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POTENTIAL EFFECTS OF FOREST POLICIES ON TERRESTRIAL BIODIVERSITY IN A MULTI-OWNERSHIP PROVINCE

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Abstract. We used spatial simulation models to evaluate how current and two alternative policies might affect potential biodiversity over 100 years in the Coast Ranges Physiographic Province of Oregon. This 2.3-million-ha province is characterized by a diversity of public and private forest owners, and a wide range of forest policy and management objectives. We evaluated habitat availability for seven focal species representing different life histories. We also examined how policies affected old-growth stand structure, age distributions relative to the historical range of variability, and landscape patterns of forest types.

Under the current policy scenario, the area of habitat for old-growth forest structure and associated species increased over time, the habitat for some early-successional associates remained stable, and the area of hardwood vegetation and diverse early-successional stages declined. The province is projected to move toward but not reach the historical range of variation of forest age classes that may have occurred under the wildfire regimes of the pre-Euroamerican settlement period. Ownership explained much of the pattern of biodiversity in the province, and under the current policy scenario, its effect increased over time as the landscape diverged into highly contrasting forest structures and ages. Patch type diversity declined slightly overall but declined strongly within ownerships. Most of the modeled change in biodiversity over time resulted from policies on public forest lands that were intended to increase the area of late-successional forests and species.

One of the alternative policies, increased retention of wildlife trees on private lands, reduced the contrast between ownerships and increased habitat availability over time for both early- and late-successional species. Analysis of another alternative, stopping thinning of plantations on federal lands, indicated that current thinning regimes improve habitat for the Olive-sided Flycatcher, but the no-thinning alternative had no effect on the habitat scores for the late-successional species in the 100-year simulation. A comparison of indicators of biological diversity suggests that using focal species and forest structural measures can provide complementary information on biodiversity. The multi-ownership perspective provided a more complete synthesis of province-wide biodiversity patterns than assessments based on single ownerships.

Key words: forest habitat; forest planning; old growth; Oregon Coast Range; wildlife habitat relationships.

Introduction

Understanding of the effects of forest management on biodiversity is based largely on empirical studies of stands and small landscapes. Given the long time frames and wide range of spatial scales affected by forest policy and management, multi-scale simulation analyses are needed to more completely understand the potential effects of alternative policies and management practices. The effects of forest management on biodiversity indicators have been simulated at landscape scales within public forest lands (Hansen et al. 1993, 1995, Carey et al. 1999, Marzluff et al. 2002), and the effects of land-use change have been modeled in multi-ownership landscapes (White et al. 1997, Pearson et al. 1999). These studies demonstrate that forest management practices and land use change can have a strong influence on some measures of biodiversity. It is not well understood how stand-level practices and landscape-level forest policies scale up to provinces and regions where a wide range of forest policies and stand management practices are in

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effect. Little research has been done at this scale, in part because assessing forest management effects at broad scales is challenging. For example, detailed spatial information about stand structure and composition that is needed to drive habitat models and to estimate the effects of forest management practices is typically absent.

Assessing the effects of forest management on biodiversity is also challenging because biodiversity can be measured in numerous ways (Silbaugh and Betters 1995, Simberloff 1998, Hansen et al. 1999, Lindenmayer et al. 2000), yet it is not practical to use large numbers of indicators. Consequently, the problem of selecting effective subsets of biodiversity indicators has generated debate among conservation biologists. Proponents of individual species approaches (i.e., fine filter) argue that individual species are the most direct and sensitive measures of environmental change and can be indicators for other species or whole ecosystems (Lambeck 1997). However, for many species, too little is known, and a set of focal species does not necessarily encompass the needs of other species (Lindenmayer et al. 2002). Consequently, many argue for using indicators based on vegetation structure and composition (i.e., coarse filter; Lindenmayer et al. 2000, 2002). While these structural indicators are relatively easy to measure, relating change in the indicators to particular effects on species or processes can be difficult. The shortcomings of fine- and coarse-filter approaches and the lack of consensus on use of indicators suggest that assessments should rely on a suite of species and structural measures (Silbaugh and Betters 1995, Lindenmayer et al. 2002). Most published studies have assessed biodiversity using either species (Hansen et al. 1993, Carey et al. 1999, Raphael et al. 2001, Marzluff et al. 2002) or structural approaches (Cissel et al. 1999, Hemstrom et al. 2001); only recently have published studies used both types of indicators (e.g., Kintsch and Urban 2002).

Assessments are also limited in the degree to which they can represent the effects of management on biological diversity. Many rely on reserve design and land management allocation rather than on specific forest management practices. Large regional federal assessments such as the one that led to the adoption of the Northwest Forest Plan (USDA and USDI 1994) were limited because (1) they were restricted to federal lands; (2) they did not have the capacity to spatially project landscape changes under the policy alternatives; and (3) they emphasized species-level measures of biological diversity rather than a combination of species, structure, and dynamics. The Interior Columbia Basin Ecosystem Management Project (Haynes et al. 2001) was an advance in regional assessments. It projected landscape change into the future for alternative federal land policies and evaluated a variety of species and landscape indicators (Hemstrom et al. 2001, Raphael et al. 2001). However, it used a relatively coarse spatial resolution (>1 km^2), used stand type classes rather than continuous measures of forest condition, and did not simulate the management practices of different non-federal landowner groups.

In an attempt to address the limitations of large assessments, we developed an approach to investigating the implications of alternative policies on fine- and coarse-filter ecological measures (Spies et al. 2002*b*). We used high resolution spatial models of vegetation (Ohmann et al. 2007) and forest dynamics models of landowner behavior (Johnson et al. 2007) to assess potential changes in indicators of terrestrial biodiversity for the Oregon Coast Range Province under current and alternative forest policies. Specifically, we had several objectives:

1) To evaluate trends in indicators of biodiversity under current and alternative forest policies within and among land ownerships.

2) To compare trends in indicators of biological diversity among several types of ecological measures including focal species, forest stand structure, landscape structure, and historical range of variation.

3) To examine the redundancy in a set of different biological diversity indicators.

4) To identify shortcomings of individual policies and/or policies in aggregate by evaluating how well trends might match expectations embodied in policies.

Methods

Study area

The Oregon Coast Range is a 2.3-million-ha physiographic province that lies to the west of the Willamette Valley, south of Washington and north of the Klamath region in southwestern Oregon, USA. The climate is characterized by mild, wet winters and cool-to-warm, dry summers (Franklin and Dyrness 1973). The western side of the province near the Pacific Ocean is wetter and cooler than the eastern side. Topography consists of relatively low, but highly dissected mountains (1248 m maximum elevation), steep slopes, and high stream densities. Bedrock consists primarily of basalts and sandstones. Soils are typically well-drained loams and silt loams and are relatively deep except on steep upper slopes. Forests are dominated by Douglas-fir (*Pseudotsuga menziesii*), western hemlock (*Tsuga heterophylla*), western redcedar (*Thuja plicata*), Sitka spruce (*Picea sitchensis*), red alder (*Alnus rubra*), and bigleaf maple (*Acer macrophyllum*). Physiognomic forest diversity is high because of strong differences in structure between conifers and deciduous trees, and because of the large amount of structural differentiation that occurs as forests develop for more than 400 years following disturbances (Spies and Franklin 1991). Extensive logging and wildfires since the mid 1800s have created a forest matrix of young and mature conifer forests interspersed with patches of hardwoods (primarily red alder and bigleaf maple) and remnant patches of old growth (structurally diverse forests >200 years old) (Spies et al. 2002*a*). Current amounts of old growth are well below levels that probably occurred historically (Ripple 1994, Wimberly et al. 2000), which probably ranged between 25% and 75% of the province. Today, less than 5% of the province is covered by old growth (Ohmann et al. 2007).

Threats to native biological diversity in the Coast Range are exemplified by the case of two terrestrial vertebrate species listed as Threatened or Endangered under the Endangered Species Act: the Northern Spotted Owl (*Strix occidentalis caurina*), and the Marbled Murrelet (*Brachyramphus marmoratus*). These species are at risk because of habitat loss or degradation associated with logging, forest conversion to agriculture and housing, and other threats including predation by humans and other species.

The Coast Range is a socially diverse province with landowners having forest management goals ranging from wood production to wilderness protection (Johnson et al. 2007). Although there are significant blocks of public lands, the province is dominated by private ownerships. Public landowners include the USDA Forest Service (10% of province), USDI Bureau of Land Management (15%), State of Oregon (12%), and Indian tribal lands (<1%). Private lands fall into two major groups, forest industry (41%; medium to large holdings that typically include mills) and nonindustrial private forests (22%). The adoption of the Northwest Forest Plan (USDA and USDI 1994) brought major changes to forest management of the federal forests in this province, shifting their emphasis toward protection of biodiversity through the creation of an extensive network of late-successional reserves and riparian reserves. This shift resulted in an 80–90% reduction of timber harvests from federal lands in the Coast Range compared with those in the 1980s. In the future, more than 75% of timber harvest in the Coast Range is expected to come from forest industry lands that are managed under regulations defined by the State of Oregon Forest Practices Act (Oregon Forest Resources Institute 2002).

Policy alternatives and landscape projections

We simulated three policy scenarios: current land policies, increased levels of green tree retention for wildlife on private lands, and no thinning on federal lands. The current policy scenario is based on the Northwest Forest Plan for federal lands, which emphasizes reserves for existing mature and old forests and thinning to restore ecological diversity in plantations (USDA and USDI 1994). Current policies on state forests emphasize a blend of ecological and commodity goals using variable rotation lengths (Oregon Department of Forestry 2001). Current policies on industrial and nonindustrial private lands are based on the State Forest Practices Act, which gives priority to timber production (Oregon Forest Resources Institute 2002). The green tree retention scenario, which changed only the private land management, was designed to illustrate an option that created more ecologically diverse early-successional stages. Under this scenario, 12 average-sized trees (40–50 cm dbh) per hectare were left at the time of harvest. The no-thinning scenario, which changed only federal land management, was designed to evaluate the effects of thinning plantations on the federal lands, a practice that is intended to diversify these stands and accelerate development of old-forest structures. These policy scenarios are described in detail in Johnson et al. (2007).

We used the Landscape Management and Policy Simulator (LAMPS), a forest management simulator (Bettinger et al. 2005, Johnson et al. 2007), to project forest conditions into the future (in five-year time steps) at a spatial resolution of ~0.06 ha. The model is largely deterministic but contains fine spatial scale stochastic elements (see Johnson et al. 2007) and a single run was used to project change under a set of forest management assumptions. LAMPS integrates harvest-scheduling routines, with stand simulation models to spatially represent forest management, growth, and succession. Measures of biodiversity were summarized for the following projection periods: 0, 25, 50, 75, and 100 years into the future. The focal species habitat models and old-growth index were programmed in C++ and run on GIS layers representing initial vegetation conditions (Ohmann and Gregory 2002) and the outputs of LAMPS.

Focal species

We selected a small set of focal species representing a wide range of habitat needs and for which we had enough information to build habitat models. Selection criteria included Threatened, Endangered, and Sensitive status; successional limitation (both early and late); dispersal distance (short and long), and sensitivity to landscape pattern (both interior species and edge species). The seven taxa were (1) Northern Spotted Owls, which are associated with late-successional forests (McComb et al. 2002); (2) Marbled Murrelets, a late-successional associate; (3) Western Bluebirds (*Sialia mexicana*), a state Sensitive Species, associated with early-successional conditions (open canopy) with an adequate number of snags for nesting (Shreiber and deCalesta 1992); (4) Olive-sided Flycatchers (*Contopus cooperi*), associated with low canopy cover and widely scattered trees or with abrupt edges that are sites of nesting and foraging (Altman and Sallabanks 2000); (5) red tree voles (*Arborimus longicaudus*), which occur in young to old closed-canopy Douglas-fir forests and are considered to be most abundant in forests with medium- to large-diameter trees (>50 cm) (Corn and Bury 1986); (6) epiphytic macrolichens of the genus *Lobaria,* which have limited dispersal capability (termed "low-mobility lichens" in this paper) and whose abundance increases with stand age (Spies 1991, McCune 1993) or with the presence of older remnant trees in young stands (Sillett and Goslin 1999) (several species of this genus had been identified as "Survey-and-Manage" species under the Record of Decision for the Northwest Forest Plan [USDA and USDI 1994]); and (7) epiphytic macrolichens of the genera *Platismatia* and *Hypogymnia*, which are able to recolonize forest stands earlier in stand development than can low-mobility lichens (McCune 1993) (termed "moderate-mobility lichens" in this paper).

A habitat capability index (HCI) was developed for each taxon (Appendices A–E). The HCI is calculated from a set of capability indices (CI) that reflect the habitat characteristics at patch and landscape levels that are important for survival and reproduction of each taxon (Table 1 and Appendices A–E). Capability indices are scaled from 0 to 1, where 0 indicates that conditions are not suitable to satisfy one or more requirements, and 1 represents theoretical optimum conditions. The selection of vegetation and physical variables to include in the HCI models depended on three factors. First, we used variables for which the relationship to reproduction or survival could be supported by empirical evidence from published studies or from the opinion of experts with whom we consulted. Second, variables were necessarily restricted to those that could be estimated from existing GIS layers, including the vegetation data layer that was based on satellite imagery, environmental data, and field data (Ohmann and Gregory 2002). Third, we selected variables that could be projected into the future by using models of forest dynamics.

We were able to empirically test the HCI models for the owl, bluebird, and flycatcher using geo-referenced data from field studies the Oregon Coast Range (McGarigal and McComb 1995). The evaluation of the owl model (McComb et al. 2002) was based on systematic surveys for owl nests. We used logistic regression analysis of habitat capability index scores on nest locations of the owl and selected the best performing HCI model (out of six to eight possible models) with the lowest Akaike's Information Criterion (AIC) value (Burnham and Anderson 1998). The best model (see McComb et al. 2002) had an AIC value that was $\geq$4.75 lower than the next competing model. We used an HCI breakpoint of 0.37 to map habitat and non-habitat; this breakpoint was selected to optimize classification accuracy, which was 76%. For the bluebird and flycatcher, we evaluated models by correlating bird abundance from 28 250–300 ha subbasins in the central Coast Range (McGarigal and McComb 1995) with the aggregated HCI scores. The Spearman's rank correlations for the bluebird and flycatcher were $r = 0.43$ ($P = 0.021$) and $r = 0.32$ ($P = 0.098$), respectively.

Empirical verification using independent data was not possible for the other taxa, but each of the models was sent to one to five published scientists and other experts in the region who provided written critiques that were used to revise the models (see Appendices A–D for model descriptions and lists of reviewers). Sensitivity analysis was conducted on all models using rank correlations with @Risk (Palisade Corporation 1997) to determine the relative importance of individual variables in model behavior. To map the habitat of these species, we examined frequency distributions of scores and used breaks at the lower and upper thirds of scores to define three arbitrary classes. Medium and high habitat-quality classes were assumed to be habitat for

TABLE 1. Variables used in habitat capability indices of focal species and old-growth index, Oregon Coast Range.

Species and variables	Max	Corr
Northern Spotted Owl		
TPH 10–25 cm dbh	219	–0.14
TPH 25–50 cm dbh	108	0.02
TPH > 75 cm dbh	>65	0.57
Diameter diversity index†	>7.5	0.47
Habitat, 300 m‡	0.3	0.78
Habitat, 800 m‡	1.0	0.69
Habitat, 2400 m‡	1.0	0.61
Marbled Murrelet		
Western hemlock 50–75 cm dbh	>70	0.32
TPH > 75 cm dbh	>35	0.62
Canopy heterogeneity	>7.5	0.87
Landscape 100 m radius§	NA	0.89
Western Bluebird		
SPH > 50 cm dbh	>5	–0.04
SPH 25–50 cm dbh	>11	–0.10
Canopy cover (%)	<10	–0.51
Olive-sided Flycatcher		
Canopy cover (%)	5-36	–0.10
TPH > 10 cm dbh	96	–0.05
SPH > 10 cm dbh	28	–0.36
Landscape 360 m radius	NA	0.20
Red tree vole		
Douglas-fir (% BA)	100	0.77
Quadratic mean diameter	>64	0.76
Canopy cover (%)	100	0.52
Canopy heterogeneity	>8.0	0.85
Low-mobility lichen		
Stand age (yr)	>400	0.81
TPH > 100 cm dbh	>30	0.87
TPH 50–100 cm dbh	>40	0.53
Landscape 100 m radius	NA	0.59
Moderate-mobility lichen		
Stand age (yr)	100	0.77
TPH > 50 cm dbh	>100	0.78
Landscape 100 m radius	NA	0.91
Old-growth structure index		
Stand age (yr)	>450	0.76
SPH > 50 cm, >5 m tall	>14	0.70
Log volume (m^3/ha)	>30	0.42
TPH > 100 cm dbh	>55	0.75
Diameter diversity index	10	0.76

Note: Abbreviations: Max, the value or range of the variable at which the index score is 1; Corr, the Spearman rank correlation coefficient from the sensitivity analysis of component variables with the HCI scores; TPH, trees per hectare; SPH, snags per hectare.

† See McComb et al. (2002) for a description of this index.

‡ Proportion of habitat at different distances around the focal pixel where habitat is defined as either "good" (large and very large tree vegetation classes) or "moderate" (remnants, broadleaf, and medium tree vegetation classes).

§ Distance around focal pixel that is evaluated for landscape effects (see Appendices A and E for details).

TABLE 2. Definitions of tree size categories, physiognomy, and remnant densities used in the landscape structure analysis, Oregon Coast Range.

Parameter and category	Definition
Physiognomy	
Open	<40% canopy cover
Closed	>40% canopy cover
Hardwoods	hardwood basal area >65%
Mixed	hardwood basal area 20–65%
Conifer	hardwood basal area <20%
Canopy tree size†	
Small	2.5–25 cm
Medium	26–50 cm
Large	51–75 cm
Very large	>75 cm
Remnant trees‡	
None	0 trees/ha
Low	1–5 trees/ha
Moderate	6–12 trees/ha
High	>12 trees/ha

Notes: Combinations of the three classifications were used to build the 34 vegetation structure/composition classes. Note that not all combinations of classes were used.

† Quadratic mean diameter for closed canopy stands only.

‡ Trees 50 cm dbh in open and small diameter stands; 75 cm in medium diameter stands; no remnants in large and very large diameter stands.

the species. Because we do not know the relationship between species viability and HCI scores for all species, we assumed that trends in predicted habitat area and pattern over time and among policies were indicative of actual area of habitat for each taxon; however, we do not know the actual relationship between the HCI scores and population viability for any of the species.

Stand structure

An index of old-growth forest structure and development was used to evaluate stand structure (Spies and Franklin 1988, Franklin and Spies 1991). The index is an average of four separate indices representing stand age and four structural features: number of large trees (>100 cm dbh), large snags (>50 cm dbh and >15 m tall), volume of large snags, and tree size diversity. The overall index ranged from 0 to 1 based on a sample of natural reference stands whose canopy dominants ranged in age from 30 to >500 years. We used the statistical distributions of the structural features in the population of stands over 200 years ($n = 25$) to set the relationship between the level of a habitat element and its index score (Spies and Franklin 1991). For each structural variable, a segmented linear curve was constructed that reaches an asymptote of 1 at the maximum value observed, 0.75 at the median, and 0.5 at the 25th percentile of the variable in the population of 25 old-growth stands. We used these breaks so that the overall scores of the index would typically be above 0.5 for stands that were identified as old growth based on field inspection and stand history and well below 0.5 for young natural stands and plantations. For stand age, the segmented line had a value of 0.8 at 200 years and a value of 1.0 at 450 years. Stands with a combined index value >0.5 were defined as late-successional-old-growth (LSOG) to match the terminology in the Northwest Forest Plan (USDA and USDI 1994). Stands with an index of >0.75 were classified as old growth (OG).

Landscape structure and dynamics

Landscape structure was evaluated in two ways. First, we used age class distributions to evaluate the state of the landscape relative to the historical range of variation (HRV) under the estimated wildfire regime of the pre-Euroamerican settlement period. HRV is an approach to assessing biodiversity that recognizes the inherent dynamics and variability of forest landscapes (Landres et al. 1999). The HRV of the Oregon Coast Range has been estimated for a variety of age classes based on paleoecological studies, dendroecological studies, and simulation modeling (Wimberly et al. 2000).

In the second approach, we evaluated landscape structure for a small set of landscape metrics using the program FRAGSTATS (McGarigal and Marks 1995). The vegetation data for the initial conditions (Ohmann et al. 2007) and for the future simulations (Johnson et al. 2007) were classified into 34 patch types based on diameter of canopy trees, physiognomy, and abundance of remnant trees in younger open and young forest stands (Table 2). The following landscape metrics were then calculated: (1) patch type diversity, (2) largest patch index, (3) contrast weighted edge density, and (4) interspersion and juxtaposition of patch types. Although numerous landscape metrics can be calculated, many are difficult to interpret ecologically and are highly correlated with each other (Ritters et al. 1995). These metrics have clear relevance to ecological phenomena: (1) diversity of all species and community types (patch type diversity), (2) large blocks of interior forest that may harbor organisms that avoid edges and contact with humans (largest patch index), (3) contrasting edges that are favored by many species and avoided by others and may be sites of blowdown (contrast weighted edge density), (4) intermixing of habitats that may benefit some species that use a variety of patch types (e.g., elk [*Cervus elaphus*]) or increase intermixing of propagules and energy (e.g., radiation, wind) among patch types (juxtaposition index).

Statistical analyses

We used linear model analysis (GLM procedure in SAS [SAS Institute 1999]) to evaluate the relative importance of ownership, distance from adjacent ownerships, and identity of nearest owner with regard to variation in HCI scores. The analysis was based on a subsample of pixel groups with uniform HCI scores and ownership. Grouping of pixels enabled us to subsample the simulation outputs without bias toward large patches that would occur if we just randomly selected individual pixels. The HCI scores were placed into 10 equal index classes between 0 and 100 that were then combined with the ownership layer (four classes) to create a habitat score/owner map. The equal HCI-ownership pixel groups served as the basic sample unit of the analyses. The centroid of each pixel group was then computed, and the Euclidean distance from these centroids to the nearest other ownership was calculated. One percent of these pixel groups were randomly selected at time 0 and 100 using fifth hydrologic unit code watersheds as strata to insure a wide distribution of samples. Analyses were done at the two times on the current policy scenario. The effects of ownership alone were evaluated for all focal species by using ownership as a dummy variable in the analysis. For the Northern Spotted Owl, whose HCI model had the longest distance of surrounding landscape influence (2.4 km), we conducted a separate analysis to evaluate the additional contribution of distance from adjacent owner and identity of owner.

We also investigated the degree to which the suite of biodiversity indicators provided similar information about the effects of forest management. We used Spearman's correlation coefficient to assess the relationship among biodiversity indicators and to determine if those relationships changed over the course of the simulation of current policy. Correlations were calculated at year 0 and year 100 from a 1% systematic sample of pixels on a 10×10 spacing. This approach was used to insure a widespread sample and because it was easier to implement with the GIS software than a complete random sample.

Results

Patterns and trends in habitat availability for focal species

Under current land management policies, habitat area for Northern Spotted Owls, Marbled Murrelets, and low-mobility lichens is projected to increase strongly over the next 100 years (Figs. 1 and 2). Habitat for Western Bluebirds is projected to decline slightly, while potential habitat for moderate mobility lichens is projected to decline and then stabilize. The area of habitat for the Olive-sided Flycatcher is projected to decrease at first and then increase. This pattern is probably a result of an initial decline in semi-open forests (canopy closure 20–40%) followed by an increase in older forest structure and snags in later decades. Of the seven species, only Olive-sided Flycatcher habitat responded differently to all three scenarios, with the least decline in habitat occurring when trees were retained on private lands, and the greatest decline occurring when no thinning occurred on federal lands. In the absence of thinning, stands remain too dense to provide acceptable foraging conditions for this species.

Habitat availability for red tree voles increased modestly over time under current policy; it did not respond to thinning on federal lands, but it increased strongly under green tree retention (Fig. 2). Western Bluebirds also increased strongly with the green tree retention scenario. Availability of habitat for moderate-mobility lichens was predicted to decrease under current policies as forests matured, but it remained fairly constant when green tree retention was practiced on private lands.

Ownership effects

Regardless of the policy scenario evaluated, public lands (initially federal, then both federal and state) provided the majority of current and future habitat for Northern Spotted Owls, Marbled Murrelets, and low-mobility lichens (Fig. 3). Red tree voles are expected to find more habitat area on public than on private lands unless green tree retention is practiced on private lands. Similarly, the contribution of private lands to habitat for Western Bluebirds and moderate-mobility lichens is projected to increase noticeably if green tree retention is practiced on private lands. Thinning on federal lands should have little if any effect on habitat availability among land ownerships compared with that under current policies for any of the seven taxa we assessed.

In general, ownership (i.e., land allocations and management practices) explained relatively little of the variation (low R^2) in habitat quality for most species, but R^2 values increased from year 0 to year 100 (Table 3). For wide-ranging species such as the Northern Spotted Owl, habitat quality on an ownership may be influenced by conditions on adjacent ownerships. However, when the identity of the nearest neighbor and the distance to the nearest neighbor were included in the regression models, the model R^2 increased by only about 1%.

Patterns and trends in stand structure

The area of structurally diverse forest (old-growth index >0.5) increased steadily over the 100-year simulation (Fig. 2). The trend differed little among the three policy scenarios. However, relative to the base policy, the green tree retention option decreased the area of the lowest index values (<0.25) by 57% by year 100 and increased the area of low-to-intermediate (0.25–0.49) index values by 141% (data not shown in Fig. 2). The distribution of structurally diverse forest was strongly concentrated on federal and state forest lands (Fig. 3).

Age class distribution in relation to historical range of variability (HRV)

Under all policy scenarios the distribution of forest age classes shifted from dominance by forests that were 20–40 years old at year 0 to dominance by forests 80–200 years old at year 100 (Fig. 4). The youngest age classes were more abundant than the HRV for that class, and the oldest age classes were less abundant than historically.

Trends in landscape structure

Under the base policy, conifer patches and patches with large- and very large-diameter trees increased over 100 years (Fig. 5) on public ownerships (see Table 2 for

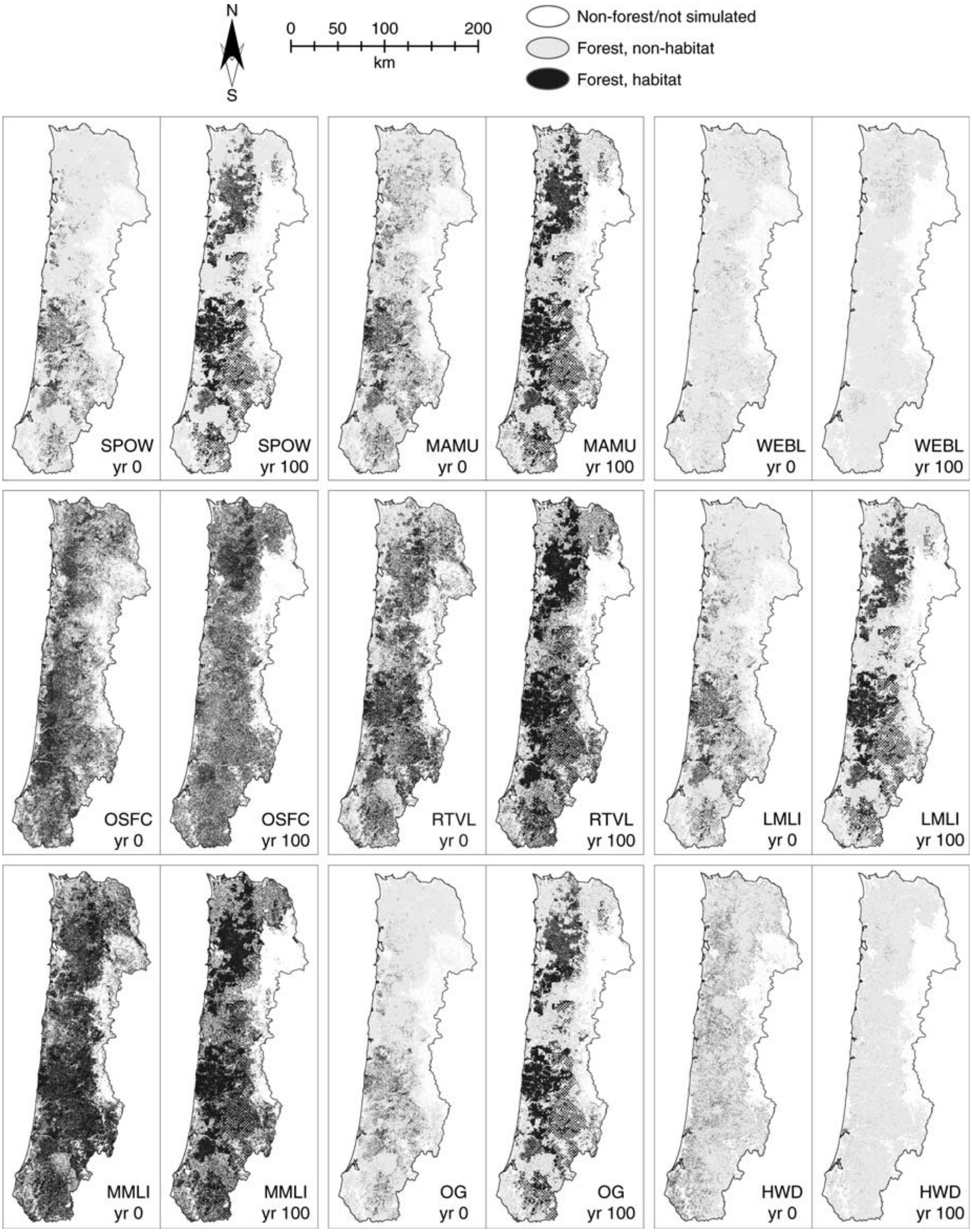

FIG. 1. Current and projected pattern of habitat for focal species and forest structure and composition classes (see *Methods* for definitions of map classes) at time 0 and 100 years for the Coast Range of Oregon under base (current) policy. Abbreviations: SPOW, Northern Spotted Owl; MAMU, Marbled Murrelet; WEBL, Western Bluebird; OSFC, Olive-sided Flycatcher; RTVL, red tree vole; LMLI, low-mobility lichen; MMLI, moderate-mobility lichen; OG, old growth; HWD, hardwood.

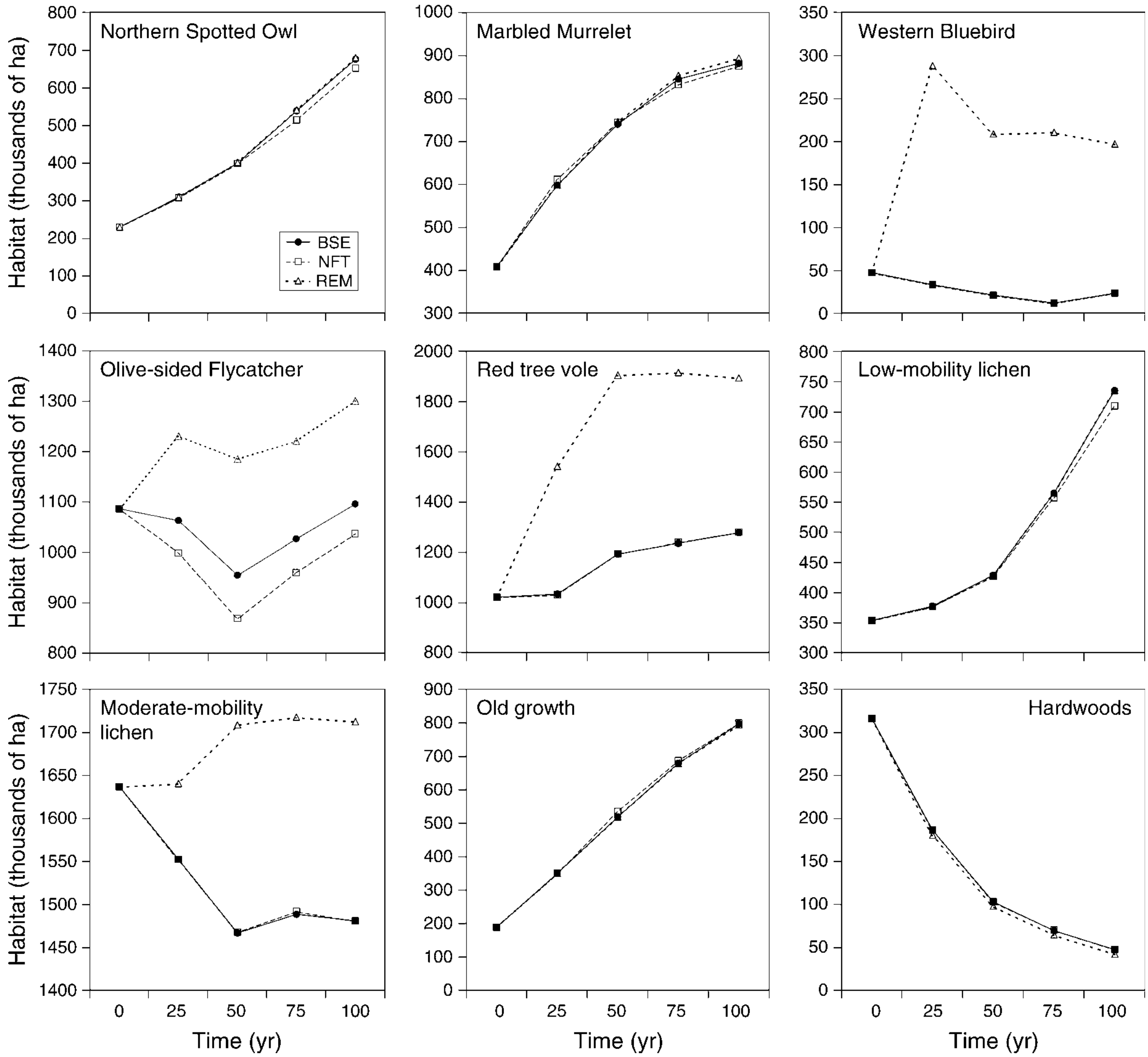

FIG. 2. Simulated changes over 100 years for three policy scenarios in amount of habitat for Northern Spotted Owl, Marbled Murrelet, Western Bluebird, Olive-sided Flycatcher, red tree vole, low-mobility lichen, moderate-mobility lichen, and old-growth and hardwood classes in the Coast Ranges Province of Oregon (see *Methods* for definition of map classes). Scenarios: BSE, base (current) policy; NFT, no thinning in plantations on federal lands; REM, leaving additional remnant wildlife trees on private lands. Note that *y*-axes do not have the same range.

definitions). The hardwood patch type declined across all ownerships. The area of small- and medium-diameter trees declined moderately, and the remaining types showed little change. The ownership distribution within vegetation types also changed between years 0 and 100 (Fig. 6). The largest ownership distribution change occurred as state lands supported a greater proportion of the very-large-diameter forest type by year 100 than in year 0. The federal lands still contained the majority of forests with large-diameter trees by year 100, and private lands contained the majority of forest with small- and medium-diameter trees. Hardwood patches occurred mainly on nonindustrial private lands and mixed conifer–hardwood patches occurred primarily on forest industry lands. State lands and private lands provided the majority of young stands with remnant trees.

The patch type diversity index declined slightly across ownerships but strongly declined within ownerships under all scenarios. By the end of the simulation for the base policy, federal lands had the lowest patch type diversity and private lands had the highest (Fig. 7). Trends for the patch type diversity index pattern were similar among scenarios (data not shown), but the retention scenario increased patch type diversity on private lands relative to the base policy, and the no-federal-thinning scenario increased diversity on federal lands; lack of thinning increases area of stands in the

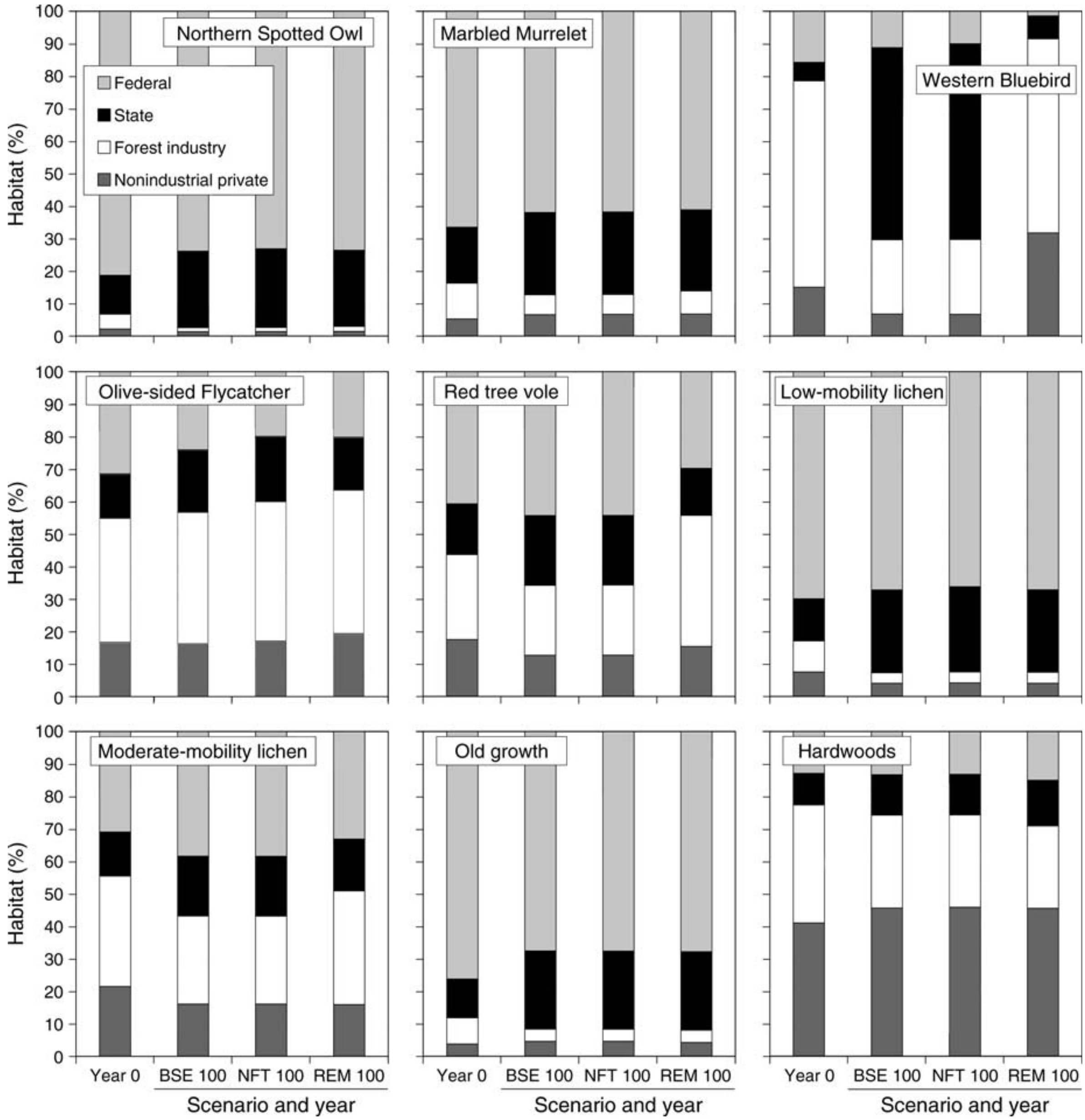

FIG. 3. Distribution of habitat for focal species and forest structure classes, by ownership, at year 0 (initial conditions) and at simulated year 100 for three policy scenarios in the Coast Ranges Province of Oregon. Scenarios: BSE, base (current) policy; NFT, no thinning in plantations on federal lands; REM, leaving additional remnant wildlife trees on private lands.

small diameter classes. Under all scenarios, the largest patch index (regardless of patch type) increased somewhat on the entire landscape and on BLM lands but increased strongly in later periods on state and Forest Service lands. Edge density and the juxtaposition index declined for the landscape as a whole and within individual ownerships in all three alternatives (data not shown for alternatives).

Comparison of indicators

The correlations among habitat scores for focal species and measures of forest structure and composition at year 0 varied from very low ($r = -0.07$) in the case of canopy cover and the Olive-sided Flycatcher index to high ($r = 0.91$) in the case of Marbled Murrelets and the red tree vole (Table 4). Most indicators were moderately and positively correlated with each other. The exceptions were the Western Bluebird and hardwoods, which had low or negative correlations with most indices. The old-growth index was moderately correlated ($r = 0.56$–0.66) with the focal species that are typically associated with late-successional forests (Northern Spotted Owl, Marbled Murrelet, red tree vole, and lichens). At year 100, the strength of association for many of the

TABLE 3. R^2 of regression models of index of habitat quality on ownership by species at time 0 and 100 for the current policy scenario, Oregon Coast Range (all models were significant at $P < 0.001$; n is the number of patches of equal habitat value sampled).

	Simulation year			
	0		100	
Species	n	R^2	n	R^2
Northern Spotted Owl	15 864	0.31	22 614	0.45
Marbled Murrelet	46 300	0.16	48 214	0.32
Western Bluebird	10 140	0.07	5751	0.32
Olive-sided Flycatcher	12 724	0.02	13 416	0.02
Red tree vole	70 533	0.11	46 722	0.33
Low-mobility lichen	38 612	0.13	45 337	0.23
Moderate-mobility lichen	99 538	0.01	34 698	0.02

indicators had increased and some changed from positive to negative. For example, the correlation between the spotted owl and old growth changed from 0.56 to 0.85 and the correlation between red tree voles and hardwoods changed from 0.14 to −0.33.

DISCUSSION

Biodiversity index trends

We expected forest policies enacted in the 1990s to greatly alter forest biodiversity patterns over the next 100 years. Our results indicate that most of this projected change is due to policies on public forest lands. Under the assumptions of our models, these policies will greatly increase old-growth structure as well as increasing habitat for all focal species associated with late-successional conditions. The province is projected to move toward the historical range of variability for most age classes of forest. The increase in mature and old forest during the simulation also reflects the presence of extensive areas of young forest in year 100 that could grow into these older classes during the 100-year simulation. However, given the long period of stand development in this province, even 100 years leaves significant differences in age class distributions between the managed forest landscape and that expected under HRV. Age classes of forest are not ideal indicators of forest habitat quality because habitat structure can vary widely within an age class (Spies and Franklin 1991). However, we lack information about historical structure of forests, so age is a surrogate for structure in these forest types.

Although federal and state forest plans were intended to increase the area of habitat for late-successional species, we observed other changes that were not clearly addressed in these plans, which seek to increase older forest habitats. These include changes in landscape structure, declines in hardwoods on all ownerships, and declines in amounts of diverse early-successional patches (semi-open areas of shrubs, trees, and herbs) and species (e.g., Western Bluebird) on federal lands. Landscape diversity is projected to decline strongly on public lands and slightly overall. At 100 years, the most diverse portion of the landscape is projected to occur on private nonindustrial and industrial forest lands.

The relatively high diversity on private lands results from the mix of open (harvested) areas and younger seral stages, which become uncommon on federal lands. Although this high diversity might seem surprising, it should be expected given the way the diversity metric is calculated, and it must be remembered that landscape diversity is only one measure of biodiversity. The range of stand types on private lands was limited by harvest rotation length compared with the range expected under a natural disturbance regime. Other indicators show that the landscape moves closer to the distribution of forest conditions under HRV of the late Holocene, as old growth, a currently rare type, increases in area. Although diversity on federal lands declines, the potential diversity or resiliency of these landscapes is much higher than that of private forest lands because when natural disturbances such as fire, windthrow, or pathogen outbreaks occur, federal landscapes will develop a full range of forest structural stages (assuming some older stages survive). Natural disturbances on intensively managed lands would not increase diversity levels as much and could result in declines in landscape diversity because these lands are primarily young forest landscapes.

The projected declines in hardwood area across all land ownerships and diverse early-successional stages (e.g., shrub fields, open stands with remnant trees, and semi-open forests) on federal lands have received little attention in recent Pacific Northwest forest biodiversity plans. These declines have two possible origins. First, many current stands with hardwoods are projected to succeed to conifer dominance (conifers >80% of basal area) over the next 50 years. Many of these hardwood

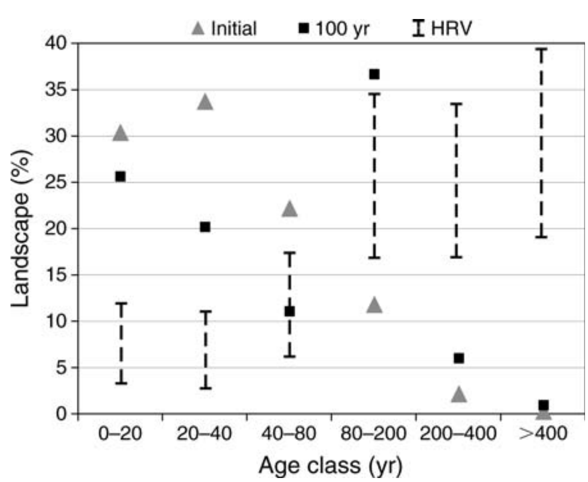

FIG. 4. Distribution of age classes in relation to percentage of landscape for initial conditions, simulated conditions at 100 years under current policies, and historical range of variation (HRV) under the pre-Columbian wildfire regime of the Coast Ranges Province of Oregon.

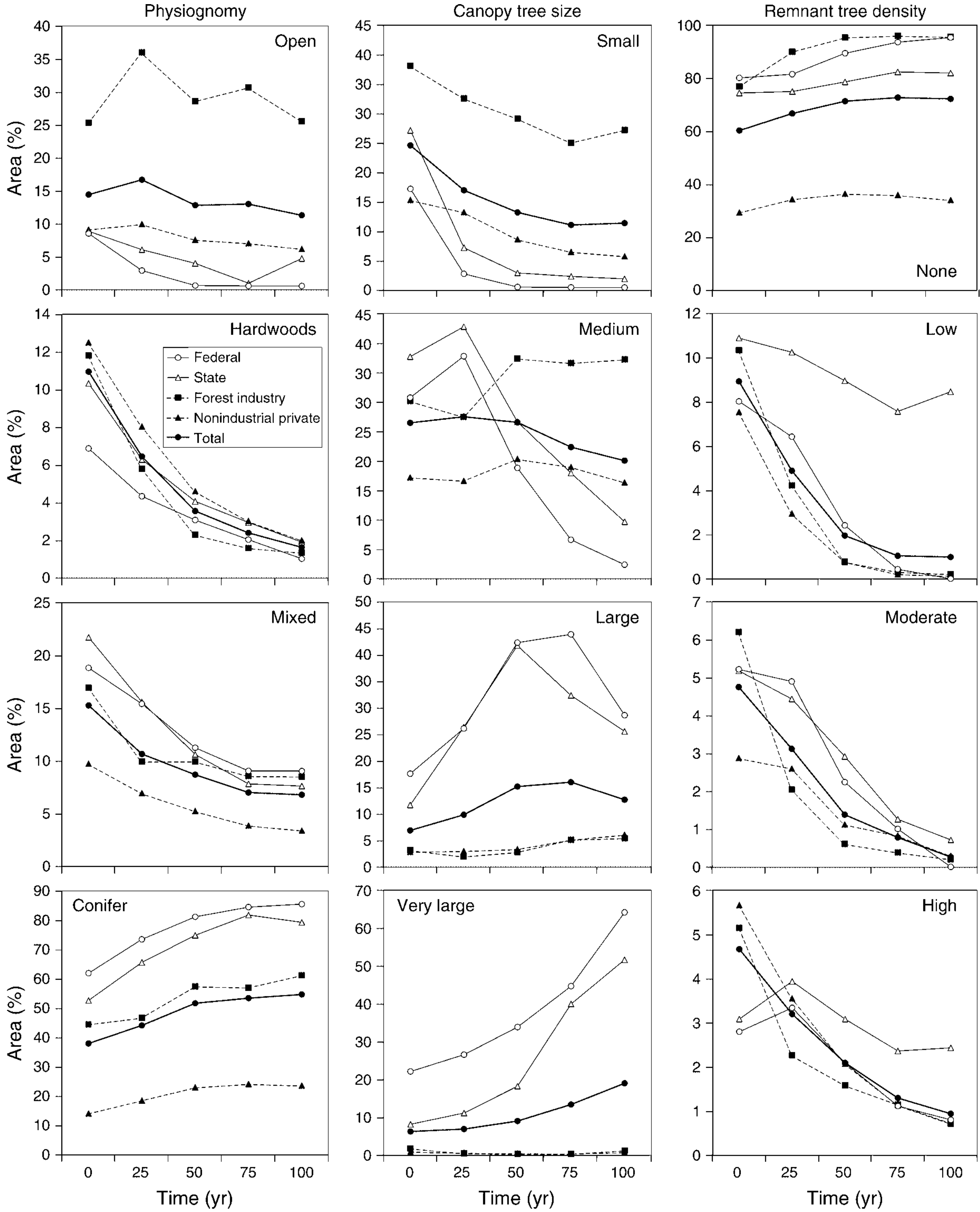

FIG. 5. Simulated changes over 100 years in the percentage of area of characteristics of physiognomy, canopy tree size, and remnant tree densities (see Table 2 for definitions of classes) under base policy, by ownership, in the Coast Ranges Province of Oregon.

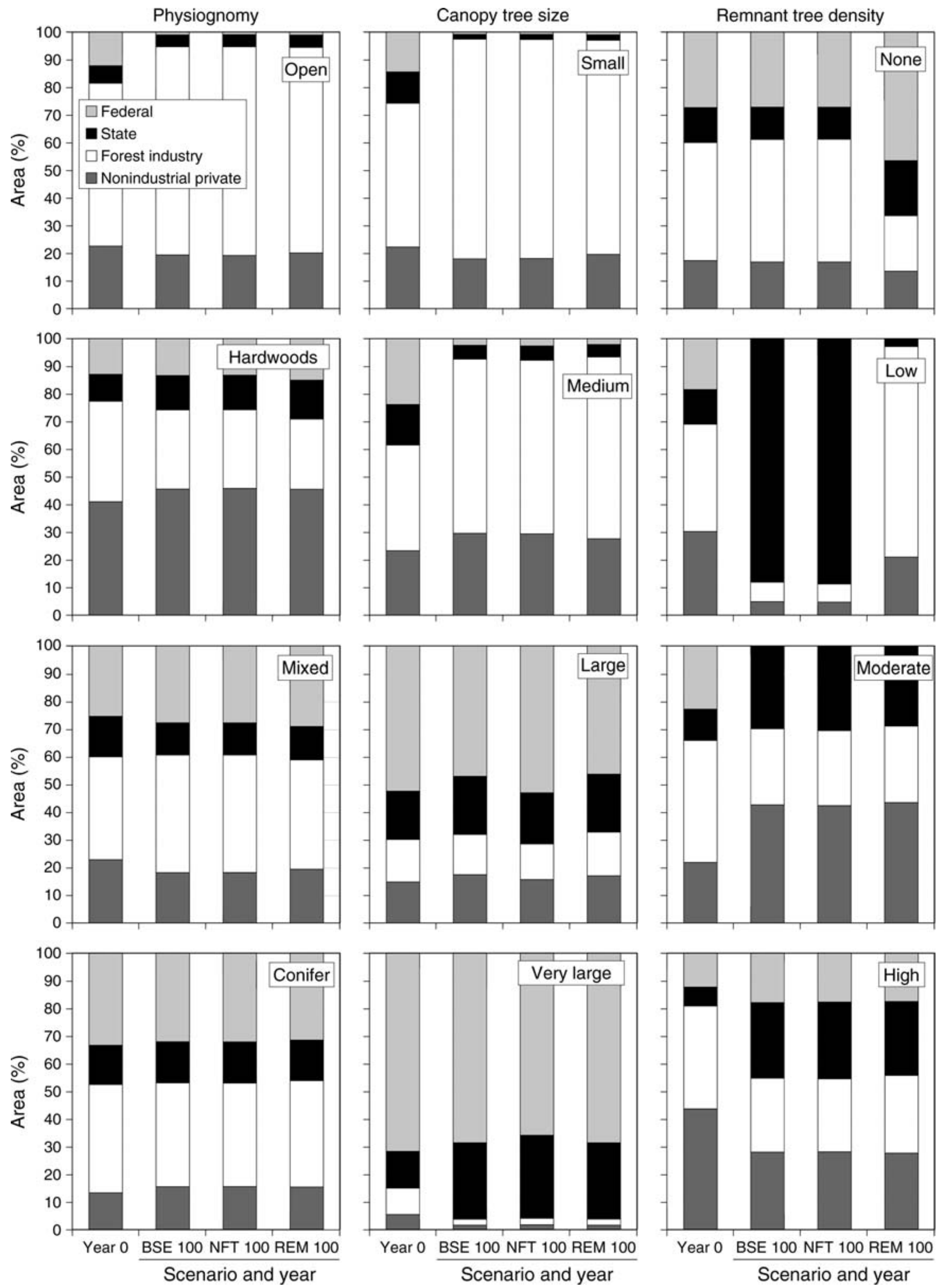

FIG. 6. Distribution of characteristics of physiognomy, canopy tree size, and remnant tree densities (see Table 2 for definitions of classes), by ownership, at year 0 (initial conditions) and at simulated year 100 for three policy scenarios in the Coast Ranges Province of Oregon. Scenarios: BSE, base (current) policy; NFT, no thinning in plantations on federal lands; REM, leaving additional remnant wildlife trees on private lands.

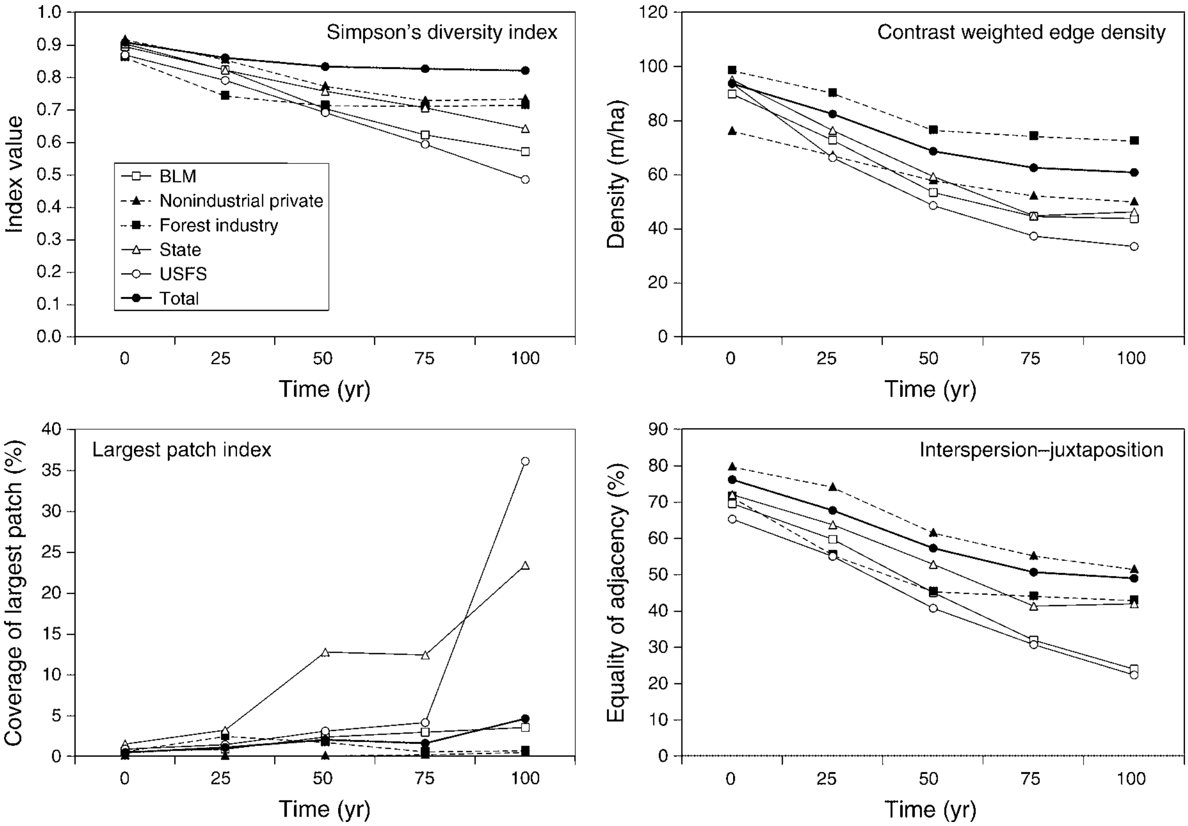

FIG. 7. Simulated changes in landscape metrics over 100 years under base policy, by ownership, in the Coast Ranges Province of Oregon (see *Methods: Landscape structure and dynamics* for description of metrics). Ownership abbreviations are: BLM, Bureau of Land Management; USFS, U.S. Forest Service.

TABLE 4. Spearman rank correlations between focal species habitat scores and selected measures of forest structure ($n = 362\,234$ groups of similar pixels) at year 0 and at year 100 in the simulation of current policy in the Oregon Coast Range (all correlations were significant at $P < 0.0001$).

	SPOW	MAMU	WEBL	OSFL	RTVL	LMLI	OG	HWD	CCOV
Year 0									
MAMU	0.77								
WEBL	−0.20	−0.27							
OSFL	0.33	0.29	0.06						
RTVL	0.73	0.91	−0.27	0.27					
LMLI	0.75	0.82	−0.14	0.37	0.83				
OG	0.56	0.65	−0.10	0.26	0.64	0.66			
HWD	0.19	0.16	0.09	0.09	0.14	0.23	0.15		
CCOV	0.26	0.44	−0.30	−0.07	0.44	0.31	0.39	0.13	
QMD	0.60	0.76	−0.25	0.25	0.75	0.69	0.63	0.14	0.13
Year 100									
MAMU	0.89								
WEBL	−0.45	−0.51							
OSFL	0.31	0.39	−0.26						
RTVL	0.88	0.91	−0.53	0.33					
LMLI	0.91	0.92	−0.43	0.36	0.88				
OG	0.85	0.86	−0.44	0.30	0.85	0.89			
HWD	−0.20	−0.25	0.41	−0.13	−0.33	−0.19	−0.22		
CCOV	0.41	0.47	−0.52	0.10	0.48	0.40	0.47	−0.29	
QMD	0.80	0.85	−0.27	0.37	0.82	0.87	0.86	−0.12	0.33

Note: Key to abbreviations: CCOV, canopy cover; QMD, quadratic mean diameter; for other abbreviations see Fig. 1.

stands are dominated by red alder, a relatively short-lived, shade-intolerant species. Second, future forestry practices and small natural disturbances are assumed to give rise to conifer-dominated stands instead of hardwoods. The shift away from hardwoods occurs more in uplands than in riparian zones, where the models assume that hardwoods have greater competitive ability than conifers. While natural disturbances may create these types, landowners work aggressively to suppress such major disturbances as fires, which could initiate hardwood patches. The decline in hardwoods probably will be reflected in other components of biological diversity as well, since many species of plants and animals, including invertebrates, are associated with hardwoods and shrubby open stages (Hibbs et al. 1994, Neitlich and McCune 1997, Johnson and O'Neil 2001). Hardwoods, other than Oregon white oak (*Quercus garryana*), are not currently a major conservation concern in the Coast Range.

The decline of patchy, semi-open forest conditions (such as might occur following natural stand replacement disturbances that are not replanted) is projected to occur for at least two reasons. First, current stands of this type are assumed to fill in with conifer and hardwood tree canopies within a decade or so. Second, when new open stands are created by timber cutting, the growth and succession models assume that stands are managed intensively and move from open, to semi-open, to closed canopy conditions (>40% canopy cover) within 10–15 years. The degree to which stands rapidly fill in with conifer cover is an area of uncertainty for future projections of forest management. The current population of semi-open stands is a legacy of past human and natural disturbances that may not be repeated. However, we may be overestimating the success of stand management practices and underestimating the future amount of semi-open forest.

The effects of ownership and alternative policies

The general patterns of biodiversity corresponded, as expected, to the management goals of the different ownership classes. It is interesting, however, that ownership did not explain a high percentage of the variation in habitat quality in the regression models. This does not mean that ownership is not an important driver of habitat patterns but it suggests that the pattern of habitat within ownerships can be noisy, resulting in a relatively low explained variation when sampled at the pixel resolution. The fine-scale variation of habitat within ownerships results from initial vegetation patterns (Ohmann et al. 2007) stemming from fine-scale ecological processes and model error and from simulated fine-scale patterns in LAMPS (Johnson et al. 2007). Within-ownership variation in forest conditions in the simulations results from harvesting, patterns of land allocation, variation in stand development, and stochastic canopy gap disturbances (<2 ha in size; Johnson et. al. 2007). The increase in variance explained in index scores by ownership by year 100 probably results from two sources. First, most of the increase probably results because current management practices increase homogeneity of forest structural conditions within ownerships. Second, some of the increase in variance explained occurs because the simulation model does not include all of the processes that created the fine-scale patterns present at year 100. One implication of spatial variation within ownerships is that landscape assessments that use ownership as a surrogate for management may miss important fine-scale forest variation that influences biological diversity.

In general, the alternative policies that we examined had little effect on measures of biodiversity associated with late-successional species and their habitats. The exception was the red tree vole, whose area of habitat was more than 50% greater under the green tree retention alternative than under the base policy. Small patches (<1 ha) of large live trees, which were part of the retention scenario, may provide residual food and cover and refugia for red tree voles in actively managed private lands. However, this finding should be viewed as a hypothesis since no studies have examined the response of the tree vole to live tree retention, either in the form of single trees or as patches. Of course, this indicator does not take into account any dispersal limitations that the species might encounter in moving into landscapes where it does not currently occur.

Early-successional patches and associated species would benefit under the green tree retention option. The lack of strong differences in late-successional indicators between the thinning and no-thinning alternatives on federal lands was somewhat surprising. Empirical studies of thinning in dense plantations indicate that habitat quality improves for many species when dense young conifer stands are thinned (Hagar et al. 1996, Hayes et al. 2003, Suzuki and Hayes 2003). However, there is no empirical evidence of the effects of thinning on the late-successional species we simulated. We projected that thinning on federal lands would open up the canopies of dense plantations and improve habitat for Olive-sided Flycatchers (Hagar et al. 1996); we would expect that indicators such as understory development and community diversity for animals and plants, which we did not examine, would be higher in the simulation under the thinning options (base policy) than under the no-thinning option (Hayes et al. 2003, Suzuki and Hayes 2003).

The lack of response of the late-successional indicators to the different thinning prescriptions is probably due to several factors. First, the period of time examined, 100 years, may be too short for thinning to appreciably affect late-successional forest development (Garman et al. 2003). Second, the thinning prescriptions we used may not be the most effective at accelerating such development. Garman et al. (2003) demonstrated that thinning regimes had varying effects on the rate of development of late-successional stands. Third, thinning results in tradeoffs among late-successional attributes. The density of large tree boles can be accelerated by thinning but at some cost to the short-term production of dead wood volume in the stand. Fourth, effects based on thinning a small number of stands may not be as extensive as when the study is scaled up to a landscape or province. Relatively low-density stands with hardwoods and shrubs may not respond as strongly to thinning as dense conifer stands. In addition, thinning effects can be diluted if the area of plantations is small compared with the landscape as a whole. For example, the federal forests occupy less than 25% of forest land, and the area of plantations thinned during the simulation was less than 14% of federal lands. Given the uncertainties around estimates of thinning effects at broad scales, further research is clearly needed.

Indicators

The correlation among species and structural indicators is not known for most ecosystems (Whitman and Hagan 2003). Yet, managers are faced with the need to develop and use indicators of various kinds to assess biological diversity. Knowledge of the strength of the association of different indicators is useful for managers seeking a parsimonious set of indicators that covers many of the dimensions of biodiversity. We found that the indicators we developed were weakly to highly correlated with each other. It is not surprising to find strong correspondence among some of the indicators because they were weighted toward late-successional species and structures and were described in terms of many of the same habitat variables. The associations between Northern Spotted Owl habitat and scores of the old-growth index were only moderate in year 0, which might be surprising to some, given the characterization of the owl as the umbrella species for old growth (Simberloff 1998). At present, owl habitat would not be a good indicator of old-growth conditions because it would greatly overestimate the area of old growth, as we defined it. This is expected, given that owls use forest patches that may contain only some of the structures of old growth (e.g., large, broken-topped trees).

The increase in degree of association among many of the indicators by year 100 results from increasing development of old-forest structure and increasing contrast between late- and early-successional conditions. The increased correlation among late-successional species and old growth results from the increase in the area of high old-growth index scores in year 100, resulting in a steeper slope for the positive relationship among old-growth scores and HCI scores of late-successional species. The stronger negative association among late-successional species and early-successional species in year 100 results from the inverse phenomena: an increased difference in HCI scores for a given pixel between early- and late-successional species. The increase in negative associations of many of the old-forest indicators and hardwoods results from the same phenomenon.

Variation in strength of association among the indicators suggests that landscape condition affects the strength of correspondence among them. This finding would argue for using multiple indicators rather than relying on a small set. Another reason for using multiple indicators, especially structural indicators in addition to focal species, is that ecological trends may be missed with a small set of focal species. In our case, the decline of hardwoods was not reflected in any of the focal species we selected. Indicators complemented each other and provided different pictures of change. For example, the diversity measures showed that private lands had higher patch-type diversity than public lands. But the forest structure analysis showed that public lands would be the location of old forests that are regionally rare. The use of the historical range of variability provided a temporal context revealing that despite the strong increase in older forests at 100 yr into the future, the province would still be considerably different from the one expected under the presettlement wildfire regime.

Scope and limitations

The assessment of policy effects on biodiversity indicators is limited in several ways. First, the biodiversity index models were developed using empirical relationships whenever possible, but they were largely based on literature and expert opinion; only a limited number of field data sets from the Coast Range were available for model verification. Second, index models assume that change in an index value relates to change in habitat quality (reflected as fitness) for each species. But empirical data were not available to test that assumption. The index is most appropriately used to allow evaluation of direction of trends and comparison of trends resulting from different management practices and policies. Third, the species habitat models were based on recent conditions in the Oregon Coast Range and may not perform similarly in other conditions of climate or landscape dynamics and structure. Fourth, complexity associated with interacting model functions prevents testing the models as a whole. The indicators are dependent on underlying models that predict initial vegetation, landscape dynamics, stand development, and coarse woody debris dynamics, all of which contain errors and constraining assumptions. Such models cannot be tested in a typical scientific experiment. However, we did evaluate the fundamental design of the policies, given their objectives and assumptions at the time they were developed. For example, we tested whether these policies are likely to achieve their goals in the future, under our assumptions, and whether different policies affected patterns of biological diversity. Despite the limitations of our models, they represent "thought experiments" that can give us insights into the possible outcomes of forest management policies (Oreskes 1997).

The appropriate scale of application of the results is the entire Coast Range or large watersheds and landscapes (probably >1000 ha) that comprise the area. While the models show considerable fine-grained detail for an area of large extent, the results should be viewed cautiously for small areas. Nevertheless, the fine-grained pattern that results from sources such as topography, logging units, and small gap disturbances appears to be a reasonable approximation of patterns that could develop; it does provide a general picture of the development of habitats and structures at the scale of small patches (e.g., ~1 ha), or narrow linear features such as riparian zones.

Conclusion

This study indicates that recently enacted forest policies could lead to major changes in terrestrial biodiversity in the Oregon Coast Range Province. Many of these changes are expected under the current policies, especially the increase in area of late-successional forest and habitat for associated species. When examined across all ownerships, however, some trends emerge that may be of concern, including the declines in hardwood forest area and structurally and compositionally diverse open and semi-open forest types. The multi-ownership perspective also shows that although forest-type diversity changes strongly within ownerships, changes in landscape diversity may be smaller at broad scales because of counteracting trends among ownerships. Comparison of alternative policy scenarios indicates that stand-level actions (e.g., live tree retention, thinning) can have either strong or weak effects at broad scales. The degree to which silvicultural practices at stand levels influence broad landscapes depends on the extent of the management action, the degree of change in stand structure, and the particular ecological measure examined. This finding suggests that it is difficult to predict the broad-scale consequences of fine-scale management actions without simulations that encompass a range of scales and a diversity of ecological measures.

We also found that a suite of biodiversity measures gives a more comprehensive picture of policy effects than a few focal species or structure measures alone. Although individual measures can be relatively highly correlated with each other, they do not necessarily function as surrogates for each other, and the degree of correspondence among them may vary with landscape condition. Further study is needed to determine whether general principles can be developed to help managers and policy makers select and effectively apply ecological measures or indicators.

This study went beyond approaches to conservation planning that rely only on land allocation design and emphasize indicators based on reserve layout criteria (e.g., maps of known locations of species or vegetation types, or spatial patterns of reserves), such as those in previous assessments (Forest Ecosystem Management Assessment Team 1993, Noss 1993). Rather, we took the existing mosaic of land uses and reserves and projected vegetation dynamics and resulting changes in multiple indicators of biodiversity based on forest stand and landscape structure and composition. The land allocation approach to conservation planning is typically the first step in any regional plan and can be carried out relatively quickly with a limited set of GIS resources (see Noss 1993). However, plans developed in this way should be considered working hypotheses that need to be tested further using multi-scale models that incorporate landscape dynamics resulting from human and natural sources. Approaches such as ours here can help policy makers, managers, and the public visualize the general appearance of habitat structure in the future, as well as show how the allocations might affect trends in species and structural indicators. Such analyses may in turn reveal gaps, as we found, in the biodiversity strategy, and help us understand the effects of management actions over time and across multiple ownerships.

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APPENDIX A

Habitat capability index for Marbled Murrelet (*Brachyramphus marmoratus*) (*Ecological Archives* A017-003-A1).

APPENDIX B

Habitat capability index for Western Bluebird (*Sialia mexicana*) (*Ecological Archives* A017-003-A2).

APPENDIX C

Habitat capability index for Olive-sided Flycatcher (*Contopus cooperi*) (*Ecological Archives* A017-003-A3).

APPENDIX D

Habitat capability index for red tree vole (*Arborimus longicaudus*) (*Ecological Archives* A017-003-A4).

APPENDIX E

Epiphytic lichen index (*Ecological Archives* A017-003-A5).