

Chapter 8

The Importance of Scale in Assessing the Compatibility of Forest Commodities and Biodiversity

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1. Introduction

The idea that forest commodities and biological diversity can be simultaneously produced from the same area in a socially acceptable manner (Haynes et al. 2002) is common in natural resources management, but is infrequently tested in natural resources science. This concept is related to the principle of multiple use, which was encoded in the Multiple-Use Sustained Yield Act of 1960 and endorsed in the National Forest Management Act of 1976. It is clear from the heated controversies and changes in forest management in recent decades that managers of the national forests have found it extremely difficult to achieve a socially acceptable mix of commodity production and biodiversity conditions. Although the idea of compatible uses is prevalent, systematic scientific approaches rarely have been used to test particular management approaches at any spatial scale (Haynes et al. 2002). Field experiments have only recently begun to evaluate consequences of alternative stand-level management activities on both commodities and measures of biological diversity (Monserud 2002). While information from silvicultural experiments is certainly needed, finding compatible mixes of different forest uses requires a multiscale perspective. Ecological, social, and economic patterns and

processes occur at different scales within spatial, temporal, and phenomenological hierarchies. What may appear compatible at one scale may not be at another. Unfortunately, our scientific understanding of compatibility problems viewed from a multiscale perspective is not well developed. For example, we lack examples demonstrating the effects of regional coarse-scale information on decisionmaking in ecosystem management at landscape and stand scales.

A better understanding of how scale influences compatibility is needed to effectively manage for diverse forest values. Because scale has different meanings and can be an abstract concept, concrete examples of scale effects are needed. The goals of this paper are to (1) identify the major components of scale from an ecological perspective, (2) examine how scale influences perceptions of ecological condition, and (3) examine how ownership and stand-level management actions can be expanded to the regional scale in terms of biodiversity and timber production. The essential question that we address is: How does scale influence measures and perceptions of the compatibility between forest biodiversity and commodities?

We emphasize assessments of ecological and socioeconomic conditions that use spatial information. Not all assessments require spatial information. For example, national and regional assessments of economic and environmental conditions under the Resources Planning Act of 1974 are based on aggregating information from forest inventory plots. In recent years, regional and landscape assessments have emphasized spatial information both to deal with ecological and socioeconomic relationships that are inherently spatial, and to better communicate complex information to general audiences (Johnson et al. 1999). As these spatial assessments have become more common, scale effects have become more apparent, and the need to understand the role of scale in assessing compatibility has become more urgent.

2. Dimensions of Scale

Scale has many different meanings, ranging from the geographer's cartographic scale (i.e., the ratio of distance on a map to distance on the ground) to the ecosystem scientist's concept of levels of scale in an ecological hierarchy (e.g., cells to organisms to biosphere) (O'Neill et al. 1986, Schneider 2001, Dungan et al. 2002). Scale concepts also have relevance to both ecological and socioeconomic sciences and management (Haynes et al. 1998). Typically, in natural resources sciences, scale concepts relate to spatial and temporal characteristics such as *extent*, the area or length of time of a study or management action, or *resolution*, the grain or finest unit of area or time that is measured or recognized. Many ecological and socioeconomic phenomena can be mapped and then characterized in terms of both extent and resolution. One general spatial ecological hierarchy is: patches—landscapes/watersheds—provinces—regions—continents—globe. The elements of this hierarchy can be defined in

various ways, including the structure and composition of vegetation, soils, topography, climate, and disturbance regimes. As one moves up this ladder, the spatial dimensions of each level increase but the exact sizes will vary with the user, except for the continents and globe, which are spatially well defined. A parallel, spatially defined socioeconomic hierarchy in forest management is: forest cutting unit—forest districts—ownership types—counties—states—nations—globe.

Scale can also refer to phenomena that are not readily mapped or defined in spatial or temporal terms. For example, ecological hierarchies, such as organism—population—community—ecosystem are defined more by levels of complexity and interaction than by particular spatial or temporal characteristics (O'Neill et al. 1986). In many cases, complexity hierarchies can be approximated with spatial or temporal units, but they are generally open systems (Waring and Schlesinger 1985) that depend on complex flows of biological or physical materials. Boundaries drawn around these units on a map are typically done for a particular study or management objective and are largely artificial. A parallel example of a socioeconomic complexity hierarchy that is difficult to define spatially is person—families—social networks—organizations—common markets. In all these cases, each level is nested in the next higher level.

An important scale question in natural resources management is, how do natural or management actions at one level in a scale hierarchy affect ecological or socioeconomic conditions at other levels? This question deals with *scaling* or the multiple scale aspect of the scale problems. Scaling effects are particularly relevant to the concept of compatibility because actions that may be compatible at one scale may not be at another. If the natural resource manager better understands how scale affects the interactions between potentially conflicting uses of the forest, then he or she may be able to more effectively produce the diverse goods and services from forests that society demands.

Ideally, management and research should consider all scales in decision or analysis but this is generally not feasible. Hierarchy theory (Allen and Starr 1988) provides some guidance for dealing with this problem. According to this concept, phenomena of interest can be treated as in a nested hierarchy, in which a focal scale (e.g., a particular watershed or landscape) for analysis is composed of a lower level of finer scale patterns and processes and is also nested in a broader scale (e.g., a subregion or region), which may constrain the range of condition or behaviors of elements at the finer scale. Thus, if we are interested in the dynamics of forest landscape, we should consider the behavior of stands or patches of trees at the next finer scale and the climatic or political context of the landscape at the next broader scale. Hierarchy theory provides a general framework for thinking about scale, but in practice it can be difficult to know which finer scale or coarse-scale elements to include in an analysis or decision.

Decisions about scale are frequently determined by the nature of the problem. In some cases, the selection of the scales of analysis (either in terms of extent or resolution) emerges from a particular ecological question such as, how do we maintain a population of an endangered species? Or, what influences water temperature in a particular stream? In these cases, information is needed about the kind of forests used for habitat, size of area required for raising offspring, the dispersal distance, the characteristics of individual fine-scale habitat elements such as nest trees or germination sites, the location of streams, riparian vegetation, and geology. In other cases, forest policies provide direction on scale. For example the Northwest Forest Plan (FEMAT 1993) specified the extent of the area (range of the northern spotted owl (*Strix occidentalis caurina*)) and some of the decision criteria (e.g., species viability) that indirectly set some bounds on scale and resolution. After the range of scales is narrowed by scientific and political considerations, other factors will come into play; these include constraints set by resources, analytical capacity, and information availability.

3. Influence of Scale on Ecological Conditions

Because ecosystems and social systems are open and hierarchical, our perception depends on our view. In this section we examine some examples of how spatial and organizational scales influence measures of ecological condition. We draw on specific examples from our work in the Oregon Coast Range.

3.1. Spatial Extent

The spatial extent of the analysis area will influence the estimate of conditions within an area. Most ecological indicators are influenced by the extent over which they are observed. For example, the number of species in an area, the proportion of different habitat or ownership types, and the historical range of variability are all dependent on the extent of the analysis area.

This phenomenon is illustrated by Wimberly et al. (2000) in a study of the historical range of variability in amount of old forest in the Oregon Coast Range. Historical range of variability is an indicator for recognizing that biodiversity of an area is not static and the dynamics of habitats and species are actually the source of long-term diversity and productivity of ecosystems (Landres et al. 1999). Using the mean of a forest condition (e.g., amount of habitat or population level of a species) to set goals for a landscape will be of limited value in characterizing the condition of dynamic ecosystems. Instead, the variation of those conditions may be a more suitable indicator than the mean. The extent of time used to estimate this range is ideally long enough to encompass natural variation in disturbances experienced by species and

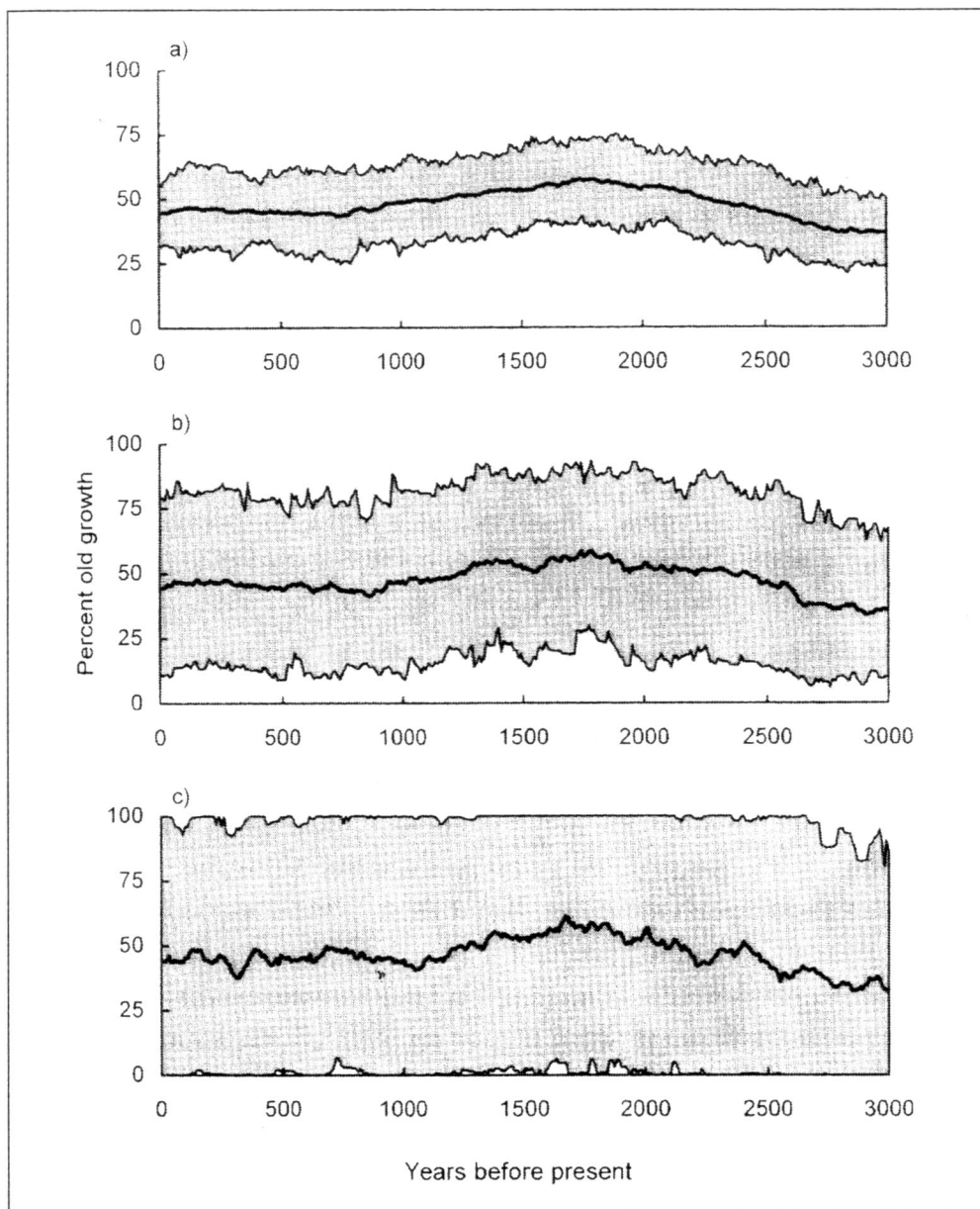


Figure 1. Simulated temporal variability of percentage of old-growth forest over the past 3,000 years in the Oregon Coast Range, summarized at three spatial scales: (a) province (2 265 000 ha), (b) national forest (300 000 ha), and (c) old-growth reserve (40 000 ha). Solid line represents the mean and the shaded area is bounded by the 5% and 95% quantiles (from Wimberly et al. 2000).

processes of an area within a period of similar climate. This may be on the order of 100s to 1,000s of years.

Wimberly et al. (2000) estimated the historical range of variability (HRV) in the Oregon Coast Range by using dendrological and paleoecological information and a spatial simulation model. In this area, the natural fire regime of the last 1,000 years was characterized by relatively large fires, a large proportion of which burned with high severity at frequencies ranging from 75 to more than 300 years, depending on the environment. They estimate the HRV (defined as the range of conditions that would be encountered 90% of the time)

for forest age classes at three different spatial extents: a forest reserve of 40 000 (hectares) ha, a national forest of 300 000 ha, and the entire province, 2 265 000 ha (Figure 1). They found that the HRV of old-growth forests (more than 200 years old) for areas the size of reserves and even national forests was so large (0 to 100%; and 10 to 90% of the area, respectively) that the measure had limited use as an indicator at these scales. At the province level however, the HRV in coverage of late successional forest conditions was about 25 to 75%, a range that was more sensitive, and thus more useful to management than a range in which any state was possible.

3.2. Spatial Resolution

All spatial assessments such as those done by the Forest Ecosystem Management Assessment Team (FEMAT 1993) and the Interior Columbia River Basin Ecosystem Management Project (ICBEMP) (USDA 1996) have a minimum mapping area or grain of analysis that can affect the nature of the assessment and its utility. For example, in FEMAT, the vegetation layers were mapped to a resolution of about 16 ha. In ICBEMP, the resolution was about 1 km or 100 ha for the entire area, although finer resolution analyses were done for subsets of the regions in both assessments. Relatively large minimum assessment units can still be valuable for regional assessments but some indicators of ecological condition cannot be evaluated at this scale. An alternative to mapping finer scale attributes is estimating the distributions of conditions within pixels or minimum mapping units. This can be done if data are available to make predictions at finer scales based on coarser scale patterns, such as geology, ownership, and forest type.

In some cases, the spatial location of fine-scale features may be important and force the assessment to use smaller minimum mapping units. For example, fine-scale information is frequently sought for questions about riparian management. Forest and stream interactions occur over relatively small areas, and the spatial position of streams, forests, landslides, and roads can make a difference to the behavior of the system. Stream gradients and junction angles can be used to predict how debris flows through drainage networks and influences riparian habitat (Miller et al., in preparation). To make this prediction it is important to know the pattern of slope gradients across an area. Digital elevation models (DEM) can be used, but if the spatial resolution is too coarse, the models will not accurately estimate the distribution of slope in a watershed, which is an important determinant of landslide risk (Miller et al., in preparation). For example, Clarke and Burnett (in press) found that 10-meter (m) DEMs were far superior to 30-m DEMs for characterizing hydrological patterns and aquatic habitat potential (Figure 2). The least agreement between the two models occurs in slope classes greater than 65%, which are the slopes with

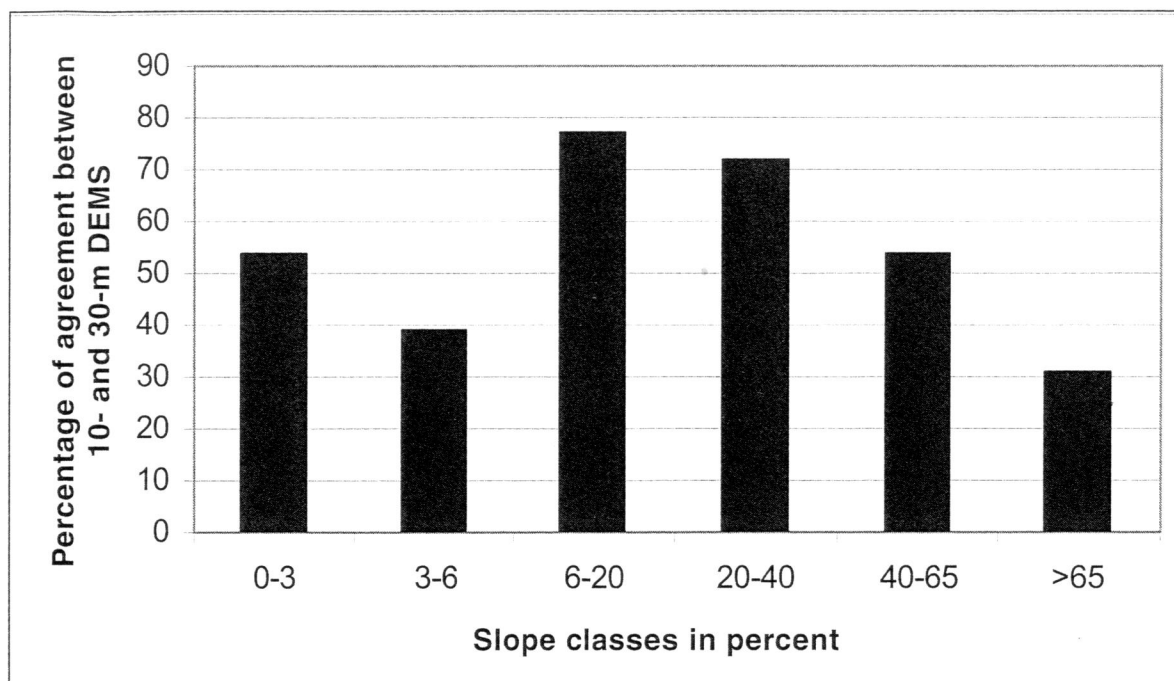


Figure 2. Percentage of agreement between area estimates of slope classes based on 10-meter DEM and 30-meter DEM from nine 7.5-minute quadrangles from the Oregon Coast Range (based on Clarke and Burnett, *in press*).

Note: DEM = digital elevation model.

the greatest probability of land sliding. Consequently, an assessment of landslide risk based on 30-m DEMs will greatly underestimate risk in steep mountainous areas.

Resolution is also important from a social perspective. When people view maps they often want to “see themselves in the landscape” to effectively participate in decisions. This often means that coarse resolution maps that can not identify stand level features (roughly at least 1 to 10 ha) are not considered relevant or precise enough for assessment purposes. As assessment and planning for forests has become more a collaborative process involving the public, it is increasingly important that the chosen spatial resolution enables citizens to relate to the representation of the forests being studied.

Despite ecological and social pressures to increase the spatial resolution of regional assessments, the full detail of natural systems can never be completely represented in an analysis. Also, the finer the detail considered, the more likely that an analysis may be overwhelmed with the sheer volume of information available. Choosing the spatial resolution for analysis can be one of the most important decisions made in an assessment, and one that will always be full of compromises. As remote sensing, GIS, and computer technology give us the capability to incorporate increasingly fine scale information, our conceptual and institutional ability to use that information (or to know when not to use it) has to increase as well.

4. Ecological Resolution

Ecological entities exist in a hierarchy of ecological scales ranging from individuals to ecosystems (Noss 1990). Fine resolution entities include species, whereas coarse level entities include vegetation types and disturbance regimes. While coarse level indicators play an important role in assessing and maintaining biological diversity, they do not necessarily capture individual species, particularly those that are rare and endemic. However, it is not possible or even necessary to track every species in an area—species are too numerous, the ecology and dynamics of their populations are too poorly known, and resources are not available to manage very many of them individually (Noss et al. 1997). Consequently, many have called for a strategy in which high-priority species, such as threatened and endangered species, are combined with coarser resolution information (e.g., vegetation types, landscape pattern, and disturbance regimes) (Noss et al. 1997).

To identify these priority species, some have proposed using focal species, not as indicators of other species, but as part of sets of indicators that collectively provide a picture of different ecosystems or elements of ecosystems (Lambeck 1997, Noss et al. 1997, and the Committee of Scientists 1999). In the range of the northern spotted owl, it is sometimes mentioned as an “umbrella” species for late-successional and old-growth ecosystems. In fact, the range of the owl set the boundaries of the FEMAT (1993) assessment area. Three small mammal species have been proposed to measure stand biocomplexity in Douglas-fir (*Pseudotsuga menziesii* (Mirb.) Franco) forests (Carey 1995, Carey and Harrington 2001). However, Lindenmayer et al. (2002) caution that focal species have the same general limitations as indicator species (e.g., species do not respond in the same way to forest structure or disturbance) and that conservation planning and assessment should not solely rely on this approach, but rather on a wide range of strategies.

We developed a set of focal species for coastal Oregon forest management assessments (Spies et al. 2002c). This set is limited, but illustrates how individual species differ in their use of the environment. Our focal species approach was based on habitat suitability models for individual species (McComb et al. 2002). We focused on habitat quality for (1) northern spotted owl, (2) pileated woodpecker (*Dryocopus pileatus*), and (3) western bluebird (*Sialia mexicana*). In addition, we developed an indicator of old-growth development that represented a forest structure-based approach (Lindenmayer et al. 2002). The first two focal species are related to closed forest conditions and are threatened by loss of older forest habitat and large dead trees. The western bluebird is a species representative of open, early-successional forest types. Populations of the bluebird in western Oregon are listed as “sensitive” because of population declines that are possibly related to competition for nest holes from the introduced European starlings (*Sturnus vulgaris*) (Csuti et al. 1997).

Each of these indicators integrates the environment differently in both space and time. For example, the foraging area for northern spotted owls is 1800 ha, while the territory for western bluebirds is less than a hectare. Pileated woodpeckers have a relatively large foraging area, over 400 ha, but require particular microsites, such as relatively large dead trees for nesting. Old-growth structure and development is primarily a stand-level phenomenon based on live and dead attributes occurring over an area of about 1 ha; the index also has a time dimension (age) because some of the biodiversity of older forests is not a product of structure but of time in which species accumulate and spatial diversity increases. In contrast to old growth, the habitat for western bluebird in this forested region is ephemeral, lasting only 10 to 15 years until conifer canopies close.

Habitat patterns of focal species are used as indicators of a larger ecosystem type. However, the habitat of a single focal species does not always match that of its associated forest ecosystem type. For example, in the Oregon Coast Range the habitat distribution of the northern spotted owl, which has been classified as "closely associated" with old growth (Ruggiero et al. 1991) overlaps 75% of the area of old growth defined by the old-growth index (Figure 3). While 75% might be considered relatively high, if the habitat of the owl were the only criterion used to evaluate old-growth habitat, a significant portion of the existing stands with moderate to high old-growth development would not be included. The habitat of the northern spotted owl is distributed differently than that of old growth largely because the owl has specific nest requirements that are a subset of the range of old-growth conditions. A spotted owl uses thousands of meters of forest in different stages of development around its nest site (McComb et al. 2002). This example illustrates the need to use multiple focal species as well as indicators based on forest structure and dynamics in ecological assessments. Given the high dimensionality of ecosystems (i.e., components of ecological variation relating to a large number of relatively independent factors such as effects of disturbance, physical environment, and dispersal capability of the species), several different types of indicators are needed. Theoretically, the number needed to completely represent the ecosystem is closely related to the number of species, interactions among species, number of habitats, variety of disturbance regimes, and variety of physical environments. Complete enumeration, however, is not possible or practical in a management context.

5. Expanding Scale from Stands to Regions

Perhaps the most interesting scaling problems involve estimating how conditions or actions at fine spatial scales affect large areas. For example, a planner may want to extrapolate plot-level inventories to create a vegetation map of larger areas (see Ohmann and Gregory 2002). Policymakers may want to know

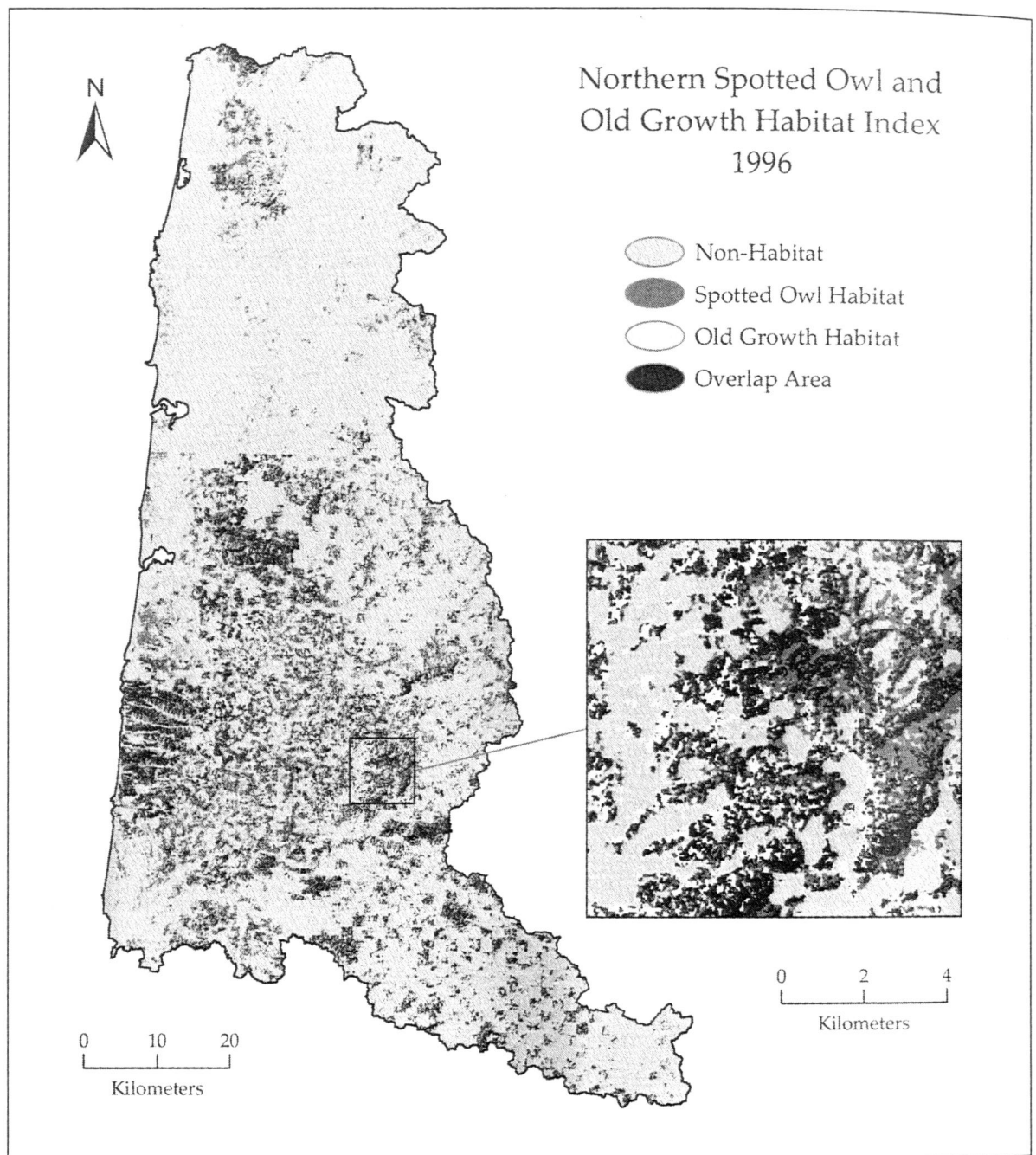


Figure 3. Area of overlap between suitable habitat for northern spotted owls (McComb et al. 2002) and moderate to high old-growth development, and areas where owl and old-growth habitat do not overlap in the central Oregon Coast Range.

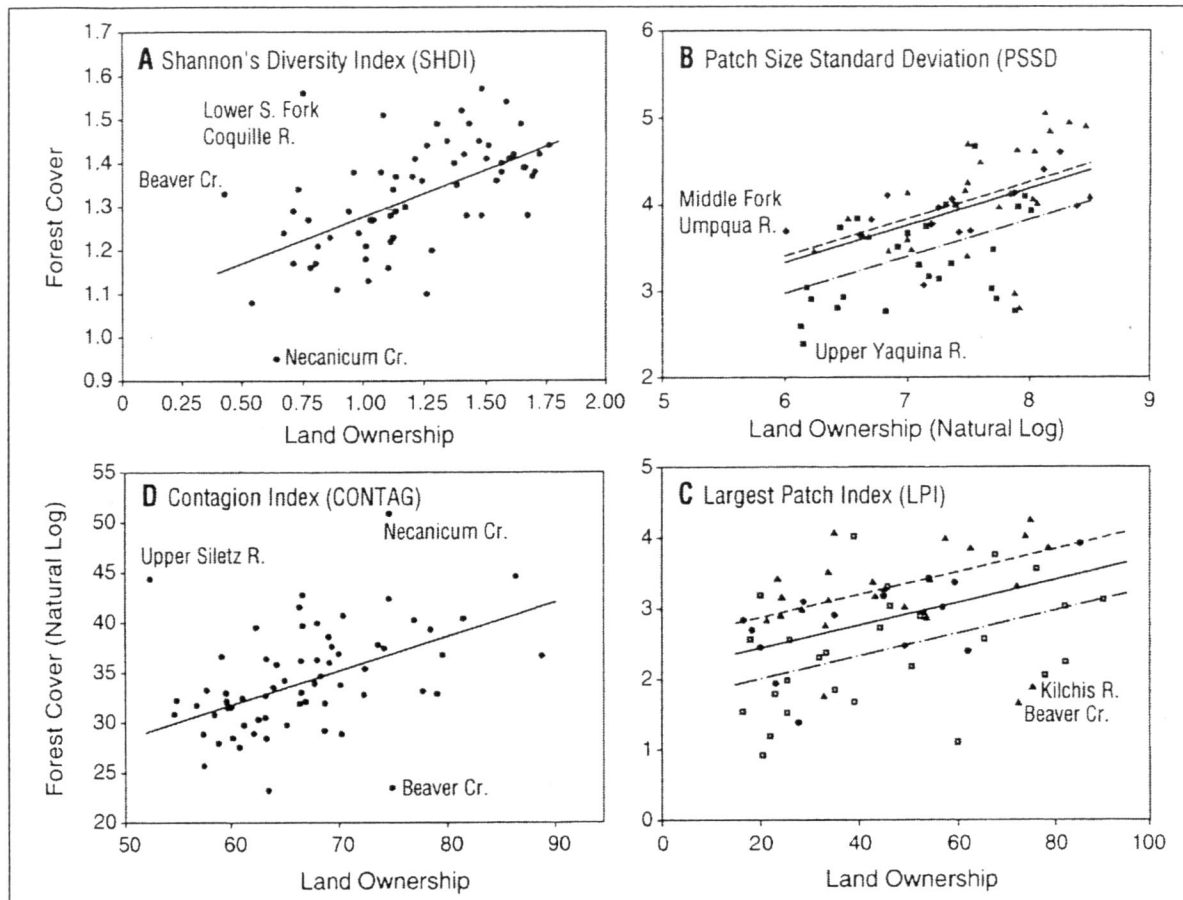


Figure 4. Relationships between landownership and landscape characteristics of forest cover types for 66 fifth-field watersheds in the Oregon Coast Range (from Stanfield et al., in press). Fifth-field watershed (Hydrologic Unit Code 5) are approximately 20 000 to 100 000 ha in size (Seaber et al. 1987). Units for figures are the same for x and y axis; A is unitless, B is hectares, and C and D are percentages.

how the practices of individual landowners affect aggregate conditions of large landscapes or regions. They may also want to know how alternative management approaches at the stand level affect biodiversity and economics over larger areas. In this section we examine these types of scaling problems.

Ownership diversity can lead to ecological diversity if the full range of forest management intensities is present in an area. For example, Stanfield et al. (in press) found that the diversity of forest cover type and spatial patterns could be explained by the underlying ownership pattern (Figure 4). The patterns they observed are a snap-shot of current conditions that are a product of more than two centuries of natural and human actions in the Oregon Coast Range (Wimberly and Ohmann, in review). The study reveals how ownership patterns and landowner behavior can help explain variation in biodiversity in a landscape or region.

Recently developed tools and scientific frameworks are useful for studying how management effects are magnified in space and time. Nonspatial forest planning tools have been used to extrapolate management effects around for many years (Davis et al. 2001, Monserud 2003). However, many ecological

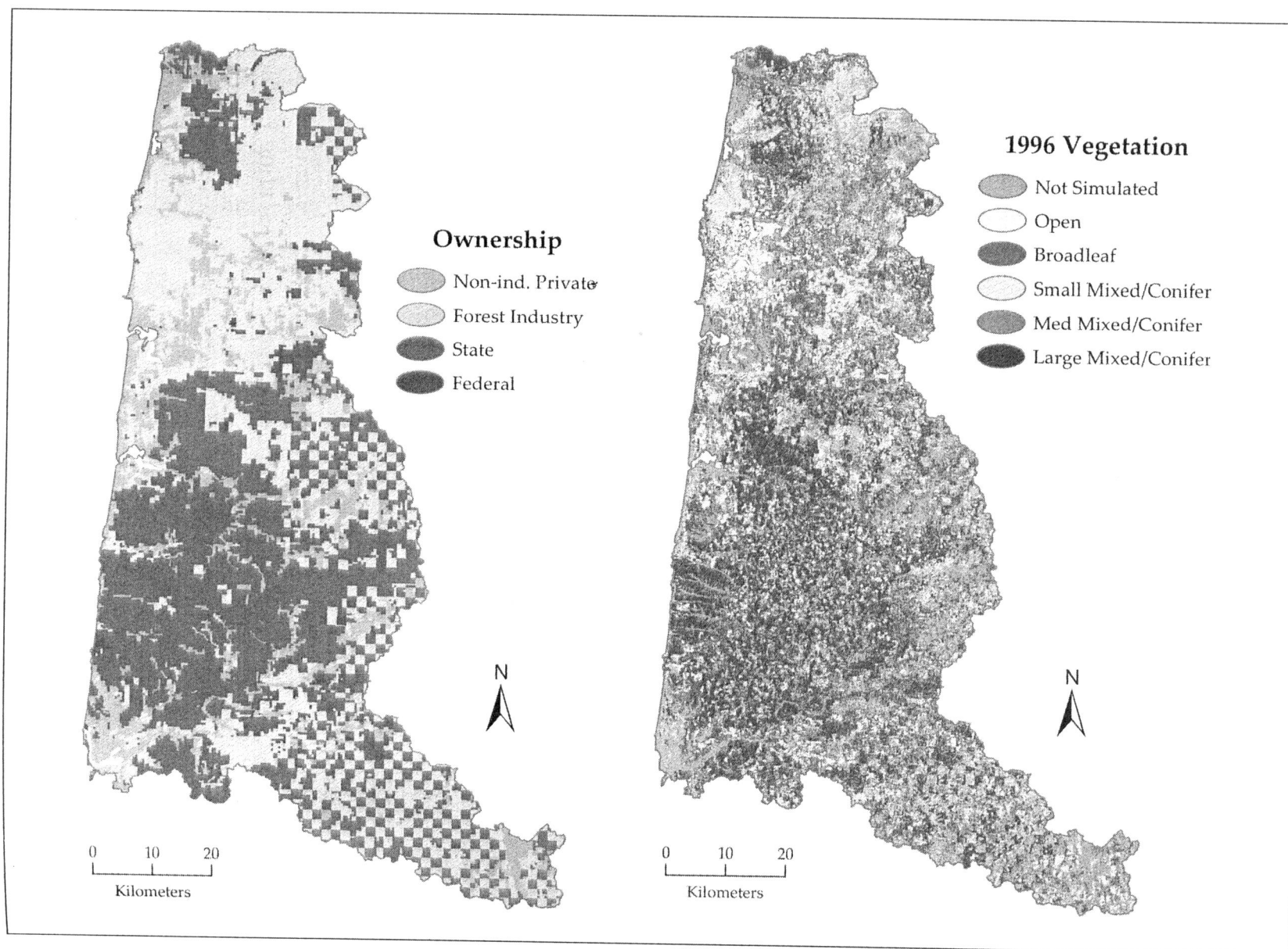


Figure 5. Vegetation and ownership patterns in 1996 across multiple ownerships in the central Oregon Coast Range (from Ohmann and Gregory 2002).

issues require spatial information such as the pattern of riparian zones or the pattern of habitat for wildlife species. New modeling systems are being developed that use spatial information at relatively fine scales to project landscape changes and their consequences (Mladenoff and Baker 1999, Davis et al. 2001). Many management and sustainability questions revolve around how actions at an individual site or stand level affect ecosystem conditions and economic outputs at broader scales. Frequently, this is done by putting forest conditions into several cover types or age classes and then projecting transitions among those classes based on management and disturbance probabilities (Cissel et al. 1999, Pearson et al. 1999). This approach is useful for understanding many landscape-scale effects and processes but it is not well suited for many forestry questions that involve projecting the effects of different silvicultural practices at broad scales. For example, the numerous combinations of initial stand conditions, management practices, and environmental conditions in a region would be difficult to incorporate into an approach based on 10 to 25 forest patch types. The problem is especially complex when landscapes include multiple owners, a situation typically characterized by non-uniform spatial information about vegetation conditions and inadequate specification of management behavior of different ownership types for model projections. In the Coastal Landscape Analysis and Modeling Study (CLAMS) (Ohmann and Gregory 2002, Spies et al. 2002b,c) we developed a set of models that can retain the diversity of initial stand conditions and management strategies to estimate forest structure and compositional diversity with enough spatial and ecological resolution to handle many different fine- and coarse-scale ecological indicators.

CLAMS examines how policies across different forest ownerships in the Oregon Coast Range affect ecological and socioeconomic outputs. The Coast Range is a fitting study area because it contains owners with a wide variety of forest management goals and approaches and patterns of ownership (Table 1, Figure 5). In the CLAMS approach, a set of different spatial simulation models are linked under a conceptual framework in which changes in landscape condition are driven by natural gap disturbances, forest management actions, and land-use changes (Figure 6). The landscape conditions are then used to estimate potential effects on various measures of habitat potential for terrestrial and aquatic organisms (McComb et al. 2002), forest structure and composition, proportion of area in each seral stage, landslides, timber volume, and recreation. The following example illustrates how the CLAMS work can be used to examine the landscape level effects of different forest owner practices and the effects of changing stand-level management practices.

The methods used to simulate landscape change as result of landowner behavior follow those described in Spies et al. (2002b). We used timber harvest volume targets from the Northwest Forest Plan for federal lands (FEMAT 1993). For state lands, we found the maximum sustainable level

Table 1. Forest policies, goals, and strategies dealing with biological diversity in the Oregon Coast Range, by major ownership categories.

Ownership	Policies	Goals ¹	Strategies
USDA Forest Service	NW Forest Plan Forest Plans	Late successional/ old-growth forests, threatened & endangered species, aquatic ecosystems, commodities	Reserves, matrix, green-tree retention, stream buffers, adaptive management areas
USDI Bureau of Land Management	NW Forest Plan Forest Plans	Same as USDA Forest Service	Same as USDA Forest Service but with different matrix prescriptions
State of Oregon	Forest Plans	Healthy forests, indigenous species, abundant timber, threatened & endangered species	"Structure-based" active management, habitat conservation plans
Private industrial	Oregon Forest Practices Act	Growth and harvest of trees, protection of environment and fish and wildlife	Limited retention of individual trees, limited stream-side protection for fish- bearing streams
Private nonindustrial	Oregon Forest Practices Act	Same as private industrial	Same as private industrial

¹ Goals are listed in approximate order of priorities.

(based on current timber volume and expected future growth) subject to achieving certain structural conditions and size distribution of interior habitat patches. For private nonindustrial lands, we used probabilities of clearcutting or partial harvesting from the work of Lettman and Campbell (1997) on historical behavior of nonindustrial private landowners in western Oregon. For forest industry landowners, we simulated the maximum sustainable level for the management intensities they provided. Timber production was calculated as the average volume of timber produced over the 100-year simulation across all ownerships including volume produced by thinning. Our long-term goal is to use the period-by-period harvest targets and management intensities for private forests from the Adams et al. (2001) analysis of economic behavior of private forest landowners in western Oregon.

Stand growth and succession were simulated by using a growth and yield model (ORGANON) (Hann et al. 1995) and an ecological succession model (ZELIG) (Urban et al. 1999) to project the development of forest stands across the entire assessment area. The growth and yield model is used for forest

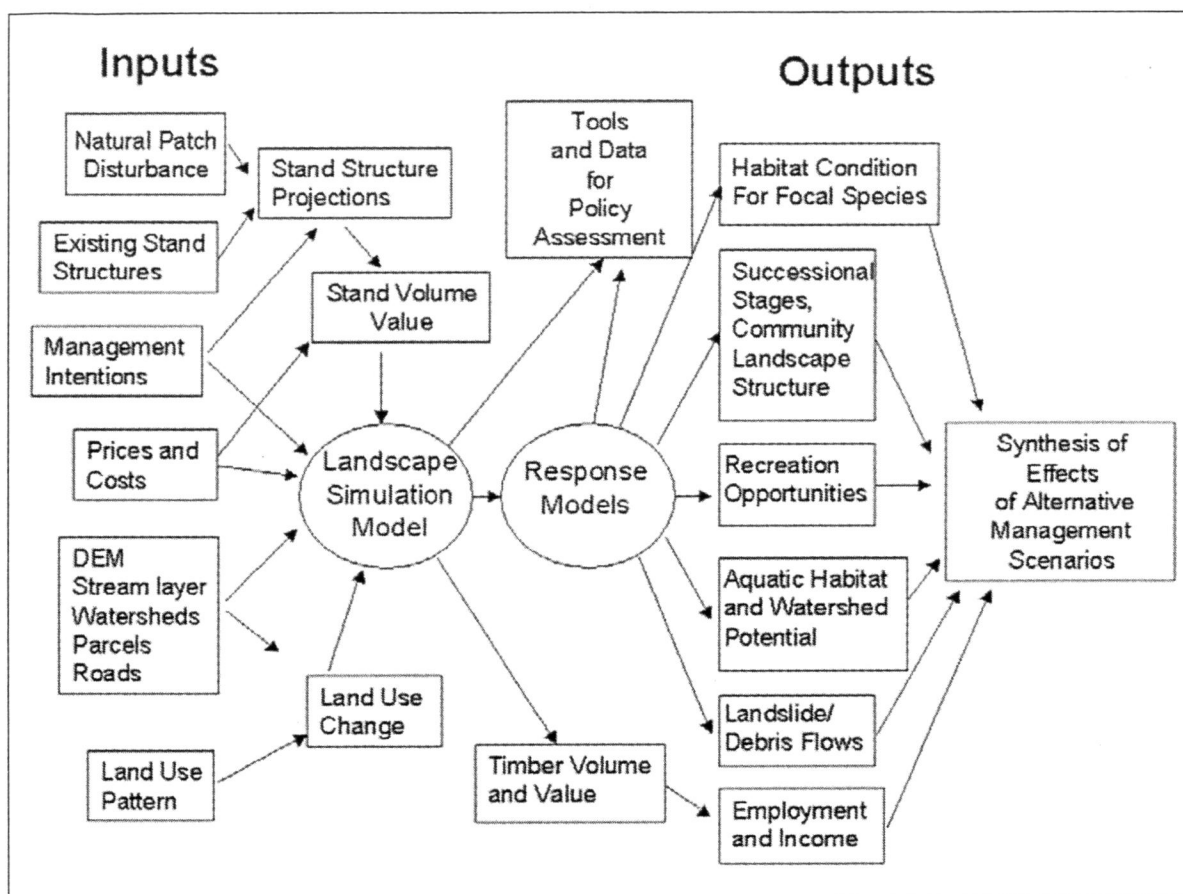


Figure 6. Conceptual model of components and links in CLAMS (Spies et al. 2002a).

management regimes characterized by young stands on short rotations (less than 80 years) and intensive forest management. The ecological succession model is used when stands are grown on longer rotations or allowed to grow without harvest or manipulation until stochastic disturbances occur. ZELIG was calibrated against ORGANON to produce about the same stand structural growth for the first 80 years, a time when mortality is largely density dependent and ingrowth of shade tolerant trees is minimal. The ORGANON model was based on stands less than 80 years old and does not estimate regeneration, ingrowth, and gap mortality that are characteristic of older stands. These models are run nonspatially on every combination of stand types (inventory plots) and management regimes. Results of each projection were then stored in a lookup table, later used in the landscape simulations.

Projections of the future landscape conditions of the Coast Range under current policies indicate that new landscape patterns will emerge over the next 100 years (Figure 7). Specifically, high-contrast landscapes will develop as federal and state agencies implement new management policies that promote the development of mature and old-growth forests to meet goals of providing habitat for late-successional species. These policies were developed largely independently of each other with relatively little effort to examine overall effects, much less coordinate strategies among ownerships. Consequently,

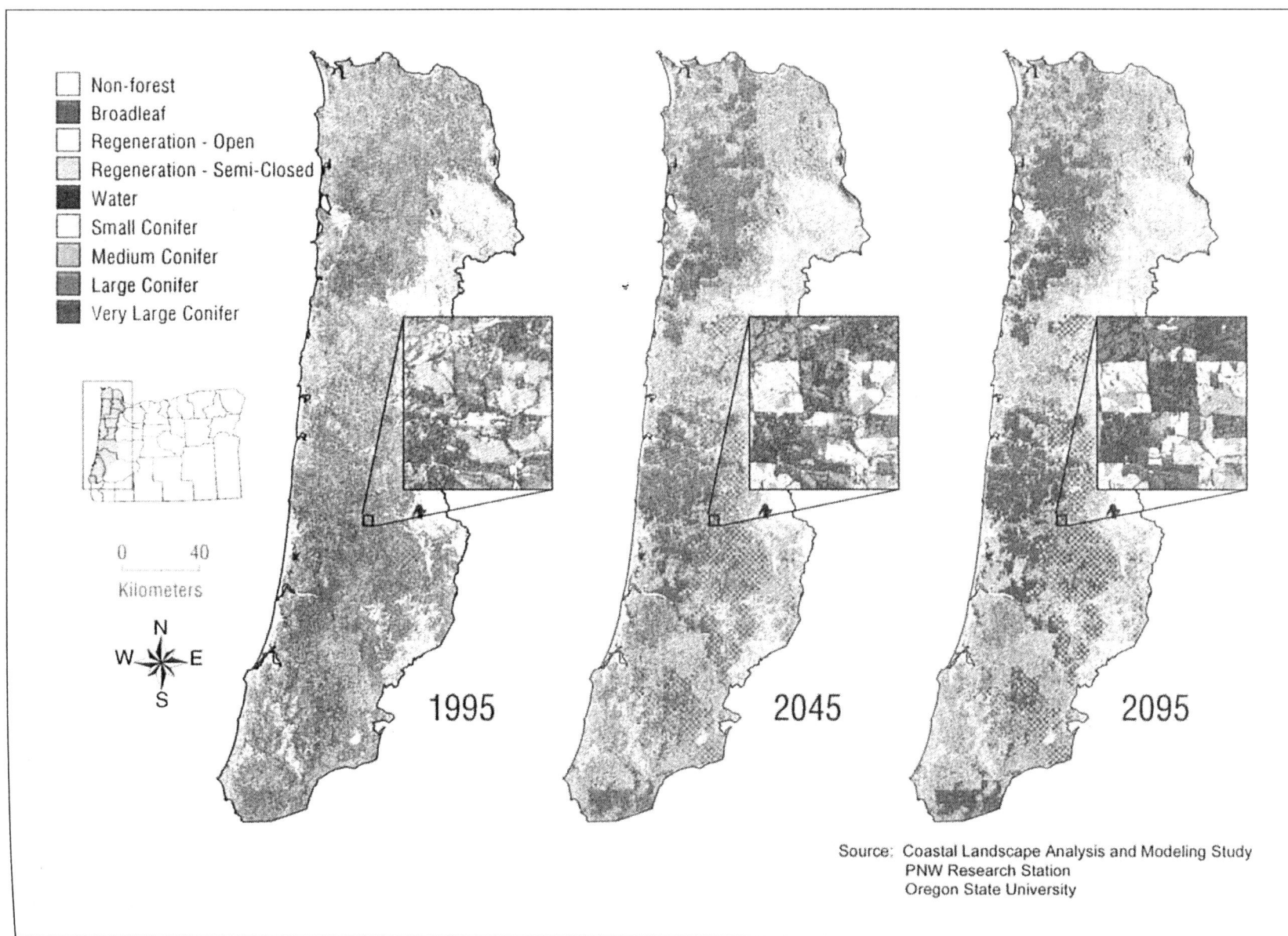


Table 2. Characteristics of alternatives used in evaluating ecological and timber production effects (from Spies et al. 2002b).

Alternative	Land ownerships	Description
Current policy	All	Current policies as indicated in federal and state forest plans and the Oregon Forest Practices Act
Increased retention	Private	Retain 5 conifers/ha > 60 cm dbh and 7 conifers/ha > 30 cm dbh
Increased thinning	Federal	Thin from below to 18 m ² /ha in stands less than 40 years old
Long rotation	Private	80-year rotation

some unexpected or unintended trends may occur. For example, the coastal landscape appears to be headed for a condition dominated by two primary states: old, structurally diverse forest and young, relatively uniform conifer plantations. Forests on nonindustrial or state lands may moderate this trend somewhat, but it still appears likely to emerge over time under current policy. This trend will result in sharp edges at ownership boundaries, decline of intermediate-aged forests (e.g., 50 to 150 years), and a decline in young ecologically diverse forests characterized by hardwoods, shrubs, and patches of live and dead large conifers. Although disturbance rates on private lands will be relatively high, particularly on private industrial land, less disturbance will occur in diverse, early-successional stages because private landowners are required to establish well-stocked conifer plantations within a few years after harvest. Federal managers are expected to suppress natural disturbances, such as fire, that reduce the area of older forest. The ecological consequences of these changes are not well understood and therefore, a subject of continuing research.

Forest management occurs at the stand level when foresters cut and plant trees and inhibit the growth of competing shrubs and herbs. We do not completely understand how these changes in silviculture affect ecological and timber production potential across landscapes. A comparison between ecological and timber-production effects at landscape scales for a set of alternative silvicultural scenarios was completed (Table 2) (Spies et al. 2002b) for a 537 700 ha area of the central Coast Range. Major watershed boundaries were used to delineate the study area, which contained a diverse set of ownerships (Figure 7). The LAMPS model (Bettinger and Lennette 2002) was used to simulate 100 years of landscape change under the different scenarios.

A composite biodiversity index was constructed by averaging the mean scores at 100 years into the simulation of four ecological indices: (1) old-growth habitat, (2) northern spotted owl, (3) pileated woodpecker, and (4) western bluebird (Figure 8). The composite index was intended only to illustrate a possible general index of ecological integrity, which has been proposed

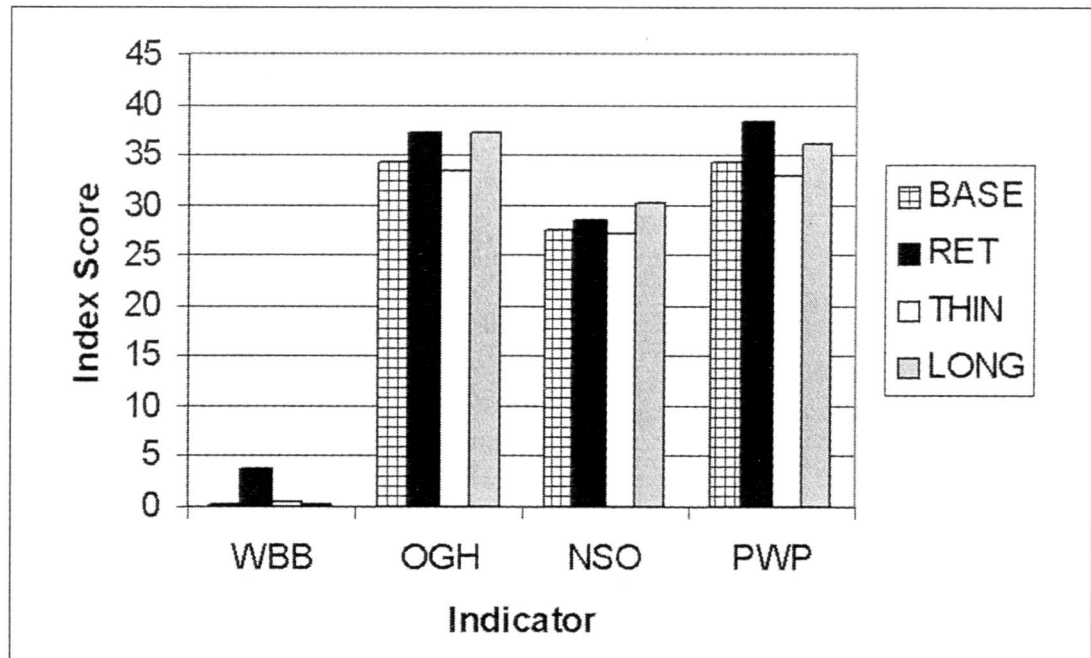


Figure 8. Mean index scores at 100 years for four different ecological indicators and four different scenarios in the central Coast Range. Index scores, which originally ranged from 0 to 1, have been multiplied by 100 to eliminate decimals. Ecological indicators: WBB = western bluebird; OGH = old-growth index; NSO = northern spotted owl; PWP = pileated woodpecker. Scenarios: BASE = base policy; RET = increased retention on private lands; THIN = thinning on federal land; LONG = 80-year minimum rotation on private lands (from Spies et al. 2002b).

as a single measure for the condition of biodiversity in an area (Karr 1991, Quigley et al. 2001). A more complete evaluation of the use of a measure for ecological integrity is needed. For example, indicators could be developed based on an expected landscape-scale distribution of index scores rather than a single mean score or a set of integrity indicators for different major components of an ecosystem.

In general, the absolute mean scores at 100 years of the individual indices differed little among the scenarios (Figure 8). However, for the western bluebird, the relative difference in mean scores was almost 10 times higher for the retention scenario than for any other scenario. This probably resulted from the increase in area of open conditions with snags, which would occur if higher densities of trees were left after clear-cutting on private lands. The retention and long-rotation scenarios resulted in slightly higher mean scores for all the biodiversity metrics when compared with the base policy and the enhanced thinning scenario. These increases in biodiversity metrics resulted from increased numbers of large trees and snags on private lands. It was somewhat unexpected that the thinning scenario did not change the biodiversity indices much over the base scenario. We expected thinning to increase the rate of development of large trees and, therefore, increase the scores of the ecological indices, which are sensitive to this attribute. Densities of large trees did increase faster with thinning, but the effects may not be large enough at 100

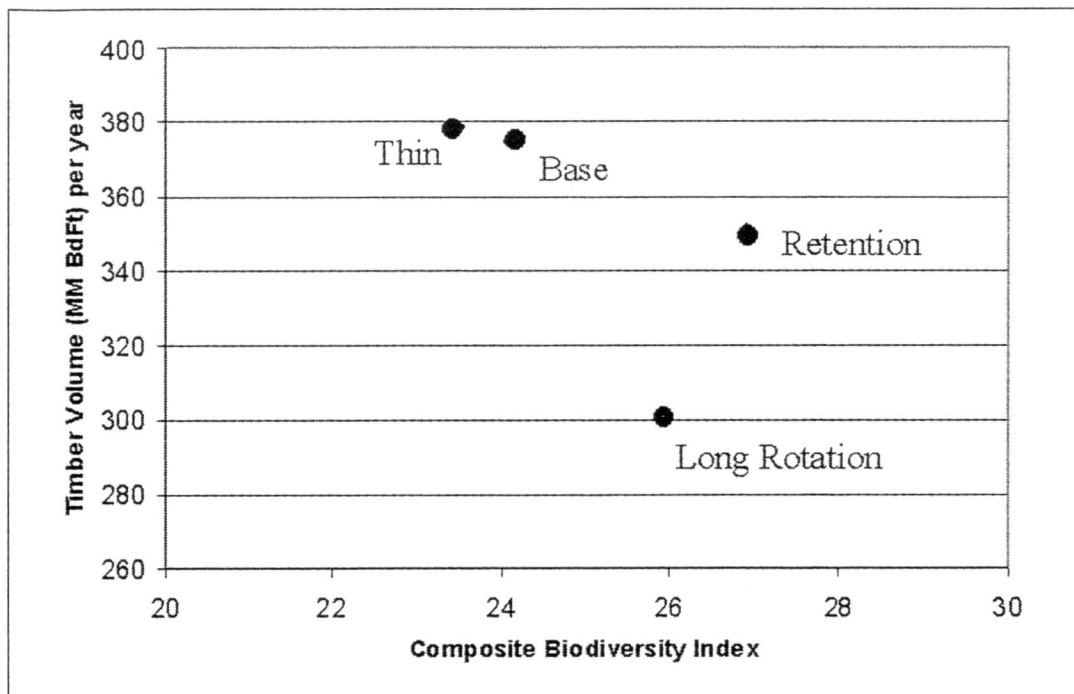


Figure 9. Relationship of timber volume production to the composite biodiversity index at year 100 for four different scenarios (see Figure 8 and Table 2 for explanations) (from Spies et al. 2002b).

years to influence the habitat models. It may be that the thinning regime we simulated did not remove enough biomass. It may also be that the thinning regime reduced amounts of standing dead and down wood, which would reduce the index scores for these measures. Finally, perhaps the thinning regime at a landscape scale was not large enough given the hectares involved. While over 40 500 ha were thinned in this scenario during the first 40 years on federal lands, this amount is only 24 300 ha more than the base policy or only about 5% of the entire central Coast Range area. Further analysis is needed to understand the reasons for the lack of effect of thinning on biodiversity measures at broad scales.

Relatively little difference existed among the scenarios in terms of timber production and the composite index of biodiversity (Figure 9). When compared to the base policy scenario, the greatest change was in the retention and long-rotation scenarios. In terms of the tradeoff between biodiversity and timber production, the retention option proved the more efficient of the two. Under the retention option, the increase in the biodiversity index came at lower cost in terms of reduced timber production. The thinning option differed little from the base policy scenario, showing a slight decline in biodiversity and a slight increase in timber volume. The long-rotation scenario resulted in the greatest decline in timber volume because during the first few decades there would be little or no timber cut that was older than 80 years. For example, during the first 25 years of the simulation, timber volume production per year from private lands would average 3074 cubic meters, whereas during the final

25 years of the study it would average 34 thousand cubic meters. While innovative commercial thinning regimes might offset some of the simulated loss in timber production, there is no doubt the long-rotation scenario significantly reduces private harvest for a number of decades.

The ecological indices are limited in at least two ways. First, they show only trends in habitat quality; this makes it difficult to say what the magnitude of the change means to other ecological criteria that require different types of analyses. However, the indices are useful to indicate direction of trends. A more sophisticated population viability analysis requires much more information (not available for most species) and many more assumptions than the habitat quality analysis we did here. A second limitation is that a single measure of ecological integrity may not change if its components are inversely related to each other. For example, early-successional indices will typically change in the opposite direction of late-successional indices. Consequently, several kinds of ecological integrity measures are needed to provide a more useful characterization of biological diversity.

The measures used to estimate timber production will also influence characterization of tradeoffs. It is relatively easy to measure timber production in terms of volume harvested over time or per time period as we did. However, landowners, especially the forest industry, are typically sensitive to the timing of the harvest or the financial attractiveness of the forest schedule. Most authors (e.g., Adams et al. 2001, Davis et al. 2001) assume that the forest industry attempts to maximize their wealth (net present value). Activities that increase the projected volume of harvest will not always increase net present value and vice versa. Thus, different measures of timber production and assumptions about future interest rates and financial risks are needed to fully evaluate compatibility of timber management and biological diversity from a variety of management viewpoints.

Compatibility is best evaluated by using a set of alternative scenarios. Without prior knowledge of the sensitivity of ecological and socioeconomic indicators to management alternatives, it can be difficult to find a range of scenarios that helps us see the possibilities for tradeoffs. The results show that a wider range of scenarios is needed to fully understand the relation between timber production and ecological conditions. This wide range of scenarios can be explored in a research setting but may prove less possible within a real policy environment where policymakers typically consider only incremental changes to current policies.

The observed lack of strong differences at landscape scales among the scenarios with different stand-level management may be a general phenomenon that occurs when fine-scale actions are extrapolated to broad scales. Stand-level effects can be diluted at broader scales if they occur in a relatively small portion of the total area, such as within one ownership or within one allocation of an ownership. Landscapes can also change slowly because of constraints on

future cutting activities or patterns that are imposed by accumulated patterns of managed stands created earlier. In addition, slow change can occur at broad scales simply because forest stands develop slowly following disturbance. Wallin et al. (1994) demonstrated that landscape patterns change slowly, and placement of cutting units early in a rotation can constrain the ability of managers to change landscape patterns in the future. Thus, the temporal extent (100 years) of our analysis may have been too short to observe the full effects of the alternatives.

Human perceptions may also mold our expectations regarding the effects of scaling stand-level activities to landscape scales. When viewing a forest area from the edge of a recent logging unit, the contrast can appear quite dramatic. It is difficult, if not impossible, for the human mind to correctly extrapolate the stand-level effects to large heterogeneous regions in the distant future.

Not all stand-level actions will be diluted at the broader scales. If the ecological process or pattern is restricted to only a small part of the total landscape, such as riparian zones, and the management action is concentrated in those areas, then the impact could be quite large, even though it affects a small portion of the total area. It is important to evaluate each ecological impact for the affected landscape area.

6. Conclusion

The scale of observation will influence the assessment of compatibility between forest commodities and biological diversity. The major hierarchies of scale include space, time, and complexity. Scale must be considered in terms of *extent* (the maximum area, timeframe, or scope of ecological and socioeconomic interactions), and *resolution* (smallest spatial and temporal or biological and ecological units). The process of aggregating fine-resolution information to broad extents is critical for examining variation in compatibility across scales; it is also influenced by decisions about extent and resolution. In this chapter we demonstrated that the extent of observation and the resolution of information will influence measures of ecological condition (e.g., historical range of variability and distribution of steep slopes).

We also demonstrated that decisions made at the scale of stands and individual landowners may have unpredictable consequences at the scale of regions and centuries. For example, the increase in contrast between ownerships and the potential decline in diverse, early-successional forests was not widely appreciated when new forest policies were established by landowners during the 1990s. We also demonstrated that individual fine-scale measures of biodiversity (e.g., habitat for the northern spotted owl) are insufficient to represent the distribution of more complex ecological entities such as old-growth forest ecosystems. At the same time we showed that aggregate measures of biodiversity based on species groups may not be sensitive to the

variations of individual species of the group (e.g., western bluebirds). The multidimensionality of ecological communities and ecosystems make it difficult to characterize biodiversity with just one or a few measures. Finally we showed that management influences compatibility (as measured by cost-efficient timber production) by increasing the aggregate habitat score of focal species. The differences among the alternatives were small, however, suggesting that a wider range of management options should be examined.

These findings have several implications for assessing compatibility in forestry. First, because measures and outcomes are influenced by scale and scale hierarchy, it is likely that some past conflicts in forest management have resulted from differences in scale of observation among planners, managers, scientists, modelers, and stakeholders. Some of these conflicts may have been avoided if the effects of scale were better appreciated and publicized. Second, in future assessments, it will be important to explicitly include scale in the analysis and conclusions. Decisions about scale are a type of assumption, reference point, or boundary condition that must be clarified when presenting the results of an assessment. Third, because assessment results can be strongly influenced by decisions about scale, a wide range of spatial, temporal, and complexity scales need to be examined when considering compatibility. Just as the image in a microscope or telescope changes with eyepieces of different magnifications, the image from our assessments changes with different scales of observation and measurement. Our analysis implies that it is not possible to achieve compatible uses by managing at one level. We cannot hope to embrace all the scales of nature and human systems in one assessment; however, we can do a better job of highlighting changing perceptions of compatibility as related to scale. It may be possible to identify compatibility outcomes that are relatively similar across a range of scales. Such outcomes may be more socially acceptable than those that are strongly influenced by the scale of observation.

The use of spatial models and tools, such as those used with CLAMS, can help identify the consequences of decisions about scale and the magnitude of effects of alternative approaches to management. If the ecological resolution is too coarse, such as with community types or indices that combine very different species or processes, the tradeoff analysis may be insensitive to important ecological differences. On the other hand, if the scale of resolution is too fine (e.g., species) assessments of compatibility will concentrate on a relatively small subset of the total species pool and may miss important dimensions of ecological variation. Consequently, a mixture of species and broad-scale indicators (e.g., forest structure, disturbance regimes) may provide a more robust assessment of ecological conditions than assessments that are confined to one type of indicator. The CLAMS work in the Coast Range indicates that large multi-ownership regions can absorb considerable fine-scale change before major changes emerge. Once changes accumulate over a large

area, however, it may take a long time to restore the area or achieve particular outcomes. Multiscale modeling approaches help us anticipate how fine-scale management actions will be expressed at broad scales. Currently, we are flying blind when we try to assess effects of actions at multiple scales. Spatial simulation models and tradeoff analysis help us visualize patterns over larger scales of space and time. These models cannot predict the future but they can help identify unintended effects of fragmented policy decisions and identify the level of risks or scales of impacts associated with particular actions.

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Compatible Forest Management

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