; even those that are not observed very frequently. This allows greater successional complexity to be captured, but rare transition parameter estimates may not be very precise. The second approach we examined—*Sub*—removed transitions that were observed fewer than 5 times: less than 1 percent of the total number of transitions. The implicit hypothesis underlying this approach is that more frequent transitions are more important to the overall successional pattern and that less frequent transitions can be ignored.

As a second part of this exercise we explored the impact of removing a model initialization artifact from the FVS results. The first approach—*Base*—includes the initial timestep in Preside's calculation of classes and transitions. The second treatment—*Drop Zero*—accounts for the fact that two kinds of initialization artifacts can incorrectly influence the estimation of transition probabilities. The first is caused by the internal calibration of FVS growth and mortality to local site conditions during model initialization.

Table 2—Stand classification variables defined for the Preside program

Variable Name	Variable Type	Class	Definition
DOMTYPE	Categorical	Classification of the dominant tree species for the landscape.	
		PP	early seral—LAOC, PICO, PIPO or TEIX leading
		DW	late seral—DF, WF or any other species leading
QMD	Continuous	Quadratic mean diameter (cm) of the >80th percentile largest-diameter trees	
		Y	<5cm—young seedling/sapling
		P	5-10cm—pole
		S	10-15cm—small tree
		M	15-20cm—medium tree
		L	20-30cm—large tree
		G	>30cm—giant tree
CANCOV	Continuous	Canopy cover (percent)	
		n	<10 percent—non-stocked
		o	10-40 percent—open canopy
		m	40-70 percent—medium canopy
		c	70 percent—closed canopy
		—	Unclassifiable
STORY	Categorical	Number of canopy layers	
		1	one story or poorly defined
		2	two story
		—	Unclassifiable

This adjustment is reported in the first simulation timestep, and can create "pseudo-transitions" that are unrelated to natural succession. The second artifact is caused by the fact that the first transition observation lacks any information about how long the stand was in its initial vegetation state prior to the transition.

Smoothing scenarios—
Classifying a landscape made up of continuous variables (such as height or QMD) into discrete variables carries the risk that the classification can change in more than one direction, especially if any variable is near a classification boundary. In this case a stand can move back and forth between two classes, creating jitter. We developed and tested two different ways to smooth the classification of FVS stand output to see if any would reduce the jitter or impact the landscape-level results. Detailed descriptions of the these methods and their results are in found in Appendix 2.

Program Configuration

FVS—
We used 500 FVS inventories (two inventories of 250 stands taken at different times) taken from mixed conifer dry site stands from the East Cascades of Washington, USA, compiled and prepared as part of the IMAP study (Miles Hemstrom, pers. comm.). These stands are principally made up of Douglas-fir and ponderosa pine. The 500 FVS inventories were projected using 10 year timesteps for 200 years using the East Cascades FVS variant (model date: 04/07/09). Ingrowth in open stands was simulated by using the Repute Event Monitor keyword package (Vandendriesche, 2009). The FVS main output file and the supplementary detailed output file created by the FVSStand keyword were then post-processed using the FVSSTAND Alone program, creating a set of input files for Preside.

After an initial analysis of the inventory, the FVS East Cascades variant was calibrated by adjusting small and large tree growth increments and maximum SDI and basal area (see Appendix 1 for details). In the unmanaged scenario, FVS was calibrated and run with the Repute-based regeneration information described above. For the managed scenario we implemented a simple management system with a restoration focus in which stands are thinned from below whenever they exceed 65 percent of the maximum SDI, removing smaller stems until the SDI is reduced to 50 percent of the maximum SDI, preferentially retaining ponderosa pine. These limits were selected based on Cochrane and others (1994), who recommend that stands be managed between 50 percent and 75 percent of full stocking. With these two thresholds, stands are re-entered every 20 to 40 years. Over time, this system creates open forests of large, older ponderosa pine, using smaller trees for low or moderate amounts of timber. Like the unmanaged scenario, management simulations were run with the calibrated model using 10 year timesteps for 200 years.

VDDT—

VDDT diagrams were defined for the landscape and included numerous pathways representing succession, natural disturbances from bark beetles, mistletoe, wild fire and management (J. Merzenich, *pers. comm.*). For the unmanaged simulation, all management pathways were disabled. Since base FVS mortality rates include mortality from background disturbance, we kept most disturbance pathways active to make the FVS and VDDT simulations more comparable. The only disturbance type that was excluded from the unmanaged simulations was stand-replacing fire, since this level of disturbance is not consistent with the FVS assumptions.

Simulations of 1000 pixels were initialized by assigning pixels evenly over most of the VDDT vegetation classes. We then simulated VDDT for 300 annual timesteps; saving information about the transitions that occurred in each timestep and printing the detailed landscape state class output every 10 timesteps to emulate the temporal resolution of FVS. We then formatted the output so that it could be provided to Preside in the same way as FVS. Finally, we ran Preside using the output generated by VDDT, allowing Preside to independently calculate the transition probabilities, in order to verify the Preside algorithms.

The VDDT management scenario was based on the same model diagram and initial conditions as the unmanaged scenario, activating some of the management pathways and simulating these pathways using a file of input multipliers which defined multipliers for a private lands scenario. In order to keep the managed scenario as similar as possible to its FVS management counterpart, we excluded the simulation of prescribed fire. As with the unmanaged VDDT runs, we ran the model for 300 years, saving transition information in all years, and state information every 10 years.

Preside—

Preside takes classification rules provided by the user and applies those criteria to the output of FVS landscape projections, classifying stand structure in a way that is compatible with VDDT. When processing is complete, FVS stand structure is classified at each timestep with user-defined rules corresponding to VDDT vegetation states, along with the probability of changing from one vegetation state to another. Preside organizes these transition probabilities into a matrix that can be compared with a matrix of transition probabilities created by VDDT for the same landscape[4] Following a comparison step, VDDT transition probabilities can be adjusted to be more FVS-like, if desired.

To faithfully recreate the VDDT classification for the mixed dry landscape, we configured Preside to classify each stand and timestep using the classification variables and breakpoints shown in table 2. Preside labelled each stand's state at each timestep, constructing a label from the concatenation of the four variables:

DOMTYPE + QMD + CANCOV + STORY

Using this coding, the label **PPGo2** indicates a pine-leading (PP) stand of giant (G) diameter class, with an open canopy (o) having two (2) stories.

The first step taken by Preside is the classification of each stand, so we began by verifying its classification algorithm. To do this, we exported results from the FVS simulation of the unmanaged landscape scenario simulation to a database using the FVS Database Extension (Forest

[4] Although stochastic transitions are commonly found in VDDT models, VDDT can also be parameterized for other modes of state change, including combinations of deterministic and stochastic rules.

Vegetation Simulator Staff 2003). We then classified those results using our own SQL queries and compared our stand classification with the classification created by Preside. Using the same stratification and breakpoints used by Preside, our checks agreed with the Preside classification. After further checks we also concluded that Preside's calculation of class-transitions were correct.

Consistency of VDDT and VDDT—Preside—VDDT Transitions

As in internal consistency check of our arithmetic and of the methodology for processing transitions with Preside, we compared the distribution of vegetation classes created directly from VDDT against VDDT output that had been processed through Preside and then simulated by VDDT. We expected that after allowing for stochastic variation, a comparison of the results these simulations would show them to be identical, and that this would verify the mathematical steps upon which the translation of FVS results are based.

We began this check by performing an Unmanaged VDDT simulation lasting 300 years, printing the landscape state every 10 years to match the output interval from FVS. This decadal output was then processed into a format that could be read by Preside, which then independently recalculated the transition probabilities using the same algorithms employed for processing FVS output. These probabilities were provided as input to a new VDDT simulation, treating all transitions as purely probabilistic. The results of the two VDDT simulations were independently simulated five times and then compared.

Transforming the Time Scale

Preside (Version 2010.06) uses decadal FVS output, conceptually creating an initial square matrix $\boldsymbol{M}$ of 10-year transition probabilities. Each diagonal element of this matrix represents the probability of remaining in vegetation class $\boldsymbol{i}$ after 10 years, and each off-diagonal element represents the probability of changing from vegetation class $\boldsymbol{i}$ to class $\boldsymbol{j}$ in the decade. Zero values are allowed, but each row must sum to 1.0. Preside converts these values to annual transition probabilities through the separate transformations for the diagonal terms of the matrix (Eqn. 1: probability of remaining in the same vegetation state) and for the off-diagonal elements (Eqn. 2), in which $\boldsymbol{M'}$ is the resulting square matrix of 1-year transition probabilities:

$$M'_{i,j \mid i=j} = e^{\ln(\lambda)\, M_{i,j}/10} \qquad \text{[Eqn 1]}$$

$$M'_{i,j \mid i\#j} = 1 - e^{\ln(1-M_{i,j})/10} \qquad \text{[Eqn 2]}$$

In cases where a diagonal entry is absent from the matrix (i.e., in which all pixels in vegetation class $\boldsymbol{i}$ change class in a single timestep), we found that row totals calculated by Preside do not always sum to 1.0, a necessary condition for accounting for all transitions over the landscape. However, the impact of this problem is small since VDDT does not actively make use of the probability of remaining in a vegetation class, and uses only the probability of changing state. Hence any imbalance in the transition calculations will be offset by the implicit value of the diagonal, which will result in an implicit row total of 1.0.

However, the annualized probabilities of $\boldsymbol{M'}$ computed using Eqn. 1 and 2 do not account for transitions to intermediate classes at the 1-year time scale, which in the case of high transition probabilities (i.e., fast transitions) can lead to different vegetation classes from those calculated at the 10-year timestep. It is therefore possible that when simulated at a faster timestep, the Preside time-scaling method will incorrectly simplify transitions that lead to different pathways.

As an alternative method, we used a classical matrix analysis approach to change from the decadal to annual time scale, to compare the matrices produced by FVS and VDDT. Beginning with the matrix $\boldsymbol{M}$ produced by Preside, we transformed to the annual probability matrix $\boldsymbol{M'}$ by transforming $\boldsymbol{M}$ to diagonal form (Eqn. 3), where $\boldsymbol{V}$ and $\boldsymbol{\lambda}$ are the matrix of eigenvectors and array of eigenvalues of $\boldsymbol{M}$, respectively.

$$M = V^{-1}\, D(\lambda)\, V \qquad \text{[Eqn. 3]}$$

Eqn. 4 then rescales the eigenvalues of the diagonal matrix $\boldsymbol{D}$ from a decadal to an annual interval leaving the eigenvectors unchanged.

$$M' = V^{-1}\, D(e^{\ln(\lambda)/10})\, V \qquad \text{[Eqn. 4]}$$

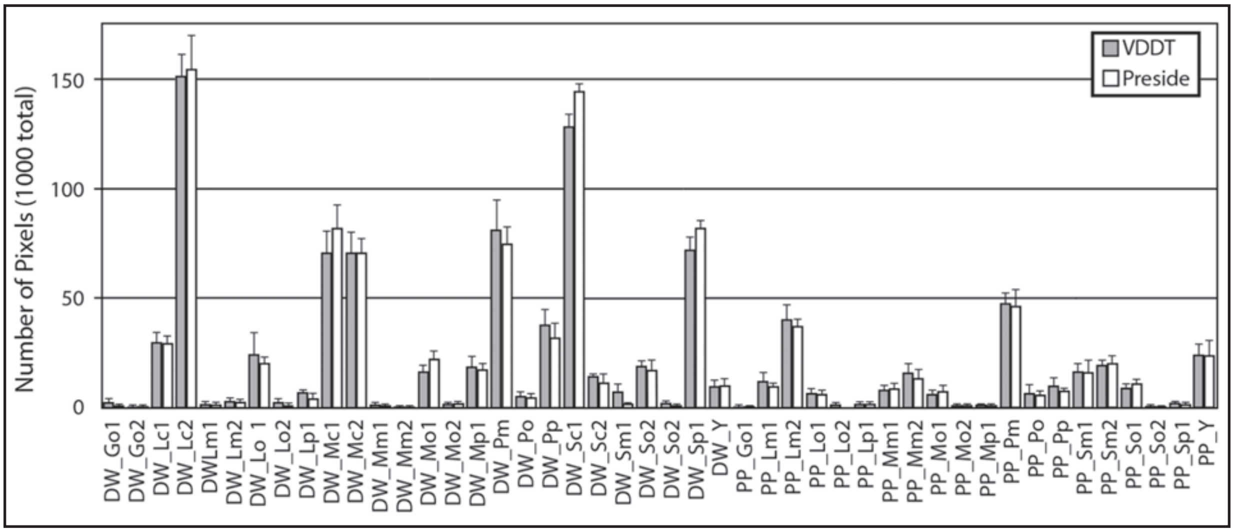

Figure 1—Comparison of the number of pixels in each state class after 300 years for the original VDDT simulation and for the VDDT simulation using transitions and probabilities exported and recalculated by Preside. Error bars are from 5 Monte Carlo simulations.

This method avoids the separate equations used by Preside, scales correctly between any two time intervals, and produces row sums of 1.0 for all possible timesteps.

Matrix Analyses

The properties of the transition matrices were used to compare VDDT and FVS simulations over the scenarios and treatments described above. Each row and column of the transition probability matrix represents a unique vegetation class (e.g., PPGo1). Depending on the scenario, there were 42 rows and columns in the FVS matrix and up to 58 rows and columns for the VDDT simulations. In the case of the VDDT simulations, the extra 16 vegetation states represent post-disturbance classes which are impossible to distinguish from other vegetation classes in FVS.

Matrix properties were characterized in two ways. First we found the dominant eigenvector, which is the theoretical equilibrium composition of the landscape. Secondly, we explored the behavior of the matrix system over time through simulation. One method of analyzing the temporal behavior was to run the matrix in VDDT. The other method we used was to characterize temporal dynamics through Monte Carlo simulation of the matrix directly (i.e., not in VDDT) through the calculation of the approach to the theoretical stable equilibrium, using the time to reach 99.9 percent of the equilibrium as a consistent measure of system dynamics. This is an arbitrary condition that will never be seen in physical landscapes. However, it provides a standardized way of comparing the behavior of different transition matrices.[5]

Results

Verification of Preside Calculations

A comparison of the results from the simulation of the unmanaged scenario in VDDT and a previous run of VDDT exported and processed through Preside shows that both runs are very similar (fig. 1). In both runs the most frequent 10 classes are identical and represent about 75 percent of the total area. Only two of the classes differ by more than 1 percent of the total landscape area. This result confirms that Preside transforms results in a consistent way, and that the transition probabilities are comparable between the two models, other things being equal. Indirectly, these results also show that for this landscape, the differences of the two

[5] Supplemental information about the simulation methodology and detailed results are available at: http://essa.com/wp-content/uploads/RobinsonBeukema2012Supplement.pdf.

Table 3—Summary of vegetation class transitions present in Unmanaged FVS and VDDT models. The two simulation systems have few vegetation states in common

	Observed Transitions		
Vegetation Class	FVS only	VDDT only	FVS and VDDT
Regular	127	145	43
Post Disturbance	0	115	0
Total	127	260	43

methods of transforming from decadal to annual timestep are not large in this landscape.

Presence and Absence of Transitions

The transition matrices from the Unmanaged VDDT and FVS simulations were analyzed using a simple count of the presence or absence of different transitions (table 3). Remarkably, FVS predicts many fewer total transitions than VDDT and only one quarter can be matched with a transition in the Unmanaged VDDT landscape. A further 115 transitions are found only in the post-disturbance classes that are unique to VDDT. Considering all transitions, there is about a two-to-one disparity. From this simplified aggregation we conclude that VDDT shows a much wider range of behavior, in spite of the fact that we might expect FVS to account more explicitly for growth, mortality and regeneration in multi-species stands. As described in more detail below, we find that much of the complex behavior of VDDT can be traced to detailed transitions driven by disturbance agents in VDDT.

FVS Transitions Caused by Calibration and Regeneration

Study of the transitions that are absent from VDDT but present in FVS shows that a number of them occur when a continuous classifier (QMD or CANCOV) switches to a smaller class. Of the 11,000 timesteps in the projection, such events take place in about 7 percent of timesteps: 226 times for QMD and 505 times for CANCOV. We found that these changes are primarily caused by regeneration in the FVS model. For QMD this happens when a pulse of regeneration grows large enough to be classified among the largest 20 percent of trees. The diameter threshold then drops to include the numerous smaller trees and the indicator declines markedly.

We examined sixteen stands that showed the most extreme decline in CANCOV or QMD. Of these sixteen cases, ten undergo the transition in the first model timestep, indicating a marked change from the inventory condition to the first projected state. Examples of results showing this behavior are shown in figure 2, which shows vegetation class over time for 10 stands. In this example, nine of the stands undergo a transition between timestep 0 and timestep 1. Our analysis shows that the first kind of declining transition is a simulation artifact produced by the initial FVS self-calibration adjustment and expressed as large tree mortality. Fortunately, transitions occurring this way can be screened out using the *Drop Zero* method, as described below.

The second kind of declining transition typically happens later in the simulation, as regeneration begins to reach a size where it can be in the top 20 percent of trees, contributing noticeably to canopy cover. It can then trigger a change in the dominant overstory layer, as the stand moves from larger older trees to include smaller younger trees. This kind of transition is not inherently unreasonable, but underscores the role of regeneration in stand dynamics and in measures related to stand maturity, and signals the need for further consideration of how the indicator is intended to be used. If QMD or CANCOV alone do not adequately capture the qualities of a mature stand and the stand truly remains in a mature state, then novel attributes (perhaps combinations of existing attributes or heuristic rules) may be called for. Alternatively, these transitions caused by regeneration also underscore the need to better consider the role of small trees when defining states and state transitions in VDDT.

Jittery Transitions

Besides declining transitions, there are frequent cases of back-and-forth switching between two classes (see two lower horizontal boxes in fig. 2), occasionally accompanied by a change in dominant cover type (PP or DW in our study landscape). We examined nine stands with jitter effects, and

Stand ID	Model Timestep																		
	0	1				5					10					15			
1238228-1	U	T	T	T	T	L	L	L	L	L	L	L	L	L	L	L	L	M	M
1238240-2	E	F	F	J	J	J	L	L	M	M	M	M	M	M	P	P	P	P	P
1252252-2	A	B	B	C	C	C	C	E	F	G	G	G	F	J	J	K	K	K	M
1278224-1	H	C	C	E	J	J	L	L	L	L	M	M	M	M	M	R	R	R	R
1284300-1	F	C	C	E	F	F	F	F	F	F	J	J	J	J	J	J	J	J	J
2201208-1	H	S	C	C	E	F	F	F	F	F	F	F	J	J	J	J	J	J	J
2207204-1	D	C	C	C	C	E	E	G	E	F	J	J	J	J	K	M	M	M	M
2213210-1	F	F	F	F	F	F	F	F	F	E	F	F	F	F	F	F	F	J	J
2277230-1	N	F	F	F	J	J	J	J	J	J	J	J	J	J	J	J	J	J	J
2278314-2	F	C	C	F	F	F	F	J	J	J	L	L	L	L	L	L	L	L	N

Figure 2—Vegetation classes are shown for 10 stands from the unmanaged FVS landscape simulation. Each class is represented by a single letter and state changes are shaded for clarity. The heavy box at left highlights the large number of transitions that take place between timestep zero (inventory) and one. The three horizontal boxes show examples of rapid state change (upper box) and classification jitter (two lower boxes).

found that jitter is also closely associated with regeneration, generally taking place when QMD or CANCOV are close to a transition threshold. In these cases the regeneration contribution, while small, is sufficient to push the indicator over the class boundary. In the examples we studied, significant regeneration mortality can occur in the following cycle, causing the classifier value to drop and return back across the threshold a second time. If there is another round of regeneration, the classification can change yet one more time. Jitter ends when there is no regeneration for two cycles, allowing the stand to grow substantially beyond the boundary condition.

A classification change due to small back and forth movement across a boundary has a large impact on the average residency time, expressed as the annual probability of changing from Class X to Class Y. A stand that alternates between two classes each 10-year timestep will have numerous predicted residence times of 10 years. If the classification was more flexible or the stand was farther from a boundary, it might have residence times of 20 or 40 years. For example, consider a stand with nearly 70 percent canopy cover for four cycles: 69.5, 70.1, 69.6 and 70.3 percent. In this example, the stand would be classified as: X-Y-X-Y. If the values were all slightly less (e.g., ranging from 68–69 percent), they would all remain in class X, resulting in a longer residence time with no transitions, and if they were slightly higher, they would all be in class Y.

This kind of classification problem is not new. A nearly identical case is addressed by Stage (1997) in the context of stand structure classification, transition probability and declining transitions. Stage's suggested approach to the problem is to move the boundary by one-sixth of the class width whenever a retrogressive (declining) transition is predicted, thus avoiding the transition. This might be reasonable for individual cases, but it often only moves the boundary problem to another value or causes it to emerge in another stand, and does not remove it entirely. Other ad hoc solutions that involve visualization of a stand's state space and its proximity to a border may be suitable for single stands, but are not feasible for the automatic classification of thousands of stands over dozens of timesteps. Ideally, what are needed are algorithms that solve the logical problem of the classification boundary and which can be implemented programmatically. In Appendix 2 we present different approaches to smoothing and further analyze the consequence of these smoothing methods on measures of the equilibrium behavior of the system. Generally, our smoothing methods resulted in landscapes that were quite

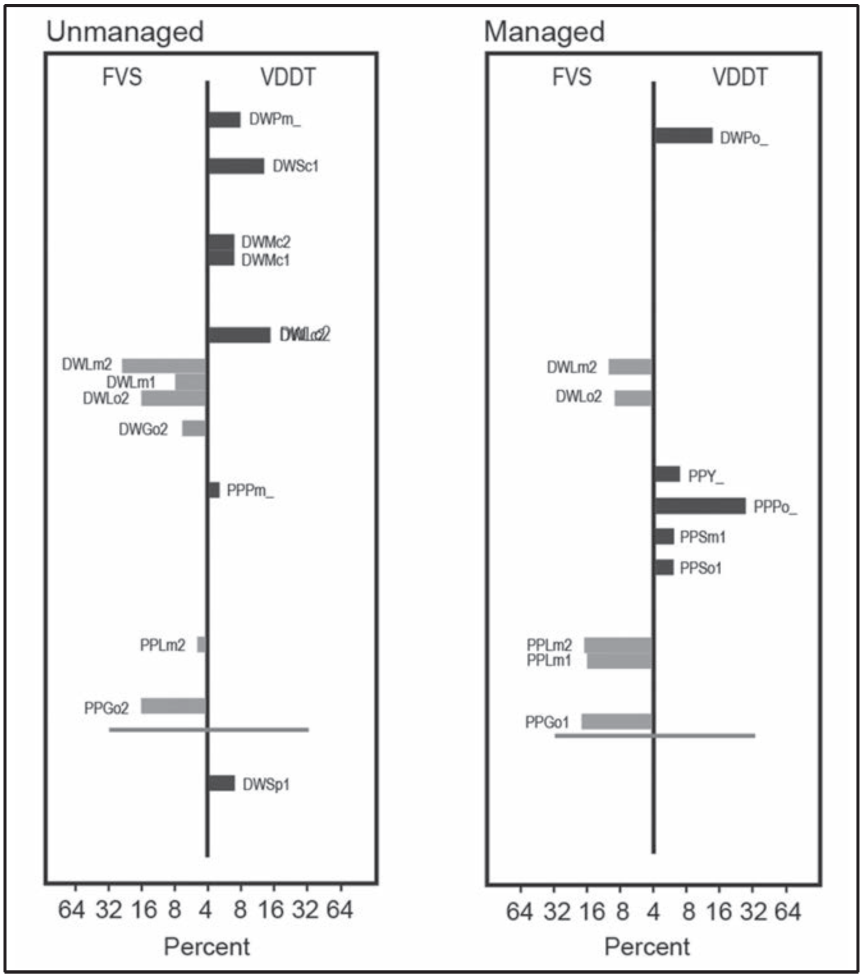

Figure 3—Left and right panels compare the composition of Unmanaged (left) and Managed (right) landscapes near equilibrium. In each panel the FVS-derived landscape composition is shown on the left (light grey bars) and the VDDT-derived composition is shown in on the right (dark grey bars). The horizontal grey line separates vegetation classes common to both models at the top, from those found in VDDT only, at the bottom. ***Key to bar labels:*** the first two letters are the dominant species; letters (Y,P,S) are small trees and (M,L,G) are medium, large or giant trees; the fourth letter represents open, medium or closed canopy, and the final number is the number of strata. Labelling details are in table 2. Vegetation classes <4 percent are not shown.

different from both FVS and VDDT landscapes; leading us to conclude that smoothing is not a panacea that will easily reconcile model differences.

Comparison of Native and FVS–Derived VDDT

Extending the analysis described earlier, we compared a Base Unmanaged VDDT simulation with an FVS simulation processed through Preside into matrix form and then simulated in VDDT. The results are notably different, but consistent with the results presented in table 3. Since VDDT defines a set of up to 16 post-disturbance classes that are not possible to define within FVS, some differences are to be expected. The key differences are unrelated to those novel classes, however, and the most common classes, accounting for 70 percent of the landscape in each model, are completely different. As shown in the left panel of fig. 3, FVS generally predicts a less complex landscape that consists primarily of “Large” or “Giant” multiple story stands. In contrast, VDDT predicts stands in which sapling and medium size trees are more common.

When the Managed FVS and VDDT transition matrices are compared, the same types of differences can be seen, with FVS predicting more big trees and VDDT predicting smaller trees (fig. 3, right panel). Both VDDT and FVS predict open to medium canopies, however.

The goal of the management regime was to preferentially retain open forests of large, older ponderosa pine. This was successful, and PP-dominated vegetation classes are more common in the managed landscape than in the unmanaged landscapes for both models. The FVS landscape contains more than 3 times as much ponderosa pine in the managed run compared to the unmanaged case, and represents just over half of the landscape, with the majority of the vegetation classes in large open or moderately open

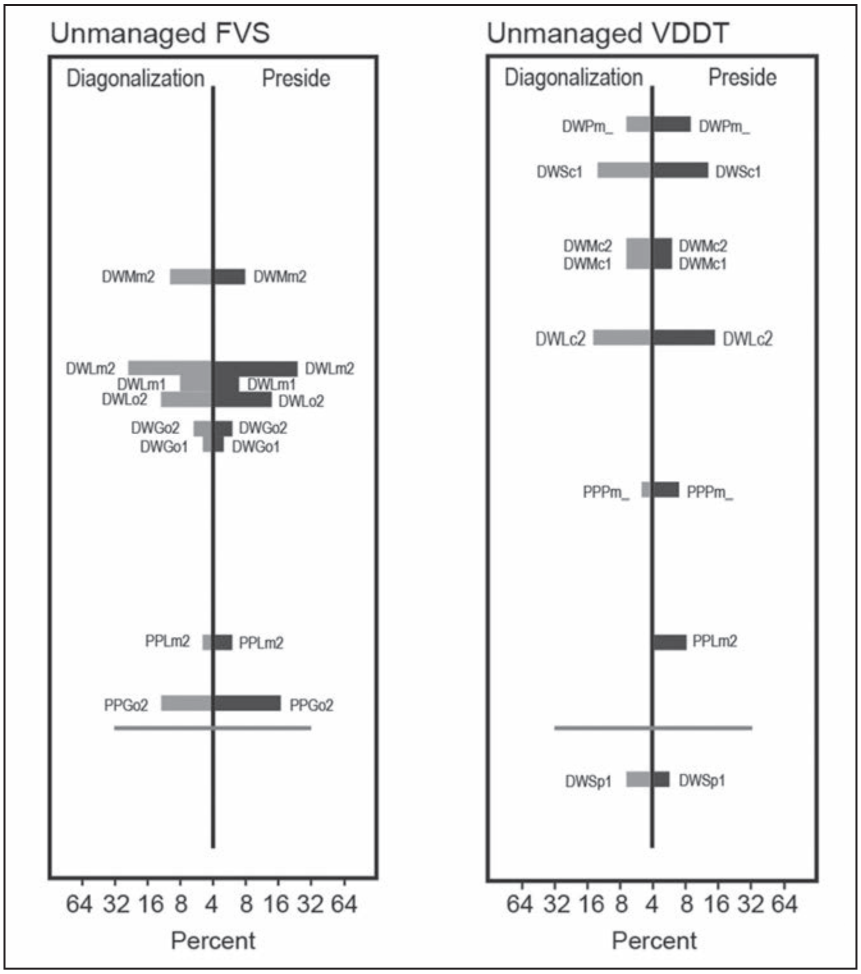

Figure 4—Decade-annual transformation differences are shown for Unmanaged FVS (left panel) and VDDT (right panel) simulations. Within each panel, horizontal bars show the landscape composition near equilibrium as calculated through matrix diagonalization on the left (light grey bars) and by Preside on the right (dark grey bars). The horizontal grey line separates vegetation classes common to both models at the top, from those found in VDDT only, at the bottom. Bar labels are defined in the preceding figure and vegetation classes <4 percent are not shown.

stands. The managed VDDT landscape also shows significantly more ponderosa pine stands. However, the managed VDDT stands are generally much younger than in either the unmanaged VDDT run or the managed FVS run.

Changes in Temporal Scale

Two methods—Preside and matrix diagonalization—exist for scaling from the decadal to annual timestep. The outcomes of these two methods are compared in figure 4, using example transition matrices for Unmanaged FVS and VDDT landscapes. The symmetry of the two examples indicates that the long-term behavior is very similar for both methods, usually with maximum differences of a few percent in vegetation categories. In the VDDT example in figure 4, there is a small excess in PP vegetation classes predicted using the Preside method, compared to the diagonalization method. In transition matrices with faster transitions or more transitions (i.e., in which the transition matrix was less sparse) the differences between the two calculation methods would probably be more significant.

Comparative Matrix Analysis of FVS and VDDT

The important results from the analysis of the equilibrium of some of the treatment matrices are summarized in table 4; with more complete results found in Appendix 2 and in supplementary material.[5] Since they show the distribution of vegetation states very near equilibrium, these results are not identical (and sometimes not even very similar) to those

[5] Supplemental information about the simulation methodology and detailed results are available at: http://essa.com/wp-content/uploads/RobinsonBeukema2012Supplement.pdf.

Table 4—The percent of the landscape in different state classes near equilibrium for Unmanaged and Managed VDDT and FVS simulations. The FVS simulations were analyzed with transitions using all years (FVS-Base), and ignoring the transition between years 0 and 1 (FVS- DZ). Within those treatments each matrix contained either all transitions (All), or the most frequent transitions (Sub) as described in the Materials section. Vegetation classes <4 percent are excluded. The transition classes shown in the two bottom rows are post-disturbance classes that have no analogue in FVS. The bottom row shows the sum within each column, and indicates the importance of rare vegetation classes in VDDT simulations

	Unmanaged						Managed					
	VDDT		FVS-Base		FVS-DZ		VDDT		FVS-Base		FVS-DZ	
	Sub	*All*	*Sub*	*All*	*Sub*	*All*	*Sub*	*All*	*Sub*	*All*	*Sub*	*All*
DWSn_							6	6				
DWPm_	7	7										
DWPo_							13	13				
DWSc1	15	15			8		4	4				
DMMo2												
DWMc2	9	9										
DWMc1	9	8										
DWLc2	19	19										
DWLc1	4	4										
DWLm2										6		5
DWLo2										6		8
DWGo2			38		39					6		6
DWGo1			30		31							
PPSn_							9	9				
PPPo_							28	28				
PPSm2							4	4				
PPSm1			9				6	6				
PPSo1							6	6				
PPMm2							5	5				
PPMo1									5			
PPLm2										13		13
PPLm1										11		11
PPGo2			23	93	18	93			76	35	80	36
PPGo1				7	4	7			19	9	20	9
DWPp1		4										
DWSp1	10	10										
Total	73	76	100	100	100	100	81	81	100	86	100	88

shown in figure 1 or figure 3 or figure 4, all of which show the distribution of vegetation classes farther from equilibrium.

The Base and Drop Zero treatments in table 4 are very nearly identical for both the managed and unmanaged FVS scenarios, regardless of whether all transitions (All) or a subset of transitions (Sub) are used. Data Simplification treatments that include rare transitions take at least a century or more to approach equilibrium (table 5).

A set of simplified complementary views of the equilibrium results is found in table 6, which regroups the data of table 4 into three groups of two strata each. Viewed through the perspective of these coarser strata, the overall disparity between the VDDT and FVS simulations is striking. When using the All-transitions treatment, VDDT is primarily small to medium Douglas-fir with medium to closed canopy at equilibrium, while FVS is primarily large, open ponderosa pine. When rare transitions are excluded,

Table 5—The median number of years required to reach near-equilibrium for the Unmanaged and Managed VDDT and FVS simulations. The FVS simulations were analyzed using All timesteps (Base) and excluding the first timestep (Drop Zero). These treatments were further divided to use all observed transitions (All) and to exclude rare transitions (Sub)

		Unmanaged		Managed	
		VDDT	**FVS**	**VDDT**	**FVS**
Base	Sub	223	38	142	138
	All	209	208	140	368
Drop Zero	Sub		165		156
	All		209		342

the species mix is more similar, but not the size or degree of canopy closure. Dropping the first transition makes little difference to the results.

Comparing simulations made using all transition probabilities and those with fewer transitions gives a fairly consistent result for timing and complexity: simpler stochastic matrices generally reach a near-equilibrium state two to three times faster than their more complex counterparts, and also tend to reach a simpler end-state with a small number of final vegetation classes. That said, the near-equilibrium (climax vegetation) end states can be remarkably different: PP in some cases and a mix of DW and PP in others. As a result of this, the attempt to simplify transitions only adds to our woes and makes it seem unlikely that FVS and VDDT simulations can be easily reconciled.

Summary and Recommendations

Although we have not been able to develop ways to easily bridge FVS and VDDT, we have learned some useful lessons. We have independently confirmed that Preside's methodology for calculating transitions and their probabilities is correct, and that identical results are obtained in VDDT when using VDDT-defined pathways and probabilities directly, and when VDDT results are output as FVS-like results which are then imported back into VDDT using Preside-calculated pathways and probabilities. This gives us confidence that if the issues caused by classification boundaries and model behavior can be resolved, accurate FVS-generated information can be provided to VDDT. Although we found that the algorithm for converting FVS decadal transition probabilities into annual VDDT probabilities is not correct, the consequences of this error are small when the transition matrix is sparse and probabilities are small. Correcting this error is conceptually straightforward.

Even after taking care to calibrate FVS and to make the FVS and VDDT simulations as similar as possible, we found two notable sources of discordance. First, the FVS simulation contains complex regeneration rules which have a significant influence on the types of transitions that emerge from the model. And for its part, VDDT also contains assumptions about regeneration that are implicit in the user-defined pathways (e.g., when moving from a

Table 6—Simplified near-equilibrium states for Unmanaged and Managed VDDT and FVS simulations, showing the percentage in different groupings by dominant species, QMD class and Cover class. Some columns do not add to 100 percent due to rounding

		Unmanaged						Managed					
		VDDT		**FVS**				**VDDT**		**FVS**			
		Base		**Base**		**DZ**		**Base**		**Base**		**DZ**	
		Sub	**All**	**Sub**	**All**	**Sub**	**All**	**Sub**	**All**	**Sub**	**All**	**Sub**	**All**
Species	PP	2	3	32	100	23	100	66	63	100	71	100	72
	DW	88	88	68		78		29	29		24		23
QMD	P,S,M	63	64	9		9		95	92	5	1	2	
	L,G	27	27	91	100	92	100			95	94	98	95
Cover	n,p,o,_	23	24	91	100	100	100	68	68	100	61	100	64
	m,c	67	67	9		1		24	24		34		31

single-story to a multistory stand), but which are nonetheless obscure. These different regeneration assumptions work themselves out in different ways and result in different stand structures that cannot easily be reconciled. Improvements could possibly be made in the VDDT regeneration arena by more careful and explicit attention to regeneration and its temporal and structural consequences, which might require additional detail in the state models. FVS would benefit from a thorough review of the Repute model which was used to drive regeneration in the FVS simulation. Secondly, VDDT contains explicit rules about low levels of natural disturbances (insects, pathogens and low-level fire) which cumulatively keep the landscape from reaching an old giant forest state. FVS has a set of assumptions about the role of natural disturbance which are embedded in its mortality model, but these rules generally have less impact on the size or structure of the forest.

Because of the underlying differences in the biological processes that are included, classifying FVS outputs into discrete VDDT-like vegetation states will always present difficulties. Simple approaches such as smoothing or moving a class boundary such as QMD, simply transfers the classification problem to another type of stand structure. Because of the way they are defined, the algorithms which define QMD and Cover remain sensitive to the appearance of any new regeneration, and the jitter issue therefore remains unresolved as long as standard definitions and categorical variables are used to stratify stand structure. There are other intrinsic difficulties for comparing the two models since in this landscape at least, VDDT contains a number of vegetation states that are undefined and therefore never observable in FVS. In the Managed case using all transitions, about 20 percent of the VDDT landscape is in vegetation classes that are out-of-scope for FVS.

Given the initial jump in stand vegetation class often produced by FVS self-calibration, combined with the censoring that takes place between the initial inventory condition and the end of the first timestep, the Drop Zero treatment should produce more reliable results than the control Base treatment. It is therefore surprising to note that the dynamics of these two treatments are very similar (table 6). Regardless, removal of the first transition is recommended to reduce initialization artifacts.

Finally, we explored a further layer of data treatments through scenarios which either retained or removed rare transitions. We discovered that the removal of the less frequent transitions results in stochastic matrices with overly simplistic pathways, along with the side effect that the landscape dynamics are sped up, often dramatically, in their approach to equilibrium. This speed seems unreasonable in forests marked by century-scale life spans, and suggests that rare transitions should always be retained.

We also examined smoothing methods that might result in less jittery FVS transition dynamics which we hoped would be more transferrable to VDDT. This exploration was neither simple nor uniformly successful. The methods we tested often resulted in system dynamics that were quite different from the comparatively similar Base and Drop Zero treatments described above. We were unable to find similarities between any smoothed scenario and the VDDT landscape dynamics that were consistent across the unmanaged and managed landscapes. A Preside option that allowed the calculation smoothed classes would provide a way to examine the benefit of reducing jumpy transitions, leaving it up to the modeler to decide whether this created better agreement between the two models.

Because of the differences in intent and design, no one model can capture all aspects of stand or landscape dynamics. Our findings show that FVS can be used to capture changes due to stand processes: tree growth, mortality, and regeneration and the associated changes in size and structure, and that these transitions can be transferred to VDDT. Modelers should consider these transitions as candidate versions of reality, just as VDDT transitions are different candidate versions of reality. Given the many possible differences between the two models, it might be helpful to develop automated methods of comparing large matrices which highlight transition probabilities that differ between the two models. Tools that allowed modelers to quickly simulate stable states might also provide insight into model differences that would highlight, for example, the role of regeneration and disturbance in creating and maintaining different pathways. When doing a landscape level analysis using VDDT however, including FVS transitions by themselves are not sufficient, and it remains necessary to add the important effects of landscape level disturbances such as beetles and fire. In this way, one is using the strengths in both models to create the best possible VDDT model.

Given the challenge of reconciling the two model paradigms, we have proposed and examined some possible approaches to aide in model comparison. Using the common framework of the transition matrix, its dynamics and stable states are a useful way of comparing two complex models, producing compact summaries—two vectors—which allow the quick identification of similarities and differences. Identifying these contrasts is the first step toward understanding the underlying mechanisms which create different projections. Although it is possible to compare the two transition matrices directly, landscape dynamics are based on the aggregate behavior of the entire system, making it misleading to consider the automated insertion of specific transitions from one model into another without careful examination of the consequences.

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References

Crookston, N.L.; Dixon, G.E. 2005. The Forest Vegetation Simulator: a review of its structure, content, and applications. Computers and Electronics in Agriculture. 49: 60–80.

ESSA Technologies Ltd. 2007. Vegetation Dynamics Development Tool User Guide, Version 6.0. Prepared by ESSA Technologies Ltd., Vancouver, BC. 196 p.

Forest Vegetation Simulator Staff. 2003. Users guide to the Database Extension of the Forest Vegetation Simulator Version 2.0. Fort Collins, CO. U.S. Department of Agriculture, Forest Service, Forest Management Service Center. (July 2009 revision). 60 p.

Moeur, M.; Vandendriesche, D.A. 2009. Calibration of state and transition models with FVS. In: T. Jain, R.T. Graham and J. Sandquist, tech. eds. 2010. Integrated management of carbon sequestration and biomass utilization opportunities in a changing climate: Proceedings of the 2009 National Silviculture Workshop; 2009 June 15–18; Boise, ID. Proceedings RMRS-P-61. Fort Collins, CO: U.S. Department of Agriculture, Forest Service, Rocky Mountain Research Station: 275–287.

Stage, A.R. 1973. Prognosis model for stand development. Res. Pap. INT-137. Ogden, UT: U.S. Department of Agriculture, Forest Service, Intermountain Forest and Range Experiment Station. 36 p.

Stage, A.R. 1997. Using FVS to provide structural class attributes to a forest succession model (CRBSUM). In: R. Teck, M. Moeur and J. Adams (comps.). Proceedings: Forest Vegetation Simulator Conference; February 3–7; 1997. Fort Collins, CO. Gen. Tech. Rep. INT-GTR-373. Ogden, UT: U.S. Department of Agriculture, Forest Service, Intermountain Research Station. 139—147 p.

Vandendriesche, D. 1997. FVSSTAND Alone: generate dynamic yield reports from the Forest Vegetation Simulator. In: R. Teck, M. Moeur, and J. Adams (comps.). Proceedings: Forest Vegetation Simulator Conference; February 3–7; 1997. Fort Collins, CO. Gen. Tech. Rep. INT-GTR-373. Ogden, UT: U.S. Department of Agriculture, Forest Service, Intermountain Research Station. 168–177.

Vandendriesche, D.A. 2009. Repute–regeneration imputation extractor. http://www.fs.fed.us/fmsc/ftp/fvs/software/pre_post/repute_z.exe. (6 August 2009).

Van Dyck, M.G.; Smith-Mateja, E.E. (compilers) 2000. Keyword reference guide for the Forest Vegetation Simulator. Revision: March 2012. Fort Collins, CO: U.S. Department of Agriculture, Forest Service, Forest Management Service Center. 119 p.

Appendix 1: Calibration of FVS

This Appendix contains the details of the parameters that were changed or added to calibrate FVS for these simulations.

As described in the body of the paper, site quality was constrained to a maximum SDI of 380 and maximum basal area of 164 ft^2 ac^{-1}. As part of the FVS calibration process, the large tree diameter increment model and small tree height increment model were also adjusted (table 7 and 8) to globally reflect pooled diameter and height calibration data from the full set of stands. Mortality was also adjusted using the TreeSzCp keyword (Van Dyck and Smith-Mateja, 2000) to reduce survival of larger trees.

Additional keywords were needed to implement the restoration management scenario (fig. 5). These keywords instruct FVS to thin the stand to 50 percent of SDI when the stand is more than 65 percent of maximum SDI, and to preferentially retain larger ponderosa pine.

Table 7—Parameters of the ReadCord and ReadCorH keywords used to adjust baseline diameter and height growth. Empty fields have coefficients of 1.0

Species	Diameter	Height
WP		
WL	0.799	
DF	1.056	2.730
SF		
RC		
GF	0.575	
LP	0.861	0.461
ES		
AF	0.637	
PP	0.824	
OT		

Table 8—Parameters of the TreeSzCp keyword used to adjust survival and dimension constraints for larger trees

Species	DB (in)	Mortality ($10y^{-1}$)	Height (ft)
WP	28	0.374	120
WL	26	0.118	120
DF	32	0.103	125
SF	12	0.129	70
RC	26	0.159	95
GF	22	0.291	120
LP	18	0.259	95
ES	16	0.223	100
AF	12	0.243	70
PP	34	0.135	120
OT	10	0.464	70

```
SpecPref                          WP        10
SpecPref                          WL        10
SpecPref                          DF        10
SpecPref                          SF        10
SpecPref                          RC        10
SpecPref                          GF        10
SpecPref                          LP        10
SpecPref                          ES        10
SpecPref                          AF        10
SpecPref                          OT        10
If              0
BSDI GT (BSDIMAX*0.65)
Then
ThinSDI
Endif           0 Parms ( (BSDIMAX8.500, 1, 0, 0, 999,1)
```

Figure 5—FVS keywords used to implement a simple Restoration Management regime that retains large ponderosa pine.

Appendix 2: Impacts of Smoothing

As described in earlier sections, the use of boundaries to classify FVS stands into VDDT classes can create cases of jitter: rapid sequences of change between different classes. We experimented with two different methods for smoothing jittery transitions. The first method—*Running Smooth*—uses an evenly weighted running smoother (*t–1, t, t+1*) to average the CANCOV and QMD variables that are

Table 9—The near-equilibrium states of the Unmanaged and Managed FVS simulations, expressed as a percentage of the landscape in each vegetation class using the two smoothing methods. FVS Base results are provided from table 4 for comparison. "All" and "Sub" indicate analyses that include all transition probabilities or exclude rare transitions, as described in table 1. Minor vegetation classes (<4 percent) have been excluded. No Managed Conditional smoothing simulations were carried out

	Base		Running		Conditional	
	All	Sub	All	Sub	All	Sub
Unmanaged						
DWSm2				5		
DWMm2				14		
DWMo2						
DWLm2				34		
DWLm1				11		
DWLo2				13		
DWGo2	38					
DWGo1	30					
PPSm1	9			100	16	
PPMo1						
PPLm2				6		
PPLm1						
PPLo2						
PPGo2	23	93			84	93
PPGo1		7				7
Total percent	100	100	100	83	100	100
Managed						
DWSm2						
DWMm2			4	5		
DWMo2						
DWLm2		6	17	15		
DWLm1			8	7		
DWLo2		6	12	10		
DWGo2		6				
DWGo1						
PPSm1						
PPMo1	5					
PPLm2		13	20	20		
PPLm1		11	16	16		
PPLo2			6	5		
PPGo2	76	35				
PPGo1	19	9				
Total percent	100	86	83	78	–	–

used to assign vegetation classes to the output of each FVS timestep. The second method—*Conditional Smooth*—uses a simple rule which is applied at each timestep *t*. The rule is: if a stand's change in QMD or CANCOV in timestep *t* causes it to be assigned to a class that differs from the one to which it was assigned in timestep *t-1*, or would be assigned in the timestep *t+1*, then the classification boundary must be exceeded by at least 1/6 of the class width for the new classification to be accepted. Thus, patterns such as A-B-A would likely become A-A-A, but patterns of A-B-B would likely stay A-B-B. We examined the impact of the application of these two kinds of smoothing methods to the behavior of the resulting transition matrices.

The results of applying these smoothing methods are shown in table 9 (the distribution of vegetation states near equilibrium), table 10 (a measure of the time required to approach equilibrium) and table 11 (a condensed comparison of the vegetation states). The Base FVS scenario is included with each table to facilitate comparison with a non-smoothed simulation. The general conclusion from these simulations and comparisons is that the smoothing methods have significant effects upon the behavior of the transition matrices, but that they are as different from the Base FVS simulation as they are from any of the VDDT-based simulations.

Table 10—The median number of years required to reach near-equilibrium for the Unmanaged and Managed FVS simulations for different smoothing methods. FVS simulations were analyzed using all timesteps (Base), subdividing those results to use all observed transitions (All) and to exclude rare transitions (Sub)

		Unmanaged	Managed
Base	Sub	38	138
	All	208	368
Running	Sub	155	183
	All	247	178
Conditional	Sub	223	–
	All	211	–

Table 11—Simplified near-equilibrium states for Unmanaged and Managed FVS simulations for different smoothing methods, using all transitions (All) and excluding rare transitions (Sub). FVS Base results are provided from table 4 for comparison. Vegetation classes are grouped as in table 4, showing the percent-age in different groupings by dominant species, QMD class and Cover class. Some columns do not add to 100 percent due to rounding; those marked with '–' were not simulated

		Unmanaged FVS						Managed FVS					
		Base		Running		Conditional		Base		Running		Conditional	
		Sub	All	Sub	All	Sub	All	Sub	All	Sub	All	Sub	All
Species	PP	32	100	100	10	36	100	100	71	51	51	–	–
	DW	68			86	64			24	47	45	–	–
QMD	P,S,M	9		100	22	4		5	1	7	12	–	–
	L,G	91	100		74	96	100	95	94	91	84	–	–
Cover	n,p,o,_	91	100		21	96	100	100	61	30	26	–	–
	m,c	9		100	75	4			34	68	70	–	–

Use of the Forest Vegetation Simulator to Quantify Disturbance Activities in State and Transition Models

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Abstract

The Forest Vegetation Simulator (FVS) has been used to provide rates of natural growth transitions under endemic conditions for use in State and Transition Models (STMs). This process has previously been presented. This paper expands on that work by citing the methods used to capture resultant vegetation states following disturbance activities; be it of natural causes or human induced. Forest Inventory and Analysis (FIA) plots were stratified by representative Potential Natural Vegetation Types (PNVTs) of the southwestern United States. Structural states within the Vegetation Dynamics Development Tool (VDDT) (i.e. STM used for forest planning) were defined based on dominant tree size, canopy cover density, and vertical story layering. A standard set of silvicultural and fire prescriptions were evaluated using FVS projections. Outputs such as post treatment vegetation states, harvest volumes, and snag generation were captured and linked to transitions in VDDT. A case study involving the ponderosa pine/bunchgrass forest ecosystem will be presented to demonstrate features of the modeling approach.

Introduction

State-and-transition models (STM) have the ability to integrate multiple interactions between ecological processes and land management strategies and as such are increasingly being used to guide land management decision making. However, owing to their complexity, some state characteristics and transition probabilities are largely constructed from expert opinion because of a lack of empirical data or lack of tools to test STM performance against empirical data. Recently, the Forest Vegetation Simulator (FVS) (Dixon 2002) has been used to provide rates of natural growth transitions under endemic mortality conditions for use in Vegetation Dynamics Development Tool (VDDT) (ESSA 2006) models (Henderson 2008, Moeur and Vandendriesche 2010, Weisz et al. 2010, Shlisky and Vandendriesche 2012). In our previous work in analyzing landscapes for forest plan revision, we divided the southwestern United States into terrestrial ecosystems that range from dry grasslands-shrublands, to semi-arid woodlands, to moist forestlands. Each ecosystem is representative of a Potential Natural Vegetation Type (PNVT) (Schussman and Smith 2006). Each PNVT, which is depicted within separate VDDT models, was then further broken into vegetation states. A vegetation state is a composite of cover type (preeminent species composition) and stand structure (dominant tree size, canopy cover density, and vertical canopy layering). The vegetation states developed for the ponderosa pine/bunchgrass (PPG) ecosystem are illustrated in table 1.

Vegetation states can transition to other states in the absence of disturbance due to natural processes. Tree establishment, growth, and mortality comprise the main components for natural succession. For example, the straight green lines connecting model states in figure 1 represent deterministic pathways (i.e., natural transitions in the absence of disturbance).

Change in vegetation states can also result from management activities, insect and disease outbreaks, and wildfire occurrence. The blue lines in figure 1 represent the myriad of stochastic pathways (i.e., probabilistic transitions due to disturbance events). State-and-transition models (STMs) such as VDDT can be used to evaluate natural succession and disturbances effects that result in vegetation change on the landscape (He 2008). The objective of this

Table 1—Stratification of figure 1 ponderosa pine/bunchgrass PNVT vegetation states A through N, according to key attributes of dominant tree size, canopy cover, and canopy layering

	Tree diameter					
GFB	0-5 in.	5-10 in.	10-20 in.	20+ in.	Canopy[a] cover	Canopy layering
A or N[b]	B	C	D	E	Open	Single
	F	G	H	I	Closed	Single
			J[c]	K[c]	Open	Multi
			L	M	Closed	Multi

[a] Except for States A and N, "Open" states have 10 to 30 percent canopy cover and "Closed" states have greater than 30 percent canopy cover. States A and N have less than 10 percent canopy cover.
[b] States A and N are grass, forbs, brush, and shrub states (GFB). State A is the characteristic state which existed in reference conditions. State N is the uncharacteristic state resulting when stand-replacing fires occur in closed canopy states. (Smith 2006)
[c] The *desired condition* is an open multi-layered (> 5 age classes) state with average diameter varying by site productivity with State J occurring on low productive sites and State K occurring on high productivity sites. (Triepke et al. 2011)

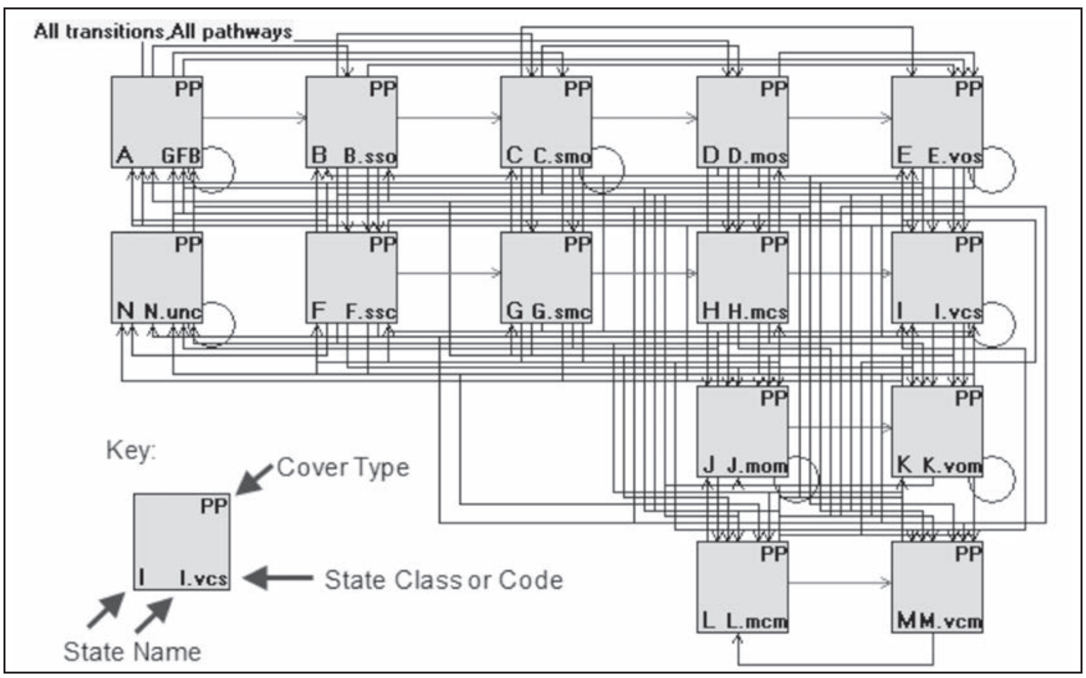

Figure 1—Conceptual pathways diagram for the ponderosa pine/bunchgrass PNVT. Boxes represent model states and arrows represent transitions due to natural growth and other natural and anthropogenic transitions such as management activities, fires, and insects and disease. In the key to the conceptual diagram we see the Cover Type (Ponderosa Pine = PP), State Name (1) and the State Class or Code (VCS = Very-large, Closed, Single story).

paper is to illustrate the expanded use of FVS for estimating outcomes from disturbance agents within the context of national forest land management planning. The general process that we used for FVS analysis is described in the sidebar. The red line between step 4 and 5 demarks the break between our initial effort to capture natural growth succession and our subsequent work to quantify resultant vegetation states due to disturbance activities. The later steps will be elaborated in the following sections.

Sidebar

The conceptual method for estimating transition rates and destination states for STMs involved using inventory data, FVS model runs, and post processing software. The analysis process culminated with formal evaluation and adjustment of model results where needed. The steps utilized were:

Step 1. Assemble inventory data for FVS projections. Regional strategic inventories such as FIA provide an excellent data source to represent vegetation stratification schemes. Classification attributes can be used to filter plot sets for finer resolution. In the absence of adequate sample, FVS can be used to grow plots into model states.

Step 2. Adjust FVS parameters to the current inventory trends (Vandendriesche 2010). The default model context for FVS is to forecast stand development to full site occupancy. If endemic or epidemic conditions are to be portrayed, the FVS model needs to be adjusted toward those ends.

Step 3. Develop natural growth projections to estimate parameters for deterministic pathways without disturbance. The goal here is to capture ecological processes that represent stand development over at least one life cycle.

Step 4. Process successional progressions through the Preside program (Vandendriesche 2009). Vegetation classification attributes are compiled into a report from which mean residence times within states and transition probabilities between states are computed.

Step 5. Develop management activity projections to estimate parameters for stochastic pathways. Construction of a treatment matrix is beneficial in assigning silvicultural prescriptions and fire activities to each model state.

Step 6. Process probabilistic transitions through the Preside program. Resident and resultant vegetation states for pre and post treatment/activity are captured. Average stand conditions and reporting attributes are summarized.

Step 7. Review model results and make adjustments in relation to conceptual expectations. When necessary, account for knowledge obtained from ancillary literature and professional judgment.

Methods

Coarse and Fine Filter Plot Datasets

During our initial phase for forest planning, “coarse filter” field data plots consisted of all Forest Inventory and Analysis (FIA) plots meeting habitat type (USDA FS 1997) specifications for each PNVT[1]. Table 2 provides a listing of the habitat types associated with the PPG PNVT. For our current work, we began by using the coarse-filter plot set. However, during the process, and under the scrutiny of silviculturists and fire ecologists, questions arose as to whether the coarse filter plots were homogeneous enough to evaluate the effects of management activities in the VDDT models. Some of the arguments against using coarse filter plots exclusively were that they contained:

- plots that were designated with incorrect habitat types,
- plots that represented inclusions or ecotones, within or between PNVTs, and
- plots that included non-stockable areas (i.e., rock outcrops, perennial water).

These contentions indicted that coarse filtered plots would add unrealistic effects to our results because they didn’t represent the type of vegetation that typically would be affected by our treatment activities. Classifying forestland with respect to its capacity for timber production has been a requirement of the National Forest Management Act and subsequent implementation regulations since the early 1980’s (Youtz 2006). Determining timber suitability is a stepwise process that separates National Forest System lands into various classification categories. (Refer to figure 2.) Separating our data set into proxies of land generally not suitable and land generally suitable for timber production or harvest is analogous to applying finer resolution to the coarse filter data set.

We therefore created fine filters to remove the non-representative plots from the coarse filtered data set using rules determined by our silvicultural and fire ecology specialists. As an example, by using the PNVT—habitat type crosswalk

[1] The terms “habitat type” and “plant association” are synonymous in the southwestern region. A PNVT is composed of several habitat types.

Table 2—Habitat type codes associated to the ponderosa pine/bunchgrass PNVT

Habitat type code	Common name
011092	Ponderosa pine/Arizona fescue/blue gramma
011093	Ponderosa pine/Arizona fescue/Gambel oak
011330	Ponderosa pine/mountain muhly
011340	Ponderosa pine/screwleaf muhly
011341	Ponderosa pine/screwleaf muhly/Gambel oak
011350	Ponderosa pine/Indian ricegrass
011380	Ponderosa pine/black sagebrush
011390	Ponderosa pine/screwleaf muhly-Arizona fescue
011391	Ponderosa pine/screwleaf muhly-Arizona fescue/blue gramma
011392	Ponderosa pine/screwleaf muhly-Arizona fescue/Gambel Oak
011400	Ponderosa pine/kinnikinnik
011470	Ponderosa pine/Arizona walnut

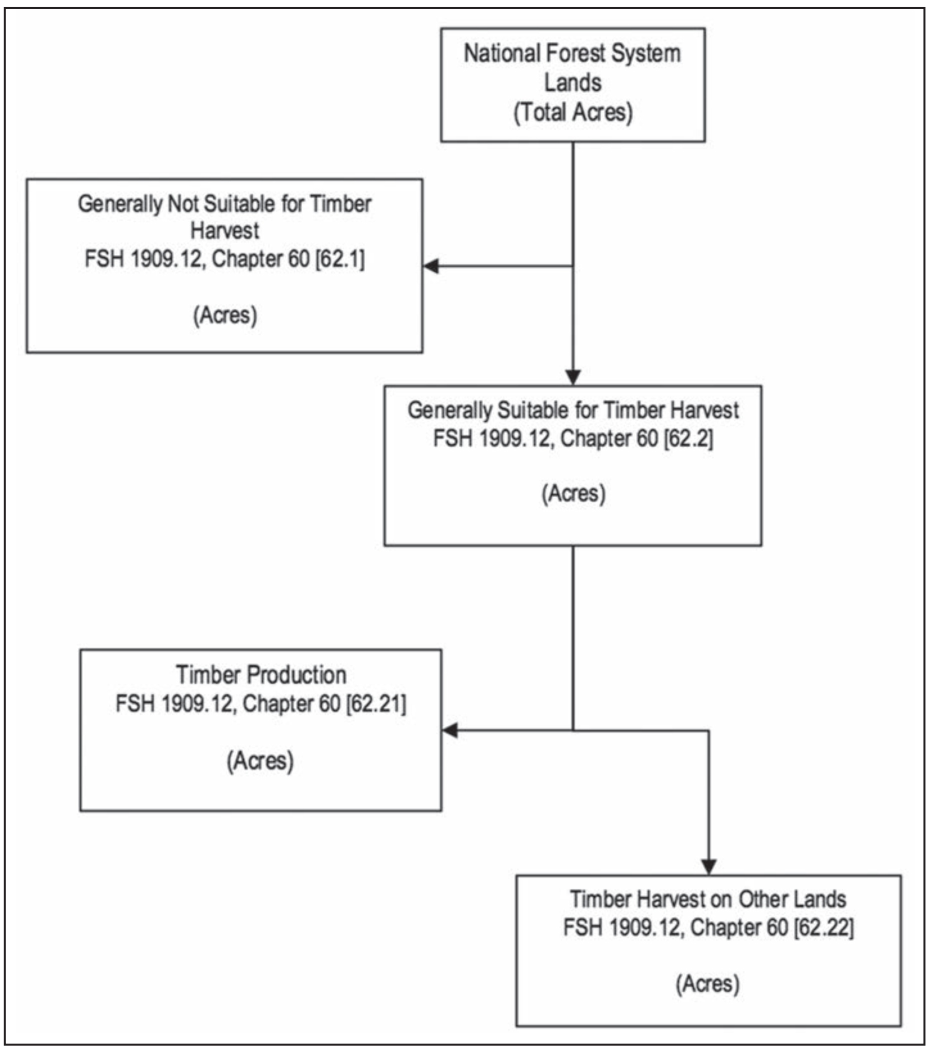

Figure 2—Timber suitability screening process.

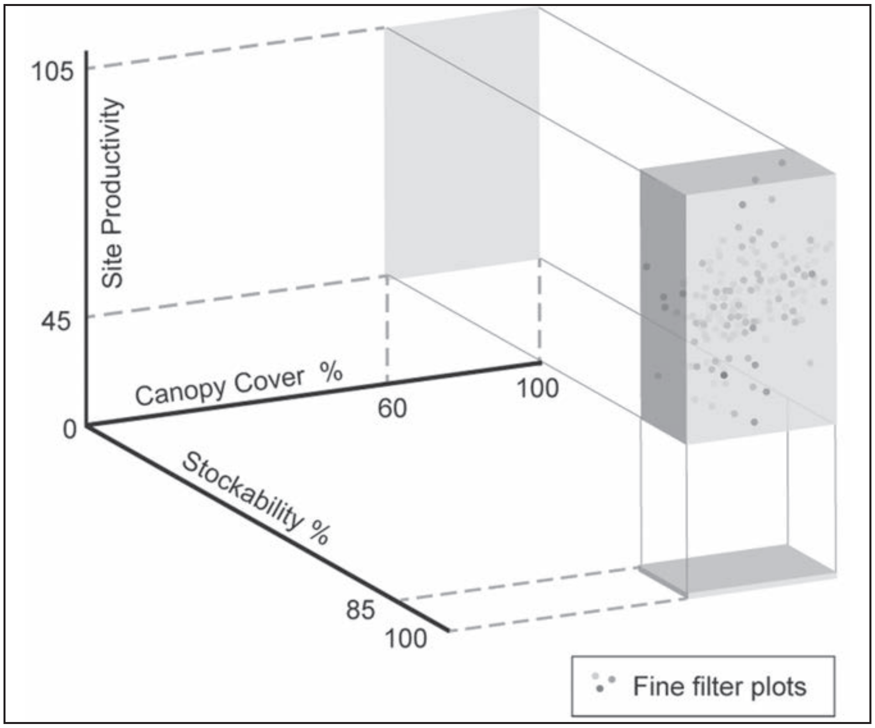

Figure 3—Use of expert opinion rules to define fine filter plots sets.

as defined for coarse filter plots, we identified 477 FIA plots associated with the ponderosa pine/bunchgrass forest ecosystem. The following fine filters were applied:

1. Must have 60 at least percent canopy cover in ponderosa pine trees
2. No canopy cover restrictions on:
   a. junipers
   b. pinyons
   c. bristecone/limber pine
   d. aspen
3. Must have less than 5 percent canopy cover for all other tree species except:
   a. Must have 0 percent canopy cover for corkbark fir, Engelmann spruce, blue spruce
   b. Must have 0 percent canopy cover narrowleaf cottonwood, other hardwoods
   c. Must have less than 2 percent canopy cover Chihuahua pine, other softwoods
4. Stockability: 85 percent or greater
5. Site Productivity: Ponderosa Pine Site Index within the range of 45 to 105.

This resulted in a subset of 283 plots meeting the fine filter criteria (approximately 60 percent of coarse file plot set). These were used in FVS projections to evaluate the effects of management activities. Figure 3 characterizes the winnowing of fine filtered plots from the coarse filtered data set.

Supplementing Plot Sample Sizes With Synthetic Datasets

When processing FIA plots to compute transition probabilities for STMs, on occasion there were either too few plots (i.e. less than 10) or no plots at all to represent particular model states. In these cases, confidence in the results from FVS simulations was relatively low. Having a small sample within a vegetation state increased the likelihood that a borderline plot (i.e. plots at the edge of the shaded box in fig. 3) would have undue effects on the resultant transition probabilities. For example, if one plot (of three) responded to treatment by transitioning to an unlikely vegetation state, the relative impact would suggest that 33 percent of representative acres within the state would transition to this suspect destination state. With a larger sample size, the

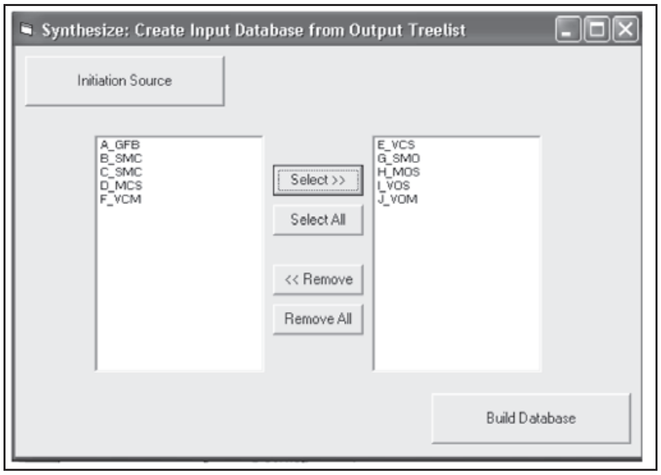

Figure 4—The Synthesize post processing program can be used to select deficient vegetation states to increase sample representation.

transition probabilities would be buffered against unlikely end states. Occasionally, this problem was encountered with coarse filtered plot sets; however, it was more likely to occur with the reduced subset of fine filtered plots.

We set the minimum sample size for a vegetation state to greater than or equal to 10 plots. If this criterion was not met, we created synthetic plots and added those to the original inventory sample. Having 30 or more representative plots in a vegetation state is desirable. An FVS post processing program (named Synthesize[2]) was developed to create modeled plot sets. The process involves first using FVS to grow existing inventory plots into vegetation states with a deficiency of existing plot data. Next, the Synthesize program reads the output tree list files created by FVS and extracts the plot and tree information of specified vegetation states (fig. 4). This extracted information is then converted into input plot and tree records in database files. These databases are used as the plot data input source for deficient vegetation states. With respect to the PPG PNVT, there were no vegetation states that lacked an adequate inventory sample. There was a need to augment the inventory sample with synthetic plots in other PNVTs in the southwestern region.

Validation of Disturbance Transitions

Two types of disturbance were considered during this phase of modeling analysis: human induced and naturally caused. Silvicultural treatments in the form of tree removal were the primary source of human intervention. Depending on the ignition source and outcome, fire can be considered either or both types of disturbance activity. Planned fires (i.e. prescribed) served as a secondary source of human induced disturbance. Unplanned fires (i.e. wildfires) for our purposes were considered the primary natural disturbance agent. Inferences regarding insects, disease, wind, and other sources of natural disturbances outside of pervasive impacts were not modeled independently due to time constraints.

The Forest Vegetation Simulator and Fire and Fuels Extension (FVS-FFE; Rebain 2010) were used to simulate the effects of using tree cutting, and planned and unplanned fires as restoration tools for the various PNVTs of the Southwest. A standard set of management activities were applied to each model state. Species composition and structure (i.e. vegetation state) were compared pre and post treatment. The resultant conditions were then used to assign the transition pathways for each disturbance type in VDDT.

Land managers should consider a full “toolbox” of treatment methods in order to work toward achieving desired conditions. However, in an effort to reduce the modeling workload, eight silvicultural prescriptions and three burning intensities were evaluated (table 3).

Silvicultural prescriptions—
Cutting parameters were adjusted in the FVS model runs to fit each PNVT and the intent of the particular treatment method. Knowledge of species silvics and ecological objectives were considered and adjusted for each PNVT as appropriate. Residual target basal areas (BAs) were assigned according to cutting prescriptions as shown in table 4 for PPG.

Tree species cut/leave preferences were set within the FVS model to fit the native species mix and emphasized

[2] Synthesize, Version 2011.01, programmed by D. Vandendriesche while working with USDA Forest Service, Forest Management Service Center, Forest Vegetation Simulator Staff, Fort Collins, CO. Inquiries should be directed to the developer.

Table 3—Standard treatments used to evaluate the effects of management activities within each VDDT Model State

Management activity	Code
Natural endemic growth in the absence of disturbance[a]	A
Free thin, all sizes to target basal area	B
Thin-from-below to target basal area	C
Thin under a 16” diameter cap to target basal area	D
Group selection with matrix thin to target basal area	E
Shelterwood seed cut to target basal area	F
Clearcut with non-regeneration objective legacy trees	G
Clearcut/coppice for hardwood regeneration	H
Prescribed fire, low intensity burning conditions	J
Prescribed fire, moderate intensity burning conditions	K
Prescribed fire, high intensity burning conditions conditions	L
Thin under a 9 inch diameter cap	M

[a] Specifying a “No Treatment” activity implies by default a management decision.

Table 4—Prescribed residual basal area targets by cutting method for the ponderosa pine/bunchgrass PNVT

Cutting method	PPG
B = Free Thin All Sizes to Target BA	50 BA
C = Thin From Below to Target BA	70 BA
D = Thin Under 16” Diam. Cap to Target BA	60 BA
E = Group Selection, with Matrix Thin to Target BA	60 BA
F = Shelterwood Seed Cut to Target BA	30 BA
G = Clearcut with Non-regen. obj. Legacy Trees	na
H = Clearcut /Coppice with Non-regen. Legacy Trees	na
M = Thin Under 9” Diam. Cap for MSO Recovery	na

restoring each PNVT to reference conditions.[3] Where certain tree species, such as aspen and oaks, were desired to retain they were either favored in the species preference settings or as a percentage of the total target residual BA. Trees with recorded dwarf mistletoe infection were selected for removal in silvicultural treatments aimed at improving overall stand vigor and health.

Cutting treatments were to be followed by a cool prescribed burn as part of the same restoration entry. These follow-up burns were not modeled in FVS-FFE. The thought being that these surface fires would treat the understory ground/shrub level and not have an impact on the resultant vegetation state. Prescribed fire treatments were modeled in FVS-FFE as the sole method of thinning trees in areas where tree cutting is not possible.

Fire modeling parameters—

Fire behavior is a combination of fuels, weather, and topography. The FVS-FFE model accepts fuel and weather parameters that mimic environmental conditions at the time of an ignition. The resulting fire behavior, such as type of fire (surface, passive, or crown fire), flame length, and torching and crowning index are then estimated by FVS-FFE. These fire behavior parameters were applied to our model states and FVS-FFE then estimated mortality and survival of the vegetation by species and size. One fire disturbance was applied at the beginning of the growth cycle with each fire modeled at low, moderate, and high conditions of weather and fuel moisture.

Environmental conditions used to simulate the low, moderate, and high fire conditions are based on historic weather data from the Alpine, Arizona, Remote Automated Weather Station (RAWS). The Alpine RAWS has the most complete and accurate data of all the weather stations on the Apache-Sitgreaves National Forests, and was used for FVS-FFE modeling of the ponderosa pine/bunchgrass PNVT.

Weather data were sorted using FireFamilyPlus[4] (v4.1) to produce a Percentile Weather Report. This percentile report was used to determine the 15th, 75th, and 90th percentile weather for the past twenty years (1990–2009). Weather data used were from the period of April 1 through October 15 each year, representing a typical fire season. The 15th percentile represents natural fire season conditions for a low intensity fire; the 75th is moderate conditions; and, the 90th is high intensity fire conditions (table 5).

[3] The concept of ‘desired conditions’ and ‘forest restoration’ strategies for the ponderosa pine and dry mixed conifer PNVTs of the southwestern region is documented in Triepke et al. 2011.

[4] USDA Forest Service, Rocky Mountain Research Station, Missoula, MT. Software and User’s Guide available online at: http://www.firemodels.org/index.php/firefamilyplus-software/firefamilyplus-downloads.

Table 5—Percentile Weather Report derived from FireFamilyPlus program

Station: 020401: ALPINE, AZ Variable: ERC Model: 7G2PE2
Data Years: 1990–2009 Date Range: April 1–October 15 Wind Directions: S, SW, W

Variable/Component Range	Low	Moderate	High
Percentile, Probabilities and Mid-Points			
Percentile Range	0-15	16-89	90-97
Climatological Probability	15	75	7
Mid-Point ERC	15-15	48-48	90-90
Number Observations	61	82	61
Calculated Spread Component	4	10	16
Calculated ERC	16	49	91
Percent Area Burned	60	70	80
Fuel Moistures/Weather			
Variable	**Low**	**Moderate**	**High**
1 Hour Fuel Moisture	11.17	4.46	2.42
10 Hour Fuel Moisture	15.39	6.15	2.81
100 Hour Fuel Moisture	19.11	10.39	4.36
1000 Hour Fuel Moisture	21.81	13.95	6.06
Herbaceous Fuel Moisture	108.83	60.23	39.72
Woody Fuel Moisture	166.06	105.34	60.00
Duff Moisture	125	50	15
Temperature	60	75	90
20’ Windspeed	10	15	20

3772 Weather Records Used, 2200 Days.

These percentile environmental conditions were used to represent both natural fire conditions, such as wildfires that may be managed to move vegetative conditions toward desired conditions, as well as burning treatments that may be used for management ignited prescribed fires. These environmental conditions approximate conditions under which a natural fire may burn. They represent a good starting point for development of a management burning prescription.

Winds are recorded at the RAWS each day at 1:00 p.m. and while they capture wind speed and direction at the average hottest time of the day, this does not represent wind gusts adequately. Consequently, the wind speeds generated from analysis of the historical weather were considered too low to reflect wind gusts effecting fire behavior, so 10, 15, and 20 mph winds were substituted for low, moderate and high wind speeds at 20’ above the main vegetation canopy (where RAWS wind speeds are measured). Based on analysis of the weather data and professional judgment, 60, 75, and 90 degrees were used respectively for air temperature. Duff moisture is also not produced by the percentile weather report. These were derived using FVS-FFE defaults for duff moisture under moist (125 percent duff moisture), dry (50 percent), and very dry (15 percent) conditions (Rebain, 2010). These conditions were used across all vegetation types to provide consistency.

Treatment Matrix

A group of silviculturists and fire ecologists was assembled from throughout the southwestern region for the purpose of providing input to the forest plan revision effort. Recall that this group influenced the decision regarding the base datasets used for modeling management activities. This same group collaboratively developed a standard set of prescriptions (eight silvicultural treatments, three burning activities) that were used to address the management alternatives put forth by planning teams on the various National Forests. For each PNVT, a “0–1 Treatment Matrix” was created that designated in which model states FVS would be used

Table 6—The "0–1 Treatment Matrix" for the ponderosa pine/bunchgrass PNVT. Model state "H" is discussed in more detail in the text

		VDDT model states													
RX	**Management activities**	**A**	**B**	**C**	**D**	**E**	**F**	**G**	**H**	**I**	**J**	**K**	**L**	**M**	**N**
B	Free thin all sizes to target Basal Area (BA)	0	0	1	1	1	1	1	1	1	1	1	1	1	0
C	Thin-from-below to target BA	0	0	1	1	1	1	1	1	1	1	1	1	1	0
D	Thin under 16-inch diameter cap to target BA	0	0	0	1	1	0	0	1	1	1	1	1	1	0
E	Group selection with matrix thin to target BA	0	0	0	1	1	0	0	1	1	1	1	1	1	0
F	Shelterwood seed cut to target BA	0	0	0	1	1	0	0	1	1	1	1	1	1	0
G	Clearcut with non-regeneration objective legacy trees	0	0	0	0	0	0	0	0	0	0	0	0	0	0
H	Clearcut/Coppice for hardwood regeneration	0	0	0	0	0	0	0	0	0	0	0	0	0	0
I	Artificially plant seedlings (modeled in VDDT alone)	0	0	0	0	0	0	0	0	0	0	0	0	0	0
J	RX fire in low intensity burning conditions	0	1	1	1	1	1	1	1	1	1	1	1	1	0
K	RX fire in moderate intensity burning conditions	0	1	1	1	1	1	1	1	1	1	1	1	1	0
L	RX fire in high intensity burning conditions	0	1	1	1	1	1	1	1	1	1	1	1	1	0
M	Thin under 9-inch diameter cap to target BA	0	0	0	0	0	0	0	0	0	0	0	0	0	0

to determine the effects of management activities. The 0–1 Treatment Matrix for the PPG PNVT is shown in table 6.

A "1" in a cell indicated that we used FVS to simulate the effects of that treatment in that model state given this was an option to be considered in the analysis of management alternatives. A "0" in the treatment matrix meant that this type of treatment was either not likely to be used in the landscape analysis process for forest planning, or that it could be readily evaluated with professional judgment. FVS was not used for these cells. A national forest could decide independently whether or not to use specific cells within the "0–1 Treatment Matrix". Adjusting the transition probabilities within VDDT for particular vegetation states allows "turning on" or "turning off" movement between model states.

"Transition probability multipliers" are values within VDDT that increase or decrease the average probability for one or more transition types. This feature simplified sensitivity testing of the probabilities and allowed exploring "what if" scenarios (for example, "what if fires were twice as frequent?"). To perform this type of sensitivity testing without multipliers, we would have needed to edit the probability of each fire transition in each model state individually. Alternatively, we specified a transition multiplier of "2" for fire transitions, and the VDDT software automatically doubled the values of all fire probabilities.

Here is an example of the computational process for determining the transition probabilities following a disturbance activity. Within the PPG PNVT, there were 40 FIA plots that represented vegetation state H (i.e. medium tree size class, closed canopy, single storied). According to table 6, silvicultural prescriptions B-F and fire intensities J-L would apply to vegetation state H. The FVS model was run eight times to represent these disturbance activities. Table 7 summarizes the destination state (post treatment) for each management activity. Dividing the total number of FIA plots (40) into the FIA plot counts by prescription (across rows in table 7) produces the proportion of plots that stayed within the original vegetation state and those that transitioned to alternant states (table 8). These transition probabilities are then used in the VDDT model.

Summary Reports by State Class

Several FVS post-processing steps were bundled together to produce aggregate summaries for each vegetation state. Table 9 provides vegetation characteristics of State H. These attributes were computed for each model state by growing the representative FIA plots forward, in the absence of disturbance, for 150 years and summarizing the results.

Table 7—Destination states of 40 plots initially residing in vegetation state H as the result of FVS simulations of management activities

		Destination states								
RX	Management activities	A	C	D	E	G	H	J	L	Total
B	Free thin all sizes to target Basal Area (BA)		4	34	2					40
C	Thin-from-below to target BA			26	2	1	10		1	40
D	Thin under 16-inch diameter cap to target BA			36	2	1			1	40
E	Group selection with matrix thin to target BA			36	3				1	40
F	Shelterwood seed cut to target BA	5	2	32	1					40
J	RX fire in low intensity burning conditions			11			25		4	40
K	RX fire in moderate intensity burning conditions			13			23	1	3	40
L	RX fire in high intensity burning conditions	16		20			1	3		40

Table 8—Transition probabilities based on the proportions of FIA plots remaining and moving to alternant states as a result of management activities in vegetation state H

	Destination states								
RX	A	C	D	E	G	H	J	L	Total
B		.10	.85	.05					1.00
C			.65	.05	.025	.25		.025	1.00
D			.9	.05	.025		.025		1.00
E			.9	.075			.025		1.00
F	.125	.05	.8	.025					1.00
J			.275			.625		.1	1.00
K			.325			.575	.025	.075	1.00
L	.4	.5				.025	.075		1.00

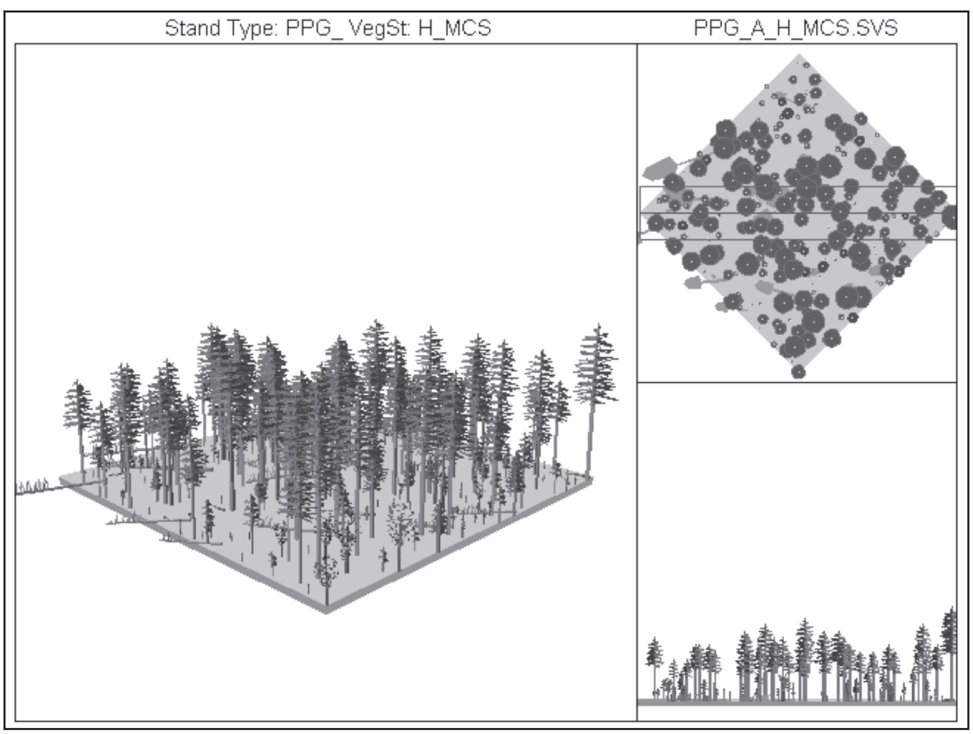

Figure 5—Stand Visualization System images of vegetation state H, the Medium Size, Closed Density, Single Story state.

Table 9—Composite characteristics of ponderosa pine/bunchgrass PNVT, vegetation state H

Vegetation characteristics	Value for State H
Vegetation structure variable examples	
Dominance type	PIPO
Canopy layers	1
Stand age in the overstory	139
Total plot activity count	1424
Proportion of stockable acres	.99
Stand-stock variable examples	
Seedlings per acre less than an inch in diameter	53.2
Trees per acre greater than an inch in diameter	249.7
Basal area per acre greater than an inch in diameter	151.6
Canopy cover percent	47.2
Annual growth: cubic feet/acre >5 inches in diameter	42.4
Annual mortality: cubic feet/acre >5 inches in diameter	20.3
Wildlife habitat variable examples	
Standing snags 8 to 12 inches in diameter	8.23
Standing snags 12 to 18 inches in diameter	6.24
Standing snags >18 inches in diameter	2.04
Pestilence disturbance examples	
Dwarf mistletoe awareness indicator (plot count)	278
Mountain pine beetle hazard	3
Wildfire risk examples	
Crown bulk density	.07
Crown bulk height	19.33
Crowning index	30.14
Torching index	32.54
Fuel load in the duff layer	4.16
Biomass and carbon examples	
Tree biomass in dry weight of live and dead boles and crown	64.5
Total stand carbon above and below ground	55.5

Figure 5 provides graphical depictions of vegetation state H using the Stand Visualization System (McGaughey 1997), which can be obtained directly from FVS output.

Timber volume calculations—

In addition to evaluating the effects of disturbance activities on post treatment states and determining average stand conditions by model state, FVS simulations also were used to quantify timber volumes required by the National Forest Management Act of 1976 (NFMA). For each combination of standard silvicultural treatment, model state, and PNVT, a variety of per acre wood volume statistics were produced in a cut-volume spreadsheet. Table 10 illustrates the harvest volumes for silvicultural prescription B in State H. When these per acre values are multiplied by the acres treated outputs from VDDT, total timber volume removal can be estimated.

Species composition—

A topic of interest was estimating the changes in species composition resulting from alternative treatment options. Relative to other management treatments, silvicultural prescription D "Thin under a 16" diameter cap to target basal area" (table 3) is controversial. Refer to Triepke et al. 2011 for a more in-depth discussion of the impasse; specifically, on page 4, Diameter Cap discussion. Each PNVT has a set of desired conditions as developed by Southwestern Regional resource specialists to guide management practices as specified in forest plan revisions. For example, the ponderosa pine/bunchgrass forest ecosystem includes

Table 10—Example of a summary of wood volume removed per acre for silvicultural prescription B in vegetation state H

Type of wood volumes per acre (cubic feet, board feet, and tons)	Net value
Softwoods	
5-9” DBH Cubic Feet (CF) for allowable sale quantity (ASQ)	153.91
> 9” DBH (CF) for ASQ	534.30
Total CF of ASQ contribution on suitable lands (excludes aspen)	688.21
0-5” DBH (tons of biomass) not ASQ	10.63
5-9” DBH (tons of biomass) not ASQ	2.22
> 9” DBH (tons of biomass) not ASQ	8.23
> 9” DBH (board feet) not ASQ	2,384.87
Nonindustrial Species > 5” diameter (PJ, etc.) (CF) not ASQ	8.14
Nonindustrial Species > 0” diameter (Tons of Biomass) not ASQ	.15
Hardwoods	
Aspen > 5” DBH (industrial CF), optional ASQ addition	8.53
Aspen > 0” DBH (tons of biomass) not ASQ	.12

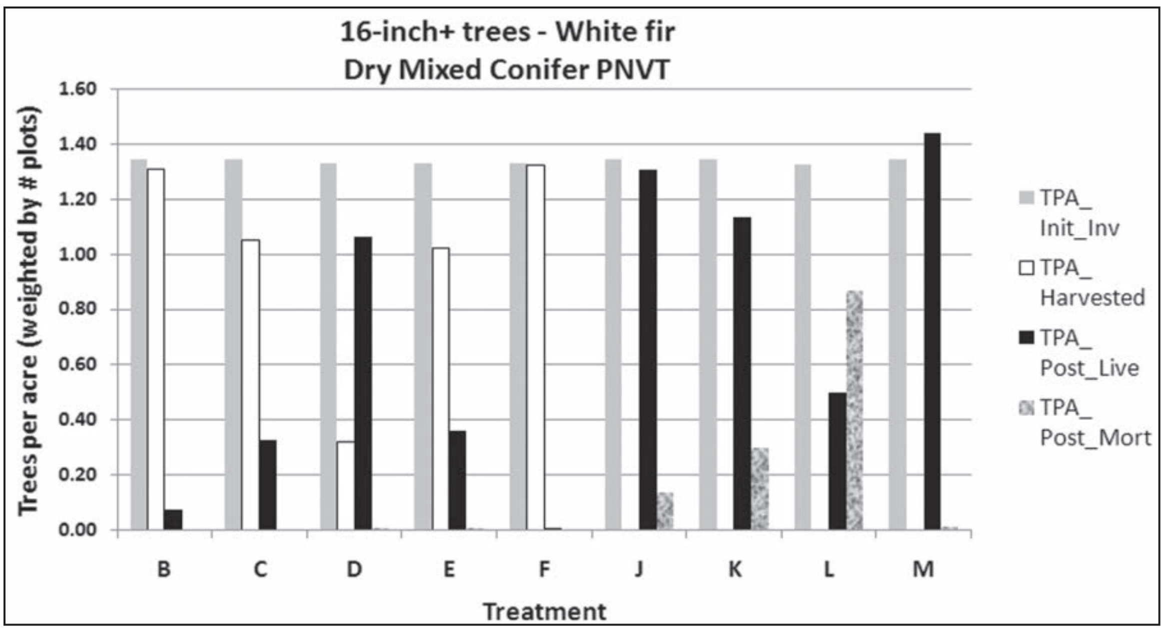

Figure 6—Tree frequency of white fir following alternative management activities in the Mixed Conifer Dry PNVT.

characteristics such as open canopy, uneven-aged structure, desired species composition, etc. In the mixed conifer dry PNVT, the goal for species composition is to mimic reference conditions under frequent, low-intensity fire regimes (i.e. open uneven-aged forests). White fir is susceptible to fire and is shade tolerant (favored by infrequent fire and closed canopy conditions). Historically, white fir was absent or rare in this forest type but over time has increased in abundance due to fire suppression and high canopy densities. FVS predicted that prescriptions B, C, E, F, and L left less white fir than did the 16-inch diameter cap prescription. Figure 6 summarizes post treatment white fir densities for the mixed conifer dry PNVT.

Discussion

Using one model’s output (FVS) as input to another model (VDDT) poses a valid question: Are the estimation error associated with the first model being passed on and perhaps compounded within the second model. Truly, inferences associated from the second model can only be as good, at the base level, as the information/parameters derived from the first model. To address this potential dilemma, we crafted the FVS model runs to forecast in accordance with observed trends from permanent plot inventory data (FIA) as based on contemporary climate. Our previously published paper (Vandendriesche 2010) stresses the importance of knowing the modeling context and making the proper adjustments in the FVS model to achieve viable estimations that are empirically based.

Based on variation in the availability of measurement data and scientific information, we recognized the importance of striking a balance between the FVS predictions, expert opinion, and research literature. This truism came into play regarding the data sets used to populate the FVS model. Based on feedback from resource specialists, criteria for fine filter rules were developed to apply against the coarse filtered, habitat type based PNVT plot sets. Additionally, when professional judgment indicated that we didn't have a sufficient number of plots in a vegetation state, we generated synthetic plots. In addition to the methods discussed above, further adjustments were made during the process based on lessons learned.

Vegetation Responses Drawn from the Literature—
Our initial intent was to quantify all STM transitions based on the results of FVS simulations using FIA plots. However, although endemic insect and disease activity (such as dwarf mistletoe) was included in our FVS projections, the probability of major insect and disease outbreaks was taken from the literature including Lynch et al. (2010) and Smith (2006). As mentioned, we did not model these effects with FVS because of time limitations in the land management planning process as implemented by the Southwestern Region. Beginning in the last quarter of 2005, each National Forest and Grassland was programmed to work through forest plan revision within a three year time period (fig. 7). This requirement focused efforts to meet required benchmarks in the planning process. Overall modeling analysis needed to be tapered to achieve the specified time goals.

Professional Judgment: Fire Effects—
Interdisciplinary teams on each national forest needed to review the results of our modeling work to differentiate acres belonging to the characteristic versus uncharacteristic "grass, forbs, brush, shrub" states (A versus N, respectively). Our mid-scale vegetation mapping products (Triepke 2005, Weisz et al. 2010) did not delineate between these two vegetation states. Consequently, when tabulating the results of fire burning under high intensity conditions (Rx L, table 3), all FVS plots destined for the "grass, forbs, brush, shrub" stage were initially assigned to state A. Following the interdisciplinary review, plots were reallocated between states A and N based on local terrestrial ecological unit data, historical fire data, and professional judgment.

As an example, FVS results of a prescribed fire burning under high intensity conditions in state H resulted in 40 percent of the plots transitioning to state A. However, a review of historical fire data and fire effects data on the Apache-Sitgreaves National Forest suggested that 27 percent the plots would go to state A and 13 percent of the plots would go to state N. It should be expected that each national forest and PNVT has a unique division of plots between states A and N resulting from fires burning under high intensity conditions in closed canopy states.

We decided that the effects of unplanned ignitions would be the same as the effects of planned ignitions. Essentially, we assumed that unplanned nonlethal ignitions would have the same destination states as Rx J; unplanned mixed severity ignitions would have the same destination states as Rx K; and, unplanned stand replacing ignitions would have the same destination states as Rx L. This was based on an interdisciplinary review of the results of the FVS-FFE modeling process.

Professional Judgment: Group Selection—
At the start of our FVS modeling process, we made two simplifying assumptions. The first assumption was that a 14-box model with states classified on the basis of cover type, dominant trees size, percent canopy cover, and storiedness (single story versus multi-story) was sufficient to represent most forest conditions and management activities (table 1). The second assumption was that for modeling

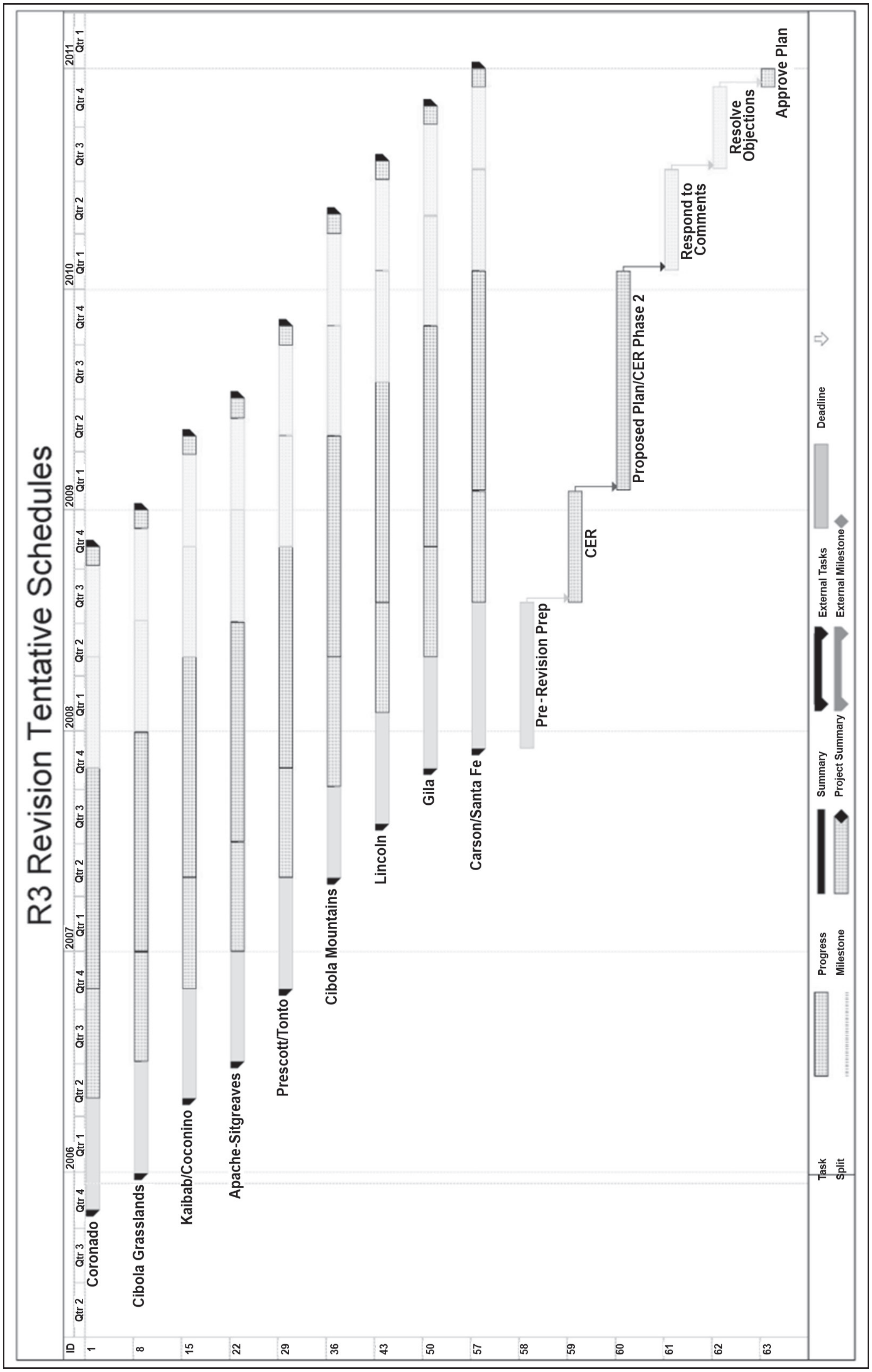

Figure 7—Southwestern Region forest plan revision timetable.

Table 11—Transition probabilities from the group selection prescription based on FVS projections

From State	A	D	E	I	J	K	Total
	To Destination States						
D		.95	.05				1.00
E			1.00				1.00
H		.90	.08		.03		1.00
I			1.00				1.00
J	.06	.41			.53		1.00
K			.50		.10	.40	1.00
L		.23	.30	.06	.23	.17	1.00
M		.09	.73	.18			1.00

Table 12—Transition probabilities from group selection prescriptions based on professional judgment

From State	A	D	E	I	J	K	Total
	To Destination States						
D					1.00		1.00
E						1.00	1.00
H					1.00		1.00
I						1.00	1.00
J					1.00		1.00
K						1.00	1.00
L						1.00	1.00
M						1.00	1.00

the effects of management activities, we only needed to use FVS to categorize conditions before and after treatment. As we completed our initial FVS process, it became apparent that these assumptions needed to be modified for the group selection treatment type (Rx E, table 3).

Table 11 lists the destination states immediately following implementation of the group selection prescription (Rx E). Post-treatment destination states and the resulting wood volume estimates describe the immediate effects of *one group selection entry* rather than a *number of entries that would create several groups* over several cutting cycles. A major objective for applying a group selection silvicultural system is to convert existing even-aged stands to uneven-aged structures. A minimum of five entries in succession on the same stand acres is needed to be successful. A more realistic modeling approach of group selection silviculture would have resulted in a more complex VDDT model with more model states and many more transitions (tracking relative age, time since disturbance, etc.).

Instead, we pursued a simplified approach. We assumed that the group selection management system would provide the "predominance of uneven-aged dynamics" which is characterized by states J and K (table 1; Triepke et al. 2011). These assumptions were implemented in the VDDT model as displayed in table 12. The results were consistent with the group selection FVS runs applied over time to develop the Long Term Sustained Yield Capacity (LTSYC) for each of our PNVTs (Youtz 2011).

Applications

The methods and models described in this paper are currently being used by the national forests in Arizona to assist in their development of revised land management plans (for example, Higgins and Kleindienst 2011). They are also currently being utilized as a point of departure for large area assessments (Hemstrom 2012) and are being linked to global climate change models to assess effects of climate change on forest ecosystems (Kerns et al. 2012). The main contribution of this work has been to demonstrate the use of the FVS model and related software to estimate the effects of disturbance activities in an STM. It is our intent that others can benefit from this process.

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References

Dixon, G.E. 2002. Essential FVS: A user's guide to the Forest Vegetation Simulator. Internal Report. Fort Collins, CO. U.S. Department of Agriculture, Forest Service, Forest Management Service Center.

ESSA [ESSA Technologies Ltd.]. 2006. Vegetation dynamics development tool user guide, Version 5.1. ESSA Technologies Ltd., Vancouver, BC, Canada. [Online] http://essa.com/downloads/vddt/download.htm. (12/31/2008).

He, H.S. 2008. Forest landscape models: definitions, characterization, and classification. Forest Ecology and Management 254: 484–498.

Hemstrom, M.A.; Salwasser, J.; Halofsky, J.S.; Kagan, J.; Comfort, C. 2012. The Integrated Landscape Assessment Project. In: Kerns, B.K., Shlisky, A.J., Daniel, C.J., tech. eds. Proceedings of the First Landscape State-and-Transition Simulation Modeling Conference, June 14–16, 2011, Portland, Oregon. Gen. Tech. Rep. PNW-GTR-869. Portland, OR: U.S. Department of Agriculture, Forest Service, Pacific Northwest Research Station: 57–71.

Henderson, E.B. 2008. Development of state and transition model assumptions used in national forest plan revision. In: Havis, R. N.; Crookston, N.L., comps. 2008. Third Forest Vegetation Simulator Conference; 2007 February 13–15; Fort Collins, CO. Proceedings RMRS-P-54. Fort Collins, CO: U.S. Department of Agriculture, Forest Service, Rocky Mountain Research Station. 89–97 p.

Higgins, B.J.; Kleindienst, H.P. 2011. Draft Vegetation and Fire Specialist Report for the Draft Forest Plan Revision DEIS. Kaibab National Forest, Williams, Arizona.

Kerns, B.K.; Hemstrom, M.A.; Conklin, D.; Yospin, G.; Johnson, B.; Bachelet, D.; Bridgham, S. 2012. Approaches to incorporating climate change effects in state and transition simulation models of vegetation. In: Kerns, B.K., Shlisky, A.J.; Daniel, C.J., tech. eds. Proceedings of the First Landscape State-and-Transition Simulation Modeling Conference, June 14–16, 2011, Portland, Oregon. Gen. Tech. Rep. PNW-GTR-869. Portland, OR: U.S. Department of Agriculture, Forest Service, Pacific Northwest Research Station: 161–171.

Lynch, A.; Anhold, J.A.; McMillan, J.D.; Dudley, S.M.; Fitzgibbon, R.A.; Fairweather, M.L. 2010. Forest insect and disease activity on the Apache-Sitgreaves N.F. and Fort Apache Indian Reservation, 1918–2009. Internal report in the forest planning records of the Apache-Sitgreaves National Forests, U.S. Department of Agriculture, Forest Service, Springerville, Arizona.

McGaughey, R.J. 1997. Visualizing forest stand dynamics using the stand visualization system. In: Proceedings of the 1997 ACSM/ASPRS Annual Convention and Exposition. April 7–10, 1997, Seattle, WA. Bethesda, MD: American Society of Photogrammetry and Remote Sensing 4: 248–257.

Moeur, M.; Vandendriesche, D. 2010. Calibration of State and Transition Models with FVS. In: Jain, Theresa; Graham, Russell T.; and Sandquist, Jonathan, tech. eds. Integrated management of carbon sequestration and biomass utilization opportunities in a changing climate: Proceedings of the 2009 National Silviculture Workshop; 2009 June 15–18; Boise, ID. Proceedings RMRS-P-61. Fort Collins, CO: U.S. Department of Agriculture, Forest Service, Rocky Mountain Research Station. 275–287 p.

Rebain, Stephanie A. comp. 2010 (revised March 20, 2012). The Fire and Fuels Extension to the Forest Vegetation Simulator: Updated Model Documentation. Internal Rep. Fort Collins, CO: U.S. Department of Agriculture, Forest Service, Forest Management Service Center. 397 p.

Schussman, H.; Smith, E. 2006. Introduction to the Historic Range of Variation: in Historical Range of Variation for Potential Natural Vegetation Types of the Southwest. Prepared for the U.S. Department of Agriculture, Forest Service, Southwestern Region by The Nature Conservancy, Tucson, AZ. 22 pp. The Nature Conservancy in Arizona. [Online] http://azconservation.org/downloads/multi/category/sw_forest_assessment/. (6/12/2009).

Smith, E.B. 2006. Historical range of variation and state and transition modeling of historical and current landscape conditions for ponderosa pine forest and woodland of the Southwestern U.S. The Nature Conservancy in Arizona.

Shlisky, A.J.; Vandendriesche, D. 2012. Use of state-and-transition simulation modeling in National Forest planning in the Pacific Northwest, U.S.A. In: Kerns, B.K., Shlisky, A.J., Daniel, C.J., tech. eds. Proceedings of the First Landscape State-and-Transition Simulation Modeling Conference, June 14–16, 2011, Portland, Oregon. Gen. Tech. Rep. PNW-GTR-869. Portland, OR: U.S. Department of Agriculture, Forest Service, Pacific Northwest Research Station: 23–41.

Triepke, F.J. 2005. Southwestern Region Mid-scale Existing Vegetation Mapping Project. Powerpoint presentation. Forest Planners Meeting, Phoenix, Arizona, March 2–4, 2005. U.S. Department of Agriculture, Forest Service, Southwestern Region, Albuquerque, NM. 14 slides.

Triepke, F.J.; Higgins, B.J.; Weisz, R.N.; Youtz, J.A.; Nicolet, T. 2011. Diameter caps and forest restoration—Evaluation of a 16-inch cut limit on achieving desired conditions. USDA Forest Service Forestry Report FR-R3-16-3. Southwestern Region, Regional Office, Albuquerque, NM. 31 p.

U.S. Department of Agriculture, Forest Service [USDA FS]. 1997. Plant Associations of Arizona and New Mexico, Volume 1: Forest, Volume 2: Woodlands. Edited and updated by USDA Forest Service Southwestern Region, Albuquerque, NM. Vol. 1: 291 p. Vol. 2: 196 p.

Vandendriesche, D. 2009. Advanced FVS topics for forest planning. Preside: process residence times. Interoffice publication. Fort Collins, CO. U.S. Department of Agriculture, Forest Service, Forest Management Service Center. 129 p.

Vandendriesche, D. 2010. FVS Out of the Box—Assembly Required. In: Jain, T.; Graham, R.T.; Sandquist, J., tech. eds. Integrated management of carbon sequestration and biomass utilization opportunities in a changing climate: Proceedings of the 2009 National Silviculture Workshop; 2009 June 15–18; Boise, ID. Proceedings RMRS-P-61. Fort Collins, CO: U.S. Department of Agriculture, Forest Service, Rocky Mountain Research Station. 289–306 p.

Weisz, R.; Triepke, J; Vandendriesche, D.; Manthei, M.; Youtz, J.; Simon, J.; Robbie, W. 2010. Evaluating the ecological sustainability of a pinyon-juniper grassland ecosystem in northern arizona. In: Jain, T.; Graham, R.T.; Sandquist, J., tech. eds. Integrated management of carbon sequestration and biomass utilization opportunities in a changing climate: Proceedings of the 2009 National Silviculture Workshop; 2009 June 15–18; Boise, ID. Proceedings RMRS-P-61. Fort Collins, CO: U.S. Department of Agriculture, Forest Service, Rocky Mountain Research Station. 321–336 p.

Youtz, J. 2006. Timber Suitability Classification Process. U.S. Department of Agriculture, Forest Service, Southwestern Region, Albuquerque, NM, USA. Interoffice publication. 17 p.

Youtz, J. 2011. White Paper K: Long-term sustained-yield calculations for forest land and resource Management planning. U.S. Department of Agriculture, Forest Service Draft Report, Southwestern Region, Albuquerque, NM, USA.

Approaches to Incorporating Climate Change Effects in State and Transition Simulation Models of Vegetation

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Abstract

Understanding landscape vegetation dynamics often involves the use of scientifically-based modeling tools that are capable of testing alternative management scenarios given complex ecological, management, and social conditions. State-and-transition simulation model (STSM) frameworks and software such as PATH and VDDT are commonly used tools that simulate how landscapes might look and function in the future. Until recently, however, STSMs did not explicitly include climate change considerations. Yet the structure of STSMs makes them highly conducive to the incorporation of any probabilistic phenomenon. The central task in making a STSM climate-sensitive is describing the relevant processes in terms of probabilistic transitions. We discuss four different approaches we have implemented to inform climate-induced changes in vegetation and disturbance probabilities in STSMs using the dynamic global vegetation model MC1. These approaches are based on our work in several landscapes in the western United States using different modeling frameworks. Developing STSMs that consider future climate change will greatly broaden their utility, allowing managers and others to explore the roles of various processes and agents of change in landscape-level vegetation dynamics. However, numerous caveats exist. Regardless of how STSMs are made climate-sensitive, they neither simulate plant physiological responses directly nor predict landscape states by simulating landscape processes mechanistically. They are empirical models that reflect the current understanding of system properties and processes, help organize state-of-the-art knowledge and information, and serve as tools for quickly assessing the potential ramifications of management strategies. As such, they highlight the need for new research, while providing projections based on the best available information.

Keywords: climate change, coupled models, dynamic global vegetation models, state-and-transition simulation model, vegetation dynamics.

Introduction

Across the globe, plant communities are already experiencing the effects of climate change: warmer temperatures, earlier springs and earlier snowmelt, reduced snowpack, changes in fire regimes, and higher concentrations of CO_2 (Parry et al. 2007). There is increasingly strong evidence that climate change will profoundly alter vegetation structure and composition, ecosystem processes, and the future delivery of ecosystem goods and services (Parry et al. 2007). Coupling climate change projections with landscape vegetation dynamics is a promising approach that involves the use of scientifically based modeling tools that are capable of testing alternative management scenarios given complex ecological, management, and social conditions. State and transition simulation models (STSMs) are one tool for simulating how landscapes might look and function in the future and thus guide decisionmaking (Daniel and Frid 2012). With vegetation STSMs, different potential vegetation types are grouped into discrete state classes. Transitions from one state class to another may occur probabilistically or are empirically based; regardless, they represent the effects of ecological processes such as succession and wildfire and management actions (Daniel and Frid 2012). Although factors such as drought and frost kill have been included as probabilistic disturbances within STSMs (e.g., Evers et al. 2011), up until recently most STSMs did not include climate change considerations. Incorporating the effects of future climate change would increase the utility of STSMs as a common platform to collectively define the roles of various processes in projecting landscape-level vegetation dynamics. To the best of our knowledge, there are only a handful of studies where changing climate has been explicitly being incorporated into STSMs (e.g., Costanza et al. 2010, Hemstrom et al. in press, Provencher et al. 2009, Provencher and Anderson 2011, Yospin 2012).

Climate change can affect vegetation by altering the future abiotic and biotic conditions under which plant species establish, survive, reproduce and spread. Increased temperature, longer growing seasons, less snow, and more frequent drought conditions may increase plant stress and decrease a species' ability to survive in the drier and warmer parts of its range (Allen and Breshears 1998, Allen et al. 2010). Changes in abiotic conditions and subsequent effects on individual species reproduction, establishment and growth may in turn substantially alter plant competitive dynamics (Pfeifer-Meister et al. 2008). Rising CO_2 concentrations will also directly affect plant growth and productivity through a variety of mechanisms (Nowak et al. 2004). But climate change modifications of disturbance regimes, such as wildfire and insect and disease outbreaks, might be the most important factors for forcing future vegetation responses (Brown and Westerling 2004, Taylor and Beatty 2005, Westerling et al. 2006, Raffa et al. 2008, Pennisis 2009). Therefore, STSMs that consider changes in successional trajectories and disturbance regimes in response to changing climate are needed.

Considerations for Creating Climate-Sensitive State and Transition Simulation Models

The structure of STSMs makes them highly conducive to the incorporation of any probabilistic phenomenon using information generated from another source, dataset, modeling framework, or even expert knowledge (Daniel and Frid 2012). The central task in making a STSM climate-sensitive is describing the relevant processes in terms of new assumptions and empirical relationships, including probabilistic transitions. This can be a challenge because it is not always clear how information from general circulation models (GCMs) and climate-sensitive vegetation models can be reduced to a set of empirical relationships, particularly to the fine spatial grain and level of detail at which many STSMs are typically developed. Information regarding potential future climate changes can be generated by more than twenty different GCMs that give variable projections of future climate (Littell et al. 2011). However, GCMs are highly complex mechanistic models that estimate potential future climatic trends on grids at resolutions of thousands or tens of thousands of square kilometers. Each grid-cell represents average conditions within its boundaries, producing daily to yearly estimates of a variety of climate attributes. Because GCMs estimate potential future climatic trends using coarse spatial grids, they are frequently spatially downscaled using a variety of quantitative techniques (Littell et al. 2011). General circulation models are all ‘‘forced’’ with scenarios of greenhouse gas emissions that reflect different assumptions about future global economic activity and fossil fuel use (Nakicenovic et al. 2000). Thus a single emission scenario can generate multiple future climate scenarios using different GCMs. Alternately, a single GCM can project multiple future scenarios under different emissions scenarios. Ideally, output from multiple GCMs and emission scenarios would be used for input into a climate-sensitive vegetation model to capture the available range of future conditions and simulate their consequences for ecosystems. However, this ensemble approach is computationally intensive (Littell et al. 2011) and furthermore requires that all climate variables needed to run the vegetation model are readily available. One approach to avoid this problem is to pair scenarios based on a gradient of risk (Kerns et al. 2009, Littell et al. 2011, Mote and Salathé 2010). For example, one could pair a scenario combination (GCM and emission scenario) at two extremes (e.g., less warming vs. lots of warming; precipitation increases vs. precipitation decreases). However, the risk framework may only be specific to a single resource issue. That is, a high risk scenario for potential changes in wildfire may be different than a high risk scenario for potential changes in an endangered species habitat.

Downscaled future climate data are becoming increasingly available at scales useful for land managers (e.g., 0.6 km^2, Rogers et al. 2011). However, it is important to assess whether downscaling has exceeded the resolution supported by observations, recognizing that finer-scale projections are not always more reliable (Littell et al. 2011). Moreover, climate change data alone are not usually useful input data for a STSM. Climate data can be used to develop the empirical relationships required to modify STSM transition probabilities (e.g., fire frequencies, insect and disease outbreaks, changes in tree growth rates) for projecting changes in potential vegetation, or as input into other models that explicitly incorporate climate information and produce output that can then be used by STSMs. Provencher and Anderson (2011) used projections of future CO_2, precipitation and temperature, to create trends in STSM disturbance transition probabilities. The authors also used information about species regeneration and simulated range shifts from the literature to develop a series of hypothetical range shifts for vegetation in Nevada. Costanza et al. (2010) modeled the effects of climate change by altering fire frequencies using the spatially explicit TELSA model (Kurz et al. 2000). Historic (1979–2010) climate and fire occurrence data were used to hindcast relationships between the acres burned and climate variables (i.e., temperature and precipitation). Those relationships were then incorporated as a multiplier on fire transition probabilities in the TELSA model runs. A statistical approach may be one method for generating projections, but may miss the important interactions between climate, disturbance, and succession that will drive changes in vegetation over the coming century.

A commonly used approach for landscape analysis is to stratify the landscape according to one or more criteria that are considered to be important external drivers of vegetation change, and then to develop a separate pathway diagram (STSM) for each stratum (the spatially stratified state-and-transition simulation model approach, Daniel and Frid 2012). Biophysical drivers, such as soils, climate, and topography are typically used to define a stratum, based on existing ecological classification systems, such as potential vegetation types (PVTs) (Chiarucci et al. 2010) or biophysical settings (Long et al. 2006). However, modelers assume that the landscape stratification is then fixed over planning horizons of several decades or centuries, although strata can move through numerous state classes according to the defined STSM transition pathways. But the assumption that the present-day landscape stratification will remain constant over time is only valid if the underlying site conditions that define the stratum boundaries on the landscape remain constant. Because this is unlikely with future climate change, climate-sensitive models must by definition incorporate many spatially-stratified STSMs with transition pathways between strata. For example, such a model would allow transitions from a broadleaf forest PVT to a mixed conifer forest PVT and in turn to a conifer forest PVT. Others have operationally referred to such STSMs as a "mega-model" (i.e., Hemstrom et al. in press). Likewise, we will refer to these large spatially stratified STSMs with transition pathways between strata as mega-STSMs.

A number of climate-sensitive vegetation models are available that can be used to create climate sensitive STSMs. These models are either empirical and typically species-specific (e.g., Rehfeldt et al. 2006, 2008; Iverson et al. 2008), or mechanistic (i.e., process-based, or physical) (Keane et al. 2004) and usually not species-specific (e.g., Bugmann 2001, Bachelet et al. 2003). Empirical models fit parameters to observations and use statistical methods to make projections. By contrast, mechanistic models typically try to represent underlying physiological processes, and thus can incorporate complex and novel interactions. The capacity to include novel interactions allows a model to yield unexpected outcomes, which is a critical consideration for planning across a wide range of potential future conditions. However, mechanistic models are highly complex and require extensive training to use them correctly. They typically cannot incorporate the current vegetation as an initial starting condition (i.e., they require an extensive 'spin-up' period), cannot directly incorporate management, and produce outputs as plant physiognomic types instead of actual species. Thus, these models provide a fairly abstract view of potential vegetation in a landscape under a set of climatic conditions. Incorporating output from mechanistic models into STSMs would better allow their results to be used for management purposes.

Using MC1 to Build Climate Sensitive State-and-Transition Simulation Models

In the following section we present challenges and four approaches for incorporating climate-induced vegetation changes in STSMs using the dynamic global vegetation model MC1: (1) modifying potential vegetation using annual probabilities and transition multipliers calculated from MC1; (2) modifying potential vegetation by developing spatially explicit changes in vegetation using regression equations between MC1 output variables and site index and the Forest Vegetation Simulator (FVS); (3) modifying wildfire probabilities using annual probabilities and transition multipliers calculated from MC1; and (4) using MC1 output to develop a simple spatially-explicit rule-base to attenuate growth potential across a landscape. These approaches are based on our collaborative work across several landscapes in the western United States (Hemstrom et al. in press, Yospin 2012) using different modeling frameworks, and described in more detail in the following section.

Dynamic global vegetation models (DGVMs) use climate projections from GCMs to simulate vegetation potential, vegetation growth, carbon and nutrient dynamics, and some natural disturbance regimes (e.g., wildfire) at relatively coarse resolution (Bachelet et al. 2003). Output is usually at a coarser spatial grain than that at which land managers make decisions. Furthermore, DGVMs do not usually include species-specific information, detailed vegetation dynamics such as seed dispersal, fire adaptations of various species, or the effects that various land management activities might have on vegetation dynamics. MC1 is a DGVM that simulates plant type mixtures and broad vegetation types; pools and fluxes of carbon, nitrogen, and water through ecosystems; and fire disturbance. MC1 routinely generates century-long, regional-scale simulations on relatively coarse-scale data grids (Bachelet et al. 2003, 2005; Lenihan et al. 2008).

MC1 (Bachelet et al. 2001) is a good candidate for incorporating climate-induced vegetation changes in STSMs because the model is mechanistic, incorporates disturbance dynamics, and projects future vegetation mechanistically based on changes in climate and biogeochemistry. MC1 combines a biogeography model (MAPSS), a model to simulate fire disturbance (MC-FIRE), and a biogeochemistry model (Century). Therefore MC1 can provide relevant output about future changes in potential vegetation and wildfire regimes that can in turn can be used to alter site potential and wildfire probabilities in a connected suite of STSMs that make up a landscape.

Although MC1 produces projections of future changes in vegetation, it does so by predicting the life form or plant functional types mixtures, which are then classified into potential vegetation classes. A common challenge for using output from DGVMs such as MC1 is that these classes are not directly comparable to most locally defined STSM strata such as PVTs (fig. 1, table 1). Thus a key methodological issue in using MC1 output to build a climate-sensitive STSM is how to relate or "cross-walk" the local strata within a study area, such as a PVTs, to the more broadly defined potential vegetation classes simulated by MC1. Typically MC1 potential vegetation classes combine numerous species and structural conditions into single entities (table 1). For example, "temperate needleleaf forest" is an important MC1 potential vegetation class simulated for many western U.S. forested landscapes. In the interior Pacific Northwest, this broad class would correspond to a variety of strata, including ponderosa pine, lodgepole pine, Douglas-fir, and grand fir. Because most STSM PVTs

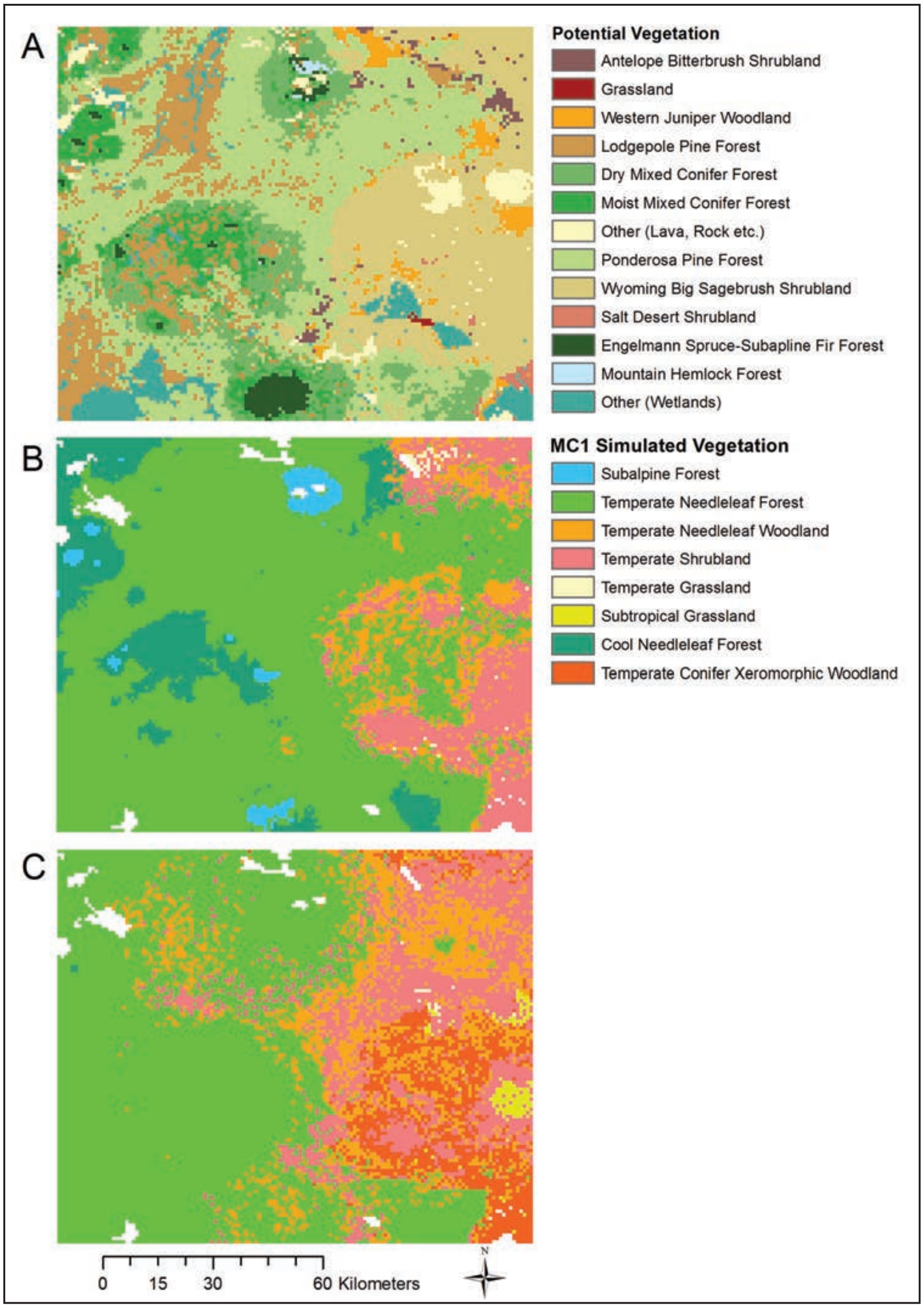

Figure 1—Model output for vegetation types in a central Oregon area (30-arc sec grid, 800-m^2): (A) STSM potential vegetation types generated from imputed data (B) MC1 potential vegetation classes projected for the historical period (30-year mode vegetation), and (C) MC1 potential vegetation classes projected for the last part of the 21st century (30-year mode vegetation, MIRCO A2 scenario).

currently in use represent relatively fine-scale ecological conditions, their relation to a MC1 potential vegetation class is often many to one (table 1).

Given this constraint, one crosswalk approach is to select a single representative STSM stratum for each MC1 potential vegetation class for a landscape. Throughout this document, we will often provide examples and illustrate processes using the PVT concept for STSM stratum. With this approach, numerous aggregated PVTs will crosswalk to a particular MC1 vegetation class (Hemstrom et al. in press). Historical and future simulations with MC1, compared to locally derived maps, can guide the selection of the most representative PVTs for each landscape. It is important to consider whether or not the representative PVT fits the MC1

Table 1—Example showing the relationship between selected MC1 potential vegetation classes and locally representative potential vegetation types from a study area in eastern Oregon

MC1 Potential vegetation class	Description	STSM potential vegetation type
Subalpine forest	Subalpine forests in cold, upper elevation environments.	• Mountain hemlock (*Tsuga mertensiana* (Bong.) Carrière) – cold, dry • Shasta red fir (Abies × *shastensis* (Lemmon) Lemmon [*magnifica* × *procera*]) – dry • Subalpine woodland
Cool needleleaf forest	Mixed conifer forests in relatively moist mid- to upper-elevation forested environments.	• Mixed conifer – moist • Lodgepole pine (*Pinus contorta* Douglas ex Louden) – wet • Mixed conifer – cold dry • Cold dry forest • Cool moist forest
Temperate needleleaf forest	Mixed conifer forests in relatively dry mid- to lower elevation forested environments.	• Ponderosa pine (*Pinus ponderosa* C. Lawson var. *ponderosa*)/lodgepole pine – dry • Mixed conifer – dry • Ponderosa pine – xeric • Mixed conifer – dry (pumice soils) • Grand fir (*Abies grandis* (Douglas ex D. Don) Lindl.) – dry • Douglas-fir (*Pseudotsuga menziesii* (Mirb.) Franco) – dry • Lodgepole pine – dry

plant functional type concept. It may also be possible to fine tune MC1 so that local vegetation is better reflected in the broad vegetation classes (Hemstrom et al. in press). When developing representative PVTs to reflect the dynamics of vegetation in the future, additional PVTs that might become more common in the future need to be added to the mega-STSM. Adjacent regions that may represent potential future conditions in the area of interest can be assessed for relevant candidates. This approach assumes that extant PVTs already approximately represent the vegetation dynamics that MC1 simulates. The selected PVTs are, therefore, surrogates for future potential vegetation types that are assumed to have generally similar successional and disturbance dynamics. The resulting climate-sensitive mega-STSM would then consist of a combination of representative PVTs based on output from MC1 from the historical and future simulation periods. Transitions in the mega-STSM could then allow portions of the landscape to move among the previously independent PVTs according to output from MC1 run with the selected climate change scenarios.

Once the strata for a mega-STSM have been selected, and incorporated into a single model, there are a number of ways in which output from MC1 can inform changes in strata. For example, MC1 projects changes in potential vegetation classes for a particular climate change scenario (an emission scenario combined with a GCM). These changes, in turn, can be converted into probabilities, which can then be used to inform the transition probabilities in the mega-model (Hemstrom et al. in press). However, MC1 vegetation types can change quickly and unrealistically from year to year, so implementing simple annual changes among strata does not necessarily lead to reasonable model output, especially at a fine spatial or temporal scales grain. A feature of STSMs is that they can be configured to change transition probabilities over time; using transition multipliers, average transition rates can be shifted up or down in any year by proportions that range from zero (no transition occurs that year) to greater than one (the transition is larger than the long-term simulation period average that year). Modelers can compute the average annual transition rate for each potential vegetation climate change transition over the entire MC1 simulation period, then develop transition multipliers to shift the average annual rate up or down according to MC1 output for that climate change transition in that year. Using this technique, it is possible to reproduce the long-term average and the year-to-year variation simulated by MC1 for transitions between PVTs in the mega-STSM. Provencher and Anderson (2011) use a different method for incorporating climate change into their STSMs, but make similar use of transition multipliers. The conditions under which the STSM allows transitions due to climate change should be carefully considered and ecological constraints may be necessary to produce plausible dynamics. One approach is to only allow transitions between PVTs within the STSM following a stand-replacing disturbance (Provencher and Anderson 2011, Hemstrom et al. in press, Yospin 2012); alternatively, transitions from one potential vegetation type to another may occur under a broader set of circumstances.

A second approach to adjust transition probabilities for successional changes among strata and states over time in a mega-STSM is to use additional MC1 output beyond just its projected plant functional types. For example, Yospin (2012) developed a regression equation between MC1 output variables (e.g., soil carbon) that correlated reasonably well ($r^2 = 0.55$, $p < 0.001$) with site index, a measurement of the height to which a Douglas-fir will grow in 50 years. Forest stands representing current and potential plant communities were run through the Forest Vegetation Simulator (Crookston and Dixon 2005) at a wide range of site index values. The rates at which trees within these stands transitioned from one STSM state to another under different site indexes were then converted to annual transition probabilities. Using the regression equation, MC1 output was used to project future site index in each location over time, which in turn was used to select the appropriate transition probabilities for each location at each time step. Because site index and MC1 data were spatially explicit, this approach allows for spatially explicit simulations of climate change effects on site productivity.

These two types of adjustments to STSM transition probabilities account for changes in plant growth potential due to climate change, but do not capture the role of other climate-related effects. The impact of climate change on other stand-replacing disturbances also needs to be accounted for, and this is the focus of our third approach. Presently, MC1 does not provide projections regarding disturbance types other than fire, although researchers are currently working to incorporate insect and disease effects. Hemstrom et al. (in press) used projections for wildfire occurrence directly from MC1 and incorporated these into a mega-STSM. First, annual trends in wildfire probabilities from MC1 were calculated using the annual fraction of cells burned each year. For simplicity, and to reduce uncertainty, output was combined for several STSM strata (e.g. forest types, arid land types). MC1 can run without or with fire suppression using a set of algorithms that only allow intense stand-replacing fires to spread (Rogers et al. 2010). If MC1 is run without fire suppression, the projections for area burned are considerably higher than would be expected with fire suppression, but future projections of fire area burned can be scaled down using empirical datasets (e.g., the Monitoring Trends in Burn Severity data-set, Eidenshink et al. 2007, http://www.mtbs.gov). Alternately, MC1 can run with fire turned off during the historical period, the future period, or both periods. In this case, a separate statistical or mechanistic fire model would be required to provide projections of fire disturbance to the mega-STSM. By using a fire model outside of MC1, carbon and biogeochemical pools simulated by MC1 are decoupled from the fire effects, missing fire mortality and biomass consumption the model normally calculates. Furthermore, an external fire model would disregard the build-up of fuel and fuel moisture variability that serve as index to trigger fires in MC1.

There are also many other parameters within a STSM or mega-model that could change dynamically in response to changing climatic conditions. Mortality probabilities may need adjustment for drought-stressed trees under some climate scenarios (for examples, Provencher et al. 2009, Provencher and Anderson 2011). Yospin (2012) used a simple spatially-explicit rule base to attenuate growth potential across a landscape. The rule base restrictions prevent forests from growing larger trees or denser forest stands when MC1 indicated that climate did not support such growth, without summarily imposing mortality for the stand. This ecological restriction on both growth and mortality is a conservative approach to making the STM climate-sensitive. Simulations using a STSM parameterized with MC1 with the fire module turned completely off are one way to test the effect of climate independently from fire disturbance. For example, the direct effects of increased CO_2 and increased plant water use efficiency may accelerate some successional pathways, or allow larger amounts of carbon to be stored on the landscape, although this effect may be only marginal for some ecosystems. Users may also need to define additional states to capture such phenomena, or allow another model to specify those transitions.

One advantage of performing spatially-explicit simulations of vegetation dynamics with a STSM (e.g., Yospin 2012) is that spatially explicit land management actions can then be simulated in conjunction with climate-sensitive ecological succession. For example, the STSM developed by Yospin (2012) is being incorporated as a module within a larger modeling system named Envision. Envision is an agent-based model of landscape change that allows individual agents, representing different types of landowners, to make probabilistic land use and land management decisions based on the availability of resources, feedbacks from past actions, and in response to user-defined behaviors (Envision is a new model based on Evolan, Bolte et al. 2006, Guzy et al. 2008). Envision is one example from the broad array of agent-based social decision simulation models, many of which rely on simple STSMs of vegetation. Incorporating climate change into STSMs of vegetation may be an effective way to bring climate change effects into simulation modeling of landscape management.

Discussion and Conclusions

Developing vegetation STSMs that incorporate the possible effects of future climate change will broaden and enhance their utility, allowing managers and other users to explore the roles of various processes and agents of change on landscape-level vegetation dynamics. The empirical basis of STSMs makes possible a variety of approaches for incorporating the effects of climate change. We describe common challenges and four approaches using output from the DGVM MC1 to create climate-sensitive STSMs (Hemstrom et al. in press, Yospin 2012). These approaches hold promise because the DGVM can mechanistically project potential vegetation changes and fire with changing climate, while the mega-STSMs can apply these changes to locally relevant potential vegetation, impose realistic management actions, and mitigate the rapid rates of change allowed under DGVMs like MC1. We are currently producing example extrapolations of possible vegetation change from several climate change scenarios in different case study landscapes using these approaches. We expect these methods to be of considerable interest to others who use STSMs as well.

However, it is essential to recognize numerous caveats about all STSM-based approaches. Regardless of how STSMs are made climate-sensitive, they neither simulate physiological responses of vegetation nor project landscape states by simulating landscape processes mechanistically. Rather they are empirical models that must draw from a combination of other models and expert judgment to reflect the current understanding of system properties. In doing so, they can help researchers organize state-of-the-art knowledge and information, and serve as tools for assessing the potential ramifications of alternative management strategies. Because STSMs are probabilistic, a series of repeated simulations can be used to bracket a potential range of future conditions under changing climate. The results from these models can be informative for land managers working at a variety of spatial grains and scales. We see these approaches as promising avenues for improving landscape planning and assessments under the projected trends and uncertainties of climate change.

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References

Allen, C.D.; Breshears, D.D. 1998. Drought-induced shift of a forest-woodland ecotone: rapid landscape response to climate variation. Proceedings of the National Academy of Sciences of the United States of America. 95: 14839–14842.

Allen, C.D.; Macalady, A.K.; Chenchouni, H.; Bachelet, D.; McDowell, N.; Vennetier, M.; Kizberger, T.; Rigling, A.; Breshears, D.D.; Hogg, E.H.; Gonzalez, P.; Fensham, R.; Zhang, Z.; Castro, J.; Demidova, N.; Lim, J.H.; Allard, G.; Running, S.W.; Semerci, A.; Cobb, N. 2010. A global overview of drought and heat-induced tree mortality reveals emerging climate 386 change risks for forests. Forest Ecology and Management. 259: 660–684.

Bachelet, D.; Lenihan, J.M.; Daly, C.; Neilson, R.P.; Ojima, D.S.; Parton, W.J. 2001. MC1: a Dynamic Vegetation Model for estimating the distribution of vegetation and associated carbon and nutrient fluxes, Technical Documentation Version 1.0. Gen. Tech. Rep. PNW-GTR-508. Portland, OR: U.S. Department of Agriculture, Forest Service Pacific Northwest Research Station. 95 p.

Bachelet, D.; Lenihan, J.M.; Neilson, R.P.; Drapek, R.J.; Kittel, T. 2005. Simulating the response of natural ecosystems and their fire regimes to climatic variability in Alaska. Canadian Journal of Forest Research. 35: 2244–2257.

Bachelet, D.; Neilson, R.P.; Hickler, T.; Drapek, R.J.; Lenihan, J.M.; Sykes, M.T.; Smith, B.; Sitch, S.; Thonicke, K. 2003. Simulating past and future dynamics of natural ecosystems in the United States. Global Biogeochemical Cycles. 17: 1045. doi:10.1029/2001GB001508.

Bachelet, D.; Neilson, R.P.; Lenihan, J.M.; Drapek, R.J. 2001. Climate change effects on vegetation distribution and carbon budget in the U.S. Ecosystems. 4: 164–185.

Bolte, J.P.; Hulse, D.W.; Gregory, S.V.; Smith, C. 2006. Modeling biocomplexity—actors,landscapes and alternative futures. Environmental Modelling and Software. 22: 570–579.

Brown, T.J.; Hall, B.L.; Westerling, A.L. 2004. The impact of twenty-first century climate change on wildland fire danger in the western United States: An applications perspective. Climatic Change. 62: 365–388.

Bugmann, H. 2001. A review of forest gap models. Climatic Change. 51: 259–305.

Chiarucci, A.; Araújo, M.B.; Decocq, G.; Beierkuhnlein, C.; Fernández-Palacios, J.M. 2010. The concept of potential natural vegetation: an epitaph? Journal of Vegetation Science. 21: 1172–1178.

Costanza, J.K.; Earnhardt, T.; Terando, A.; McKerrow, A. 2010. Modeling vegetation dynamics and habitat availability in the southeastern U.S. using GAP data. In Maxwell, J.; Gergely, K.; Aycrigg, J.L., eds. Gap Analysis Bulletin No. 18. Moscow, ID: USGS/BRD/Gap Analysis Program.

Crookston, N.L.; Dixon, G.E. 2005. The forest vegetation simulator: a review of its structure, content, and applications. Computers and Electronics in Agriculture. 49: 60–80.

Daniel, C.J.; Frid, L. 2012. Predicting landscape vegetation dynamics using state-and-transition simulation models. In: Kerns, B.K.; Shlisky, A.J., Daniel, C.J., tech. eds. Proceedings of the First Landscape State-and-Transition Simulation Modeling Conference, June 14–16, 2011, Portland, Oregon. Gen. Tech. Rep. 869, Portland, OR: U.S. Department of Agriculture, Forest Service, Pacific Northwest Research Station: 5–22.

Evers, L.; Miller, R.F.; Hemstrom, M.; Merzenich, J.; Neilson, R.; Doescher, P. 2011. Estimating historical sage-grouse habitat abundance using a state-and-transition model. Natural Resources and Environmental Issues. 17: 16.

Guzy, M.R.; Smith, C.L.; Bolte, J.P.; Hulse, D.W.; Gregory, S.V. 2008. Policy research using agent-based modeling to assess future impacts of urban expansion into farmlands and forests. Ecology and Society. 13: 37.

Hemstrom, M.A.; Halofsky, J.E.; Conklin, D.R., Halofsky, J.S., Bachelet, D; Kerns, B.K. In press. Developing climate-informed state and transition models: In: Halofsky, J.E.; Creutzburg, M.; Hemstrom, M.A., tech. eds. Mid-scale landscape assessment: Methods used in the Integrated Landscape Assessment Project. PNW-GTR-XXX, Portland, OR: U.S. Department of Agriculture, Forest Service, Pacific Northwest Research Station.

Iverson, L.R.; Prasad, A.M.; Matthews, S.N.; Peters, M. 2008. Estimating potential habitat for 134 eastern US tree species under six climate scenarios. Forest Ecology and Management. 254: 390–406.

Keane, R.E.; Cary, G.J.; Davies, I.D.; Flannigan, M.D.; Gardner, R.H.; Lavorel, S.; Lenihan, J.M.; Li, C.; and Rupp, T.S. 2004. A classification of landscape fire succession models: spatial simulations of fire and vegetation dynamics. Ecological Modelling. 179: 3–27.

Kerns, B.K.; Naylor, B.J.; Buonopane, M.; Parks, C.G.; Rogers, B. 2009. Modeling tamarisk (*Tamarix* spp.) habitat and climate change effects in the Northwestern United States. Invasive Plant Science and Management. 2: 200–215.

Kurz, W.A.; Beukema, S.J.; Klenner, W.; Greenough, J.A.; Robinson, D.C.E.; Sharpe, A.D.; Webb, T.M. 2000. TELSA: the tool for exploratory landscape scenario analyses. Computers and Electronics in Agriculture. 27: 227–242.

Lenihan, J.; Bachelet, D.; Neilson, R.; Drapek, R. 2008. Response of vegetation distribution ecosystem productivity, and fire to climate change scenarios for California. Climatic Change. 87: 215–230.

Littell, J.S.; McKenzie, D.; Kerns, B.K.; Cushman, S.; Shaw, C.G. 2011. Managing uncertainty in climate-driven ecological models to inform adaptation to climate change. Ecosphere. 2: art102. http://dx.doi.org/10.1890/ES11-00114.1.

Long, J.L.; Miller, M.; Menakis, J.P.; Keane, R.E. 2006. Chapter 6—Developing the LANDFIRE Vegetation and Biophysical Settings Map Unit Classifications for the LANDFIRE Prototype Project. In: Rollins, Matthew G.; Frame, Christine K., tech. eds. 2006. The LANDFIRE Prototype Project: nationally consistent and locally relevant geospatial data for wildland fire management Gen. Tech. Rep. RMRS-GTR-175. Fort Collins, CO: U.S. Department of Agriculture, Forest Service, Rocky Mountain Research Station. 123–179 p.

Mote, P.W.; Salathe´, E.P. 2010. Future climate in the Pacific Northwest. Climate Change. 102: 29–50.

Nakicenovic, N.; Alcamo, J.; Davis, G.; de Vries, B.; Fenhann, J.; Gaffin, S.; Gregory, K.; Grubler, A.; Jung, T.Y.; Kram, T.; La Rovere, E.L.; Michaelis, L.; Mori, S.; Morita, T.; Nowak, R.S.; Ellsworth, D.S.; Smith, S.D. 2000. Special report on emissions scenarios: a special report of Working Group III of the Intergovernmental Panel on Climate Change. Cambridge, United Kingdom: Cambridge University Press. 599 p.

Nowak, R.S.; Ellsworth, D.S.; Smith, S.D. 2004. Functional responses of plants to elevated atmospheric CO_2– do photosynthetic and productivity data from FACE experiments support early predictions? New Phytologist. 162: 253–280.

Parry, M.L.; Canziani, O.F.; Palutikof, J.P.; van der Linden, P.J.; Hanson, C.E., eds. 2007. Contribution of Working Group II to the Fourth Assessment Report of the Intergovernmental Panel on Climate Change. . Cambridge, United Kingdom: Cambridge University Press.

Pennisi, E. 2009. Western U.S. forests suffer death by degrees. Science 323: 447.

Pfeifer-Meister, L.; Cole, E.M.; Roy, B.A.; Bridgham, S.D. 2008. Abiotic constraints on the competitive ability of exotic and native grasses in a Pacific Northwest prairie. Oecologia. 155: 357–366.

Provencher, L.; Anderson, T. 2011. Climate change revisions to Nevada's Wildlife Action Plan: vegetation mapping and modeling report to the Nevada Department of Wildlife. Reno, NV: The Nature Conservancy. 254 pages + electronic appendices.

Provencher, L.; Low, G.; Abele, S.L. 2009. Bodie Hills conservation action planning: final report to the Bureau of Land Management Bishop field office. The Nature Conservancy, Reno, NV.

Raffa, K.; Aukema, B.H.; Bentz, B.J.; Carroll, A.L.; Hicke, J.A.; Turner, M.G.; Romme, W.H. 2008. Cross-scale drivers of natural disturbances prone to anthropogenic amplification: the dynamics of bark beetle eruptions. BioScience. 58: 501–517.

Rehfeldt, G.E.; Crookston, N.L.; Warwell, M.V.; Evans, J.S. 2006. Empirical analyses of plant-climate relationships for the western United States. International Journal of Plant Science. 167: 1123–1150.

Rehfeldt, G.E.; Ferguson, D.E.; Crookston, N.L. 2008. Quantifying the abundance of co-occuring conifers along inland northwest (USA) climate gradients. Ecology. 89: 2127–2139.

Rogers, B.M.; Neilson, R.P.; Drapek, R.; Lenihan, J.M.; Wells, J.R.; Bachelet, D.; Law, B.E. 2011. Impacts of climate change on fire regimes and carbon stocks of the U.S. Pacific Northwest. Journal of Geophysical Research. 116: G03037. doi:10.1029/2011JG001695.

Taylor, A.H.; Beaty, R.M. 2005. Climatic influences on fire regimes in the northern Sierra Nevada mountains, Lake Tahoe Basin, Nevada, USA. Journal of Biogeography. 32: 425–438.

Westerling, A.L.; Hidalgo, H.G.; Cayan, D.R.; Swetnam, T.W. 2006. Warming and earlier spring increase western U.S. forest wildfire activity. Science. 313: 940–943.

Yospin, G. 2012. Historic and simulated vegetation dynamics in former Oregon white oak savanna, southern Willamette Valley, Oregon, USA. Eugene, OR: University of Oregon. 128 p. Ph.D. dissertation.

Modeling the Dynamic Responses of Riparian Vegetation and Salmon Habitat in the Oregon Coast Range With State and Transition Models

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Abstract

Interactions between landuse and ecosystem change are complex, especially in riparian zones. To date, few models are available to project the influence of alternative land-use practices, natural disturbance and plant succession on the likely future conditions of riparian zones and aquatic habitats across large spatial extents. A state and transition approach was used to model the effects of various management and restoration practices on conditions of riparian forests, channel morphology, and salmonid habitat. We present results of model analyses for the Wilson River in the Oregon Coast Range. We focus on critical habitat for spawning and rearing salmon and how habitat quality might be influenced by alternative land-use practices over the next 50 years, especially contrasting the outcomes of passive vs. active habitat restoration strategies. Results of our simulations suggest that active restoration of large wood in streams may accelerate habitat improvement relative to recovery projections under a passive restoration strategy. Active restoration seems to be a more viable approach for species such as coho salmon in the Wilson River watershed, which has limited potential spatial distribution in the drainage network, and where a significant proportion of the available habitat is in poor condition. In contrast, using active restoration techniques to improve habitat for a widely distributed species such as steelhead seems less feasible. Steelhead habitat is abundant throughout the basin and at least some of it is currently in good or excellent condition. Thus, large portions of the Wilson River would need to be restored to substantially increase the proportion of the stream network that is in good or excellent condition for steelhead.

Very little data are available with which to validate models at this scale. Results of our model simulations appeared reasonable wherever field and lidar data were available for comparison, however we caution that these comparisons do not validate all the factors simulated in our models. Consequently, the results of the model simulations should be interpreted as hypotheses of likely outcomes from management strategies at the scale of a large watershed (one or several 5th-field hydrologic units or HUC5s) or a large portion of a USFS ranger district. Nevertheless, the approach holds promise for simulating physical and biological responses of aquatic organisms and their habitats to alternative restoration approaches.

Keywords: state and transition models, riparian management, stream habitat, Oregon Coast Range, Tillamook burn, large wood addition, coho, steelhead, lidar.

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Introduction

Pacific salmon and steelhead have declined in abundance or have been eliminated from large parts of their historical range (Nehlsen et al. 1991), and many populations are now listed under the U.S. Endangered Species Act (USDA and USDI 2000). Multiple factors have contributed to these declines, including the degradation of spawning and rearing habitat in tributary streams (Federal Caucus 2000). Maintenance of existing high-quality habitat and restoration of degraded habitat have become cornerstones of many salmon recovery efforts (NRC 2002). However, there is a critical need for information relating how management activities in riparian areas will interact with natural processes to create and maintain salmon habitat and how this habitat will change over time—particularly over the broad spatial extents that are relevant to recovering salmonid populations. Landscape-scale perspectives of historical, current, and potential future conditions of upland, riparian, and aquatic systems resulting from plant succession, natural disturbances, and land-use practices can help inform policy directions. Further, such information is essential to developing strategic restoration policies for large regions (e.g., Oregon Plan for Salmon and Watersheds; http://www.oregon-plan.org/ accessed 15 November 2011) that make the most of limited funds. However, providing relatively detailed information on current and future riparian conditions for large watersheds (e.g., one or several 5th-level hydrologic units sometimes referred to as HUC5 or HUC6 watersheds; USGS and USDA, 2009) over large areas poses substantial logistical and technical challenges.

Mapping and classifying riparian zones using a combination of remotely sensed and field-collected data offers a means of assessing the current condition of riparian areas at fine detail over large areas. However, these assessments only provide a static "snapshot" of a watershed at a single point in time. The objective of this project was to develop state and transition models that could use this "snapshot" as a starting point and then project changes in salmonid habitat resulting from ecological processes that shape streams, riparian vegetation, and the upland systems to which they are connected.

The state and transition models developed here were designed to forecast changes at the watershed scale (one to several 5th-field HUCs) resulting from plant succession, hydrogeomorphic processes, and natural disturbances. Simulations using a background natural disturbance regime were used to "hindcast" the historical condition and to forecast the outcomes of "passive restoration", i.e., the strategy of allowing natural ecosystem processes to dictate the pace and trajectory of habitat recovery without human intervention. The models also accommodate land use activities such as logging, stream and riparian restoration activities (e.g., large wood additions to streams or planting of conifers in riparian areas to facilitate conversion from hardwoods to conifer dominated stands), and episodic disturbance events (e.g., debris flows, windthrow, and wildfires) that shape stream channels and valley floors. The models were used to forecast the outcome of alternative management policies, specifically contrasting a passive restoration strategy to an active strategy of restoring large wood to forested stream reaches where large, in-stream wood is currently lacking.

Methods

The aquatic-riparian state and transition models described in this paper were designed to simulate the temporal dynamics of riparian vegetation in the mountain stream networks of the Oregon Coast Range. To apply these models, the stream network must be delineated into relatively homogeneous stream reaches classified into "potential geomorphic types." The riparian zone around each reach must then be mapped and assigned to a "potential vegetation type." Each reach polygon must also be attributed with the current vegetation type to provide a starting point for the model simulation. These steps are described, below. Additional information on the methods we used, along with the aquatic-riparian state and transition models and supporting data are available from the project website (accessed 7 November, 2011): http://www.fs.fed.us/pnw/lwm/aem/projects/ar_models.html.

Stream Network Delineation and Classification

The stream network was delineated with the NetStream

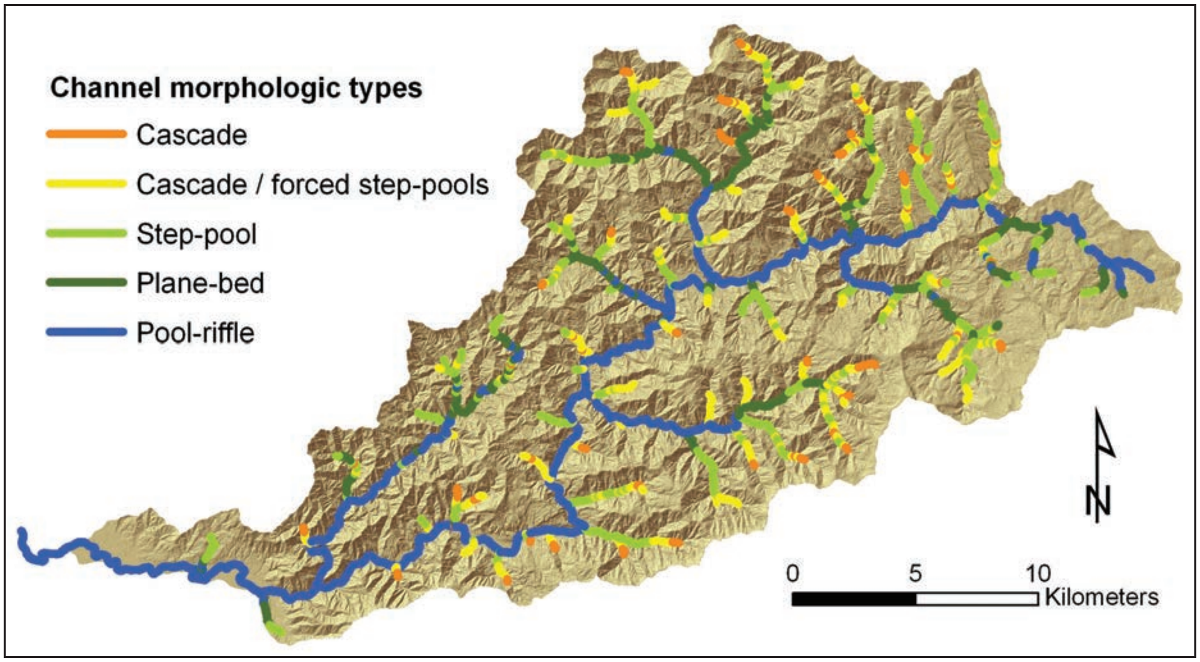

Figure 1—Classification of stream reaches into potential geomorphic types following the stream classification system of Montgomery and Buffington (1997 and 1998).

tool available from Earth Systems Institute[3] and hydraulic geometry coefficients for Pacific maritime mountain streams (Castro and Jackson 2001). A 5-m Lidar DEM and literature derived threshold values for drainage area and channel gradient were used to trace the stream network and provide a preliminary classification of channel reaches into six geomorphic types described by Montgomery and Buffington (1997, 1998): (1) colluvial, (2) cascade, (3) cascade with wood-forced step-pools, (4) step-pool, (5) plane-bed, and (6) pool-riffle (fig. 1). We dropped non-fish-bearing stream reaches from our analysis (i.e., bedrock and colluvial reaches and those with channel gradient > 20 percent or drainage areas < 1.5 km^2). We compared our preliminary classification with field-classified geomorphic types from 29 sampled reaches (described below). The final classification correctly assigned 86 percent of the 29 sampled reaches in the Wilson River watershed into their respective Montgomery and Buffington potential geomorphic types. The final stream network was generated with field-derived coefficients for bankfull width and depth and the boundary of the riparian zone around each classified stream reach was then delineated using path-distance thresholds generated from ArcGIS path-distance tools. This method identifies a geomorphically delineated riparian zone based on a "cost threshold" evaluated from a combination of distance and elevation from the active channel. We verified riparian zone delineations by comparing the path-distance derived boundaries to boundaries mapped at our 29 field sites based on distance, elevation, and slope breaks to adjacent hillslopes.

The geomorphically defined riparian zone boundary is likely to underestimate the influence of stream-adjacent trees in areas where steep hillslopes bound narrow valley floors. In these locations, trees growing on lower hillslopes may significantly affect riparian conditions in the stream. Thus, the limits of the riparian polygons were defined in two ways: the actual valley-floor delineation was done using the path-distance methods described above and a 30-m

[3] These tools are available from the NETMAP website: http://www.netmaptools.org.
Note: The use of trade or firm names in this publication is for reader information and does not imply endorsement by the U.S. Department of Agriculture of any product or service.

Table 1—Cross-walk between upland potential vegetation types (PVTs), plant association groups (PAGs), and the final riparian PVTs used for our model development within the Wilson River watershed

Riparian PVT	PAG description	PAG code	Upland PVT description
not modeled	Sitka spruce/ wet nonforest	991	nonforest
not modeled	W. hemlock/dry non-forest	1971	nonforest
not modeled	W. hemlock/wet non-forest	1991	nonforest
Sitka spruce/W. hemlock wet	Sitka spruce/oxalis-swordfern-moist	902	Sitka spruce (coastal)
Sitka spruce/W. hemlock wet	Sitka spruce/salal-mesic	901	Sitka spruce (coastal)
Sitka spruce/W. hemlock wet	Sitka spruce/salmonberry-wet	903	Sitka spruce (coastal)
Sitka spruce/W. hemlock wet	W. hemlock/oxalis-swordfern-moist	1907	W. hemlock moist (cascades)
Sitka spruce/W. hemlock wet	W. hemlock/salmonberry-wet	1908	W. hemlock wet (coastal)
W. hemlock intermediate	Pacific silver fir/Alaska huckleberry-wet	2207	silver fir intermediate (high)
W. hemlock intermediate	Pacific silver fir/oxalis-high precipitation	2208	silver fir intermediate (high)
W. hemlock intermediate	W. hemlock/Alaska huckleberry/oxalis	1909	W. hemlock cool (cascades)
W. hemlock intermediate	W. hemlock/Oregon grape-salal	1906	W. hemlock intermediate (cascades)
W. hemlock intermediate	W. hemlock/vanilla leaf-cool	1905	W. hemlock cool (cascades)
W. hemlock intermediate	W. hemlock-warm, transitional to Douglas fir	1903	W. hemlock hyperdry (SW)

buffer was also drawn on both sides of the stream. The final “riparian” polygon used for vegetation classification was defined as the larger of the two methods.

Potential Riparian Vegetation

We based our classification of riparian potential vegetation types (PVT) on the adjacent plant association groups (PAGs obtained from the Northwest Oregon Ecology Group, Corvallis, Oregon) and upland PVTs from the Siuslaw National Forest, Oregon (table 1) because potential vegetation type spatial data with previously classified riparian vegetation were not available. The dominant PVTs in the Wilson River watershed are Sitka spruce/western hemlock wet PVT and Douglas fir/western hemlock PVT. We added a meadow riparian PVT to characterize riparian areas in the northeast portion of the Wilson watershed where deep-seated landslides created low gradient valleys with high water tables, and present-day flooding by beaver dams tends to prevent forested vegetation types. The meadow riparian PVT is characterized by PAGs in the Douglas-fir/western hemlock PVT as well as alder, willow and sedge functional groups. Combining the 3 PVTs with the potential geomorphic types (figure 1) resulted in a suite of 11 separate state and transition models for the Wilson River watershed (table 2).

Field Sampling of Selected Reaches

Sampling was designed to characterize three aspects of selected study reaches: (1) channel, steambank, and valley floor geomorphic conditions of the entire reach; (2) vegetation zones within the reach; (3) vegetation composition and structure in quantitative sub-plots. We randomly selected a stratified sample of 30 stream reaches from the preliminary delineation and classification. We generated fine-scale hillshade and canopy height maps for each reach and field sampling protocols were designed around these maps. We sampled 29 of these reaches during the summer of 2009 to provide data from which current conditions could be classified and mapped for the entire stream network within the study watershed.

Channel and valley floor geomorphic conditions— At each reach, the channel morphology was classified into channel types following the Montgomery and Buffington (1997, 1998) classification. Similarly, the overall plant association group and potential vegetation type was determined and the current vegetation structure was classified following state-classes as defined in the state and transition models (described below). Current land use, evidence of herbivory, and the overall condition of the channel were also noted. The longitudinal gradient was measured using an auto-level

Table 2—List of the 11 aquatic-riparian state and transition models developed for the Wilson River watershed.

Model	Potential vegetation type	Channel type	Length (km)	Riparian area (ha)	Percent total length	Percent total riparian area
sx_cscd	Sitka spruce	Cascade	4.1	27.4	2	2
sx_cscd_sp	Sitka spruce	Cascade / w-f step-pools	28.1	174.1	12	13
sx_sp	Sitka spruce	Step-pool	43.0	262.1	19	19
sx_pb	Sitka spruce	Plane-bed	26.8	162.6	12	12
sx_pr	Sitka spruce	Pool-riffle	35.1	212.9	16	15
me_pr	Meadow	Pool-riffle	11.7	70.4	5	5
df_cscd	Western hemlock	Cascade	6.2	40.2	3	3
df_cscd_sp	Western hemlock	Cascade / w-f step-pools	18.7	114.2	8	8
df_sp	Western hemlock	Step-pool	31.4	192.4	14	14
df_pb	Western hemlock	Plane-bed	12.4	75.2	6	5
df_pr	Western hemlock	Pool-riffle	7.4	44.4	3	3
		Grand total	224.9	1,375.9	100.0	99.0

Note: "w-f" in "Cascade / w-f step pools" denotes "wood-forced" – steep channels where steps and pools are formed where large wood obstructs the channel and that, lacking wood, would otherwise have a cascade morphology.

and stadia rod. Measurements were taken over a 100-m long length of channel wherever possible although dense vegetation sometimes limited the survey length.

The limits of both the active valley floor and the total valley floor were delineated on the hillshade and canopy height maps. We defined the active valley floor as the area with flood return intervals of 2 to 5 years, and determined the limits of this zone based on evidence of scouring, sediment deposition, and vegetative change outside of the bankfull width of the channel. We defined the total valley width as the limit of valley floor inundation which might occur in a 100-year return interval flood. Practically, the total valley floor width was determined by distance from, and height above, the bankfull channel and often coincided with obvious slope breaks with adjacent hillslopes or terraces.

Within each reach, we established five cross-channel transects which were marked on the hillshade and canopy height maps. The first transect was located randomly, using a random number generator, and located within the first 20 percent of the reach length. The remaining transects were spaced at equal intervals along the remaining 80 percent of the reach length. All pools, pool-structure, large wood, and over-hanging vegetation was inventoried between transects 1 and 5. The length, type, forming agent, total depth, tail-out depth (residual depth by difference) was recorded for each pool. All pieces of large wood were also inventoried. Lengths and diameters of each piece of large wood, in or suspended above, the bankfull channel, were estimated and a subset was measured. Also, we recorded percent of total length of large wood within the bankfull channel and the proportion that would be in the water at bankfull flows.

We measured the bankfull width, right and left bank elevation, thalweg depth, and two additional bed depths to provide a coarse cross-sectional profile at each cross-channel transect. Bankfull depth was calculated as the difference between the bankfull elevation and the thalweg depth. The elevation measurements were made using a stadia rod and inclinometer and provided reasonably accurate measurements (a system we field tested at the beginning of the field season and provided repeatable elevation measurements accurate to within a few centimeters). Bank characteristics were recorded for both the left and right banks, for a 2-m wide swath centered on the transect tape (1-m upstream and 1-m downstream). Within this zone, the stream bank was ranked either as stable or unstable. The percent length of undercut bank and average undercut depth was also recorded as was the ground cover on the bank and the percent of the swath with overhanging vegetation. Stream shade was evaluated at the center of the channel by estimating the percent of sky obscured by vegetation within a 20-cm diameter ring, held at arm length, 60° above a level horizon facing due south (the approximate location of the sun at solar noon on the solstice at the latitude of the study sites).

Reach and sub-reach summaries of vegetative conditions—

While in the field, discrete, homogenous vegetation zones within each study reach were delineated and mapped based on composition, size and structure of vegetation. We distinguished the following vegetation strata within each zone: (1) upper or overstory canopy layer; (2) secondary canopy layer; (3) sapling and tall-shrub layer (2 to 6-m height); (4) short shrub layer (0.5 to 2-m height); (5) herbaceous layer (<0.5 m); and (6) exposed ground. The total canopy cover and the relative cover by species were recorded for each layer. For tree layers the mean diameter breast height (DBH) was recorded from a subsample of trees present; for the shrub layers, average shrub heights were recorded. Herbaceous cover was characterized by functional groups (grasses, sedges, rushes, forbs, and ferns) rather than species. Ground cover was the percent of area not covered by herbaceous vegetation (soil, litter, rock/boulder, wood, water). Herbaceous and ground cover sum to 100 percent.

Quantitative sub-plots of vegetation composition and structure—

Eight to ten quantitative sub-plots were located in each study reach and data from these were used to develop lidar classification techniques for vegetation mapping. To allocate field plots, lidar canopy height rasters were segmented using eCognition software and classified into height classes. Field plot locations were then randomly selected based on height classes. These locations were then mapped onto the hillshade and canopy height maps for use in the field. Plots were located in the field using handheld GPS units and the detailed maps and then sampled to characterize the structure and composition of the vegetation within each sub-plot. The actual number of plots sampled was limited by the time available to sample each reach, but at least 5 of the 10 sub-plots were always sampled. We used sub-plots of 3 sizes: 5-m radius for tree plots; 2-m radius for shrub plots; 1-m radius for herbaceous plots. On tree plots, we recorded the species and DBH of all trees with DBH greater than 12.5 cm. We measured tree heights of at least 3 trees within each of three canopy layers (when present): (1) the upper or overstory canopy layer; (2) the secondary canopy layer; (3) the sapling layer. For trees between 2.54 and 12.5 cm DBH, all stems were counted and the mean height of the trees was estimated. For shrub plots, the mean height, canopy cover and species composition was recorded. For herbaceous plots, the height, cover, and composition were recorded. Dominant species were identified to species if possible.

Mapping Current Vegetation

To derive detailed information on current vegetation composition and structure for each riparian polygon we followed a two-step approach. First, using discrete-return lidar data (>8 pulses m^{-2}), a high-resolution raster map was created with a spatial resolution of 5 m. The map classification consists of 12 classes: 5 non-tree classes (water, barren, herbaceous vegetation, shrubs) and 7 tree classes distinguishing between conifer and hardwood, and tree sizes. For the tree classes an additional sub-class was derived separating stands with and without understory. In the second step, we calculated the area proportions of each (raster) class within each riparian polygon, and then labeled each polygon using a defined classification logic. Here, we briefly describe the classification methods and results. A more detailed description lies outside the scope of this paper which is focused on using state-and-transition models to simulate the dynamics of riparian vegetation.

The vegetation raster map was classified using measurements from the quantitative vegetation sub-plots for model training and validation. For each field plot, we extracted lidar point clouds and calculated 41 potential predictor variables from the lidar height and intensity distributions similar to Hudak et al. (2008). For estimation of continuous variables such as tree DBH we used multiple linear regression and for classification of categorical map attributes we used RandomForest (Breimann 2001). First, we used a canopy height model (fig. 2) and a height threshold of 0.5 m to separate woody vegetation (shrubs and trees) from herbaceous vegetation, barren, and water. Then we classified herbaceous vegetation, barren surfaces, and water

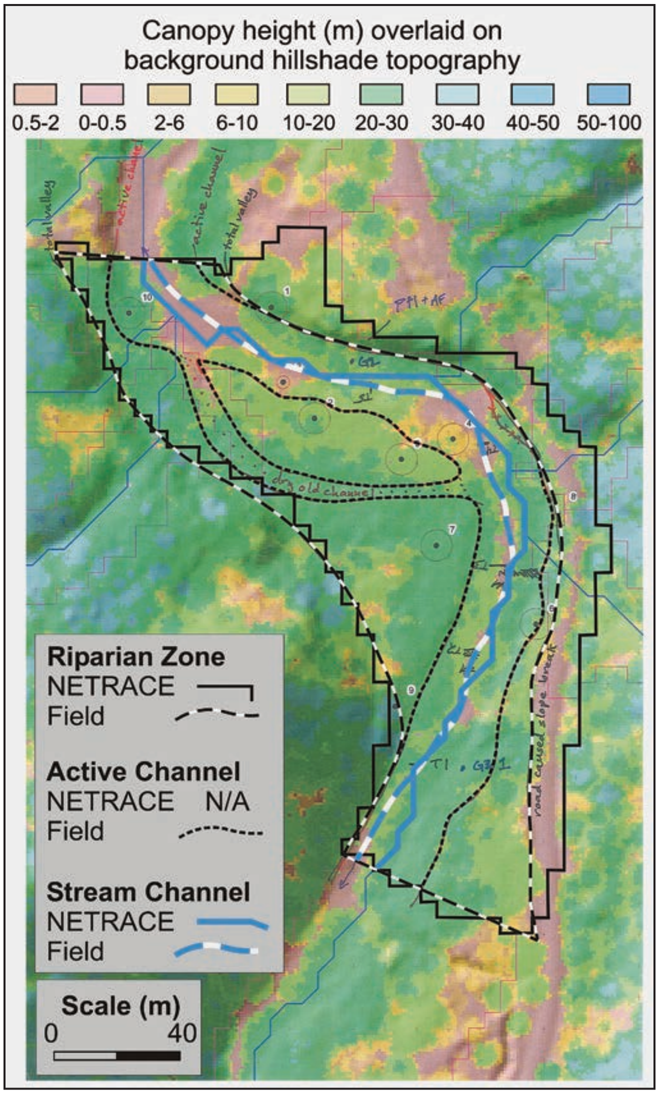

Figure 2—Example of a lidar-derived canopy height map overlaid on a hillslope-shaded topographic map prepared for field sampling a selected reach on the South Fork Wilson River. Lines contrast preliminary NetStream delineations (solid) with field-drawn delineations (white-dashed). Note, the right boundary of the riparian zone is caused by a road grade.

using lidar height and intensity metrics. However, low-stature shrubs (< 0.5 m) could not be reliably distinguished from herbaceous vegetation.

We then used the lidar-point data to estimate tree height and compared those estimates to the field-measured tree heights which showed a strong correlation for both conifers (r = 0.91) and hardwoods (r = 0.85). Because structural states in the state-and-transition models are based on tree diameters rather than height, tree height and canopy cover data derived directly from lidar were converted to tree diameter at breast height, using the following equations for hard woods and conifers:

$$dbh_H = \exp(0.537+0.147 * \mathrm{H95PCT}_{log} + 0.809 * \mathrm{HMEDIAN}_{log} + 0.003 * CANCOV)$$

$$dbh_c = \exp(0.2127 + 1.1045 * \mathrm{H95PCT}_{log})$$

Where dbhH and dbhC is maximum plot-level dbh for hardwoods and conifers, respectively, H95PCT is the 95th percentile of lidar vegetation heights, HMEDIAN, median lidar height and CANCOV is lidar canopy cover (number of returns above 2m divided by total number of returns). Model accuracy was acceptable (RMSE = 30-36 percent), but generally lower than observed in studies of more uniform forest stands. When converted to categorical size classes, overall classification accuracy was 64 percent (63 percent for hardwoods and 64 percent for conifers). For conifer trees, confusion (omission/commission error) between medium and large tree classes was roughly 40 percent, but balanced. For hardwood plots, there was a tendency of large trees to be mapped as medium tree class. Further, tall shrubs were misclassified as young hardwood trees. Thus, we combined the two classes into a single class.

A classification of hardwood and conifer forests was derived from lidar metrics using a combination of field and photo-interpretation plots. Analysis of the field data alone showed high omission errors for conifers, partly resulting from an unbalanced data set dominated by hardwood plots. Thus, to increase sample size, we collected additional reference data by means of photo-interpretation of near-infrared airphotos from the National Agriculture Imagery Program (NAIP). We generated 150 random plots (5 m diameter), 75 within and 75 outside the riparian zone. The best RandomForest model showed an overall accuracy of 85 percent (out-of-bag, boot-strapping estimate). Classification of tall understory shrubs showed an overall accuracy of 67 percent.

To estimate canopy density, we used lidar-derived estimates of canopy cover (vegetation returns above 2 m divided by all returns) without any transformation. We were unable to derive an acceptable relationship between tree cover estimates from the quantitative sub-plots and the lidar-derived canopy cover. However, at the polygon level the agreement between simple (uncalibrated) lidar cover and field-based estimates was high (80 percent).

To derive the current vegetation layer for the state-and-transition modeling, the vegetation raster map was summarized to describe the vegetation within each riparian polygon. Polygons were classified into bare (<=15 percent herbaceous) or herbaceous (>15 percent) when the proportion of shrub pixels within the valley floor polygon was less than 5 percent, or open (5–15 percent), medium (15–40 percent), and dense shrub (40–100 percent) otherwise. To account for a wider influence zone for trees we summarized forest structure attributes using a buffer zone of 30 m for stream orders 1–4, and a 30-m buffer in addition to the valley floor boundary for stream orders greater than 5. Here, we distinguished between nonforest (<15 percent), open forest (>15 percent) and dense forest (>40 percent). Forest polygons were classified as mixed conifer/hardwood when neither of the two forest types made up greater than 65 percent of the forest area. Further, we defined multi-layer stands when more than one tree size class occupied greater than 10 percent of the area. Canopy density was derived from lidar estimates of canopy cover (the number lidar returns above 2 m divided by the total number of returns) without any transformation.

The classification description for each riparian polygon was compared to the classification made during the field visit (table 3). The attributes of the overstory tree canopy were classified correctly approximately two-thirds of the time. However, the lidar-based methods had difficulty classifying understory attributes. The density of the understory shrub canopy agreed with the field observations in only 38 percent of the cases. Similarly, the agreement in the number of canopy layers was only 48 percent at the polygon level. In eight of the 10 erroneous classifications, the lidar failed to detect a multistory canopy observed in the field.

It is quite difficult to assess the overall classification accuracy. The lidar-based classification of the overstory tree types were quite accurate for homogeneous forest types (either conifer or hardwood) as evidenced by the 85 percent accuracy for the quantitative sub-plots. Summarizing the 5-m raster map to describe the heterogeneous riparian polygons was more challenging. For example, in only 3 of the 29 sampled plots (10 percent) did the lidar classification of all 5 attributes completely agree with the field classification. In only 8 of 29 plots (41 percent) did all 4 canopy attributes (excluding shrub canopy density) agree between the lidar and field classifications (table 3).

While the overall classification accuracy appears quite low, it does not distinguish between classification errors among ecologically similar classes (e.g., medium hardwood and large hardwood) and distinctly different classes (herbaceous versus forest). Overall, misclassified polygons tended to be classified into relatively similar classes (table 3). For example, in the hardwood plots, the regression equations relating height and diameter resulted in a tendency to map large diameter alders as medium diameter. While the lidar data were quite accurate in assessing canopy height, converting height to diameter classes added additional uncertainty caused by large variations in tree physiognomy. Similarly, the field sampling included 4 polygons with mixed conifer-hardwood overstory, of which, 3 were correctly classified by lidar. But the lidar also classified 6 hardwood or shrub dominated polygons as mixed. Thus, misclassified polygons were often classified into relatively similar classes.

The assessment of overall classification accuracy is further complicated because we do not have an accuracy assessment of our field classifications. Some attributes were easily observed in the field, but others were quite difficult to estimate. Estimating average canopy density for multiple canopy layers was especially problematic in large riparian polygons with heterogeneous vegetation. We used a 40-percent canopy cover threshold to distinguish between open and closed canopies for both overstory and understory tree layers, as well as the shrub layers. It is possible that the lidar-based classification, trained from the relatively homogeneous quantitative sub-plots, made more accurate classification than was possible in the field.

Large In-Stream Wood and Missing State Classes

To fit the current conditions to the VDDT state class structure (described below), reaches were also assigned to one of three in-stream wood classes: large wood (>20

Table 3—The classification accuracy for current vegetation in the 29 sampled riparian polygons based on a comparison of the lidar-based classification versus the classification made during field sampling

Reach ID	Lidar class[a]	Field class[a]	Type	Size	Canopy	Strata	Shrub	All attributes	Canopy attributes
1169	HMD2D	HMD2D	1	1	1	1	1	1	1
1371	XMD1O	HSO1O				1	1		
6872	XMO1D	HMD2D		1			1		
8808	XSO1D	HSD1O		1		1			
9280	XLO2D	S---O							
10202	XMO1O	XMD2D	1	1					
10444	XMD1O	XMD2D	1	1	1				1
10929	HMD1O	HMD2D	1	1	1				
11448	HMD1O	HMD1O	1	1	1	1	1	1	1
11749	XMD1O	CMD2D		1	1				1
11805	HLD2D	HMD1D	1		1		1		
14925	HMD1O	HMD1D	1	1	1	1			
14957	HLD2D	HMD2D	1		1	1	1		1
15588	HMD1O	HMD2D	1	1	1				1
15764	HMD1O	HMD1D	1	1	1	1			1
15816	HMD2D	HMD2D	1	1	1	1	1	1	1
15898	HMD1D	HMD1O	1	1	1	1			
16594	HMD1O	HMD1D	1	1	1	1			
17049	HMD1D	HMD2D	1	1	1		1		
17061	HLD2D	HMD1D	1		1		1		
17548	HMO1D	HSO2O	1		1				
17807	CLD2D	XMD2D			1	1	1		
18206	HMO2O	HMD2O	1	1		1	1		
18615	XMD1D	S---O							
19058	XSO1D	S---O							
19266	XLD2D	XMD2O	1		1	1			
19726	HMD1O	HMD1D	1	1	1	1			1
19843	HSO1O	S---O					1		
20152	CMO1D	S---D					1		
Number correct			19	17	19	14	13	3	8
Percent correct			66	59	66	48	45	10	28

[a] Classification codes are as follows for each character in the 5 character code:
First character: S=shrub; H=hardwood; C=conifer; X=mixed conifer and hardwood. Second character: S=small; M=medium; L=large.
Third character: O=open; D=dense.
Fourth character: 1=single-story; 2=multi-story.
Fifth character: O=open; D=dense.

in. dbh), small wood (≤20 in. dbh), or no wood. Unfortunately, we were unable to determine the abundance and size of wood present in each stream reach from lidar data. Therefore, reaches within each potential geomorphic type were assigned to each class in the proportion of each wood class determined from field sampling. Additionally, we assumed that local recruitment of in-stream wood would occur in large and giant tree state classes, and therefore, these structural states were assigned to the large wood state classes. Overall, this would tend to overestimate the amount of in-stream wood present in the stream network because little wood was present in many of the sampled reaches even where medium- and large-sized trees were growing in the riparian zone or on lower hillslopes. However, very little of the stream network currently falls into the large or giant state classes. These were seldom observed in the field and none of the sampled plots were field classified into large or giant tree size classes. Thus, errors in the classification

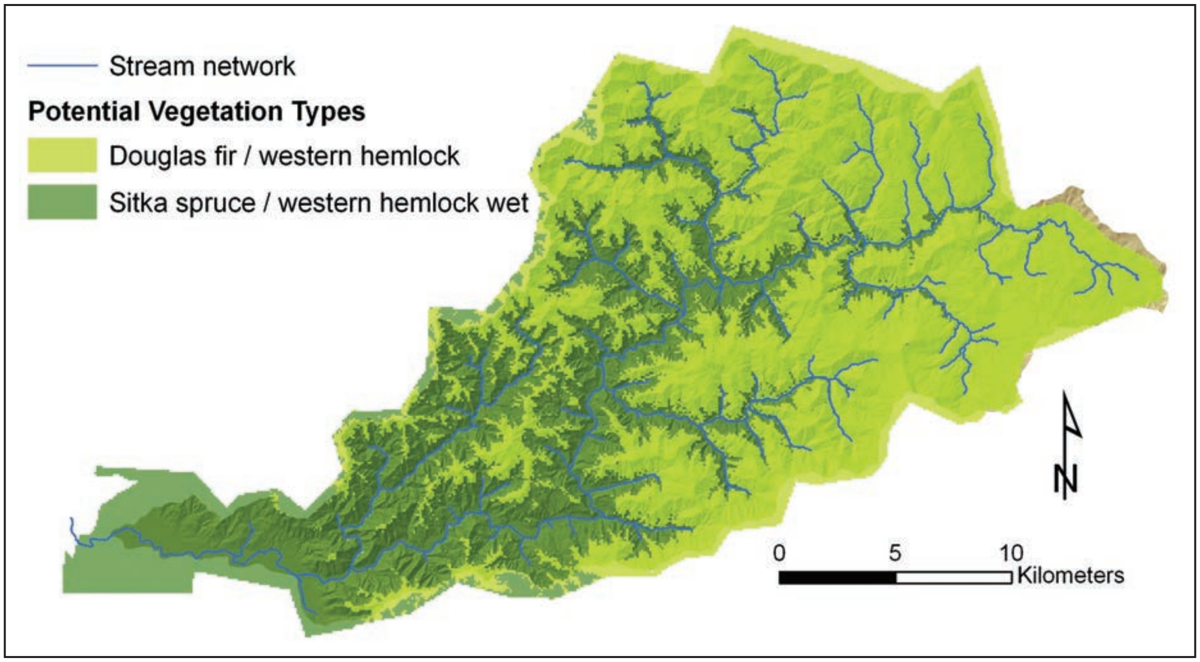

Figure 3—Classification of upland vegetation into Potential Vegetation Types as used in the state and transition models developed for the Wilson River watershed.

of the current conditions caused by assuming that large in-stream wood present in large and giant stands were likely quite small.

The lidar classifications generated a number of structural state classes that were not present in the models we developed. For example, we only simulated open canopy, single-storied stands and closed canopy, multi-storied stands for conifer dominated riparian polygons. Hardwood dominated stands were all simulated as closed canopy, single-storied stands and all mixed conifer and hardwood stands were simulated as closed, multi-storied structural states. Such simplifications are necessary to minimize the complexity of our models and are reasonable approximations of the potential forest structural state classes. The lidar methods we used to classify current conditions, however, were not restricted to just those structural states present in our models. Thus, the lidar classified 144 mixed conifer and hardwood polygons as either open-canopy or as single-storied. These polygons were reclassified into closed canopy multi-storied states. Similarly, the lidar classification generated 194 hardwood-dominated polygons classified either as open canopy or multi-storied all of which were reclassified as closed canopy single-storied states. In total, current conditions in 422 polygons out of the 1554 polygons (27 percent) needed to be reclassified into the most similar state class that occurred in the models.

Aquatic-Riparian State and Transition Models

We intersected the classified stream network map with the potential vegetation map of the Wilson River watershed to identify all possible combinations of channel and vegetation types (table 2). We excluded the coastal plain in the lower watershed from our analyses because the area had been drastically altered by agriculture and development. Models for the remaining area included five geomorphic types in three PVTs: (1) the Sitka spruce/western hemlock wet PVT in the lower parts of the watershed and low elevation riparian zones, (2) the Douglas-fir/western hemlock PVT in the upper parts of the watershed and high elevation riparian zones (fig. 3), and (3) the meadow PVT present only in the extreme eastern portion of the watershed (area of deep-seated landslides in Devils Lake Fork not shown in figure 3 because their spatial extent is too limited to be visible at the scale of the figure). Conceptually, the models do not limit the overstory canopy to a single dominant tree species. For example, in the Sitka spruce/western hemlock wet PVT, we recognize that either mixed species stands, or stands dominated by either Douglas-fir or western hemlock are likely to occur thus we abbreviate this PVT as "sx", denoting a Sitka spruce—miXed conifer overstory.

We built separate state and transition models for each of the 11 combinations of PVT and geomorphic channel types using the Vegetation Development Dynamics Tool (VDDT; Beukema et al. 2003; http://essa.com/tools/vddt/ accessed 15 November 2011). These VDDT models were developed from upland models for the northern Oregon Coast Range and the Washington Coast Range (please see the ILAP project web page: http://oregonstate.edu/inr/ilap/ as well as the following link for downloads of data, models, and documentation ftp://131.252.97.79/ILAP/Index.html). We expanded these models to include riparian shrub and hardwood state classes, hydrogeomorphic processes, and riparian restoration practices (table 4). We also expanded the pathways and transition probabilities to characterize the historic and current land use activities present in the watershed. The completed models simulated a large number of possible states of a stream reach, from recently disturbed states resulting from stand replacing disturbances such as logging, landslides or wildfire to states where stand replacing disturbances have not occurred for long periods of time.

Individual states within each model are defined on the basis of the potential vegetation type (PVT), the dominant overstory trees, the vegetation structure, and the presence and size of in-stream wood. Tree sizes are based on diameter breast height, in inches, as follows: young 0-1; small 1-10; medium 10-20; large 20-30; giant > 30. In-stream wood is recruited from forested states in two size classes. Small wood constitutes pieces with large-end diameters ≤ 20 inches; large wood is > 20 inches. Thus, in-stream wood recruited from medium-sized tree stands falls into the small-wood class whereas wood from large- or giant-sized tree stands falls into the large-wood state class. Nonforest state classes have tree canopy cover < 15 percent; open canopy state classes have tree canopy cover between 15 percent and 40 percent; closed canopy state classes have tree canopy cover > 40 percent. Open shrub states have shrub cover < 40 percent; closed shrub states have shrub cover > 40 percent. Shrub cover in Sitka spruce/western hemlock wet PVT and Douglas-fir/western hemlock PVT is dominated by *Rubus spectabilis* (salmonberry), while *Salix* spp characterize shrubs in the riparian meadow PVT. In forested states, we assume that conifer-dominated states have open shrub understories whereas alder-dominated or mixed conifer-alder stands have closed shrub understories.

Sitka spruce/western hemlock wet and Douglas-fir/western hemlock PVTs—
Nonforested early seral states used in the models are either barren from post-debris flow conditions or shrub states. Barren conditions are rapidly colonized by salmonberry, transitioning into an open shrub state after 3 years (table 4). Salmonberry grows rapidly and these open shrub states transition to dense shrub states after 2 more years. Wildfires transition directly to the open shrub states which then require 5 years before they reach the dense shrub state class. Forested states can only be initiated from barren or shrub states through tree regeneration. Alder is highly favored early seral tree species in these riparian models (table 4). Conifer regeneration is much lower than that of alder. Further, once salmonberry grows into a dense shrub layer, regeneration rates are reduced by 50 percent for alder and nearly a factor of 10 for conifers. Following conifer regeneration, successional development is simulated as a deterministic process with intermediate seral states defined on the basis of dominant tree sizes rather than age (see table 4). We only simulate open single-story and closed multi-story conifer stands where canopy growth in small, medium, large and giant tree sizes leads to canopy closure (table 4) which is assumed to be accompanied by regeneration of shade-tolerant tree species that form a secondary canopy layer.

Alder regeneration results in a successional sere dominated by alders (table 4). We use a probabilistic transition in year 20 of the small-alder state that forces 25 percent of these state classes into a mixed conifer-alder successional sere. We also simulate relatively slow rates of conifer regeneration in alder-dominated medium-sized tree states.

Table 4—Table of annual transition probabilities and probability multipliers used in the models. (continued)

	Nonforest vegetation			Conifer-dominated forest vegetation				
Vegetative structural state	**Barren**	**Open shrub**	**Dense shrub**	**Young tree**	**Small tree**	**Medium tree**	**Large tree**	**Giant tree**
Age (years)	0-3	0-5	6-200	0-10	11-35	36-65	66-110	111-200+
Base probabilistic transitions								
Tree regeneration (alder)	0.6	0.6	0.3					
Tree regeneration (conifer)	0.04	0.04	0.005					
Canopy growth (open to closed)					0.05	0.03	0.03	0.01
Mixed severity wildfire (open canopy)						0.0027	0.0019	0.0009
Mixed severity wildfire (closed canopy)				0.0045	0.0045	0.0018	0.0027	0.0036
Stand replacing wildfire (open canopy)						0.0023	0.0023	0.0009
Stand replacing wildfire (closed canopy)				0.0045	0.0045	0.0023	0.0023	0.0036
Small wood decay to no wood			20 years					
Large wood decay to small wood			100 years					
Windstorm[1]					$M_{Low,High}$	$M_{Low,High}$	$M_{Low,High}$	$M_{Low,High}$
Transitions controlled with multipliers								
Debris flow source channels		M_3	M_2	M_2	M_2	M_1	M_1	M_1
Debris flow receiving channels	$M_{No,S,L}$	$M_{No,S,L}$	$M_{No,S,L}$	$M_{No,S,L}$	$M_{No,S,L}$	$M_{No,S,L}$	$M_{No,S,L}$	$M_{No,S,L}$
Management controlled with multipliers								
Post-wildfire salvage logging	M	M						
Post-wildfire or logging conifer planting		M						
Pre-commercial thinning				M				
Partial harvest					M	M	M	M
Regeneration harvest					M	M	M	M
Large wood addition				M	M	M		

Table 4—(continued) Table of annual transition probabilities and probability multipliers used in the models.

		Nonforest vegetation		Alder-dominated forest vegetation				
Vegetative structural state	**Barren**	**Open shrub**	**Dense shrub**	**Young tree**	**Small tree**	**Medium tree**	**Large tree**	**Giant tree**
Age (years)				0-5	6-20	21-50		
Probabilistic transitions								
Tree regeneration (conifer)						0.0125		
Alternate succession to mixed conifer – alder stand				0.25 in YR 20				
Stand-replacing wildfire (closed canopy)				0.0045	0.0045	0.0045		
Windstorm					M_{High}	$M_{Low,High}$		
Transitions controlled with multipliers								
Debris flow source channels				M_2	M_2	M_2		
Debris flow receiving channels				$M_{No,S,}$	$M_{No,S,}$	$M_{No,S,L}$		
Management controlled with multipliers								
Hardwood conversion – Plant conifers				M	M			

		Non-forest vegetation		Conifer-alder mixed forest vegetation				
Vegetative structural state	**Barren**	**Open thrub**	**Dense thrub**	**Young tree**	**Small tree**	**Medium tree**	**Large tree**	**Giant tree**
Age (years)						21-65	66-110	110-200
Probabilistic transitions								
Stand-replacing wildfire (closed canopy)						0.0045	0.0045	0.0036
Windstorm						$M_{Low,High}$	$M_{Low,High}$	$M_{Low,High}$
Transitions Controlled with Multipliers								
Debris flow source channels						M_1	M_1	M_1
Debris flow receiving channels						$M_{No, S,L}$	$M_{No, S,L}$	$M_{No, S,L}$

[a] High severity windstorms only occur in the Sitka spruce / western hemlock, wet PVT. See text for details.

Note: the transition probabilities shown here are from the Douglas-fir / western hemlock PVT. The Sitka spruce PVT will use slightly different probabilities. Also, a number of transitions (especially natural disturbance processes where underlying rates are poorly documented and all management-based transitions) rely on a file of multiplier values (shown here as an "M") to determine the rates of these processes in the models. A multiplier of zero effectively turns off the process.

Mixed stands eventually become entirely conifer-dominated after 200 years.

Riparian meadow PVT—
This model characterizes areas of the northeastern portions of the watershed where beaver might be active and contribute to the development of wetlands and ponds. The main difference between this model and the two other PVTs are additional wet meadow/beaver pond states (MeW). The rest of the model (structure and probabilistic transitions) is very similar to the Douglas-fir/western hemlock PVT except that we assume that the non-MeW states are more mesic than structurally similar Douglas-fir/western hemlock states.

The MeW wet meadow/pond state persists when beaver are present and is characterized by grasses/sedges/forbs or riparian shrubs. Floods or beaver extirpation transition wet meadows to relatively drier conditions (MeD). Without reoccurrence of dam building by beaver, MeD states transition to forested stands of Douglas-fir/western hemlock or alder. As in the other PVTs, alder is an early seral tree species with much higher probability of regeneration than conifers. *Salix* spp. dominate in younger alder or conifer states while salmonberry is more abundant in older tree states.

Natural disturbances in all PVTs—
We simulate a variety of probabilistic disturbances (table 4). We simulate several stand replacing disturbances that cause state transitions to the open shrub state. These include stand replacing wildfire and clearcut logging. The overall annual wildfire probability in any forested state is 0.0045 which is equivalent to a 220-year fire return interval. In young- and small-sized tree stands, only stand-replacing wildfire occurs. In stands with larger trees, both mixed-severity and stand-replacing fires occur, and the actual probabilities vary with tree size and canopy closure. Mixed-severity fires transition closed-canopy state classes to open canopy states or maintain pre-existing open-canopy states. Mixed-severity fires also drive recruitment of wood into the streams—recruiting small wood from the medium-sized states and large wood from the large- and giant-sized states.

High severity wind storms are also simulated as stand replacing disturbances but only occur in the low-elevation, Sitka spruce/western hemlock wet PVT—thus preferentially affecting wet sites in low-elevation mountain valleys located nearest to the Pacific coast. We parameterized the models with high-severity windstorm probabilities equal to 0.0083 per year (for a return interval of 120 years). We assume the shrub layer will be minimally disturbed by high-severity windstorms so that the postdisturbance stand inherits an open or dense state class from the predisturbance state class.

State transitions resulting from low-severity wind storms are similar to those caused by mixed-severity fire in that these wind storms transition closed-canopy state classes to open canopy states or maintain preexisting open-canopy states. They also drive recruitment of small wood from the medium-sized states and large wood from the large- and giant-sized states. We parameterized the models with high-severity windstorm probabilities equal to 0.0100 per year (for a return interval of 100 years).

We only simulate decay and loss of large, in-stream wood in the dense shrub state class where we require 100 years for the large-wood state class to transition into a small-wood state, and 20 years for a small-wood state class to transition into a no-wood state class. Wood decay cannot be easily tracked in the other state classes because there are multiple transition pathways and disturbances that can recruit additional wood and the VDDT modeling software prevents tracking the age of specific attributes independently of the age of the underlying state class.

The occurrence of debris flows are controlled using a multiplier file. We start by assuming that the probability of landslide-caused debris flows is limited by the potential rate of hillslope hollow refilling which suggests that landslides might occur approximately 1 in 500 years from colluvial hillslope hollows. This would give an expected base rate of debris flows of 0.002 per year. There is substantial evidence, however, that debris flows are more likely to occur soon after stand-replacing disturbances. Thus, for channel types where debris flows are likely to originate (cascade and cascade with wood-forced step pools), we created three categories: Debris Flow1 (multiplier M_1 in table 4) occurs in mature and old-growth forest and accounts for 10 percent of all debris flows; Debris Flow2 (multiplier M_2 in table 4) occurs in dense shrub, young, and small forest state classes and accounts for 30 percent of all debris flows; Debris Flow3 (multiplier M_3 in table 4) occurs only in the open shrub state class which results from wildfire or clearcutting and accounts for 60 percent of all debris flows. Because the underlying base probability given to all debris flows in the state-and-transition models is 0.01, we use the following multipliers to drive debris flow transitions in models for cascade or cascade-with-wood-forced-step channels: M_1=0.02; M_2=0.06; M_3=0.12.

Larger channels lower in the network (step-pool, planebed, and pool-riffle) have longitudinal gradients too shallow for origination of landslides. Instead, they are impacted by debris flows moving down the stream network. Because many headwater channels converge to form larger streams, and because the larger stream can be impacted by debris flows in any of the headwater channels, the probability of a debris flow increases. Consequently, we assumed that debris flows impacted step-pool channels 1 in 100 years and 1 in 50 years for planebed and pool-riffle channels.These debris flows may originate in an area lacking large wood and deposit only sediment, or may travel through the reach removing all large wood. Alternatively, they could deposit either small wood or large wood. Thus we created three categories: Debrisflow no wood (multiplier M_{No} in table 4) where the debris-flow-impacted channel is left free of wood; Debrisflow small wood (multiplier M_S in table 4) where the debris flow impacted channel is left with abundant small wood; Debrisflow large wood (multiplier M_L in table 4) where the debris-flow-impacted channel is left with abundant large wood. In the simulations reported here, these three classes of debris flows were given equal weightings. Again, because the underlying base probability given to all debris flows in the state-and-transition models is 0.01, we use the following multipliers to drive debris flow transitions in models for all step-pool channels: $M_{NO} = M_S = M_L$ = 0.3300. We use the following multipliers for all planebed and pool-riffle channels: $M_{NO} = M_S = M_L$ = 0.6700. Future model runs can be made iteratively, and the relative proportion of no wood, small wood, and large wood debris flows could be based on the tree size and large wood abundance in the debris flow source areas.

Management transitions in all PVTs—
The models we developed include a large number of possible management practices. Logging transitions (post-wildfire salvage logging, precommercial thinning, partial harvest and regeneration harvest, or clearcutting) were inherited from the upland models we used as a starting point for the aquatic-riparian models (table 4). In all cases, these transitions are initialized with a base probability of 0.0100 and must be controlled using an external multiplier file. We have not simulated forest harvest in any riparian stands so in the simulations reported here, all forest harvest transitions probabilities were set to zero.

The models also include several planting or restoration transitions, including planting of conifers in the open shrub state class following stand-replacing disturbances, large wood additions to the stream, and hardwood conversion of small alder stands by planting of conifers. These restoration transitions are also initialized with a base probability of 0.0100 and must be controlled using an external multiplier file. Additional management practices or prescriptions could be added to the models if need is demonstrated by managers or model users. Similarly, natural disturbance pathways could be edited and transition probabilities altered with new probabilities, or modified by using static or temporal multipliers.

Channel Conditions and Aquatic Habitat Quality

We used a 4-factor scale for each state class in the models to qualitatively rank their channel morphologic conditions: shade, erosion, undercut banks, large wood, pools, large pools, off-channel habitat, width-depth ratio, and riparian shrub abundance. We inferred the relative abundance of large wood, pools, undercut banks, and erosion from the cover type and structural stage of each state class in the models. These variables were then used in an expert systems model to rank the habitat quality (poor, fair, good, excellent) for migration, spawning, summer and winter rearing of coho salmon and steelhead. We assumed that coho habitat would be characterized by low gradient reaches (< 3 percent gradient), abundant large pools, and low erosion (low concentration of fine sediment) for spawning.

We used the abundance of large pools as an indicator of pool-riffle morphologies that would provide appropriate spawning riffles. Where pools were lacking, low -gradient reaches were assumed to be in a plane-bed state, with few distinct riffles. We assumed that Coho rearing habitat would be characterized by abundant pools, a combination of either undercut banks or large wood, and abundant off-channel habitat in areas with low erosion (low concentration of fine sediment). Thus, states with low gradient (i.e. pool-riffles) and abundant in-stream large wood, either recruited by stand-altering disturbances or self-recruited from large and giant trees, were assumed to provide the most favorable coho habitat.

We assumed that steelhead habitat would encompass nearly all coho habitat in low gradient reaches (i.e. pool-riffle morphologies) but would also extended into higher gradient reaches (including step-pool reaches) as long as pools and in-stream wood were abundant. We assumed that steelhead spawning requirements would be similar to those of coho, that is, they would utilize riffles in pool-riffle reaches where large pools were abundant and erosion (concentration of fine sediment) was low. We assumed that steelhead rearing habitat would be much more extensive within the watershed because this species was better able to occupy steeper reaches (i.e., > 3 percent gradient) with fast flowing water and little off-channel habitat. Also, steelhead typically spend two years rearing in freshwater instead of the single year that is the norm for coho. Thus we assumed that steelhead would be larger bodied in their second year, and would require larger pools and deeper water than in their first year.

We combined our qualitative habitat rankings with an independent evaluation of potential habitat quality, using Intrinsic Potential (IP) models (Burnett et al. 2007). IP scores are based on channel gradient, valley floor width, and drainage area and thus reflect the underlying physical "potential" to support the species of interest. The IP scores do not change over time, regardless the type and magnitude of disturbances that may alter other channel characteristics that influence the current quality of the stream habitat. We used the IP scores as a coarse filter through which we could identify portions of the stream network that had the potential to provide quality habitat for rearing, i.e., reaches where coho IP ≥ 0.60 (~18 km of the stream network) and steelhead IP ≥ 0.75 (~150 km of the stream network). Thus, large portions of the watershed, especially the coastal plain which has been converted to agricultural uses, mainstem reaches confined by bedrock gorges or high alluvial terraces, and headwater reaches too steep to support fish were excluded from our analyses.

Analysis and graphical display of model output— We developed the 4-factor ranking scale for both channel attributes and fish habitat quality in MS Excel but using spreadsheet templates from VDDT and then imported the ranked attributes into the VDDT models. Thus, the native graphic capabilities of VDDT or PATH (Path Landscape Model, VDDT's sister state-and-transition simulation model; http://www.apexrms.com, accessed 15 November, 2011) and other analytical tools can be used to analyze the model outputs. However, we found it easier to conduct these analyses outside of the VDDT/PATH platform where we could code SAS (Statistical Analysis Software) to perform specific detailed analyses needed to answer the questions in which we were interested. Further, we coded SAS to format output datasets that could be pasted directly into a graphic template in SigmaPLOT and thus quickly analyze and display results of individual model runs.

Model Application

The state and transition models were applied to the mountainous portion of the Wilson River watershed (500 km^2) in the northern Oregon Coast Range to examine: (1) current conditions relative to the historic condition; (2) likely trajectories of aquatic and riparian habitats given current and expected land-use practices; (3) the potential of passive restoration to meet recovery goals; and (4) the potential of active restoration to accelerate recovery. As a first step in our analysis, we simulated the Tillamook Burns to evaluate the ability of the models to successfully "hindcast" current conditions resulting from forest regrowth following these large, stand-replacing fires. We then used the models to compare future projections of habitat quality resulting from

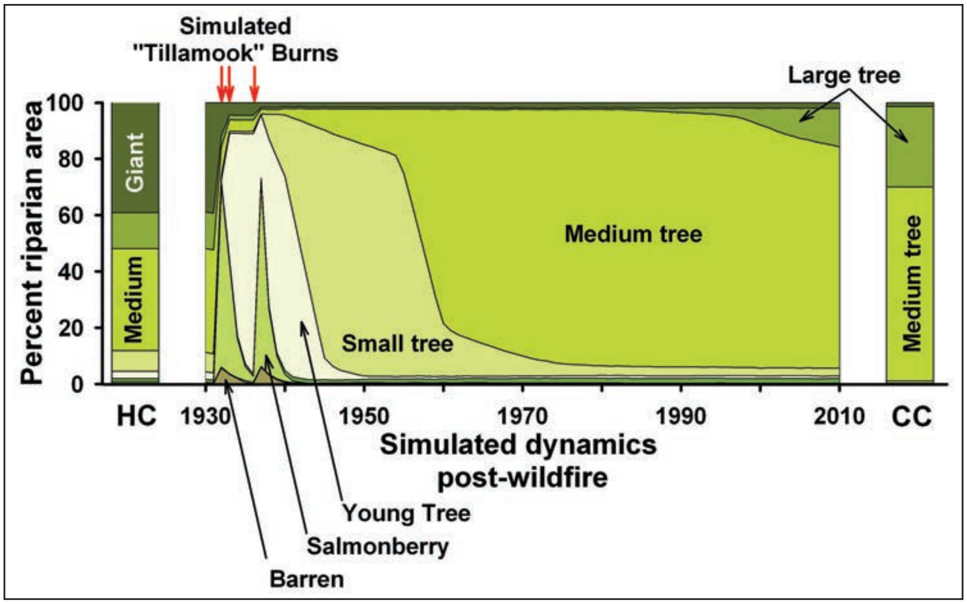

Figure 4—Comparison of historic condition (HC), the time series of changes resulting from the Tillamook Burns and subsequent forest regrowth through 2010, and the lidar-derived current condition (CC) for approximately 2010. The model projection includes a 10-year period from 1930 to 1940 where the watershed burned three times (red arrows), with each burn covering approximately 75 percent of the watershed. During the post-fire recovery (1940–2010), stand-replacing wildfires did not occur, but the models continued to simulate background rates of wind disturbance, mixed-severity wildfire, and debris flows.

active restoration with large wood addition to that resulting from passive restoration.

Simulating the Effect of the Tillamook Burns

The history of the Tillamook Burns—a series of large stand replacing wildfires that burned large portions of the northern Oregon Coast Range in 1933, 1939, 1946, and 1951—provides an opportunity to examine the ability of our state and transition models to simulate a time series of episodic disturbance and subsequent recovery of the riparian forests. We lack detailed records of the conditions of the Wilson River watershed prior to these major fires. However, large portions of the watershed were burned in high-intensity fires that essentially reset vegetation structure and composition to post-wildfire states. Thus, we initialized our model runs with initial conditions reflecting the long-term average historical condition of riparian vegetation within the watershed. The historic condition was projected from a 1000-year model run in which all anthropogenic effects were turned off and natural disturbance rates reflected the presumed historical rates of wildfires, windstorms, and debris flows. We lack detailed records of the exact portions of the Wilson River watershed burned in the Tillamook fires as well as historical records of postfire salvage logging and efforts to replant the burned areas. We do know that the 1933 fire was the largest of the "Tillamook Burns," so we used a series of temporal multipliers to force major wildfires in 1933 and 1934 and again in 1938. Each of these fires burned approximately 75 percent of the watershed so that by the late 1930s the structure and composition of the watershed had been nearly entirely reset to the earliest seral stages.

We simulated postfire recovery (1940–2010) by preventing stand-replacing wildfires (which did not occur over this time period), but continued to simulate background disturbances from wind, mixed-severity wildfire, and debris flows. Because salmonberry is a highly successful early-seral shrub throughout the northern Oregon Coast Range and resprouts readily after wildfire, in our models, wildfires forced transitions to salmonberry-dominated state classes (fig. 4). Subsequently alder rapidly colonized these states

Table 5—Summary table showing the percent of the riparian area in each vegetation composition and structural group within the Wilson River watershed

							Non-	Salmon-		Alder	Alder	Alder	Mixed Mixed
Mixed							**Conifer**	**Conifer**	**Conifer**	**Conifer**	**Conifer**		
Time step	**Forest**	**berry**	**young**	**small**	**medium**	**medium**	**large**	**giant**	**young**	**small**	**medium**	**large**	**giant**
HC	1.1	0.8	2.4	6.80	18.96	16.5	11.12	12.86	0.2	0.5	0.8	1.76	26.18
Post-burn	6.9	66.2	21.6	0.3	0.8	0.8	0.6	0.6	0.9	0	0.1	0.2	1.1
1940	1.6	3.2	65.2	21.6	1.2	1	0.5	0.4	3.8	0.2	0.1	0.2	1.1
1950	1.3	0.4	0.9	78.5	8.7	3.9	0.5	0.5	0.3	3.8	0	0.2	1.1
1960	1.5	0.4	0.9	14.7	60.4	15.7	0.5	0.5	0.1	4	0.1	0.1	1.2
1970	1.6	0.5	0.8	4.5	61.7	24.2	0.5	0.5	0	2.5	1.7	0.1	1.2
1980	1.6	0.5	0.9	2.7	56.3	31.1	0.5	0.5	0	0.8	3.6	0.1	1.2
1990	1.7	0.4	0.9	2.7	50.4	36	1.4	0.6	0.1	0.5	4.1	0.1	1.3
2000	1.7	0.4	0.9	2.6	45	38	4.6	0.6	0	0.3	3.4	1.3	1.3
2010	1.6	0.4	0.9	2.6	40.3	36.1	11.1	0.6	0.1	0.2	2.2	2.7	1.3
LIDAR-CC	0.3	0	0	0.1	42.2	11.4	29.4	0.5	0	0.5	5	10.1	0.6

Note: The data shown here subdivide the coarser vegetation classes shown in figure 4. HC denotes the simulated long-term average condition of the stream network resulting from a long-term (1,000-year) simulation of natural disturbance and plant succession. Post-burn denotes the composition and structure of the riparian zone in the year following the last simulated burn. LIDAR-CC denotes the current condition of the riparian zone, based on the lidar-derived vegetation structural classes and subsequent classification into model state classes.

resulting in a rapid shift toward alder dominance, followed by growth and aging of established trees over time (table 5). By the end of the simulation period in 2010, forest structure and composition resembled the 2010 lidar-derived current conditions (fig. 4).

Current vegetation (both simulated and lidar-derived), however, is markedly distinct from the long-term average historic condition (fig. 4, table 5). Current conditions (CC; fig. 4) were determined through classification of lidar imagery. Giant conifer (DBH > 30 inches) dominated stands are notably lacking in the riparian zone. Historically, our simulations project that some 25 percent of the stream network would have supported riparian vegetation dominated by conifers larger than 30 inches DBH. Conversely, alder-dominated stands and mixed conifer-alder stands of medium- or large-sized trees are overrepresented in the riparian zone (fig. 4). These trends are expected, given the history of large, stand-replacing wildfires within the watershed.

A more detailed analysis of forest composition, separating the forest types into alder-dominated forest, conifer-dominated forests, and mixed forests where conifers are beginning to over-grow previously alder-dominated forests, is shown in table 5. There are some discrepancies between the simulated structure and composition in 2010, compared to the lidar-derived current vegetation. For example, we under predict the amount of large, conifer-dominated forest (table 5). However, much of the area within the Tillamook Burn was planted with conifers in the decade following the burns. We did not simulate postdisturbance planting in our model runs. First, our models only allow postdisturbance planting over a short time window following disturbance which is typical of Burned Area Emergency Response (BAER) treatments currently employed within the region. Second, we do not know if the riparian zones were successfully planted in the Tillamook Burn area. Certainly, our observations in the field and the composition of the lidar-derived current vegetation are generally in agreement with the simulation results which indicate that large areas of the riparian zone remain in alder-dominated stands. Overall, we conclude that the models, as parameterized, provide an

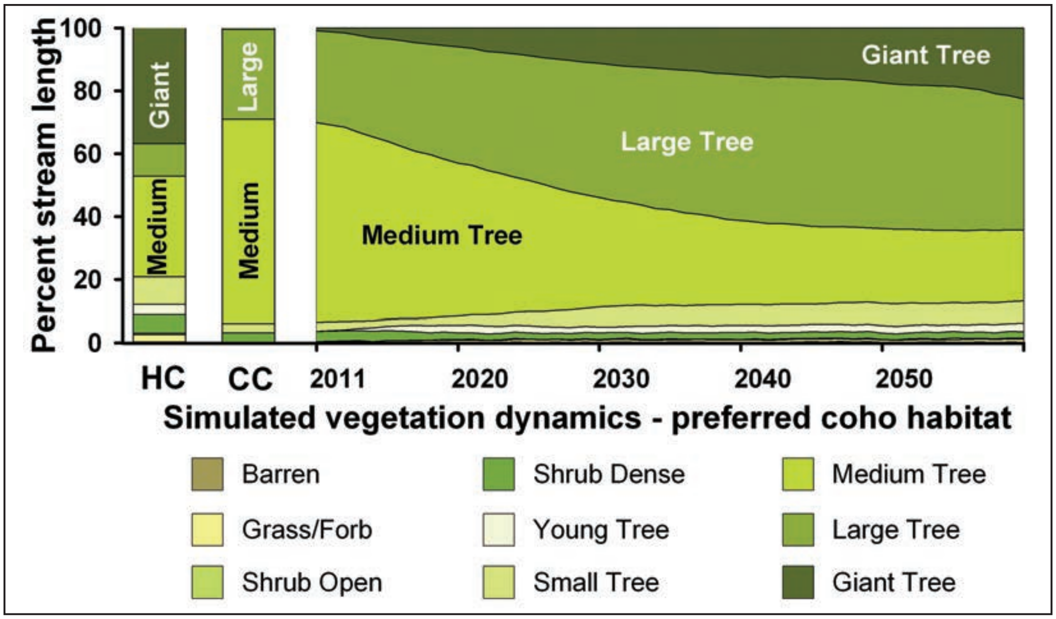

Figure 5a—Comparison of historical condition, current condition, and projected future conditions of the riparian vegetation resulting from passive restoration (i.e., all anthropogenic activities turned off) in areas where the intrinsic potential score for coho is greater than 0.60.

acceptable simulation of the riparian vegetation dynamics in the riparian zones of the Wilson River watershed.

Changes in Coho and Steelhead Habitat

Coho—

Only 18 km of the stream network within the mountainous portions of the Wilson River watershed have high intrinsic potential to provide quality rearing habitat for juvenile coho salmon (IP score $\geq$ 0.60). Comparisons of the simulated historical condition and with lidar-derived current condition of riparian vegetation show large departures from historical conditions. The historical condition (HC) was projected from 500-year model runs in which all anthropogenic effects were turned off but natural disturbances continued to occur. Current conditions (CC) were determined through classification of lidar imagery. Under current conditions, giant-tree structural states are almost entirely lacking and nearly two-thirds of the riparian areas are dominated by medium-sized tree structural states (fig. 5a). Simulations of the historical condition suggest that nonforest vegetation comprised as much as 10 percent of the riparian zone. Today, nonforest vegetation comprises less than 5 percent of the riparian zone. Changes within the watershed have also substantially influenced the quality and abundance of rearing habitat available for coho salmon (fig. 5b). Our historical simulation suggests that nearly two-thirds of the potentially useable habitat would have been ranked as good or excellent quality. Today, less than 25 percent of that stream habitat is ranked as good or excellent.

Future conditions were projected from the aquatic-riparian VDDT models with all anthropogenic activities turned off (i.e., no forest harvest in riparian zones, no salvage logging, and no riparian planting or other restoration treatments). This simulation is effectively a "passive restoration" scenario where no active management occurs. Simulations suggest that, over time spans of 5 to 10 years, changes in vegetation structure are relatively slow. Over the longer term, in the absence of episodic disturbances, the growth of conifers in mixed stands will lead to a slow but steady increase in the abundance of large trees. However, even after another 50 years (by ca. 2060) riparian forest conditions will still remain distinct from their historical conditions (fig. 5a). Similar patterns are seen in the projections for habitat abundance and quality for coho salmon, with changes accumulating slowly, but steadily, so that conditions are markedly improved after 50 years. However,

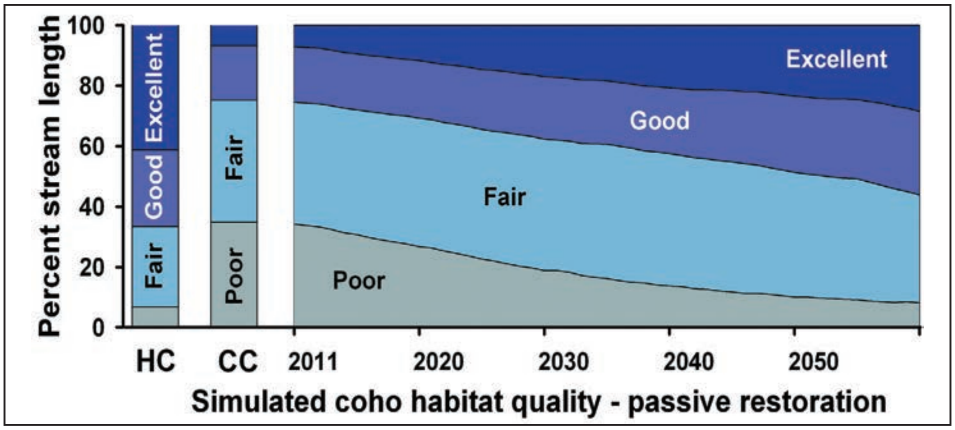

Figure 5b—Comparison of rearing habitat quality under historicall, current, and projected future conditions resulting from passive restoration in areas where the intrinsic potential score for coho is greater than 0.60.

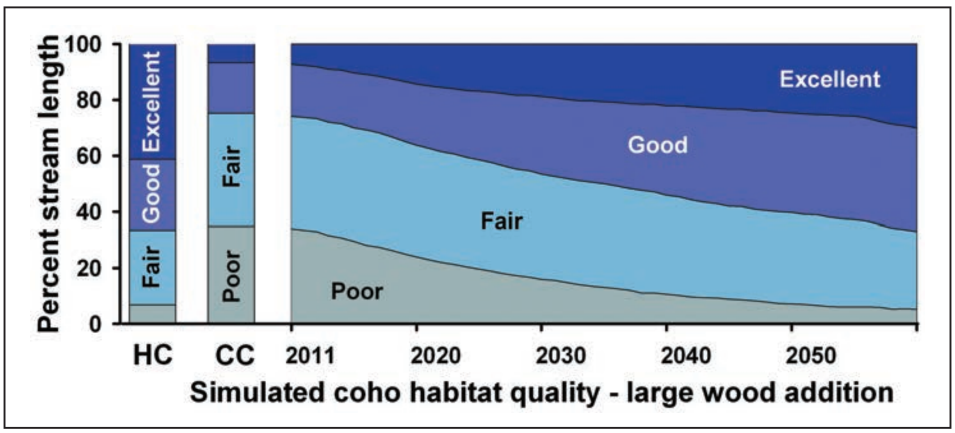

Figure 5c—Comparison of rearing habitat quality under historic, current, and projected future conditions resulting from an active restoration from large wood addition in areas where the intrinsic potential score for coho is greater than 0.60. Note that large wood addition has no effect on the structure and composition of the adjacent riparian vegetation, thus vegetation structure is not shown for this simulation.

even after 50 years of "passive restoration," comparison with the historical condition shows that there is substantially less habitat ranked as excellent and more ranked as fair (fig. 5b) than in the simulated historical condition.

Steelhead—

The portions of the stream network with the highest potential to provide high-quality rearing habitat for steelhead (IP score > 0.75) are much more extensive than for coho, encompassing most of the modeled reaches of the mainstem Wilson River as well as all of the larger tributaries within the watershed. Collectively, more than two-thirds of the modeled stream network has the potential to provide high-quality rearing habitat for steelhead.

Comparisons between the simulated historic condition and lidar-derived current condition of riparian vegetation for steelhead are very similar to those for coho salmon in which giant-tree structural states are almost entirely lacking and nearly two-thirds of the riparian areas is dominated by medium-sized tree structural states (fig. 6a). The quality and abundance of rearing habitat available for steelhead under current conditions is also substantially different from the simulated historical condition (fig. 6b). Our historical simulation suggests that nearly two-thirds of the high IP

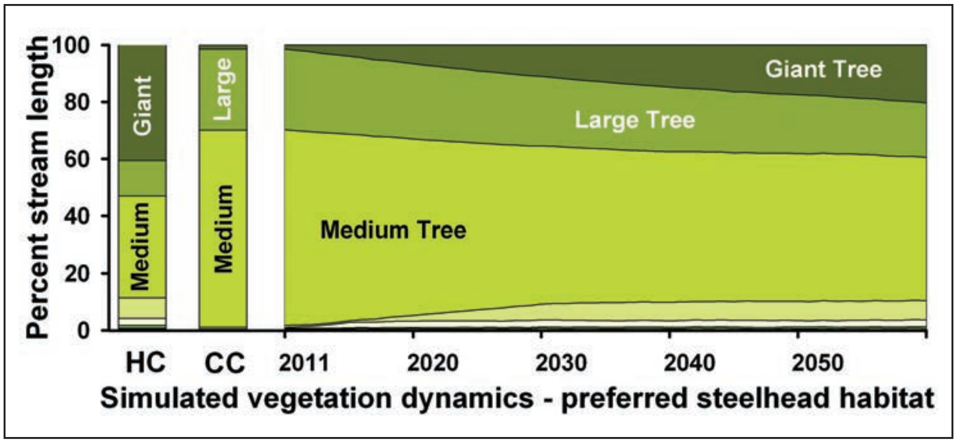

Figure 6a—Comparison of historic condition, current condition, and projected future conditions of the riparian vegetation resulting from passive restoration (i.e., all anthropogenic activities turned off) in areas where the intrinsic potential score for steelhead is greater than 0.75. Legend follows figure 5a.

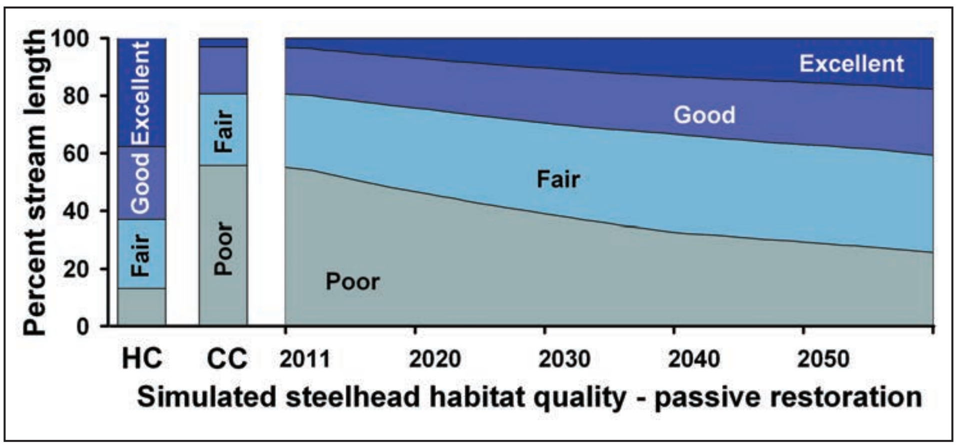

Figure 6b—Comparison of rearing habitat quality under historic, current, and projected future conditions resulting from passive restoration in areas where the intrinsic potential score for steelhead is greater than 0.75.

habitat would have been ranked as good or excellent habitat quality. Today, only 20 percent of that stream habitat is ranked as good or excellent.

Simulations of passive restoration suggest that, over the short term, changes in vegetation structure are relatively slow. Over the longer term, the models simulate substantial growth of riparian trees, however, even after another 50 years (by ca. 2060) riparian forests remain distinct from their historical conditions (fig. 6a). As with coho, similar patterns are seen in the projections for habitat abundance and quality for steelhead, with changes accumulating slowly, but steadily, so that conditions are markedly improved after 50 years. However, even after 50 years of "passive restoration," comparison with the historical condition shows that there is substantially less habitat ranked as excellent and more ranked as fair or poor (fig. 6b) than in the simulated historical condition.

Stream Restoration Through Large Wood Augmentation

Our model simulations project substantial recovery is likely to occur over the next 50 years in the absence of major episodic disturbance, however, even after 50 years, the quality and abundance of stream habitat for coho and

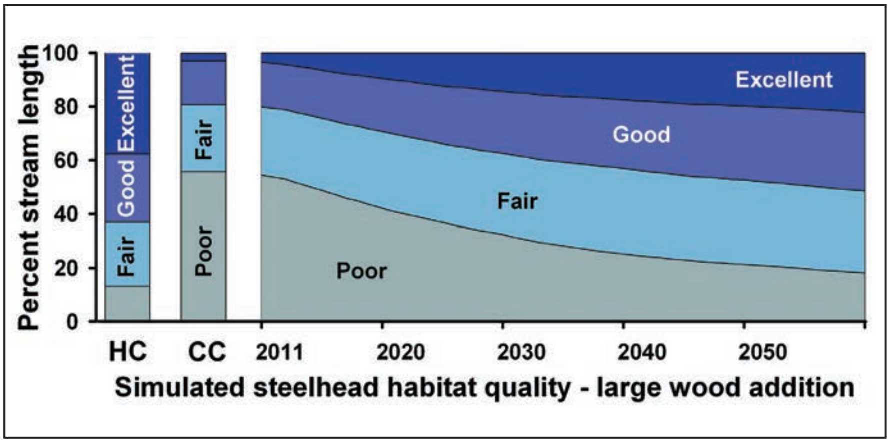

Figure 6c—Comparison of rearing habitat quality under historic, current, and projected future conditions resulting from active restoration scenario with large wood addition in areas where the intrinsic potential score for steelhead is greater than 0.75. Note that large wood addition has no effect on the structure and composition of the adjacent riparian vegetation, thus vegetation structure is not shown for this simulation.

steelhead remains distinct from the simulated historical condition. One active restoration approach would be to add large wood to stream reaches where it is currently lacking to accelerate the recovery of habitat quality. The initial conditions we used to start our model runs for the 18 km of river with high intrinsic potential for coho salmon indicated that 61 percent of those 18 km had little or no stream wood and 24 percent had abundant large wood at the beginning of our simulation (ca. 2010). In the active recovery scenario, we simulated active large wood addition, treating approximately 1 km of stream network per decade. After 50 years, the portion of those 18 km where large wood was abundant increased from 53 percent to 64 percent. These large wood addition treatments, however, led to a small increase in the availability of coho rearing habitat ranked good or excellent after 50 years. Some 57 percent of the stream network was ranked good or excellent under the passive restoration scenario and 67 percent ranked in those categories in the active restoration scenario (fig. 5b versus 5c). Because the availability of high-quality habitat for coho is presently very limited within the Wilson River watershed, treating only 5 km of stream channel through large wood addition results in a modest improvement in simulated habitat quality.

We conducted a similar series of model simulations for the 150 km of stream network with high intrinsic potential to support steelhead. Our simulations showed that to get an improvement similar to coho in the amount of habitat ranked good or excellent required treatment of approximately 44 km of stream channel which increased the amount of the stream network ranked good or excellent from 41 percent under passive restoration to 52 percent under active restoration over the 50-year model simulation (fig. 6b versus 6c). The active restoration scenario for steelhead would be much more expensive than for coho because it would require adding large wood to approximately 9 km of stream per decade.

The abundance of large wood was not readily quantified from the remote sensing techniques employed in this study. We did make estimates of large wood abundance from our field sampling plots (n = 29) which allowed us to specify the initial conditions for our model simulation. However, we emphasize that our 29 sample plots are too few to realistically estimate the abundance of large wood. Actual stream inventory data would help set more realistic initial conditions for our model simulation.

Discussion

The results presented here demonstrate the utility of our state and transition models for exploring a variety of questions related to landuse decisions and the effect of alternative management scenarios on riparian vegetation, channel morphology, and stream habitat condition for salmonids.

The restoration scenarios examined here represent only a few of the large number of management questions that could be explored using these models.

State and transition models are relatively "transparent" in that other users can pick up a package of existing models, and with a minimum effort begin working with those models. This ease of use is facilitated by the fact that the VDDT and PATH software (http://www.apexrms.com/path, accessed 15 November 2011) needed to run these models are publicly available and relatively easy to use. And while the size of some of our aquatic-riparian state and transition models may make them appear daunting at first, the software user interface makes it relatively easy to revise the models to meet a wide variety of alternative assumptions. The true value of these models is that alternative assumptions or "alternative management scenarios" can then be readily tested and the model outputs used to provide hypotheses of likely outcomes to explore the ways in which policy decisions may influence the future condition of riparian zones.

Our models are broadly portable. We developed models specifically for the Wilson River watershed in the northern Oregon Coast Range. However, similar potential vegetation and geomorphic types occur throughout the central and northern Oregon as well as the southern Washington Coast Range. Consequently, our models are likely to be directly applicable to these larger regions. Further, the general model structures and the rather exhaustive list of transition processes included in the models provides a template from which models for other areas within the region could be readily constructed.

Although the results of our model simulations have appeared reasonable wherever data were available for comparison, we caution that these comparisons do not provide detailed validation of all the factors simulated in our models. Consequently, the results of the model simulations should be interpreted cautiously. The models are not intended to provide detailed predictions of specific outcomes at the scale of a single reach or for a specific restoration project involving 100s or a few kilometers of stream channel. Rather, the results of the model simulations should be interpreted as hypotheses of likely outcomes from management directions at the scale of a large watershed (one or several 5^{th}-field hydrologic units or HUC5s) or a large portion of a USFS Ranger District.

References

Beukema, S.J.; Kurz, W.A.; Pinkham, C.B.; Milosheva, K.; Frid, L. 2003. Vegetation Dynamics Development Tool, User's Guide, Version 4.4c. Vancouver, BC, Canada: ESSA Technologies Ltd. 239 p.

Breiman, L. 2001. Random Forests. Machine Learning, 45:5–32. Burnett, K.M.; Reeves, G.H.; Miller, D.J.; Clarke, S.; Vance-Borland, K.; Christiansen, K. 2007. Distribution of salmon-habitat potential relative to landscape characteristics and implications for conservation. Ecological Applications. 17: 66–80.

Castro, J.M.; Jackson, P.L. 2001. Bankfull discharge recurrence intervals and regional hydraulic geometry relationships: patterns in the Pacific Northwest, USA. Journal of the American Water Resources Association. 37: 1249–1262.

Federal Caucus. 2000. Vols. 1–2. Conservation of Columbia basin fish, final basinwide salmon recovery strategy. Portland, OR: Federal Caucus/Bonneville Power Administration. http://www.salmonrecovery.gov/Final_Strategy_Vol_1.pdf, http://www.salmonrecovery.gov/Final_Strategy_Vol_2.pdf. (June 9, 2003).

Gatziolis, D.; Fried, J.S.; Monleon, V.S. 2010. Challenges to estimating tree height via LiDAR in closed-canopy forests: A parable from western Oregon. Forest Science. 56: 139–155.

Hudak, A.T.; Crookston, N.L.; Evans, J.S.; Hall, D.E.; Falkowski, M.J. 2008. Nearest neighbor imputation of species-level, plot-scale forest structure attributes from LiDAR data. Remote Sensing of Environment. 112: 2232–2245.

Montgomery, D.R.; Buffington, J.M. 1997. Channel-reach morphology in mountain drainage basins. Geological Society of America Bulletin. 109: 596–611.

Montgomery, D.R.; Buffington, J.M. 1998. Chapter 2: Channel processes, classification, and response. In: R.J. Naiman and R.E. Bilby, eds. River Ecology and Management—Lessons from the Pacific Coastal Ecoregion. New York: Springer-Verlag: 13–42.

National Research Council (NRC). 2002. Riparian areas: functions and strategies for management. Washington, DC.: National Academy Press. 428 p.

Nehlsen, W.; Williams, J.E.; Lichatowich, J.A. 1991. Pacific salmon at the crossroads: Stocks at risk from California, Oregon, Idaho and Washington. Fisheries. 16: 4–21.

Pflugmacher, D.; Cohen, W.B.; Ohmann, J.L.; Kennedy, R.E. 2011. Final report to the Oregon Watershed Enhancement Board, for grant OWEB # 208-8007-5781. Mapping Current Conditions and Modeling the Dynamic Responses of Riparian Vegetation and Salmon Habitat in Oregon: Mapping Component. Submitted 6 June 2011.

U.S. Department of Agriculture, Forest Service; U.S. Department of the Interior, Bureau of Land Management. 2000. Interior Columbia basin supplemental draft environmental impact statement. The Interior Columbia Basin Ecosystem Management Project. Washington, DC: U.S. Government Printing Office. [Irregular pagination].

U.S. Geological Survey and U.S. Department of Agriculture, Natural Resources Conservation Service. 2009. Federal guidelines, requirements, and procedures for national Watershed Boundary Dataset: U.S. Geological Survey Techniques and Methods 11-A3, 55 p.

Landscape Composition in Aspen Woodlands Under Various Modeled Fire Regimes

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Abstract

Quaking aspen (*Populus tremuloides*) is declining across the western United States. Aspen habitats are diverse plant communities in this region and loss of these habitats can cause shifts in biodiversity, productivity, and hydrology across spatial scales. Western aspen occurs on the majority of sites seral to conifer species, and long-term maintenance of these aspen woodlands requires periodic fire. We use field data, remotely sensed data, and fire atlas information to develop a spatially explicit landscape simulation model to assess the effects of current and historic wildfire regimes and prescribed burning programs on landscape vegetation composition in the Owyhee Mountains, Idaho. The model is run in the Tool for Exploratory Landscape Scenario Analyses (TELSA) environment. Model outputs depict the future structural makeup and species composition of the landscape at selected time steps under simulated management scenarios. Under current fire regimes and in the absence of management activities, loss of seral aspen stands will continue to occur. However, a return to historic fire regimes, burning 12–14 percent of the modeled landscape per decade, maintains the majority of aspen stands in early and mid seral woodland stages and minimizes the loss of aspen. A fire rotation of 70–80 years was estimated for the historic fire regime while the current fire regime resulted in a fire rotation of 340–450 years. Implementation of prescribed burning programs, treating aspen and young conifer woodlands according to historic fire occurrence probabilities, are predicted to prevent conifer dominance and loss of aspen stands.

Keywords: Aspen, *Populus tremuloides*, VDDT, TELSA, succession, disturbance, fire regime

Introduction

Region-wide decline of quaking aspen has caused concerns that human alteration of vegetation successional and disturbance dynamics jeopardize the long-term persistence of these woodlands. (Bartos 2001, Kay 1997, Shepperd et al. 2001, Smith and Smith 2005). Aspen are an important component that provides ecosystem diversity in the conifer dominated western mountains. Aspen ecosystems provide a disproportionately diverse array of habitats for flora and fauna for its relatively small area on the landscape (Bartos 2001, Jones 1993, Kay 1997, Winternitz 1980). In the semi-arid western U.S., aspen commonly occurs as a disturbance-dependent species, seral to conifer species (Bartos 2001, Kaye et al. 2005, Smith and Smith, 2005). It is well known that in mixed aspen and conifer stands, periodic fires prevent conifer dominance and possible loss of the aspen stand (Baker 1925, Bartos and Mueggler 1981, DeByle et al. 1987). Although the aspen is a prolific seed producer, the conditions required for successful seed germination and establishment are rare in the American West (Mitton and Grant 1996). Aspen clones in the region reproduce primarily via vegetative suckering and therefore it can be concluded that an aspen clone lost is not likely to re-establish via seed. An example of recent successful establishment of aspen seedlings occurred in response to the severe fires in 1988 in Yellowstone National Park (Romme et al. 2005). All aspen stands are however not seral to conifers. Aspen stands in certain biophysical settings and away from a conifer seed source have been observed to exist as self-regenerating even and uneven aged stands that do not appear to be at risk of

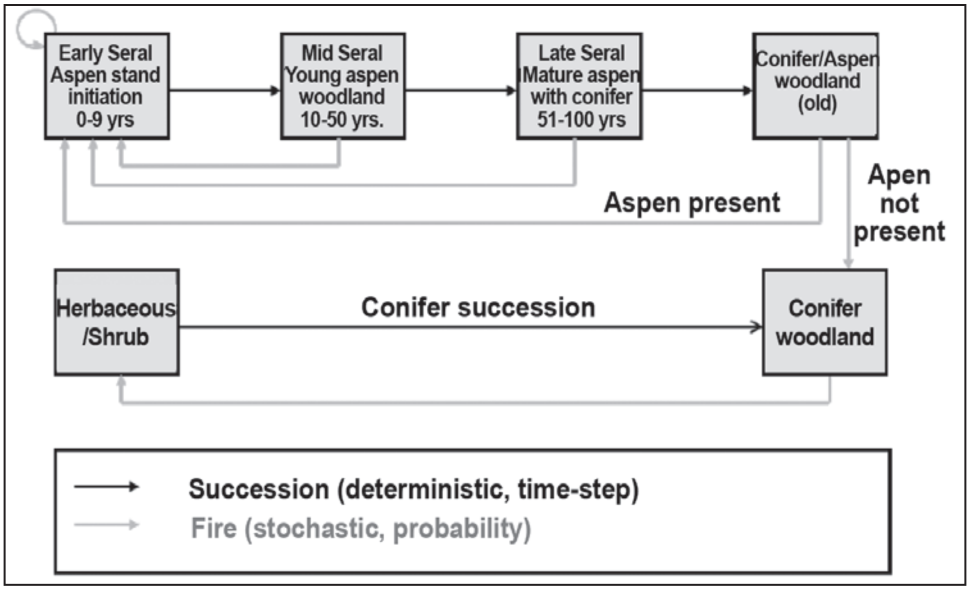

Figure 1—Simplified pathway diagram for upland aspen/conifer communities that served as the conceptual model for vegetation dynamics in the Owyhee Mountains.

rapid decline due to conifer expansion even in the absence of fire (Mueggler 1989, Rogers et al. 2010, Strand et al. 2009). Mortality in these stable aspen stands has however been observed over the past decade (Worall et al. 2008). This mortality has been correlated with rising temperatures and drought in the southwestern U.S. (Huang and Anderegg 2011; van Mantgem and Stephenspn 2007, van Mantgem et al. 2009) potentially caused by hydraulic failure of roots (Anderegg et al. 2012).

Successional rates within pure and mixed aspen stands and interactions with fire and herbivory have been studied at the stand level, however, little work has examined these dynamics across larger landscapes over decades. Computer simulation models may be a means to better understand these dynamics in landscapes where aspen is present. Such landscape level succession/disturbance models have been used for evaluating habitat patterns in forests and woodlands (e.g., Klenner et al., 2000; Bunting et al. 2007) and assessment of fire regimes and management scenarios (Bunting et al. 2007, Franklin et al. 2001, Keane et al. 1997).

In response to the need for better understanding of interactions between aspen/conifer succession and fire regimes across larger landscapes over decadal time scales, we simulated a number of aspen management scenarios using a conceptual state-and-transition model developed for aspen/conifer woodlands (fig. 1, Strand et al. 2009) and the Tool for Exploratory Landscape Scenario Analyses (TELSA, ESSA Technology 2003). We utilized field and remotely sensed data combined with spatially explicit modeling to estimate the effects of current and historic fire regimes on landscape vegetation composition and structure, emphasizing aspen woodland dynamics. Although prescribed fire has been suggested and applied to mitigate the frequent fire events common in the western mountains of the past, with the goal of maintaining and restoring aspen woodlands (Bates et al. 2004, Brown and DeByle 1989, Miller et al. 2005, Shepperd, 2001), little is known about how such management affects the vegetation composition and structure spatially and temporally. We therefore also incorporate prescribed burning scenarios into our modeling runs. In particular, we address the following four research questions: (1) Can we simulate the fire regime that maintained aspen stands prior to Euro-American settlement?; (2) What extent and frequency of fire is required to stabilize the current land cover composition within aspen woodlands?; (3) What is the structural composition of aspen woodlands under historic and current fire occurrence probabilities, and under prescribed burning scenarios?; and (4) What is the effect of fire size on the long-term maintenance of aspen woodlands?

Methods

Site Description

The mountain ranges of the Owyhee Plateau in SW Idaho (116.4° W, 43.0° N) contain vegetation communities representative of many semi-arid mountains of the western U.S.A. We include two study areas in this research: the South Mountain study area encompassing 17,000 ha and the Silver City Range covering 20,000 ha. Western juniper woodlands (*Juniperus occidentalis* ssp. *occidentalis*) and sagebrush (*Artemisia* spp.) steppe dominate the landscape above 1700 m altitude, interspersed with pockets of aspen, mountain shrub species, and meadows. Western juniper is gradually replaced by Douglas-fir (*Pseudotsuga menziesii* ssp. *glauca*) above 1850 m in both mountain ranges. Aspen stands are commonly located on cool northeast facing slopes, in concave snow and moisture accumulation areas. Soils that support aspen include deep fine-loamy and loamy-skeletal mixed pachic or typic cryoborols, rich in organic material with high water-holding capacity (USDA NRCS 1998). In the area, aspen occurs in three distinctly different biophysical settings with different successional trajectories and rates; pure aspen on south-facing aspects above 1900 m, aspen on wet micro sites, and aspen/conifer stands on mountain hillsides (Strand et al., 2009). Areas that support aspen receive 400-1000 mm annual precipitation (Oregon Climate Service 1999) in the form of rain in the spring and fall, and snow during the winter. Summer and early fall are warm and dry with an average high temperature in July of 26.7º C (WRCC 2003).

Field Data Collection

A total of 82 aspen clones along elevational and succesional gradients were sampled across the study areas. Site characteristics were recorded: slope, elevation, aspect, and Universal Transverse Mercator (UTM) coordinates. We further collected stand characteristics: canopy cover of aspen and conifers in the crown and below 2-m height, increment cores from the five tallest mature aspen and conifer trees (thought to be among the oldest), stem counts of aspen and conifers in three height classes (< 2 m, 2 m up to 75 percent of the stand height, and trees taller than 75 percent of the stand height). The increment cores were mounted and sanded, and the annual growth rings counted in a stereo-microscope for the age estimate. Faint annual rings in aspen were stained with phloroglucinol solution before ring counting (Patterson 1959).

Model Requirements and Assumptions

TELSA (Essa Technology 2003) is a spatially explicit landscape dynamics model environment, allowing the user to explore the effect of natural and anthropogenic disturbances on landscape composition. Input data to this model include potential natural plant communities, initial vegetation types and structural stages, along with natural and anthropogenic disturbance agents and pathways. Succession is treated as a deterministic variable with a constant pre-determined time period between successional states.

Successional rates in upland aspen stands are based on models developed by Strand et al. (2009). They discovered that the successional development in upland aspen/conifer woodlands on the Owyhee Plateau can be characterized with a positive exponential function where the proportion conifer in the stand is fit against time since conifers were introduced to the stand:

$$f(t) = A\ e^{kt}\ (0 < f(t) < 1) \tag{1}$$

where f(t) is the proportional cover of conifers in the aspen stand (e.g. conifer cover divided by total cover of all tree species), which is close to 0 at $t = 0$ and approaches 1 at complete conifer dominance, and the constant k represents the successional rate. The best model estimate ($R^2 = 0.63$, F=114.4, p<0.001) was:

$$f(t) = 0.0177\ e^{0.0315 * t}\ 0 < f(t) < 1 \tag{2}$$

where the model constant A = 0.0177 and successional rate k = 0.0315. Time since the initiation of conifer establishment was the only variable that significantly affected the successional rate in this data set although environmental variables such as terrain attributes, soil and climate data were included in model development (Strand et al. 2009). This model was developed using only upland aspen/conifer stands, and does not apply to aspen in riparian areas nor

Table 1—Areas of mapped cover types within the South Mountain and Silver City Range study sites on the Owyhee Plateau in SW Idaho

Cover type	South Mountain Area (ha)	Silver City Range Area (ha)
Aspen woodland (pure aspen)	496	236
Aspen/Douglas-fir woodland	1371	2002
Aspen/Western juniper woodland	745	527
Bare/Rock	2	72
Ceanothus/Mesic shrub	299	365
Douglas-fir	298	923
Juniper woodland/Low sage open	1635	787
Juniper woodland/Low sage closed	1056	141
Juniper woodland/Mountain big sage open	4062	3321
Juniper woodland/Mountain big sage closed	3451	1259
Curlleaf mountain-mahogany	227	1983
Low sagebrush steppe	1335	2343
Mountain big sagebrush steppe	1729	5992
Wet meadow	42	189

areas around meadows and springs. An exponential increase in the conifer dominance occurs 50–60 years after conifers were initiated to the aspen stand, as prolific conifer seed production and spread begins (see Strand et al. 2009). This exponential increase in conifer dominance marks the transition of mid seral aspen into late seral aspen (fig. 1).

In TELSA, disturbance is treated as a stochastic variable driven by user-defined probabilities. This stochastic component in landscape models results in many possible landscape configurations given the same input variables, allowing the range of variability in landscape composition to be explored statistically.

Spatially explicit simulations in TELSA require information in the form of GIS data layers (digital maps) of the study area. Each landscape unit in the map must be classified hierarchically in a potential vegetation type (PVT), current cover type, and current structural class. PVTs are groupings of habitat types or ecological sites with similar overstory composition in the absence of a disturbance and similar environmental requirements. For the sagebrush steppe/juniper woodlands we employed the PVT classification developed by Bunting et al. (2007) in the same general study area. As mentioned earlier, aspen woodlands are potentially present in three PVTs (Strand et al. 2009): pure aspen, aspen/western juniper, and aspen/Douglas-fir.

In the simulation, aspen stands on pure aspen PVTs represent stands that can be expected to self-regenerate and persist as uneven aged aspen stands for decades into the future. Over time, aspen on aspen/western juniper and aspen/Douglas-fir PVTs become outcompeted by western juniper and Douglas-fir, respectively, and in the absence of a disturbance within a certain time period will permanently convert to pure conifer stands (Wall et al. 2001, Strand et al. 2009). Aspen/conifer stands that burn prior to permanent conversion to conifer stands are assumed to return to stand initiation aspen stands (fig. 1).

Each landscape unit is characterized by its PVT, but also by the current cover and structure. The current cover map represents the vegetation currently present on the ground and includes the climax vegetation classes represented by the PVTs with the addition of seral cover types such as grasslands, shrublands, and young woodlands. The structural classes within aspen succession include: stand initiation aspen, young aspen woodlands, mature aspen woodlands, aspen woodlands with conifers, and conifer woodlands. We used input GIS layers previously developed for the Owyhee Plateau by Strand (2007) depicting the PVT, current vegetation and structural stages, see table 1 for landscape distribution of cover types and table 2 and figure 2 for distribution of PVTs.

Table 2—Areas of mapped potential vegetation types (PVT) within the study area

Cover type	Silver City South Mountain Area (ha)	Range Area (ha)
Aspen woodland	496	236
Aspen/Douglas-fir woodland	1669	2925
Aspen/Western juniper woodland	745 5	27
Bare/Rock	2 7	2
Ceanothus/Mesic shrub	299	365
Juniper woodland/Low sage	4028	3272
Juniper woodland/Mountain big sage	9240	10571
Curlleaf mountain-mahogany	227	1983
Wet meadow	42	189

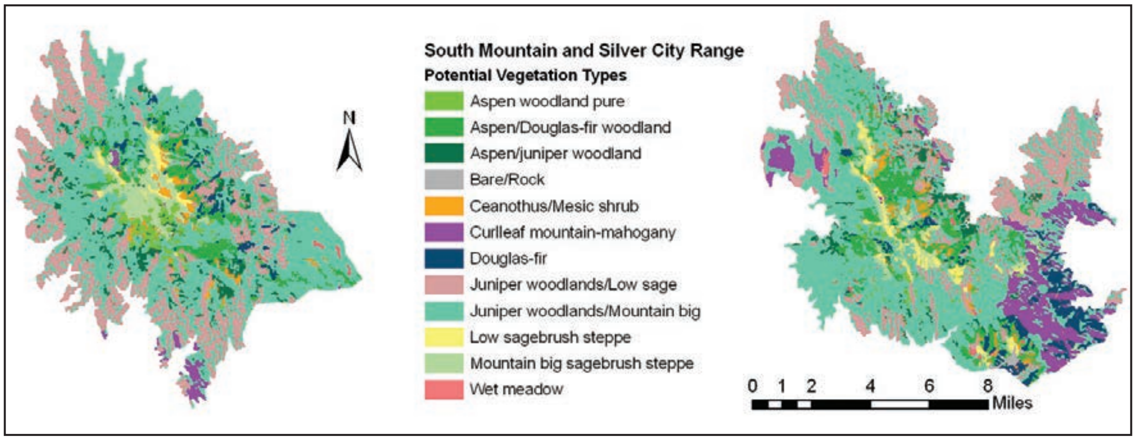

Figure 2—Potential vegetation maps of the South Mountain (left) and the Silver City (right) 856 areas of the Owyhee Mountains in SW Idaho.

In general, we make the assumption that PVTs are static, and consequently a landscape unit occupied by a PVT at the beginning of the simulation will stay within that PVT throughout the simulation. The land cover and structural vegetation stage within the landscape unit may change via the successional time step or revert to an earlier seral stage via disturbance (i.e. fire). This static view of PVT works well in most ecosystems within reasonable time periods. In the aspen ecosystem, however, this static view is limited for two reasons. First, aspen has been observed to expand into adjacent areas with low canopy cover such as grasslands and sagebrush steppe. Such expansion of aspen clones was observed during field assessments during this study and has also been reported by other researchers (Manier and Laven 2001). Expansion of aspen could not be incorporated directly in the TELSA simulations, but upper limits of aspen expansion were estimated based on expansion rates and the length of currently available aspen/sagebrush edge. The rate of aspen expansion into adjacent cover types, was estimated by recording the decrease of aspen stem age along four transects perpendicular to the aspen/sagebrush steppe ecotone during the 2006 field season in the nearby Jarbidge Mountains. The four transects show similar expansion rates of approximately 0.5 m per year (20 m expansion in 40 years). We assume here that the aspen expansion rates are similar in the Jarbidge and Owyhee Mountains, because the two mountain ranges are located at similar latitudes and span similar altitudes. Second, it is currently not known how long and under what conditions an aspen clone can persist after conifers dominate a site. It has been suggested that aspen clones can be sustained for decades in the absence of mature ramets maintained only by transient suckers

(Despain 1990). This hypothesis has not yet been tested (Hessl 2002); and we assume here that old mixed aspen/conifer stands permanently transition to conifer stands 120 years after aspen regeneration has diminished due to conifer dominance within a stand (Strand et al., 2009). In such stands we do not expect a fire event to return the landscape unit to young aspen woodland but rather to young conifer woodlands, resulting in permanent loss of aspen within the landscape unit (fig. 1).

The current wildfire size distribution was calculated from a fire database provided by the Interior Columbia Basin Ecosystem Management Project (http://www.icbemp.gov/) for the interior Columbia River basin between 1986–1992. The maximum allowable area burned in prescribed fires was set to 1000 ha per year in scenarios that included prescribed fire.

Current wildfire probability of occurrence in each PVT and structural stage was computed from an overlay analysis in a GIS (ESRI 1999–2005) of digital fire atlas data from 1957–2002 and a recently developed landcover map for the Owyhee Plateau (Roth 2004). Historic wildfire probabilities were estimated based on the 40–60 year fire interval suggested by Jones and DeByle (1985a) for aspen woodland with increasing fire probability later in succession where flammable conifers are present. The fire occurrence probability for juniper woodlands at their initiation was derived from the 40–50 year mean fire return interval suggested by Burkhardt and Tisdale (1976). As western juniper woodlands mature, there is a decrease in understory productivity resulting in lower amounts of fine fuels and a reduced ability to carry fire in these older woodlands (Miller et al. 2005, Bunting et al. 2007). For mid- and late seral juniper woodlands, we employed fire occurrence probabilities used by Bunting et al. (2007).

During a TELSA simulation, fires start in random locations according to the assigned disturbance probability. A fire that starts in a landscape unit can spread into an adjacent landscape unit if that unit is eligible for fire disturbance. The size of wildfires and prescribed fires were randomly assigned to each fire based on the pre-defined fire size probability distribution.

Six major assumptions and simplifications relating to aspen ecology and succession are important parts of this model. They are:

1) Aspen reproduction from seed is not included in this model.
2) Aspen are not allowed to spread laterally into other PVTs (e.g. sagebrush).
3) Adjacency between vegetation types does not affect succession.
4) Fire will convert a conifer dominated aspen stand to a young aspen stand regardless of the pre-disturbance conifer cover in the stand, i.e. no legacy effects are considered.
5) Aspen stands are permanently converted to conifer stands 120 years after aspen suckering has ceased due to conifer dominance (i.e. ~230 years after conifer initiation into the stand).
6) Effects of insects, disease, and animal use are not included in this model.

The potential effects of these assumptions and simplifications on model outcome and interpretation are addressed in the Discussion section.

Model Scenarios

To determine whether the assigned model parameters were realistic, we tested the model by subtracting 100 years from the age of each landscape unit followed by a simulation 100 years into the future using assigned successional rates, fire probabilities, and fire size distributions. The actual current landscape composition was then compared to the modeled composition. Future landscape compositions for the two study areas were evaluated at 25, 50, 100 and 200 years from current time. Fire management regimes included:

Scenario 1: Current fire management i.e. suppressed wildfire only.

Scenario 2: Historic wildfire probabilities.

Scenario 3: Historic wildfire probabilities with larger fires.

Scenario 4: Prescribed fire in aspen/conifer woodlands according to historic fire probabilities, no prescribed fire applied in other cover types.

Table 3—The TELSA model requires estimates of the disturbance size distribution as part of the input. This table describes the percent of fires in each size class for the five simulation scenarios. The current wildfire size distribution was estimated from the Interior Columbia Basin Ecosystem Management Project geographic database (ICBEMP 1995)

Scenario	Fire size 0-1 ha	Fire size 1-10 ha	Fire size 10-100 ha	Fire size 100-1000 ha
1	90	5	3	2
2	90	5	3	2
3	50	20	15	15
4	1	4	25	70
5	1	4	25	70

Table 4—This table described the current and historic probability of wildfire occurrence in the major PVTs and structural stages on the Owyhee Plateau

PVT	Structural stage	Current wildfire probability	Historic wildfire probability
Low sagebrush steppe	Grassland	0.00064	0.002
	Low sagebrush steppe	0.00064	0.005
Mtn big sagebrush steppe	Grassland	0.001	0.002
	Mtn big sagebrush steppe	0.001	0.02
Juniper woodlands/Low sagebrush steppe	Grassland	0.00064	0.002
	Low sagebrush steppe	0.00064	0.02
	Stand initiation juniper	0.0008	0.01
	Open young woodland	0.0008	0.001
	Young multistory woodland	0.0005	0.002
	Old multistory woodland	0.0004	0.006
Juniper woodlands/Mtn. big sagebrush steppe	Grassland	0.001	0.005
	Mtn. Big sagebrush steppe	0.001	0.02
	Stand initiation juniper	0.001	0.02
	Open young woodland	0.0007	0.01
	Young multistory woodland	0.0002	0.002
	Old multistory woodland	0.00009	0.001
Aspen woodlands/conifer	Young woodlands	0.0002	0.0002
	Mature woodlands	0.0002	0.005
	Woodlands with conifer	0.0002	0.01
	Conifer/aspen woodland	0.0002	0.0

Scenario 5: Prescribed fire in aspen/conifer woodlands and young juniper woodlands according to historic fire probabilities.

Although succession in TELSA is treated as a deterministic variable with a pre-determined time period between transitions, fire starts and fire size are stochastic components in the model. Because of this stochastic element, the model results will vary between runs even though the input variables and landscape maps are identical. Simulations were therefore run 10 times for each management scenario to quantify the variability between runs. Means and variances were calculated and displayed as error bars in the resulting graphs.

Results

Fire Occurrence, Size, and Probabilities

Fire perimeter data from the Bureau of Land Management (BLM) 1957–2002 show that only 94 ha of the combined

Table 5—This table provides a comparison of the current cover type distribution and the 100-year simulated current cover type distribution for South Mountain

Cover type	Current area ha	Simulated current ha
Aspen	2611	2610
Ceanothus / Mesic shrub	477	362
Curlleaf mountain-mahogany	223	117
Douglas-fir	298	284
Grasslands/Meadow	70	402
Juniper woodland	10193	11831
Sagebrush steppe	3053	1136

37 000 ha study region has burned in wildfires within this time period. Overlay analysis in GIS reveals that none of these fires occurred on soils that support aspen woodlands. Fire records prior to 1957 are not available. Prescribed fire in aspen stands has occurred on the Owyhee Plateau, but to this date not in modeled areas.

The current wildfire size distribution was estimated from the Interior Columbia Basin Ecosystem Management Project database (ICBEMP 1995, table 3), which indicates that most wildfires within the region become less than 1 hectare in size. Information about the historical wildfire size distribution is not available for the study area and we therefore simulated two historical wildfire scenarios with two different fire size distributions (scenarios 2 and 3, table 3) to test the sensitivity of fire size within the model. In scenario 2 we used the same fire size distribution as scenario 1 (90 percent of fires become < 1 ha in size) while in scenario 3 the proportion of fires larger than 1 ha was increased (see table 3 for more detail). Commonly, prescribed fires are in the size class 10-1000 ha (scenarios 4 and 5, table 3). Current wildfire probabilities were estimated via overlay analysis between current cover types (Roth, 2004) and the digital fire atlas obtained from the BLM for the time period 1957–2002. Historical wildfire probabilities were based on literature references (DeByle et al. 1987, Bunting et al. 2007; see table 4).

Management Scenarios

To evaluate the input model parameters, we tested the model by subtracting 100 years from the age of each landscape unit followed by a simulation 100 years into the future using assigned successional rates, fire probabilities, and size distributions. We compare the resultant modeled landscape composition to the actual current landscape composition in table 5. The model accurately simulated the current area of aspen using the inputs from 100 years back in time. The simulated area of juniper woodlands was larger, and the area in sagebrush steppe and grasslands was smaller than observed. These results suggest that the simulated successional rates within the juniper PVTs are slightly overestimated in the model. We attribute this to the fact that the juniper successional models were developed in a different study area on Juniper Mountain south of South Mountain.

Future landscape composition of aspen seral stages was predicted under varying management scenarios for South Mountain and the Silver City Range (figs. 3 and 4). Under current wildfire regimes the early, mid, and late seral woodlands are predicted to decrease within the next 100 years while the old woodlands are predicted to increase. Continuation of current fire management is predicted to result in loss of aspen woodlands within the next 100 years, with additional losses in the following century.

Modeled historical fire regimes predicted an increase in early and mid seral woodlands while the area in late seral woodlands decreased and old woodlands remained at current levels. Scenarios 2 and 3, historic fire probabilities with smaller and larger fire size distributions, yielded similar results with an increase in the mean area of the early and mid seral aspen classes for the scenario with larger fire size compared to the smaller fire size. This difference, however, falls within the variability of the 10 runs (figs. 3 and 4).

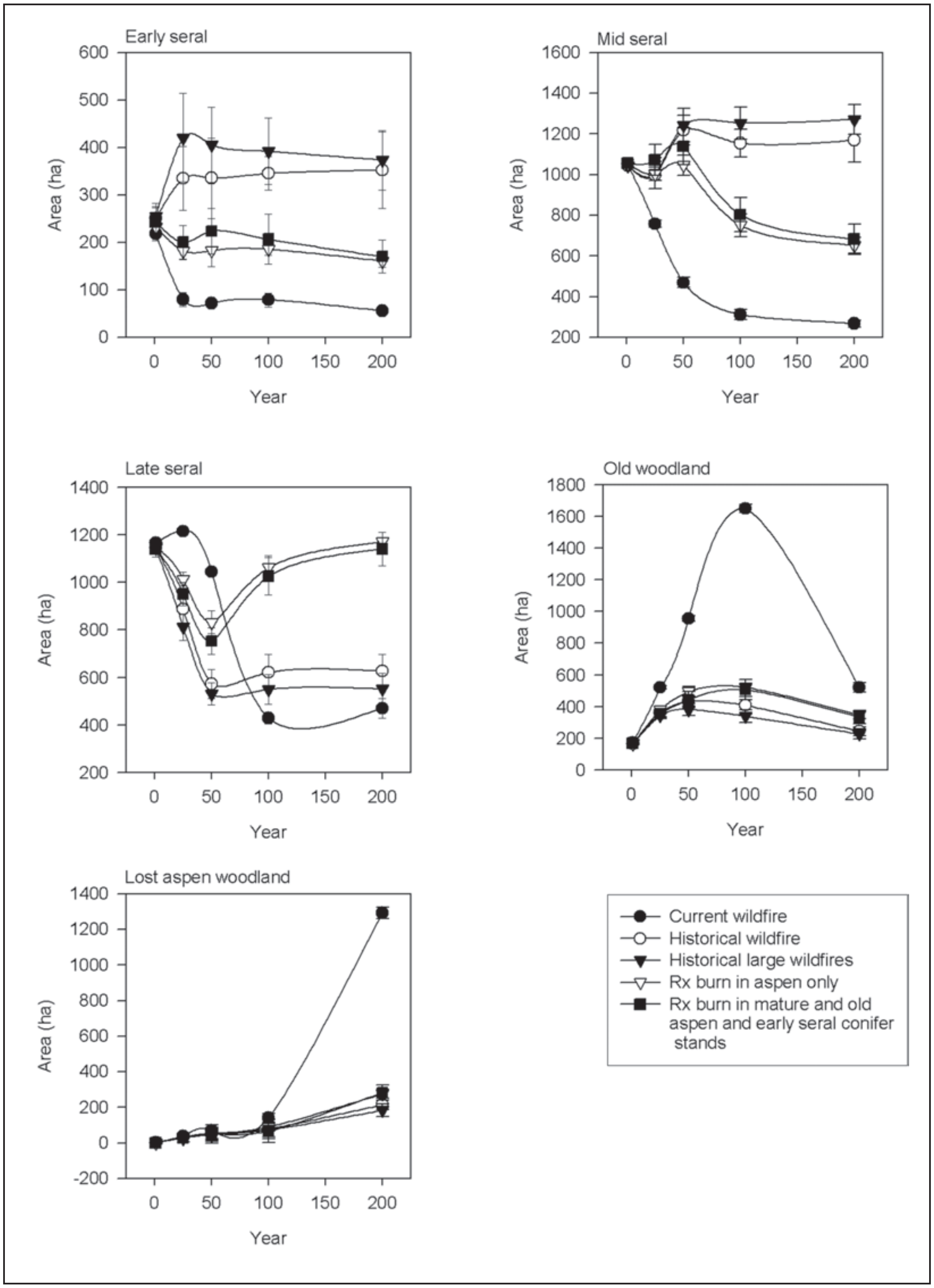

Figure 3—Area of aspen woodland in different seral stages under five simulated management scenarios on South Mountain. The total area in aspen vegetation is currently 2610 ha.

Prescribed fire applied in aspen only (scenario 4) and in aspen and young juniper (scenario 5) resulted in a decrease in early and mid seral aspen woodlands. The area in late seral aspen woodlands initially decreased but reached a stable level, similar to the current area, approximately 100 years into the future. The area in old aspen and the loss

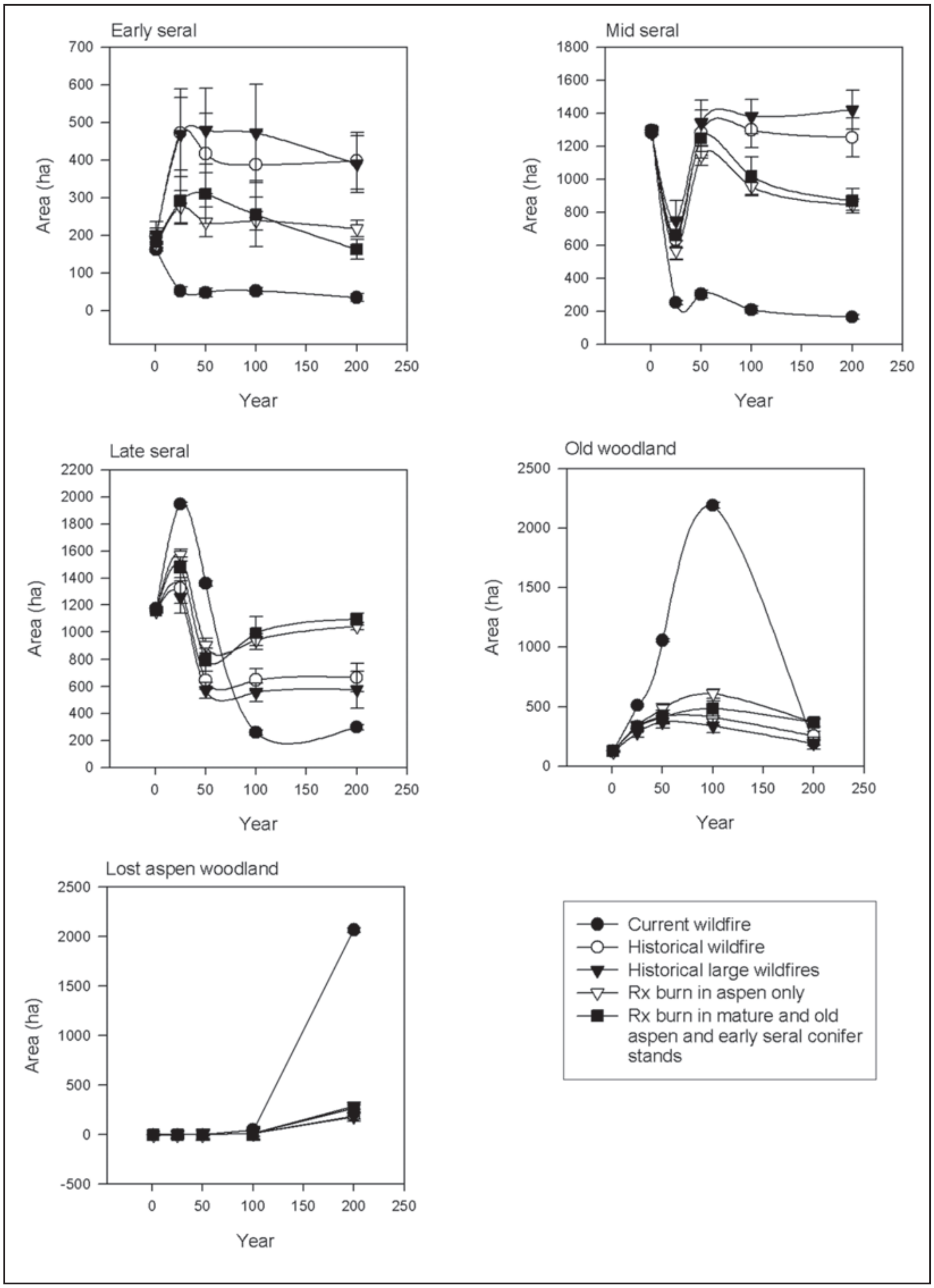

Figure 4—Area of aspen woodland in different seral stages under five simulated management scenarios in the Silver City Range. The total area in aspen vegetation is currently 2765 ha.

of aspen is similar for the prescribed fire and historical fire management scenarios. Under historical fire regimes a larger portion of the landscape was stable in mid seral woodlands, while for the prescribed fire simulations a larger portion of the area stabilized in late seral woodlands. These predictions indicate that the aspen loss can largely be

Table 6—Fire rotation and decadal proportion of the landscape burned under modeled fire regimes

Study area	Scenario	Fire rotation (years)	Fire area per decade (percent)
South Mountain	Current wildfire (1)	340	2.9
South Mountain	Historic fire probabilities (2)	82	12.2
South Mountain	Historic prob. large fires (3)	72	13.9
South Mountain	Prescribed fire in aspen (4)	466	2.1
South Mountain	Prescribed fire in aspen+young juniper (5)	192	5.2
Silver City	Current wildfire (1)	449	2.2
Silver City	Historic fire probabilities (2)	79	12.7
Silver City	Historic prob. large fires (3)	66	15.1
Silver City	Prescribed fire in aspen (4)	448	2.2
Silver City	Prescribed fire in aspen+young juniper (5)	178	5.6

mitigated by implementing appropriate prescribed burning programs.

Fire rotation is a measure of how many years it would take to burn an area equal to the study area under a given fire regime. Under historical fire probabilities, our simulations indicate that the fire rotation for the two study areas was 70–80 years, while at current fire management the estimated fire rotation was 340 years on South Mountain and 449 years in the Silver City area (table 6). Fire rotations were also computed for the prescribed fire scenarios, although these numbers may not be meaningful for aspen management because the simulated prescribed fire programs here target aspen stands. According to this model, the historical fire regimes—which are able to maintain the majority of aspen stands in early and mid seral woodlands—required that approximately 12–14 percent of the area burns per decade. Currently, only 2–3 percent of the landscape burns per decade, of which the majority of the burned area is sagebrush steppe rather than juniper or aspen woodlands.

Aspen Expansion

Given the aspen expansion rate into sagebrush of approximately 0.5 m per year (20 m expansion in 40 years) and the length of the aspen/sagebrush steppe boundary within the South Mountain study area, the maximum area gained by aspen clones in 100 years would be 340 ha, corresponding to 13 percent of the current aspen cover. These results indicate how much assumption 2, "Aspen is not allowed to spread laterally in the model", affects the interpretation of the model results. Although we realize that the expansion rate likely varies with annual precipitation, site productivity, and other environmental conditions, the average expansion rate estimated here provides a guideline for assumptions made regarding the importance of aspen expansion for landscape composition.

Discussion

Fire Disturbance and Landscape Dynamics

Modeling results suggest that under a continuation of current fire regimes, aspen will continue to decline on both South Mountain and in the Silver City Range. Current mid- and late seral aspen/conifer stands will continue to age over the next 50–100 years and eventually become permanently converted to conifer woodlands in the absence of disturbance (figs. 3 and 4). Through simulations of succession-disturbance dynamics in TELSA under current and historic fire regimes and prescribed fire scenarios, we are able to address the four questions posted in the introduction.

1) Can we simulate the historical fire regime that maintained aspen stands prior to Euro-American settlement?

Results produced under the historical fire conditions yield a landscape where over half of the aspen area is in early or mid seral successional classes and the loss of aspen is low. The distribution between successional stages is:

14 percent in the early seral stage, 45 percent in mid seral and 35 percent in late seral (late seral and old combined, see figs. 3 and 4). We predict an ~ 6 percent loss of aspen (compared to the current area occupied by aspen) over the 200 year simulated time period even under historic fire regimes, which is likely due to caveats in the model assumptions. Within the model there is no avenue for aspen recruitment via seed or expansion of aspen into previously aspen free habitats. Under stochastic and randomly distributed application of fire, by necessity, some aspen stands will by chance escape fire for a long enough time period to convert to conifer woodlands. Sexual reproduction of aspen is not likely to occur in the West, although such infrequent severe fire events enabling seedling establishment may be important for aspen regeneration long term. This model also did not include expansion of aspen into shrub and grasslands. We here estimate that the maximum estimated expansion rate for aspen on South Mountain (340 ha in 100 years or 13 percent of the current aspen area) would more than counteract the predicted loss of 6 percent in our model.

Whether this model scenario is indeed a fair representation of fire regimes prior to European settlement is difficult to assess, but comparisons can be made to independent estimates from other researchers. Our simulated historical fire regime resulted in a fire rotation of 70–80 years, which is somewhat longer than the mean fire frequency of 50 years suggested by Jones and DeByle (1985a). We also compared the area in successional classes to predictions presented as part of the LANDFIRE Rapid Assessment Reference Condition Models. For the aspen biophysical setting in mapping zone 18, which includes southern Idaho, the suggested distribution among successional stages is 14 percent in early seral, 40 percent in mid seral and 45 percent in the late seral class, which is very similar to our modeled results. Loss of aspen is avoided in the LANDFIRE reference models by including an insect/disease outbreak every 200 years, which reverts aging aspen stands to earlier successional stages.

2) What extent and frequency of fire (burned area per decade) is required to stabilize the current land cover composition within aspen woodlands?

Under historical conditions we predict that 12–14 percent of the landscape burned per decade and that this amount of fire largely maintained the aspen stands in early and mid seral stages. Current fire regimes, resulting in approximately 2 percent of the landscape burned per decade, is (according to model predictions) clearly not enough to avoid aspen loss or to maintain aspen in early and mid seral stages. Prescribed fire applied in aspen and young juniper woodland results in 5–6 percent of the landscape burned per decade while application of fire in aspen stands only results in 2 percent of the landscape burned per decade. By targeting only aspen/conifer stands, aspen could theoretically be kept on the landscape with minimal burning efforts. In reality this may not be a feasible management scenario considering that all surrounding conifer woodlands would be allowed to mature to late successional stages providing an increasing source of conifer seeds and probability for conifer establishment. Application of prescribed fire in both aspen and young juniper according to historic fire occurrence probabilities would both maintain aspen in a younger stage and minimize the source of conifer seeds. Prescribed fire applied also in mature juniper woodlands was not considered due to the practical difficulty of burning such areas. In both prescribed fire scenarios, all conifer woodlands that currently exist in mature successional stages would therefore continue to mature and remain on the landscape.

3) What is the structural composition within aspen woodlands under historical and current fire probabilities? What is the structural composition under prescribed burning scenarios?

Landscape composition at user selected times is reported by TELSA at defined disturbance regimes and initial landscape composition. The initial landscape composition is only important to gain understanding about a certain study area over a relatively short period. As the model is allowed to run for a sufficiently long time period the landscape composition at the equilibrium state is independent of the initial landscape composition. Under historic fire regimes approximately 60 percent of the aspen woodlands exist in an early or mid successional stage, while this proportion is ~10 percent for current fire regimes and ~30 percent for the prescribed burning scenarios. Under prescribed burning scenarios ~45 percent of the aspen develop into late 406 seral woodlands, of which the majority is the self-regenerating pure aspen stands where 407 prescribed fire was not applied. The amount of aspen in the old successional class and lost aspen woodlands is similar in the historic and prescribed burning scenarios (figs. 3 and 4).

4) What is the effect of fire size on the long-term maintenance of aspen woodlands?

Historical fire regimes (scenarios 2 and 3) were simulated with two fire size distributions (table 3). Although the scenario with larger fires (scenario 3) results in a larger area in early and mid seral woodlands, the difference is within the error bar generated for multiple runs. Based on these results we conclude that there is marginal effect of fire size on the structural composition of aspen woodlands and the long-term maintenance of aspen woodlands. It is important to note that these results in the "model world" do not necessarily apply to the "real world". A closer evaluation of the model assumptions leads us to believe that this model is not well suited to answer question 4. One could speculate that larger fires would benefit the fire dependent aspen woodlands in several ways. Larger fires would reduce the conifer seed source and probability of conifer establishment within newly established aspen stands. Modeling of this phenomenon would require the spatial model to account for seed dispersal to adjacent stands such that aspen stands that are closer to conifer woodlands would be more likely to experience conifer establishment and eventually dominance. Larger fires would also clear larger areas, into which aspen could expand. Aspen clones surrounded by closed conifer woodlands have no means of extending their area. The ability for aspen to expand into adjacent grass and shrub lands was not incorporated in this model. An improved model where the distance to seed source and expansion of existing aspen stands were included would likely show different results with regards to the importance of fire size.

Model Assumptions and Their Potential Effects on Model Outcomes

The full complexity of ecosystem interactions is neither feasible nor necessary to capture in a model to improve the understanding for how the system functions. The model presented here is a form of deductive reasoning where the model results are a product of the input data and model assumptions. In the following section, we discuss the major assumptions and their potential effect on model outcomes.

1) *Aspen reproduction from seed is not included.* Although aspen in the western mountains reproduce primarily via vegetative suckering (Baker 1925, Barnes 1975, Mitton and Grant 1996, Romme et al. 2005), recruitment via sexual reproduction has occurred after severe fires such as the 1988 fires in Yellowstone National Park (Romme et al. 2005). We did not include the occurrence of such infrequent and severe fires because the occurrence probability and the probability of aspen establishment are unknown. Also, such a fire is unlikely to occur within the modeled time period due to the stochastic nature of these events combined with fire suppression. Such large infrequent fire events represent non-equilibrium conditions (Turner and Romme 1994) over the spatial and temporal extents addressed in this model. Including infrequent severe fires leading to aspen regeneration by seed would require modeling over a much longer time period and extent.

2) *Aspen cannot spread into other potential vegetation types.* Expansion of aspen into adjacent shrub- or grasslands has been observed (Manier and Laven 2001). We calculated that aspen on South Mountain could expand as much as 340 ha in 100 years (13 percent of the current aspen cover) in the absence of fire if all aspen along aspen/sagebrush boundaries were expanding. This expansion would to some extent counteract the small aspen loss predicted under historical fire regime scenarios.

3) *Adjacency between vegetation types does not affect succession.* In our model, the presence of a conifer seed source near an aspen stand does not affect the rate of succession. Incorporation of such effects would result in variability in successional rates between stands far away and close to conifers. Considering adjacent conifer seed sources would increase successional rates in scenarios where only

aspen stands are burned while conifer stands are left to mature and become a neighboring seed source.

4) *Fire will convert a conifer dominated aspen stand to a young aspen stand regardless of the pre-disturbance conifer cover in the stand, i.e. no legacy effects are considered.* It can be expected that an aspen stand with high cover of seed producing conifers is more likely to experience more rapid succession after a fire than a stand with only a few conifer seedlings pre-fire. Western juniper seeds, for example, are persistent in the seed bank (Chambers et al., 1999) and may survive a low severity fire and hence become an immediate source of juniper seedlings after a fire.

5) *Aspen stands are permanently converted to conifer stands 120 years after aspen suckering has ceased due to conifer dominance, i.e. ~230 years after conifer initiation into the stand.* Reduced vegetative reproduction in aspen stands that are becoming dominated by conifers has been observed by several researchers (Bartos and Campbell 1998, Kaye et al. 2005, Strand et al. 2009). It is however not known how long an aspen clone can remain dormant in a non-reproductive state and still return to an aspen woodland after a fire, hereafter referred to as the persistence time. The actual time an aspen clone can remain under conifer dominance could be significantly different from 120 years. The 120-year time period was selected because this can be considered the life expectancy of existing mature aspen ramets in the conifer-dominated stand. When all mature ramets are gone and the stand is no longer regenerating, permanent loss of the stand is assumed to occur resulting in a change from an aspen/conifer PVT to a conifer PVT. Strand et al. (2009) show that the length of the persistence time only affects the starting point of rapid aspen decline (see figs. 3 and 4). The length of the persistence time is also extremely important when considering the possibility that one avenue for aspen rejuvenation is infrequent intense wildfires creating a substrate suitable for aspen seedling establishment. In a scenario of effective fire suppression where large intense fires (ones not possible to suppress) occur at an interval longer than the persistence time for all aspen clones in the area, local extinction of aspen will occur in aspen/confer PVTs.

6) *Effects of insects, disease, and animal use on aspen and conifers are not included in this model.* Fire is the only disturbance included in this model, although previous work has demonstrated that insects, disease, animal browsing, and wind felling are examples of other disturbances affecting aspen and conifer succession (Jones and DeByle 1985b, Jones et al. 1985, Kay and Bartos 2000, Kaye et al. 2005). We deliberately omitted these disturbance agents in the model to gain a clearer understanding of the effects of fire disturbance alone on the ecosystem. The LANDFIRE rapid assessment program (http://www.Landfire.gov) has produced a series of reference condition (RC) models, which provide an estimate of the expected distribution of successional classes under pre-European settlement conditions. The LANDFIRE RC model for aspen in the northern Great Basin incorporates an insect/disease disturbance in aging aspen/conifer stands every 200 years which restores aspen to an earlier successional state and maintains aspen on the landscape. Regardless of whether the infrequent catastrophic event is a large severe fire promoting sexual reproduction of aspen, an infrequent disease outbreak, or a land- slide, it is questionable whether managers of aspen resources can rely on such infrequent stochastic events for ecosystem maintenance. Kulakowski et al. (2006, p. 1397) state that "*human perceptions of ecosystems are often on time scales that are shorter than the cycles of natural variation within ecosystems*". With the help of field observations, mapping, and modeling we can begin to comprehend aspen ecosystem succession and disturbance dynamics at multiple spatial and temporal scales. The question is, can we manage aspen and other resources at such broad temporal scales?

Management Implications

Over long time periods (i.e. centennial) aspen will most likely remain a part of the western landscape unless the climate changes drastically such that it is unfavorable for the species. Quaking aspen are apparently tolerant to a variety of fire frequencies and severities; vegetative reproduction occurs when fires are less severe and more frequent. Reproduction via seed can occur after extensive severe fire events if the soil moisture and weather conditions are within the 'window of opportunity' for aspen regeneration (Romme et al. 2005). Therefore, even if aspen that is seral to conifers are eliminated from the landscape due to fire suppression, eventually a large-scale disturbance event will likely occur and pure aspen stands, riparian aspen, and aspen occurring on microsites may provide seed for aspen recruitment and establishment. This optimistic outlook does not offer a solution to the immediate concern over the current aspen declines across the West. Human activity and needs, and current fire policy makes it unlikely that aspen woodlands within the West will return to historic fire regimes and active management has been proposed in locations where maintenance of aspen is a priority. Before engaging in management activities it is naturally important to make appropriate ecological field assessments to evaluate the current state of the aspen stands, their successional trajectories in a landscape context, and the presence of possible stressors.

In this analysis we show via modeling that the historical fire frequency suggested by Jones and DeByle (1985a) maintains aspen on the landscape. In many areas it is not feasible or desirable to return to historic fire regimes, and prescribed burning may be an alternative. Model predictions suggest that in theory prescribed burning programs can mitigate aspen loss and maintain aspen woodlands in younger seral stages. Restoration of aspen woodlands has been suggested (Bartos et al. 1991, Brown and DeByle 1989, Miller et al. 2005) and such restoration projects (e.g., Bates and Miller 2004, Bates et al. 2004, Brown and DeByle 1989) have been carried out by managers. Ecological factors that must be considered prior to burning are the fuels composition and structure, current understory composition, presence of weeds, and the successional stage of aspen woodland development (Miller et al. 2005). Other concerns are post-fire wildlife and animal use (Bartos and Campbell 1998, Kay and Bartos 2001, Kaye et al. 2005), which can jeopardize aspen suckers and prevent the aspen clone recovery. Post-treatment monitoring is recommended to better understanding the browsing pressure on the treated aspen clone.

Where fire is undesirable for restoration, Shepperd (2001) has suggested a series of alternative management activities including commercial harvest, mechanical root stimulation, removal of competing vegetation, protection of regeneration from herbivory and regeneration from seed. Cutting of conifers followed by prescribed fire has also been applied (Bates and Miller 2004). The felled conifers provide a fuel ladder that help carry the fire in aspen stands which are commonly difficult to burn.

Ecosystem management requires assessment of interactions among succession, natural disturbance regimes and management activities. Landscape dynamics models such as TELSA provide an avenue for managers, scientists, and stakeholders to evaluate the long-term effect of changing natural disturbance regimes and management activities on landscape vegetation composition. All models have limitations. It is important to clearly understand the model assumptions during interpretation of model results and during the decision making process. The ultimate test of a model is not how accurate or truthful it is, but only whether one is likely to make a better decision with it than without it (Starfield 1997).

The modeling results presented here indicate that active management is necessary in areas where aspen are seral to conifers and aspen maintenance is a management goal unless we rely on infrequent severe disturbance events to maintain these aspen resources. Reliance on severe disturbance will likely lead to continued decline of aspen in our study region and across the western U.S.

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References

C.B. 2012. The roles of hydraulic and carbon stress in a widespread climate-induced forest die-off. PNAS. 109: 233–237.

Baker, F.S. 1925. Aspen in the central Rocky Mountain region. USDA Bull. 1291, 47 p.

Barnes, B.V. 1975. Phenotypic variation of trembling aspen in Western North America. Forest Science. 22: 319–328.

Bartos, D.L., 2001. Landscape Dynamics of Aspen and Conifer Forests. In: Sustaining Aspen in Western Landscapes: Symposium Proceedings; 13–15 June 2000. Grand Junction, CO: USDA Forest Service Proceedings RMRS-P-18: 5–14 p.

Bartos, D.L.; Campbell, R.B. 1998. Decline of Quaking Aspen in the Interior West – Examples from Utah. Rangelands. 20: 17–14.

Bartos, D.L.; Mueggler, W.F. 1981. Early succession in aspen communities following fire in western Wyoming. Journal of Rangeland Management. 34: 315–318.

Bartos, D.L.; Mueggler, W.F.; Campbell, R.J. 1991. Regeneration of aspen by suckering on burned sites in western Wyoming. Forest Research Paper INT-448. Ogden, UT: USDA Forest Service, Intermountain Research Station. 10 p.

Bates, J.D.; Miller, R.F. 2004. Restoration of aspen woodland invaded by western juniper: Applications of partial cutting and prescribed fire. 16th International Conference, Society for Ecological Restoration, August 24–26, Victoria, BC, Canada.

Bates, J.D.; Miller, R.F.; Davies, K.W. 2004. Restoration of Quaking Aspen Woodlands Invaded by Western Juniper. Rangeland Ecology and Management. 59: 88–97.

Brown, J.K.; DeByle, N.V.1989. Effects of prescribed fire on biomass and plant succession in Western aspen, Ogden UT: United States Department of Agriculture, Forest Service, Intermountain Research Station, Research Paper INT-412.

Bunting, S.C.; Strand, E.K.; Kingery, J.L. 2007. Landscape characteristics of sagebrush- steppe/ juniper woodland mosaics under varying modeled prescribed fire regimes. In: Masters, R.E.; Galley, K.E.M., eds. Proceedings of the 23rd Tall Timbers Fire Ecology Conference: Fire in Grassland and Shrubland Ecosystems, Tall Timbers Research Station. Tallahassee, FL, US: 50–57.

Burkhardt, J.W.; Tisdale, E.W. 1976. Causes of juniper invasion in southwestern Idaho. Ecology. 57: 472–484.

Chambers, J.C.; Vander Wall, S.B.; Schupp, E.W. 1999. Seed and seedling ecology of pinyon and juniper species in the pygmy woodlands of western North America. Botanical Review. 65: 1–38.

DeByle, N.V.; Bevins, C.D.; Fischer, W.C. 1987. Wildfire occurrence in aspen in the interior western United States. Western Journal of Applied Forestry. 2: 73–76.

Despain, D.G. 1990. Yellowstone vegetation: Consequences of environment and history in a natural setting. Roberts Rinehart Publishers, Inc., Boulder, CO.

Essa Technology, 2003. Tool for Exploratory Landscape Scenario Analyses (TELSA), version 3.1a, ESSA Technologies Ltd., Vancouver, B.C., Canada.

Franklin, J.; Syphard, A.D.; Mladenoff, D.J.; He, H.S.; Simons, D.K.; Martin, R.P.; Deutschman, D.; O'Leary, J.F. 2001. Simulating the effect of different fire regimes on plant functional groups in Southern California. Ecological Modeling. 142: 261–283.

Hessl, A. 2002. Aspen, elk, and fire: The effects of human institutions on ecosystem processes. Bioscience. 52: 1011–1022.

Huang C.-Y.; Anderegg, W.R.L. 2011. Large drought-induced aboveground live biomass losses in southern Rocky Mountain aspen forests. Global Change Biology DOI: 10.1111/j.1365-2486.2011.02592.x.

ICBEMP, 1995. Interior Columbia River Basin Ecosystem Management Project, Disturbance - Fire Locations, Intermountain Fire Science Lab - Missoula, MT, Online Linkage: http://www.icbemp.gov/spatial/disturbance/ (last accessed December 2011).

Jones, J.R.; DeByle, N.V., 1985a. Fire. In: DeByle and Winokur. eds. Aspen: ecology and management in the Western United States. Gen. Tech. Rep. Fort Collins, CO: U.S. Department of Agriculture, Forest Service, Rocky Mountain Research Station: RM-119: 77–81.

Jones, J.R.; DeByle, N.V., 1985b. Other physical factors. In: DeByle and Winokur, eds. Aspen: ecology and management in the Western United States. Gen. Tech. Rep. RM-119. Fort Collins, CO: U.S. Department of Agriculture, Forest Service, Rocky Mountain Research Station: 83–86.

Jones, J.R., DeByle, N.V., Bowers, D.M., 1985. Insects and other invertebrates. In: DeByle and Winokur, eds. Aspen: Ecology and Management in the Western United States. Gen. Tech. Rep. Fort Collins, CO: U.S. Department of Agriculture, Forest Service, Rocky Mountain Research Station: RM-119. 107–114.

Kay, C.E. 1997. Is aspen doomed? Journal of Forestry. 95: 4–11.

Kay, C.E.; Bartos, D.L. 2000. Ungulate herbivory on Utah aspen: Assessment of long term exclosures. Journal of Range Management. 53: 145–153.

Kaye, M.W.; Binkley, D.; Stohlgren, T.J. 2005. Effects of conifers and elk browsing on quaking aspen forests in the central Rocky Mountains, USA. Ecological Applications. 15: 1284–1295.

Keane, R.E.; Long, D.G.; Basford, D.; Levesque, B.A. 1997. Simulating vegetation dynamics across multiple scales to assess alternative management strategies: Proceedings for GIS '97; 1997 February 17–20. Vancouver B.C. Fort Collins, CO; GIS World Books. 640 p.

Klenner, W.; Kurz, W.A.; Beukema, S. 2000. Habitat patterns in forested landscapes: management practices and the uncertainty associated with natural disturbances. Computers and Electronics in Agriculture. 27: 243–262.

Kulakowski, D.; Veblen, T.T.; Kurzel, B.P. 2006. Influences of infrequent fire, elevation and pre-fire vegetation on the persistence of quaking aspen (*Populus tremuloides* Michx.) in the Flat Tops area, Colorado, USA. Journal of Biogeography. 33: 1397–1413.

Manier, D.J.; Laven, R.L. 2001. Changes in landscape pattern and associated forest succession on the western slope of the Rocky Mountains, Colorado. In: Sustaining Aspen in Western Landscapes: Symposium Proceedings; 13–15 June 2000. Grand Junction, CO. USDA Forest Service Proceedings RMRS-P-18: 15–22.

Miller, R.F.; Bates, J.D.; Svejcar, T.J.; Pierson, F.B.; Eddleman, L.E. 2005. Biology, ecology and management of western juniper. Technical Bulletin 152, Oregon State University Agricultural Experiment Station, Corvallis, OR.

Mitton, B.J.; Grant, M.C. 1996. Genetic variation and the natural history of quaking aspen. Bioscience. 46: 25–31.

Mueggler, W.F. 1989. Age distribution and reproduction of intermountain aspen stands. Western Journal of Applied Forestry. 4: 41–45.

Oregon Climate Service 1999. Precipitation for Idaho; Average Monthly and Annual (1961–90), digital raster data, http://inside.uidaho.edu, University of Idaho Library, Moscow, ID, USA.

Patterson, A.E. 1959. Distinguishing annual rings of diffuse porous tree species, Journal of Forestry 57: 126.

Rogers, P.C.; Leffler, A.J.; Ryel, R.J. 2010. Landscape assessment of a stable aspen community in southern Utah, USA. Forest Ecology and Management. 259: 487–495.

Romme, W.H.; Turner, M.G.; Tuskan, G.A.; Reed, R.A. 2005. Establishment, persistence, and growth of aspen (*Populus tremuloides*) seedlings in Yellowstone National Park: Ecology. 86: 404–418.

Roth, A. 2004. Fire patterns within a successional gradient of a sagebrush steppe/juniper woodland, M.S. Thesis, University of Idaho, Moscow, ID, USA.

Shepperd, W.D. 2001. Manipulations to regenerate aspen ecosystems. p. 355–365. Sustaining Aspen in Western Landscapes: Symposium Proceedings; 13–15 June 2000; Grand Junction, CO. USDA Forest Service Proceedings RMRS-P-18.

Shepperd, W.D.; Bartos, D.L.; Mata, S.A. 2001. Above- and below-ground effects of aspen clonal regeneration and succession to conifers. Canadian Journal of Forest Research. 31: 739–745.

Smith, A.E.; Smith, F.W. 2005. Twenty-year change in aspen dominance in pure aspen and mixed aspen/conifer stands on the Uncompahgre Plateau, Colorado, USA. Forest Ecology and Management. 213: 338–348.

Starfield, A.M. 1997. A pragmatic approach to modeling for wildlife management. Journal of Wildlife Management. 61: 261–270.

Strand, E.K.; Vierling, L.A.; Bunting, S.C.; Gessler, P.E. 2009. Quantifying successional rates in western aspen woodlands: Current conditions, future predictions. Forest Ecology and Management. 257(8): 1705–1715.

Strand, E.K. 2007. Chapter 4: Landscape dynamics in aspen and western juniper woodlands on the Owyhee Plateau, Idaho. Moscow, ID: University of Idaho. Ph.D. dissertation.

Turner, M.G., Romme, W.H., 1994. Landscape dynamics in crown fire ecosystems. Landscape Ecology. 9: 59–77.

U.S. Department of Agriculture, National Resouscer Conservation Service [USDA NRCS], 1998. Owyhee County digital soil survey, SSURGO, Natural Resources Conservation Service, U.S. Department of Agriculture. On line: 164 http://www.ncgc.nrcs.usda.gov/branch/ssb/products/ssurgo/.

van Mantgem, P.J.; Stephenson, N.L. 2007. Apparent climatically-induced increase of mortality rates in a temperate forest. Ecology Letters. 10: 909–916.

van Mantgem, P.J.; Stephenson, N.L.; Byrne, J.C.; Daniels, L.D.; Franklin, J.F.; Fulé, P.Z., Harmon, M.E.; Larson, A.J.; Smith, J.M.; Taylor, A.H.; Veblen, T.T. 2009. Widespread increase of tree mortality rates in the western United States. Science. 323: 521–524.

Wall, T.G.; Miller, R.F.; Svejcar, T.J. 2001. Juniper encroachment into aspen in the Northwest Great Basin. Journal of Range Management. 54: 691–698.

Western Regional Climate Center [WRCC]. 2003. Division of Atmospheric Sciences. Desert Research Institute. Reno, NV. http://www.wrcc.dri.edu.

Winternitz, B.L. 1980. Birds in aspen. Management of western forests and grasslands for nongame birds. Gen. Tech. Rep. INT-86. Ogden, UT: U.S. Department of Agriculture Forest Service, Intermountain Forest and Range Experiment Station.

Worrall, J.J.; Egeland, L.; Eager, T.; Mask, R.A.; Johnson, E.W.; Kemp, P.A.; Shepperd. W.D. 2008. Rapid mortality of *Populus tremuloides* in southwestern Colorado, USA. Forest Ecology and Management. 255: 686–696.

English Equivalents

When you know:	Multiply by:	To find:
Millimeters (mm)	0.039	Inches
Centimeters (cm)	.394	Inches
Meters (m)	3.28	Feet
Kilometers (km)	.621	Miles
Hectares (ha)	2.47	Acres
Square meters (m^2)	10.76	Square feet (ft^3)
Square kilometers (km^2)	.386	Square miles
Cubic meters per second (m^3/sec)	35.3	Cubic feet per second (cfs)
Degrees Celsius	1.8 °C + 32	Degrees Fahrenheit

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