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*Research*, part of a Special Feature on Crossing Scales and Disciplines to Achieve Forest Sustainability

# Sustaining Biodiversity in the Oregon Coast Range: Potential effects of Forest Policies in a Multi-ownership Province

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ABSTRACT. To understand the potential effects of forest policies on sustaining biological diversity at broad scales, we used spatial simulation models to evaluate current and potential future habitat availability over 100 yr for three focal species: Pacific Fisher (*Martes pennanti*), Pileated Woodpecker (*Dryocopus pileatus*), and Warbling Vireo (*Vireo gilvus*). The habitats of these species represent a broad range of spatial scales and forest types. Area of habitat for fishers and Pileated Woodpeckers is predicted to increase over time under current forest land management policies. Habitat for Warbling Vireos is predicted to decline. These patterns are consistent with past analyses that predicted declines in diverse early successional forests and hardwood forests and increases in late-successional forests under current and two alternative policies. Land ownership influenced the spatial arrangement of habitat for all three focal species. Public lands subsidized habitat for wide-ranging species on adjacent private lands. A land use policy that required greater green tree retention on private lands seemed to result in modest increases in habitat quality over 100 yr for Pileated Woodpeckers. Thinning of plantations on federal lands had little effect on these focal species. Policy analyses such as these highlight incongruities between historic habitat patterns and contemporary spatial and temporal scales of habitat in managed landscapes. This information can be used to assess risks and inform the policy debates surrounding biodiversity conservation.

Key Words: *forest habitat; forest planning; habitat scaling; landscape spatial scale; wildlife habitat relationships.*

## INTRODUCTION

Society values biodiversity (Gowdy 1997). Recently developed management plans on public forest lands place primary importance on biodiversity conservation and secondary importance on commodity production (USDA and USDI 1994). Even many private landowners are seeking third-party certification, which reflects greater balance between biodiversity conservation and commodity production (Cashore and Lawson 2003). The effectiveness of biodiversity conservation approaches to meet societal needs for a variety of resources remains unclear, particularly with regards to policy analysis and development (Grumbine 1994).

The science of forest sustainability is in its infancy as concepts and indicators are still being developed (Lindenmayer et al. 2000, Haynes et al. 2003). Effective approaches to sustaining biodiversity are scale dependent. For instance, when managing species of concern, populations may change over a few generations or less but entire communities or ecosystems typically change more slowly (Loreau et al. 2001). Forest management effects may appear strong at the stand level but be diluted at landscape and regional levels by the cumulative effects of activities on multiple ownerships (Spies et al. 2007). Advancing sustainability theory and practice requires an understanding of how ecological domains of space and time intersect with domains imposed through land management and land ownership (Wiens 1989). The spatial scaling characteristics of many organisms have evolved or are pre-adapted to domains of scale produced by thousands of years of disturbance and regrowth and these scaling properties are reflected in the life history characteristics of species comprising native faunal communities (Wiens 1989). As human-imposed disturbances and regrowth activities cause

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a departure in spatial domains from those seen historically, we predict increasing risk to those species whose life history characteristics are most incongruent with scaling properties of contemporary landscapes (Lambeck 1997). Although the concept of the historical range of variability has been used as a conceptual framework for conserving biodiversity (Landres et al. 1999), less attention has been paid to the effects of contemporary and novel landscape patterns on biodiversity sustainability.

Current efforts at balancing social goals of biodiversity protection with timber production usually use a filter approach to reduce the risk of losing a species from an ecosystem (Hunter 1999). Three management strategies are often used: coarse-, meso- and fine-filters. In theory, the combination of these three strategies is used to “capture” species during implementation of land management plans and policies (Lambeck 1997, Landres et al. 1999, Kintsch and Urban 2002, Spies et al. 2002*b*, Hunter 2005). Typically ecosystem indicators are used in coarse- and meso-filter analyses, but there are many possible ecosystem indicators (Whitman and Hagan 2003). High-risk species such as the Northern Spotted Owl (*Strix occidentalis caurina*) (McComb et al. 2002, Olson et al. 2006) are typically chosen for a fine filter analyses. More recently focal species have been used in association with meso-filter analyses (Lambeck 1997, Hunter 2005). We know too little about the ecology of individual species to use this approach for all species, and any set of focal species does not necessarily encompass the needs of other species (Lindenmayer et al. 2002). But examining likely responses of habitat for species representing very different life history strategies allows us to understand how contemporary and novel landscape patterns might influence these species and associated ecological processes. Past efforts at sustaining biodiversity have used these filter strategies but have focused on single land ownerships. FEMAT (1993) developed a comprehensive land allocation strategy to provide protection to species associated with late-successional conifer forests in the Pacific Northwest, but did not explicitly consider state or private land contributions to meeting the needs of these species. Similarly, State Forest management plans in Oregon such as the Tillamook and Elliott State Forest plans developed structure-based management objectives specific to state lands but did not explicitly consider contributions of adjacent private lands (Sessions and Bettinger 2001). But plants and animals occupy habitat without regard to ownership boundaries unless those boundaries reflect changes, i.e., domains, in habitat quality, so these past approaches represent incomplete assessments. The few examples of past cross-boundary species conservation strategies suggest that the complex mosaic of landowners can have a profound influence on the potential for species such as Northern Spotted Owls (USDA and USDI 1994) and Coho salmon (*Oncorhynchus kisutch*) (Spies et al. 2002*b*) to persist in an ecoregion. In an attempt to address the limitations of past approaches, we developed an approach to understand the implications of policy change on a suite of forest values (Spies et al. 2002*b*).

Analyses of the potential advantages and risks associated with alternative policies are often facilitated by use of habitat models. Initial efforts at model development were habitat suitability index models (HSI), pattern recognition models (PATREC) and statistical relationships, i.e., regression models. All of these approaches contributed to the understanding of habitat availability, but all had shortcomings. HSI and PATREC models usually did not relate directly to the hierarchical manner in which many birds and mammals are assumed to select habitat (Johnson 1980). Statistical models are often constrained by the range of predictor variables used to develop the models; novel future conditions that extend beyond current ranges in these variables can lead to unrealistic predictions of habitat availability (*personal observations*). Wildlife habitat relationships (WHR) models also have been used widely to aid in management decisions in managed forests. Many WHR models have been tested and seem to be useful for predicting species occurrence or abundance in a habitat condition (e.g., Johnson and O’Neill 2001). Unfortunately, WHR models often are limited to those relationships that span a predetermined set of classes of vegetation or physical conditions, and are not flexible when considering new conditions that may develop under proposed or novel management techniques. To address this concern, we developed a modeling approach that could perform the following functions:

1. Identify high quality habitat locations for selected species in the present landscape. This information can be used by land managers to prioritize land management actions, including protection, to improve habitat quality over time.

2. Provide spatially explicit estimates of habitat suitability required for mapping habitat distribution across current and predicted future landscapes. Landscape-scale habitat quality information is a prerequisite to understanding the impacts of land management on animal survival, reproduction, and dispersal among metapopulations.

3. Guide the selection of alternative land policy scenarios. The method needed the following qualities:

- Adaptable to goals and constraints that might be imposed by land managers,
- Founded on principles of wildlife conservation and landscape ecology,
- Allow assessment of forest landscape patterns to estimate if any are superior for selected wildlife species than are the existing conditions.

We used the Oregon Coast Range as a case study to examine how forest policies might affect various measures of biodiversity over a multi-ownership region (Fig. 1.; Spies et al. 2007). Our goal was to assess how current and alternative policies influence the likelihood of biodiversity conservation over mixed ownership areas over time. Our specific objective was to evaluate the trends in habitat availability for three focal species under current and alternative forest policies in the Oregon Coast Range that are associated with meso-filter goals.

## METHODS

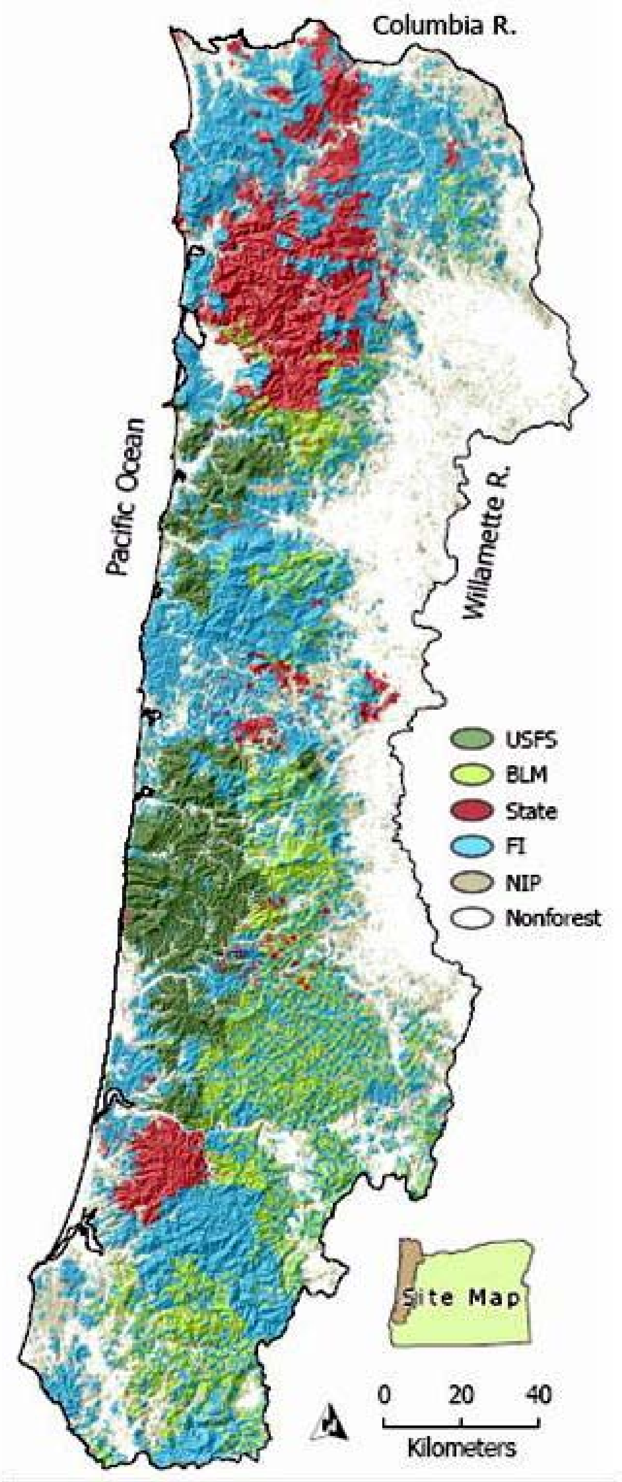
### Study area

The Oregon Coast Range is a $2\times10^6$ ha physiographic province in Oregon with a climate characterized by mild, wet winters, and dry cool summers (Franklin and Dyrness 1973). The coastal side of the province is wetter and cooler than the eastern side. Topography consists of relatively low but highly dissected mountains (1200 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 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 over 400 or more years following disturbances (Spies and Franklin 1991). Extensive logging and wildfires since the mid-1800’s 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 forests, i.e., structurally diverse forests >200-yr 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). Today less than 5% of the province is covered by old growth (Ohmann et al. 2007).

Past work assessed forest composition and seral stages as well as habitat patterns for a few selected species (Spies et al. 2007), but the focal species used in past studies represented either species with very large or very small home ranges and did not include species associated with hardwood forests. Projections of likely future forest composition in the region under current policies revealed potential declines in hardwoods over the next 100 yr (Johnson et al. 2007, Spies et al. 2007). The historical range of variation in the amount of hardwoods in the region is unknown. Wimberly (2002), however, estimated that the range of variation in forests less than 80-yr old was about 10 to 50%, which sets an upper bound on the amount of hardwood forest that might be expected, since the dominant hardwood tree, red alder (*Alnus rubra*), has a maximum age of around 100 yr (Burns and Honkala 1990). The amount of alder would probably have been much less than 50% since conifers would have a competitive advantage on most of the drier upslope sites.

Policy scenarios evaluated in the Oregon Coast Range to date have been driven by questions revolving around the Northwest Forest Plan, e.g., contributions of private lands, federal thinning efforts, but recent analyses predicted marked declines in hardwood forests in the future. We selected three focal species for consideration based

**Fig. 1.** Patterns of land ownership in the Oregon Coast Range.



on a range of home range sizes and associations with forest structure and composition. Because these species selected patches and resources over a range of spatial scales they represented a range of spatial scaling properties, i.e., from a species with fairly extensive area requirements to one with relatively small area requirements, and associations with hardwoods and conifers.

To examine the likely future trends in habitat availability under current and alternative policies, we chose a species that is associated with hardwood forests, but which can also use early successional forests. Warbling Vireos (*Vireo gilvus*) have home ranges of 1–2 ha (Gardali and Ballard 2000). In the Oregon Coast Range, Warbling Vireos typically nest in hardwood stands from 1–37 m above ground (Gardali and Ballard 2000). Although most common and abundant in mature hardwood stands, they may also be found in young deciduous stands that develop following a clearcut (Ward and Smith 2000). Structural characteristics of the stand, with the exception of having tall hardwoods, are quite variable among nest sites (Gardali and Ballard 2000). Further the species is of additional importance because Ballard et al. (2003) and Nott et al. (2005) recently reported 3–9% declines/yr in Warbling Vireo abundance in western North America, including the Oregon Coast Range.

Pacific Fishers (*Martes pennanti*) have very large home ranges. The home range size of Pacific Fishers is approximately 4000 ha for males and from 1500–2500 ha for females (Zielinski et al. 2004, Aubry and Raley 2002). In the western United States, Fishers are associated with mid- or late-successional conifer forests often with a hardwood component (Buskirk and Zielinski 2003, Powell and Zielinski 1994, Jones and Garton 1994, Carroll et al. 1999, Zielinski et al. 2004). They usually select large hollow trees or snags as den sites (Lyon et al. 1994, Aubry and Raley 2002). Fishers are rare in the Oregon Coast Range and probably are only found in the southern extent of the region either as a result of a reintroduction effort or range expansion from California (Zielinski et al. 2004). We included Fishers in our analyses because of the potential for the species to reoccupy former portions of the Coast Range should suitable habitat become available and because large, wide-ranging carnivores have been proposed as biodiversity indicators (Carroll et al. 1999).

Pileated Woodpeckers (*Dryocopus pileatus*) also are associated with older forests, but will forage in openings as well (Mannan 1984). They have a home range of approximately 480 ha during the breeding season (Bull and Jackson 1995) and they require dead wood within their home range for nesting and foraging. They construct nests in trees or snags having a mean diameter of 67 to 71 cm dbh (Nelson 1989, Mellen et al. 1992). The diet of Pileated Woodpeckers consists primarily of wood-boring insects, particularly carpenter ants (Genus *Camponotus*) (Bull 1975, McClelland 1979). In western Oregon, snags and logs are the most frequently used foraging substrates (Mannan 1984).

## Modeling habitat availability

Habitat suitability index (HSI) models are a class of theoretical models specifically developed to facilitate the consideration of wildlife species in multidisciplinary natural resource assessments (Schamberger and O’Neil 1986, Roloff et al. 2001). We extended the HSI approach developed several decades ago to a spatially explicit assessment of habitat quality. The models we have developed are similar to traditional HSI models in that they index habitat quality on a scale from 0.00 to 1.00, and they consider life requisites of the organism. They differ from the original HSI approach in that species-specific multiple spatial scales are represented; HSI models typically have a minimum area for model applicability. Further, observed empirical relationships are considered in model structure. Hence we use the term ‘habitat capability index’ (HCI) to differentiate our approach from the traditional HSI models.

Each HCI model includes a set of indices associated with the foraging and reproduction requirements of each species. Capability indices are scaled from 1.00 to 0.00, where 1.00 represents optimum habitat conditions and 0.00 indicates no habitat. The value for a capability index at a given location is determined by an estimate of vegetation or other environmental variables in a patch and in biologically meaningful neighborhoods of patches surrounding the focal patch. In most cases, the selection of a particular variable and its relationship to a capability index is supported with an amalgamation of information rather than by inference based on a strict probability model.

We developed species-specific habitat capability models for the three focal species based on existing literature and empirical relationships following the approach established by McComb et al. (2002) and Spies et al. (2007) (Table 1). The habitat capability models were based on habitat selection at multiple spatial scales. The models were designed to operate on rasterized GIS layers with information about vegetation structure and composition at the scale of 25×25 m. Each model predicts a Habitat Capability Index (HCI) that includes a set of capability indices (CI) associated with the characteristics of a focal patch for reproduction (Nesting Capability Index, NCI) and the potential suitability of the surrounding landscape for foraging (Landscape Capability Index, LCI).

The selection of vegetation and physical variables to include in the HCI models depended on four factors. First, we used variables for which the relationship to reproduction or survival could be supported by empirical evidence from published studies or from expert observations. 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 using stand-level forest succession models. Finally, we only retained variables that had a noticeable influence on HCI values as a result of sensitivity analysis.

An important assumption underlying the HCI modeling approach is that the optimum value of a measured variable for satisfying a life requisite of a species is known. However, the specification of an optimum value for any measured variable is complicated by conflicting definitions of “optimum” and an almost complete lack of direct, empirical data to support such a specification (Van Horne and Weins 1991). Optimum values of measured variables were estimated by examining the range of variation among observations made in relatively unmanaged Oregon Coast Range forests (Landres et al. 1999) and selecting the mean or median value for the variable estimated in habitat types selected by the species.

We conducted verification for Pileated Woodpeckers and Warbling Vireos, but lacked independent data with which to verify the Fisher model. Verification was assessed using geo-referenced animal abundance data from ten 250-ha sub-basins in each of three regions of the Oregon Coast Range (McGarigal and McComb 1995). We selected the best performing HCI model out of 2 to 7 possible models using Akaike’s information criterion (Burnham and Anderson 1998). In addition, each model received external peer review by at least two regional experts.

Threshold HCI scores were used to classify habitat for each species into low, medium, and high habitat quality levels. 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 represent habitat availability for the species. We assumed that trends in predicted habitat area and pattern over time and among policies were indicative of actual area of habitat available to each species. We assumed that the relationship between species viability and HCI scores for each species would be strongest when related to high quality patch availability in a sub-basin. Consequently we used high quality patch area per sub-basin as the basis for model verification against independent field data.

### Linking biodiversity measures to forest landscape change

The models were programmed in C++ and run on GIS layers representing initial vegetation conditions (Ohmann and Gregory 2002) and the outputs of the Landscape Management Policy Simulator (LAMPS), a spatial forest management simulator (Bettinger et al. 2005) for each 5-yr time step. Following clear cutting, the composition of regeneration is assigned a conifer, hardwood, or mixed species list that is based on a probability distribution. This distribution varies with climate region, distance from larger streams and land management intensity. In addition, we model fine-scale stochastic elements, e.g., small natural patch disturbances, to incorporate uncertainty and heterogeneity at fine scales. Again, we turn rates of disturbance into probability distributions for application spatially, with the probability of disturbance a function of position on the landscape. After regeneration, the structural and species compositional change in stands was modeled using ZELIG, an ecological successional model that was calibrated to the conditions of the Oregon Coast Range (Johnson et al. 2007).

Habitat availability for each species was summarized for projection periods 0, 25, 50, 75, and 100 yr into the future. We chose a simulation time horizon of 100 yr as a compromise between the realities of long-term ecological processes and

**Table 1.** Variables used in habitat capability indices of focal species, Oregon Coast Range. Maximum is the value or range of the variable at which the index score is set equal to 1. The correlation values represent the rank order correlation between each variable data set and the HCI data set. Rank is assigned by ordering the data from lowest to highest. Details of each model are given in Appendix A for the Pacific Fisher, in Appendix B for the Pileated Woodpecker and in Appendix C for the Warbling Vireo. †QMD = quadratic mean diameter, i.e., diameter of the tree with the average basal area, ‡Distance around focal pixel that is evaluated for landscape effects (see Appendices for details).

| Species | Variables | Maximum | Correlation with HCI |
|---|---|---|---|
| **Pacific Fisher** | | | |
| | Number of snags/ha >100 cm dbh‡ | 100 | 0.11 |
| | Number of trees/ha >100 cm dbh‡ | 100 | 0.37 |
| | Percent of home range in mid or late-seral forest‡ | 60 | 0.98 |
| **Pileated Woodpecker** | | | |
| | Number of snags/ha 50–75 cm dbh | 3.5 | 0.36 |
| | Number of snags/ha >75 cm dbh | 1.9 | 0.56 |
| | Percent of home range with QMD† >22.5 cm‡ | 75 | 0.60 |
| | Percent of home range with QMD† >15 cm‡ | 80 | 0.53 |
| | Number of trees/ha >25 cm dbh | 25 | 0.51 |
| **Warbling Vireo** | | | |
| | Hardwood canopy closure (%) | 70 | 0.24 |
| | Tree species subindex (proportion of hardwood basal area to total basal area) | 0.65 | 0.66 |
| | Number of trees/ha >25 cm dbh | 124 | 0.20 |

management time horizons. The time frame associated with current federal policy is 100 yr. We could have used a longer time frame, but under a longer simulation period we would need to consider fire and climate change, as well as make even longer extrapolations of human population change, a parameter that influences the rate of loss of forest land to development. Because fire was relatively infrequent in this landscape, i.e., natural fire rotation of 150 to 300 yr, it is plausible that 100 yr could pass without a large impact from wildfire (Wimberly et. al. 2000).

Projections were developed for three policy scenarios: current land policies, increased levels of green tree retention on private lands, and no thinning on federal lands (Spies et al. 2007). A green tree retention scenario was designed to illustrate an option that reduced the contrast in vegetation structure between public and private lands. Higher levels of live and dead tree retention on private lands could promote a more diverse early successional condition (Schreiber and DeCalesta 1992). The no-thinning scenario was designed to evaluate the effects of thinning plantations on federal lands. Thinning of mid-aged stands on federal lands has been proposed to promote more rapid tree growth and increase the rate of development of late-seral forests (Hagar et al. 1996, Suzuki and Hayes 2003, Garman et al. 2003). The policy scenarios are described in detail in Johnson et al. (2007).

## RESULTS

HCI scores for Pileated Woodpeckers and Warbling Vireos were associated ($R^2$ = 0.33–0.59) with independent field data in two of the three regions sampled by McGarigal and McComb (1995) (Table 2). Models for both species performed poorly in the Nestucca River drainage. It is unclear why model predictions were not associated with field data in this region. For Warbling Vireos, in particular, the model performed quite well in the other two basins and the model structure was very similar to a previously developed model (Banks et al. 1999). Although there are several possibilities why the model performed poorly for Warbling Vireos in the Nestucca Basin, it remains unclear why this basin differed from the other two. One possibility is related to the detectability of the species. Warbling Vireos are easily overlooked in field studies (Gardali and Ballard 2000). Second, breeding bird survey data from the region indicates that detections vary by up to 30% from year to year between 1966 and 2005 (Sauer et al. 2005). Indeed, Nott et al. (2005) did not report Warbling Vireos from their MAPS station in the Siuslaw National Forest from 1992–2001, so this species was not particularly abundant where they sampled. Mills et al. (1995) also reported year-to-year variability in results when verifying their HABCAP model in North Dakota.

Predictions for Pileated Woodpeckers were also more poorly correlated with observed abundance in the Nestucca than the other basins. The McGarigal and McComb (1995) study, upon which verification was based, used a combination of air photo and ground-based measures of habitat conditions. In their study, the variance explained for Pileated Woodpecker abundance was 52%, so there is a limit to which habitat can explain the abundance of a species. Indeed, field data collected by McGarigal and McComb (1995) indicated a poor association between abundance of Pileated Woodpeckers and late-successional forest in the Nestucca basin, but high correlations in the other two basins.

We also exhaustively analyzed the data to assess this apparent anomaly from the Nestucca basin. The untransformed field data for both species were not different from normal in their distributions. Regressions with the square-root transformed values did not improve model $R^2$'s. Further, we identified potential outliers using Cook's Distance and Studentized Residual analysis. Removing outliers did not improve the fit of the field data to the Warbling Vireo model. In contrast, removing outliers substantially improved the $R^2$ for the Pileated Woodpecker models in two basins where outliers were detected. Outlier removal in the Lobster basin increased HCI $R^2$ from 0.46 to 0.81, and in the Nestucca Basin HCI $R^2$ increased from 0.004 to 0.20, indicating that for species with low densities, observations in one of the ten sub-basins that could be classified as an outlier can change the relationship substantially.

Nonetheless, despite exhaustive data exploration, we could not identify a cause for the departure of data in the Nestucca Basin from the patterns in the other two basins. Because our associations were strong in two of the three regions, we assumed that our models represented a reasonable estimate of habitat availability for these two species over two-thirds of the Oregon Coast Range, or more.

Area of habitat for Fishers and Pileated Woodpeckers is predicted to increase over time under current policies and habitat for Warbling Vireos is predicted to decline (Figs 2 and 3). Although Fishers are currently not known to occur in much of the area that we analyzed, they likely did occur there at one time, with extirpation caused by overtrapping and habitat loss (Aubrey and Lewis 2003). Hence our predictions of potential habitat recovery may provide insight into possible avenues for range expansion in the Oregon Coast Range. Habitat availability for Pileated Woodpeckers is predicted to increase more rapidly if increased retention levels were applied to private industrial lands in the region (Fig. 3). A more modest effect of green tree retention was observed for Fishers, probably because den trees of adequate size would

**Table 2.** Regression coefficients ($R^2$) between independent field data (McGarigal and McComb 1995) collected in three regions of the Oregon Coast Range ($n$=10,250 sub-basins in each region) and calculated HCI scores for two focal species. Values in bold are significant at $P$>0.05.

| Species | Region | Combined habitat capability (HCI) | Foraging habitat capability (LCI) | Reproduction habitat capability (NCI) |
|---|---|---|---|---|
| Pileated Woodpecker | Drift Creek | **0.589** | **0.355** | **0.463** |
| Pileated Woodpecker | Lobster Creek | **0.457** | **0.366** | **0.448** |
| Pileated Woodpecker | Nestucca River | 0.004 | 0.005 | 0.010 |
| Warbling Vireo | Drift Creek | **0.591** | **0.572** | **0.774** |
| Warbling Vireo | Lobster Creek | **0.332** | **0.320** | **0.374** |
| Warbling Vireo | Nestucca River | 0.052 | 0.068 | 0.004 |

take >100 yr to develop. Thinning of plantations on federal lands had little effect on habitat availability for any of these focal species. Habitat availability for Warbling Vireos declined under all policies because no alternative policy directly addressed the projected loss in hardwood forests in the region. Given the uncertainty in the accuracy of the model predictions based on model verification results, precise estimates of the area of habitat available for each species is not possible. Rather, the patterns observed among policies should be viewed relative to one another.

Ownership had a major effect on habitat patterns for most species and vegetation classes across the region (Fig. 4). In particular, habitat for species with very large home ranges, e.g., Fishers, occurs largely on federal lands with habitat on private lands being supplemented from adjacent federal lands. Although 55% of private lands occur within 1.1 km of public lands, nearly all Fisher habitat occurring on private lands was within 1.1 km of public lands now (99.9%) and 100 yr into the future (97%) under current policies. Where habitat for Fishers occurred in mixed ownership areas it was represented by a combination of high quality denning habitat on public lands in combination with periodic availability of foraging habitat on intermingled private lands.

Pileated Woodpecker habitat on private lands also was subsidized by adjacent public lands, but to a lesser degree than Fishers, and was more influenced by leave tree levels on private lands (Fig. 4). Over 70% of the habitat for Warbling Vireos was on private land, and habitat availability decreased to a much greater extent on private industrial lands over the 100-yr projection period than on public lands (Fig. 4). Federal landowners did not subsidize habitat on private lands for this species probably due to the small area represented by a territory on the landscape. Hence, as home range sizes increase, the ability of one landowner to subsidize habitat for others in a planning area increased.

## DISCUSSION

We expected forest policies enacted in the 1990's to lead to dramatic changes in habitat availability for the focal species we examined. Our results combined with results from previous work (Spies et al. 2007) indicates that increases in habitat availability for Fishers and Pileated Woodpeckers as well as Northern Spotted Owls, Marbled Murrelets (*Brachyramphus marmoratus*), and Red Tree Voles (*Arborimus longicaudus*) should be expected based on current federal policies. The Northwest Forest Plan is designed to increase

**Fig. 2.** Patterns of habitat change over a 100-yr simulation in the Oregon Coast Range under current land management policies on federal, state, and private forest lands for three focal species: Fisher = Pacific Fisher, WAVI = Warbling Vireo, and PIWO = Pileated Woodpecker.

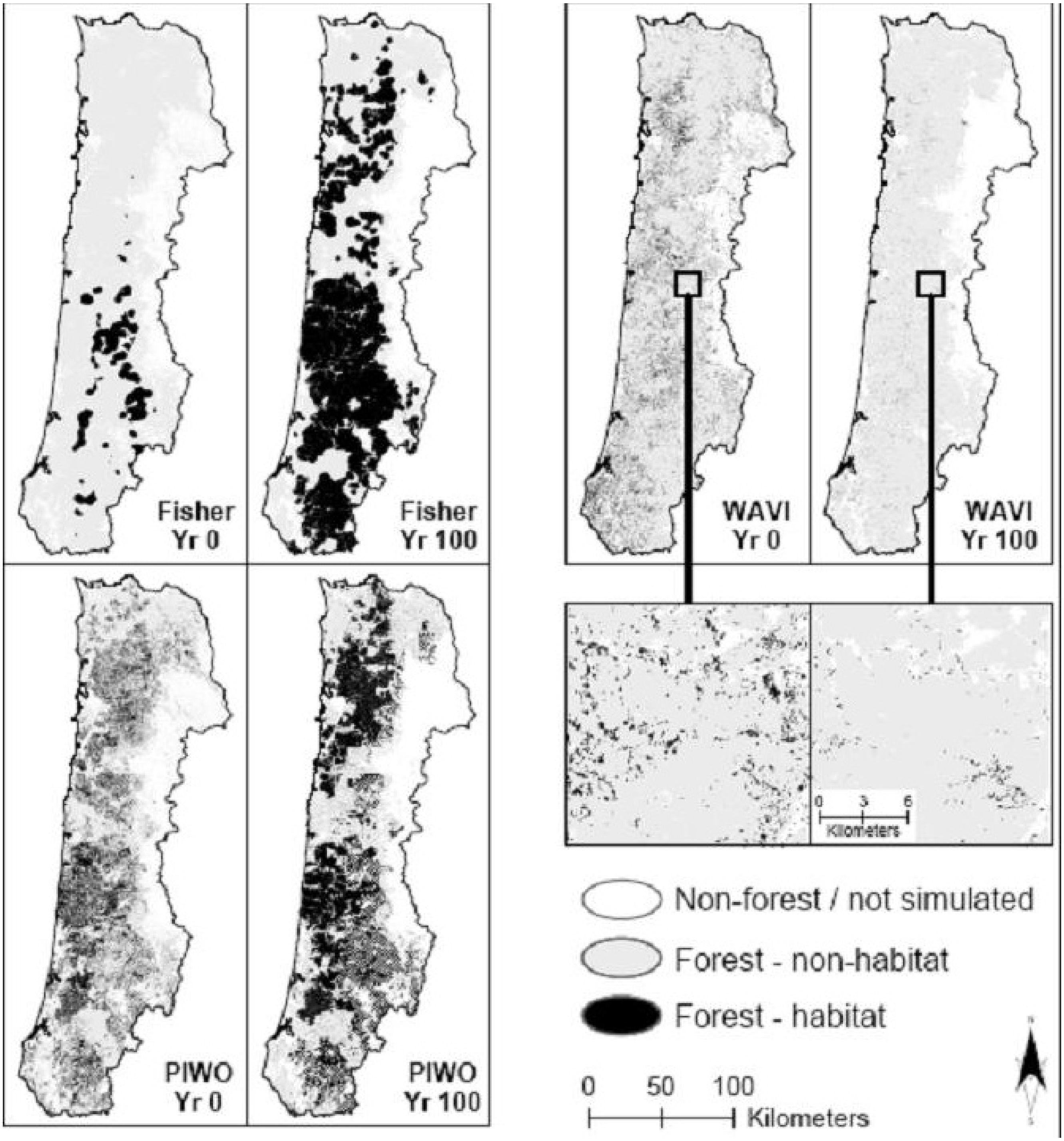
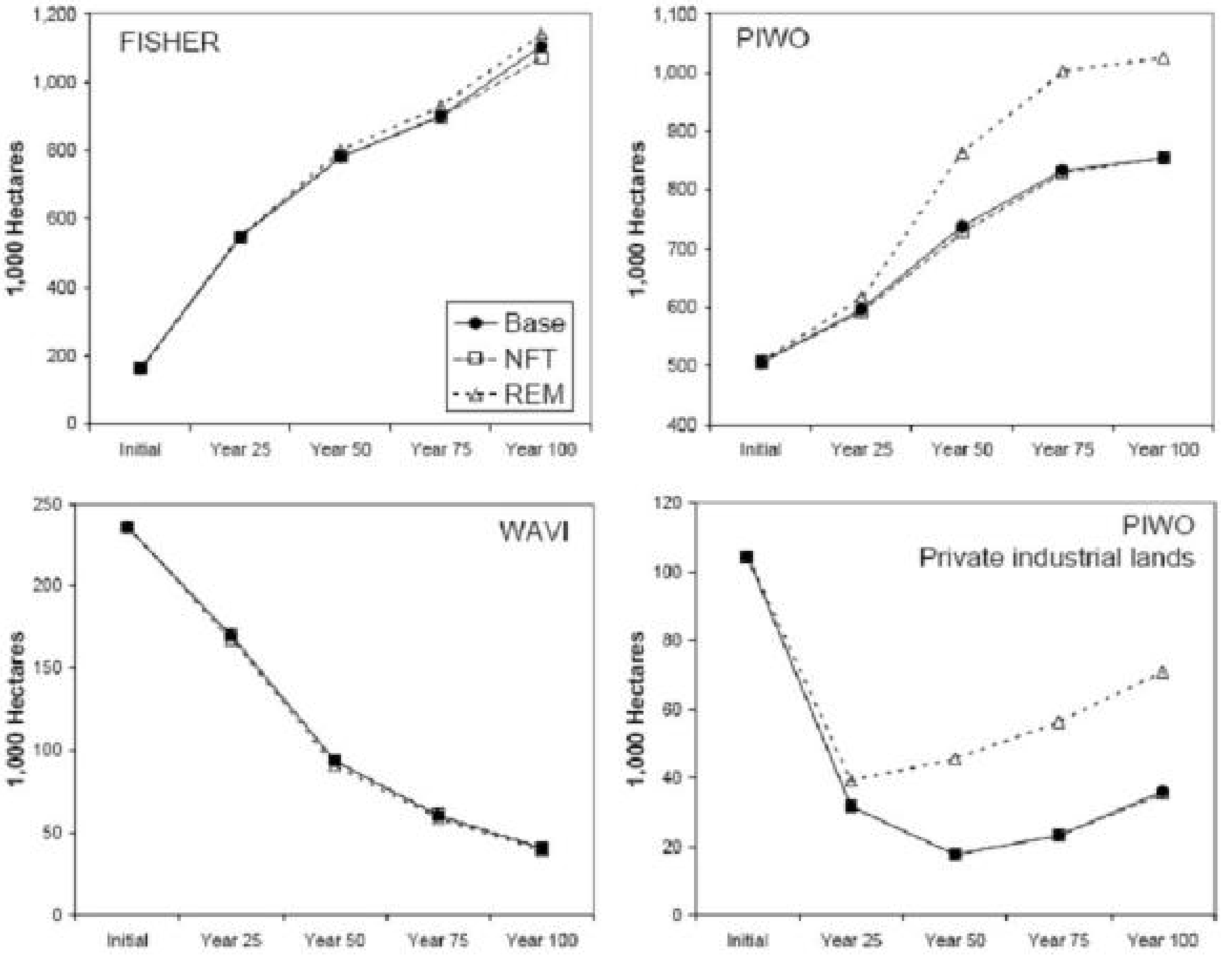


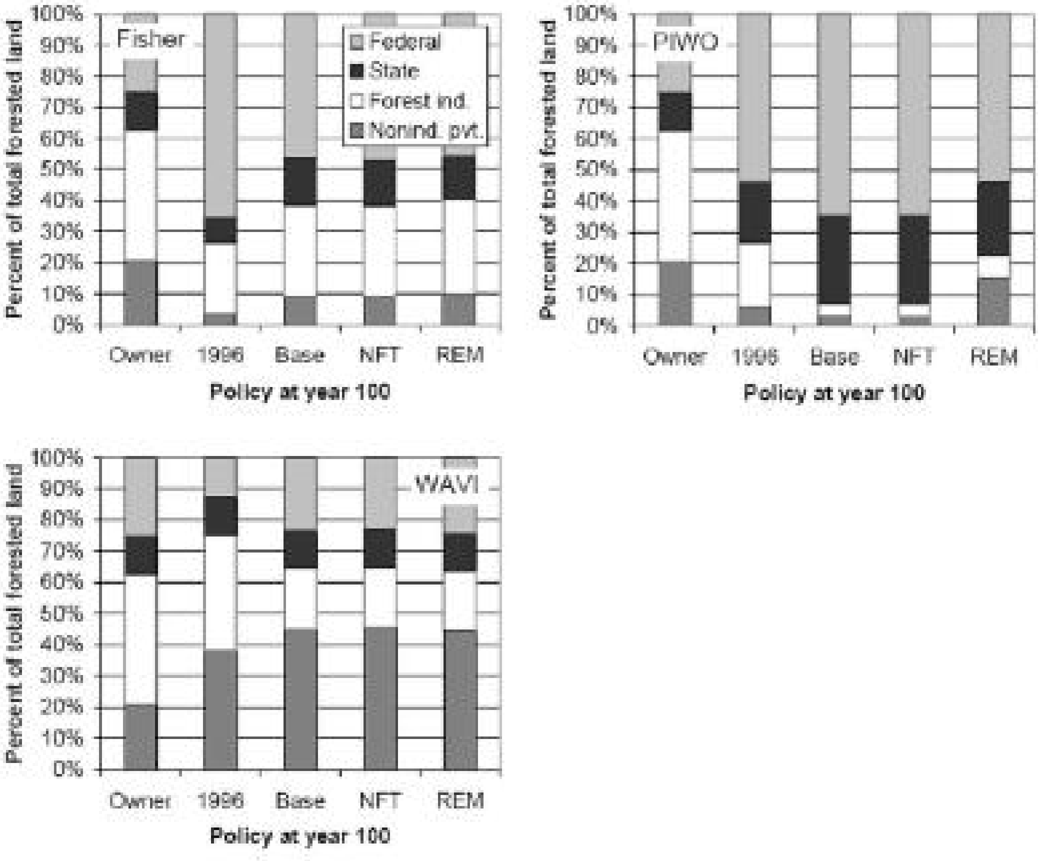
**Fig. 3.** Estimated changes in habitat availability over a 100-yr simulation in the Oregon Coast Range under current (BASE) and two alternative land management policies on federal, state, and private forest lands for three focal species: Fisher = Pacific Fisher, WAVI = Warbling Vireo, and PIWO = Pileated Woodpecker. Alternative policies are NFT = no forest thinning in young stands on federal lands and REM = increased retention of live and dead trees on private lands. Because trends resulting from the REM policy were so apparent for pileated woodpeckers on private lands, a separate chart is provided.



habitat for species associated with late successional forests on federal lands (USDA and USDA 1994). However, given the long period of stand development, even 100 yr leaves significant differences in age class distributions between the managed forest landscape and that expected under the historical range of variation (Spies et al. 2007).

We observed other changes that were not clearly addressed in current federal and state forest management plans. The projected declines in area of hardwoods and associated species such as Warbling Vireos, as well as declines in diverse early successional stages, e.g., open stands with remnant trees and semi-open forests, have not been addressed in any of the biodiversity plans that have

**Fig. 4.** Forest ownership area in the Oregon Coast Range and comparisons among current and two alternative policies in the contributions of four forest ownerships to providing habitat for three focal species 100 yr into the future.



been enacted over the past 10 yr. The decline in hardwoods probably will influence other components of biological diversity because 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, Kennedy and Spies 2004). The area of hardwood forests in the Oregon Coast Range has been declining from a relatively high point in the mid 20th century to present. Large areas of hardwoods were established from wildfires and extensive logging in the late 1800s and early 1900s (Kennedy and Spies 2004). Although it is true that many species associated with this vegetation type are mobile, it is also true that hardwoods add a significant component of species diversity to conifer stands in terms of lichens and invertebrates, and ecosystem processes such as nitrogen fixation. The trends in management on private industrial lands and federal lands are leading to a strong decline in hardwoods. The conservation significance of this decline is not well understood, but it is a change that can affect ecosystem diversity and function and one that we should be aware of. It could be that on private industrial lands, large areas will be devoid of hardwoods or at least have amounts that were below the long-term variation in the cover of this forest type.

A policy reflecting increased levels of green tree and snag retention on private lands would be more typical of residual levels of living and dead wood following natural disturbances than is currently practiced. Increased retention levels seem to have direct benefits not only for species associated with diverse early successional conditions, e.g., Western Bluebird (*Sialia mexicana*) and Olive-sided Flycatchers (*Contopus virens*), Spies et al. 2007, but also those associated with mature and old-growth forests, such as Pileated Woodpeckers. Consequently the alternative policies that we examined may address species associated with diverse early and late successional conditions, but not those associated with hardwood forests. Alternative policies should be examined that address projected declines in hardwoods and hardwood associated species.

### Influence of spatial and temporal domains on biodiversity sustainability

Species with home ranges smaller than the average land ownership parcel clearly can have their habitat needs more easily addressed on individual ownerships. Providing hardwood habitat to aid in reversing the regional declines in Warbling Vireo abundance may be quite easily achieved by having a set of standards, guidelines, or incentives associated with particular ownerships. This approach however can lead to landscapes with patches of habitat that might represent highly bimodal distributions of habitat quality. If these patches also become isolated within a mosaic of low quality patches over time, then connectivity may need to be given additional consideration for these species (Turner 1989).

Sustaining biodiversity in a mixed ownership landscape can be partially achieved by having federal lands supplement habitat on private lands for species with very large home ranges, such as Pacific Fishers, but it may come with a price. In a mixed ownership landscape habitat quality can be compressed to median values due to contributions from adjacent owners. Patches of old forest that likely represented high quality habitat for Fishers, e.g., large den trees, abundant dead wood, occurred over large areas under historic disturbance regimes (Wimberly et al. 2000), although these areas were not uniform. Nonaka and Spies (2005) demonstrated that within large patches of old forest there was structural heterogeneity providing much more patchiness of resources than managed forests. These historically large patches of high quality habitat were sufficiently large to encompass the home range of several Fishers (Wimberly et al. 2000, Zielinski et al. 2004) with an internal structure that provided den and foraging sites. In contemporary landscapes, areas comprised of low and high quality patches are likely to be smaller, except on portions of the Siuslaw National Forests, and more segregated except at the scale of about 2 km where Bureau of Land Management Lands are interspersed with private lands. The net effect of decreasing high quality patch sizes and increasing segregation of high and low quality patches is to depress the habitat capability scores for Fishers compared to what might have occurred in the past. By altering these spatial domains of habitat patterns, sustainability of various components of biodiversity may depend much more on cross-ownership coordination of management that addresses both patch sizes and connectivity across the region (Barten et al. 1991).

There also are potential issues associated with changes in the temporal patterns of habitat across complex landscapes. These effects may be severe where a species requires a condition that takes time to develop or for which the longevity of a condition on a landscape is reduced through active management. The lag time between policy implementation and measurable changes in habitat availability is significant and may not be apparent for 25–50 yr on industrial forest lands for species such as Pileated Woodpeckers (Fig. 3).

The combination of departure from historic patterns of habitat over both space and time may lead to changes in the ability of landscapes to sustain these three focal species, and biodiversity in general. Past patterns of habitat for these species were often represented by large patches (Wimberly et al. 2000). These patches were not static on the landscape historically, but in the absence of infrequent wildfires, they become much more static due to land ownership (Spies et al. 2007). The amplitude of change in habitat dynamics for a species is depressed in a more static landscape and generally reflects slow increases or decreases in habitat availability, especially on some ownerships (Fig. 3). Conceptually there are advantages to species to go through fluctuations in availability of high quality habitat (Kawecki 1995). During periods of habitat restriction and resulting population declines, allelic expression can be restricted to that portion of the gene pool best adapted to surviving these types of fluctuations if they are not too rapid (Kirkpatrick and Barton 1997).

### Implications for a social range of acceptability

Although we can assess potential effectiveness of alternative policies on indicators of biodiversity, policies must be both biologically effective and socially acceptable in order to conserve biodiversity in a region. The most appropriate use of the results of our simulations is to view them as representations of alternative decisions. We can then see which policy or decision set is not only most likely to achieve biodiversity sustainability goals, i.e., minimize risk, but also compare the policies to the conditions that society may find acceptable (Shindler et al. 2002). Just as we can estimate the historical range in variability of ecological indicators in ecosystems, we could attempt to estimate the range of indicator values likely to be accepted by society. Hence, we have both an ecological reference point and a social reference point driven by culture and values when making policy decisions (Fig. 5). This approach would allow us to better understand not only which decision might offer a result more consistent with the cultural range of acceptability now and into the future, but also to understand issues such as lag times associated with decisions, or unexpected abrupt changes in trajectories.

For instance, given a departure from historic conditions in availability of habitat for a species such as Pileated Woodpeckers, e.g., we used late successional conditions from Wimberly 2002 as a surrogate for historic habitat availability, segments of society may call for a policy change (Fig. 5, time <0). The policy change may not produce the desired responses, i.e., recovery, until some years after the policy is implemented due to lag times associated with forest growth (Fig. 5, time = 100 yr or more). During this lag time, the cultural range of acceptability may remain constant, increase, or decrease (Fig. 5, brackets). We can make decisions now based on comparisons of future trends among alternative policies, but if the cultural range of acceptability changes then we are moving the landscape forward toward a continually moving target (Fig. 5).

### Scope and limitations

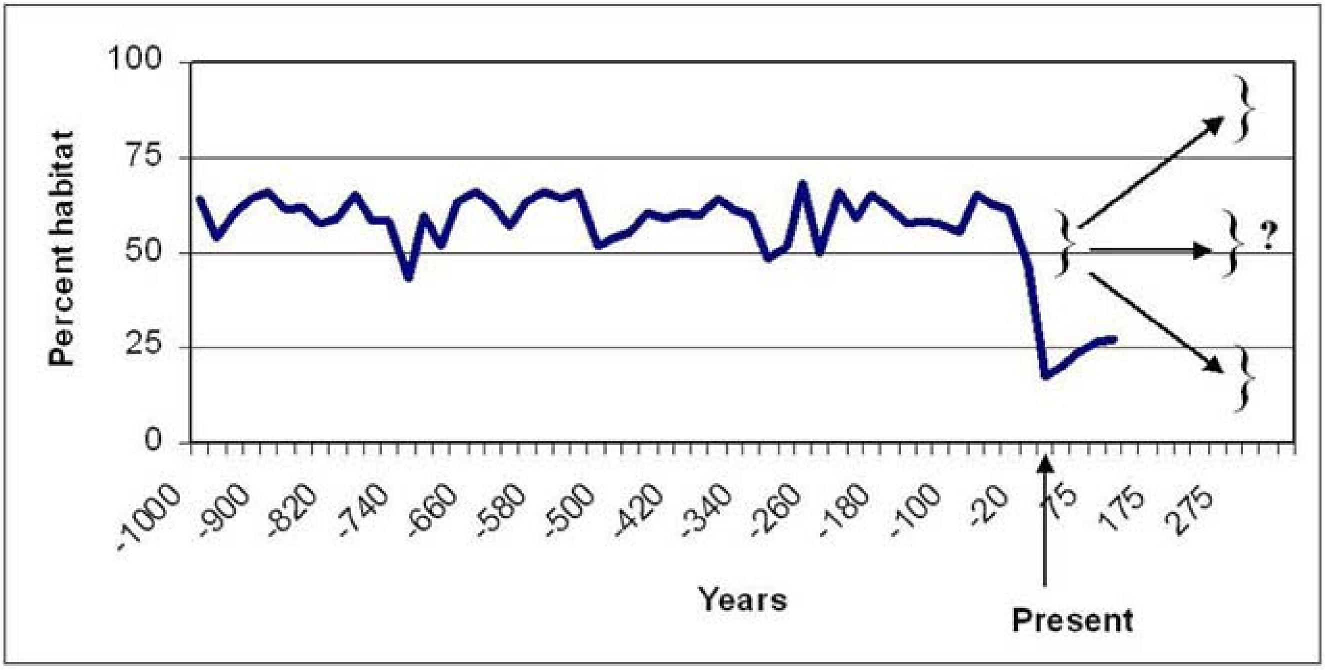
Model verification was both informative and frustrating. Although field data indicated strong model performance in two Oregon Coast Range basins, performance was not strong in another basin. It is not clear if the poor relationships in the Nestucca Basin are a result of poor model formulation, atypical data collected in that one basin in one year, or some combination of both. Clearly the results do suggest caution in interpretation of the results because there is uncertainty as to the cause of these poor relationships. Consequently we strongly encourage readers to view the results in a relative rather than an absolute sense.

Further, the assessment of policy effects on focal species is limited in several ways. First, the habitat capability models were developed using empirical relationships whenever possible, but were largely based on literature and expert opinion; only a limited number of field data sets from the Oregon 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 (McComb et al. 2002), but empirical data were not available to test that assumption. The index is most appropriately used to allow a relative comparison of trends resulting from different management practices and policies. Third, the models were developed on recent conditions in the Oregon Coast Range and may not perform similarly in other conditions of climate or landscape dynamics and structure. Fourth, the indicators are dependent on underlying models that predict initial vegetation, landscape dynamics, and stand development, all of which contain errors and constraining assumptions. Such models cannot be tested in a typical scientific experiment. Despite limitations of our models, they represent useful “thought-experiments” that can give us insights into the possible outcomes of forest management policies (Oreskes 1997).

## CONCLUSIONS

Approaches such as ours provide a framework for assessment of potential effects of current and alternative policies on forest types and conditions and focal species habitat availability. The results may help to initiate dialogue leading to a reduction in the risk of future policies having unintended consequences on Warbling Vireos, Pileated Woodpeckers, and Fishers, and more generally on biodiversity in a region. It is difficult to predict the broadscale consequences of fine-scale management actions without simulations that encompass a range of scales and diversity of ecological measures. Such approaches can help planners visualize the general

**Fig. 5.** Estimated changes in a pileated woodpecker habitat availability under historic conditions (time < 0, based on Wimberly 2002) and future conditions (time > 0) compared to cultural range of acceptability (brackets) at present and into the future.



appearance of habitat availability in the future and also understand how their actions might affect relative trends in these focal species and associated structural indicators. Such analyses may in turn reveal gaps, as we did, in the regional biodiversity strategy, and give policy makers and managers a better sense of the effects of their actions over time and space.

Given that owners have different goals for their forests, the overall potential for sustaining biodiversity will depend on the diversity and spatial pattern of those ownerships. Opportunities for addressing incongruities between domains of scales for species at risk of decline and inherent domains of scale in contemporary landscapes exist through cross-boundary cooperation among landowners (Barten et al. 2001). Many social, legal, and political barriers exist to coordinating forest management practices across owners to achieve particular sustainability goals (Thompson et al. 2004). Nonetheless, considering sustainability as a three-legged stool of ecological, economic, and social components; without consideration of all three parts, the whole cannot be achieved.

*Responses to this article can be read online at:*
*http://www.ecologyandsociety.org/vol12/iss2/art29/responses/*

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# Appendix A. Habitat capability index for Pacific Fisher (*Martes pennanti*)

**Developers: Brenda C. McComb, David Vesely**

**Reviewers: Bill Zielinski, Keith Aubry**

## *Background*

In the western U.S., fishers are associated with mid or late-successional conifer forests often with a deciduous component (Buskirk and Zielinski 2003, Powell and Zielinski 1994, Zielinski et al. 2004). They are generalized predators (Powell and Zielinski 1994) that usually select hollow trees or snags as den sites (Lyon et al. 1995, Marshall 1992, Aubry and Raley 2002).

Presently, fishers are rare in the Oregon Coast Range, probably only found in the southern extent of the region either as a result of a reintroduction effort or range expansion from California (Zielinski et al. 2004). This species may once have been more common across the region prior to trapping and settlement. We have elected to include the fisher in the Coast Range analysis because of the potential for the species to reoccupy former portions of the Coast Range should suitable habitat become available and if any future harvest does not limit viability of the species (Marshall 1992). Although Buskirk and Zielinski (2003) provided a recent comprehensive overview of habitat use by fishers in western forests, we are not aware of any scientific study of habitat associations from the Oregon Coast Range physiographic province. Therefore this habitat model based on general relationships described by Buskirk and Zielinski (2003) and recent work in California and in the Rogue River National Forest of Oregon (Zielinski et al. 2004, Aubry and Raley 2002).

## *Habitat use information*

Natal den sites and resting sites tend to be in large (> 100cm dbh) hollow trees and snags (Zielinski et al. 2004, Aubry and Raley 2002). They seem to require multiple resting sites distributed throughout their home range (Zielinski et al. 2004).

The primary foods of fishers are snowshoe hares (*Lepus americanus*), porcupines (*Erethizon dorsatum*), mice, voles, and carrion (Jones 1991, Powell 1977). Fishers forage in a variety of vegetation types, and seem to select areas with 60-100% canopy cover (Powell and Zielinski 1994, Zielinski et al. 2004).

Fishers are associated with large blocks of mid- and late-successional conifer and mixed conifer-hardwood forests (Jones 1991, Jones and Garton 1994, Carroll et al. 1999). They usually select stands with dense canopy closure, vegetation structure characterized by fine-scale heterogeneity, abundant amounts of woody detritus (Powell and Zielinski 1994), and avoid forest openings (Buskirk and Ruggiero 1994, Jones 1991, Jones and Garton 1994). In western Oregon, fishers are most often associated with mid- to low-elevation Douglas-fir (*Pseudotsuga menziesii*) and western hemlock (*Tsuga heterophylla*) forests (Powell and Zielinski 1994). However, on the east side of the Cascades, fishers occur in high-elevation, true fir (*Abies* spp.) forests (Powell and Zielinski 1994).

The average home range size of male fishers is 4,000 ha, nearly three times the size of female home ranges (Powell and Zielinski 1994). Powell and Zielinski (1994) reported the average home range for female fishers is 15 $\mathrm{km}^2$ ; range 4.2-32 $\mathrm{km}^2$ ; $n = 12$ studies). Zielinski et al. (2004) reported female home range sizes of 1500 ha in Coastal California, and Aubry and Raley (2002) reported that female home ranges averaged 25 $\mathrm{km}^2$ in the Cascades of Oregon. Studies of fishers over large landscapes have not been conducted (Lyon et al. 1994) although the relationship between fisher occurrence and forest fragmentation, patch size, or other measurements of forest landscapes have been hypothesized for the Klamath region of California by Carroll et al. (1999) to predict distribution.

### *Habitat capability index*

The virtual extirpation of fishers in the Oregon Coast Range increases the uncertainty about precise habitat requirements for fishers in this Province. However, using information about fishers from other regions, large blocks of mid- and late-seral forest seems to be critical to their persistence. Within those large blocks of forest, large hollow trees and snags are needed for dens and a closed canopy may also increase foraging habitat quality. Fishers seem to avoid clearcuts and open areas (Buskirk and Zielinski 2003). Based on work from Coastal California by Zielinski et al. (2004):

- An average of 14 resting sites per female fisher were detected; these were distributed throughout the females' home ranges.
- Resting sites in California and natal den sites in Oregon were in live trees and snags > 100 cm dbh with natal sites in hollow trees.
- Female home ranges average 15-25 $\mathrm{km}^2$ in size
- Approximately 60% of the home range is in mid to late seral forest (Figure A1).

Hence,

$$HCI_i = \text{Minimum}\ (DEN_i\ , FSI_f) \qquad \text{(A.1)}$$

Where

$HCI_i$ = Habitat capability index
$DEN$ = Den site index
$LCI$ = Landscape capability index
$_i$ = pixel

### *Den site index*

$$DEN_i = (S1_i * S2_i\ )^{1/2} \qquad \text{(A.2)}$$

Where

$DEN$ = A den site subindex that is calculated for each 4- x 4-pixel window (1 ha) and assessed over a 2.2-km radius window because den and resting sites must be well distributed across a home range.
$S1$ = index to snags > 100 cm dbh

$S2$ = index to trees > 100 cm in dbh
$_i$ = pixel

$$IF\ S100T_i \geq 100\ THEN\ S1_i = 1.0,\ ELSE\ S1_i = S100T_i *0.01 \qquad (A.3)$$

where

$S100T$ = number of 4- x 4-pixel windows within a 2.2-km radius area with > 2 snags > 100-cm dbh and > 5-m in height / ha

$$IF\ T100_i \geq 100\ THEN\ S2_i = 1.0\ ELSE\ S2_i = T100_i *0.01 \qquad (A.4)$$

where

$T100$ = number of 4- x 4-pixel windows within a 2.2-km radius with > 30 trees per hectare > 100 cm dbh. E chose a threshold of 30 trees per hectare of this size based on the distribution of densities of this size trees in unmanaged old forests in the Coast Range (T.A. Spies).

### *Landscape capability index*

Mid and late-seral forest is defined as areas where canopy cover is > 60% and the quadratic mean diameter of all trees is > 25 cm dbh based on field data interpretation by Dr. T.A. Spies.

*If mid and late-seral forest is > 60% of a 2.2 km radius analytical window THEN* $L2_i = 1.0$

*ELSE If cover of a 2.2-km radius window by mid seral and late seral forest < 30% then* $L2_i = 0$

*ELSE if cover of a 2.2-km radius window by mid seral and late seral forest is 30-60% then* $L2_i$ *= (% mid and late-seral forest – 30) *0.0333* (A.5)

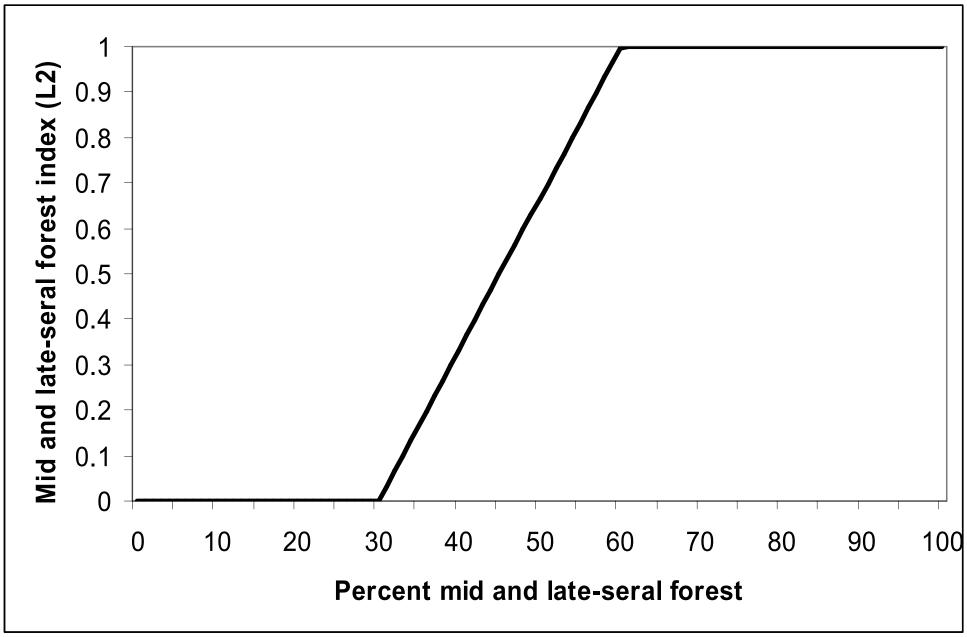

**FIG. A1**. Relationship between index score and percent mid and late-seral forest within a 2.2-km radius analytical window.

where

$LCI_f = L2_i$ averaged within a 2.2-km radius analytical window

$_i$ = pixel (A.6)

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# Appendix B. Habitat capability index for Pileated Woodpecker (*Dryocopus pileatus*)

**Developers: Brenda McComb, David Vesely, Michael McGrath**

**Reviewers: Joan Hagar, S. Kim Nelson**

## *Background*

Pileated woodpeckers occur in every physiographic province of western Oregon (Csuti et al. 1997). The species is generally absent from landscapes dominated by non-forested plant communities and agricultural lands.

Pileated woodpeckers lay three to five eggs in nests they excavate in dead or dying trees. (Harris 1980) found that cavity selection is based upon the structural quality of the wood and that they will occasionally excavate cavities in live trees with soft heartwood. Mellen et al. (1992) reported that pileated woodpecker nests occur in trees or snags having a mean diameter of 71 cm dbh and a minimum diameter of 40 cm. Nelson (1989) reported similar nest tree diameters: the mean diameter was 67 cm dbh and minimum was 47 cm dbh. In Oregon, pileated woodpeckers usually nest in forest stands >80 years in age (Mellen 1992, Nelson 1989).

The diet of pileated woodpeckers consists primarily of wood-boring insects, particularly carpenter ants (Genus *Camponotus*) which they extract from snags and stumps (Bull 1975, McClellard 1979). In western Oregon, snags and logs are the most frequently used foraging substrates (Mannan 1984). Bull (1975) reported that the abundance of coarse woody debris (CWD) was significantly greater in stands used by pileated woodpeckers than was generally available. Pileated woodpeckers do forage in young stands (age <40 years) but shrub cover in clearcuts may limit accessibility to stumps and logs (Bull 1975).

## *Habitat capability index model*

The pileated woodpecker habitat capability index model assessment includes two sub-indices for nesting and foraging habitat. The lower score of the two sub-indices determines the overall HCI score in a potential home range. We assume that high quality foraging habitat cannot compensate for lesser quality nesting habitat, and vice versa. The model assumes that nesting habitat quality is most limited by the availability of snags >50 cm dbh to construct a nest cavity. The relationship between snag density (snags per ha) and nesting habitat quality is assumed to be linear up to a threshold of 3.5 snags/ha for snags having a dbh from 50 - 75 cm (Figure B1) and 1.9 snags/ha for snags >75cm (Figure B2) based on upper 95% confidence limits of median snag densities in the region. Large snags (dbh > 75 cm) receive greater weight in scoring than snags $50 < \text{dbh} \leq 75$ cm because we assume woodpeckers can construct cavities having greater volume in larger snags.

The quality of pileated woodpecker foraging habitat is assumed to be limited by the proportion of the home range that is has > 25 trees per ha > 25 cm dbh, and that the quadratic mean diameter of trees in the home range is of sufficient size for foraging (> 15 cm dbh). The foraging habitat sub-index is an additive function with truncation that assumes components of the sub-index do not modify one another and are compensatory unless the sum is above the optimal threshold (Van Horne and Wiens 1991). Foraging habitat was assessed at the scale of the home range, approximately 480 ha (1236 m radius) (Bull and Jackson 1995).

### *Habitat capability index*

$$pileated\ HCI_f = minimum\ (NCI_f\ ,\ LCI_f) \tag{B.1}$$

Where

$HCI$ = Habitat capability index
$NCI$ = Nesting capability index
$LCI$ = Landscape capability index
$f$ = focal pixel

### *Nesting capability index*

All metrics for this index are calculated for a focal pixel at the center of a 3 x 3 “moving window.” This moving window of pixels averages conditions for the 0.5625 ha surrounding the “focal” pixel (i.e., 3 x 3 pixels). The averaging is done to: (1) smooth inter-pixel variation; and (2) provide a “patch” or “stand” level summary, which is consistent with the scale of the stand modeling and stand inventory data.

$$NCI_f = (0.4 * S1_f)+(0.6 * S2_f) \tag{B.2}$$

Where

$NCI$ = Nesting capability index
$S1$ = medium snag score
$S2$ = large snag score
$f$ = focal pixel

$$\text{IF } S5075T_f \geq 3.5 \text{ THEN } S1_f = 1.0\text{, ELSE } S1_f = S5075T_f * 0.2857 \tag{B.3}$$

Where

$S5075T$ = number of snags ($50 < \text{dbh} \leq 75$ cm) / ha
$S1$ =medium snag score
$f$ = focal pixel

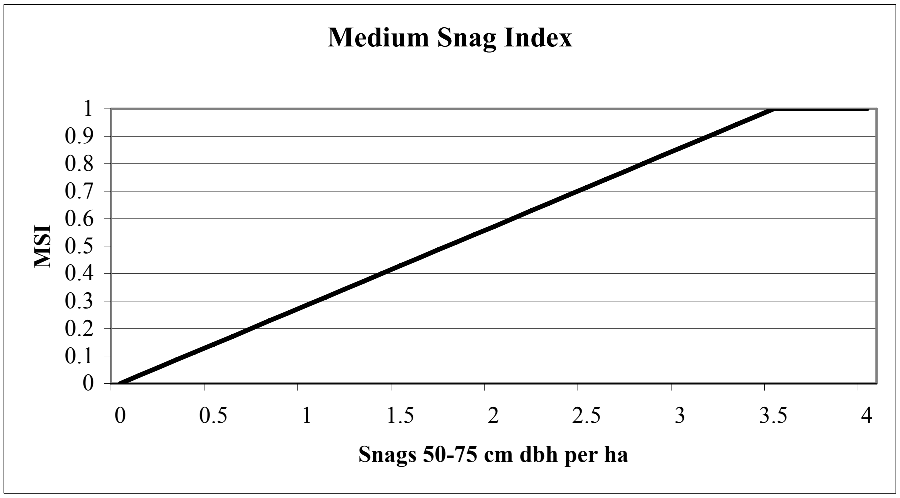

**FIGURE B1.** Relationship between medium snag index and snags 50-75 cm dbh per ha.

IF $S75T_f \geq 1.9$ THEN $S2_f = 1.0$, ELSE $S2_f = S75T_f * 0.5263$ (B.4)

Where

$S75T$ = number of snags (dbh >75-cm and height >5-m) / ha
$S2$ = large snag score
$f$ = focal pixel

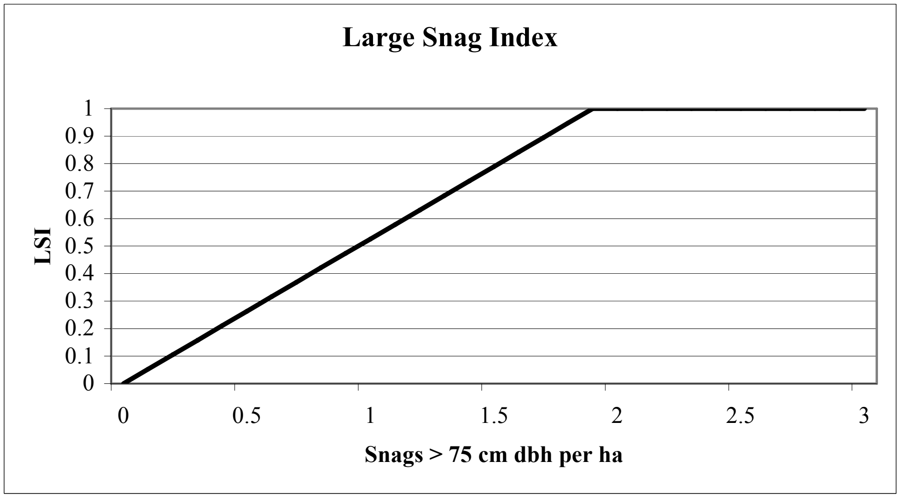

**FIGURE B2**. Relationship between large snag index and snags >75 cm dbh per ha.

*Landscape capability index*

$$LCI_f = \sqrt{NR_f * F_f * MinDen_f} \tag{B.5}$$

Where:

$LCI_f$ = The Landscape capability index

$NR_f$ = Nesting and roosting habitat index
$F_f$ = Foraging habitat index
$MinDen_f$ = The minimum tree density index
$f$ = focal pixel

$$NR_f = \frac{1.4165}{1+e^{-\left(\frac{P_n-0.6052}{0.1664}\right)}}\left(\textit{truncated to}\ 1.0\right) \qquad \text{(B.6)}$$

Where:

$NR_f$ = Nesting and roosting habitat index
$e$ = Euler's constant
$P_n$ = the proportion of nesting and roosting habitat located within 1236 m of the focal pixel. Nesting and roosting habitat is defined as forested pixels with QMD of dominant conifers $\geq$ 22.5 cm.
$f$ = focal pixel

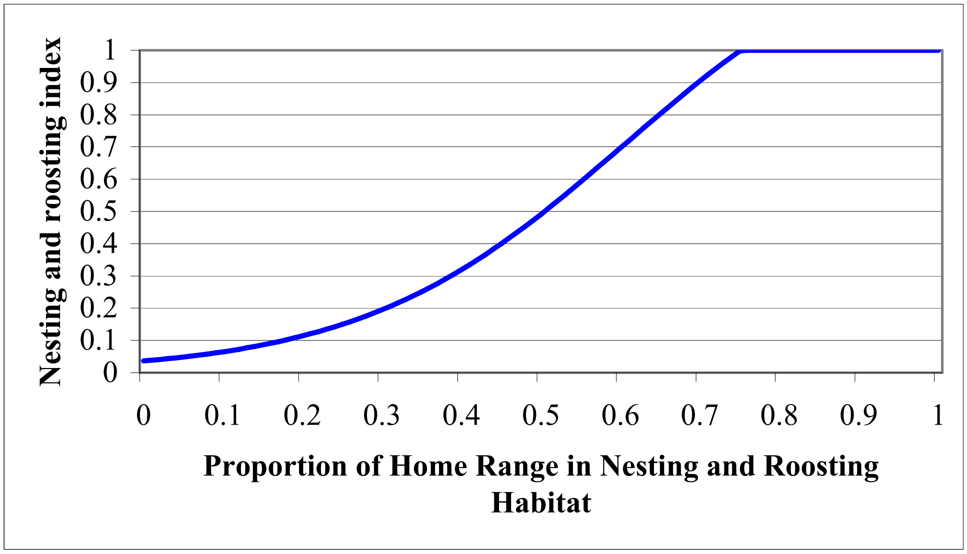

**FIG B3**. Relationship between nesting and roosting index and the proportion of forested pixels with QMD of dominant conifers > 22.5 cm within a 1236-m radius analytical window.

$$\text{(L2)}\ F_f = 1.0866 * e^{\left(-0.5\left[\frac{\left|P_f-0.7763\ \right|}{0.1344}\right]\right)}\left(\textit{truncated to}\ 1.0\right) \qquad \text{(B.7)}$$

Where:

$F_f$ = Foraging habitat index

$e$ = Euler's constant
$P_f$ = the proportion of foraging habitat located within 1236 m of the focal pixel. Foraging habitat is defined as forested pixels with QMD of dominant conifers $\geq$ 15 cm.
$f$ = focal pixel

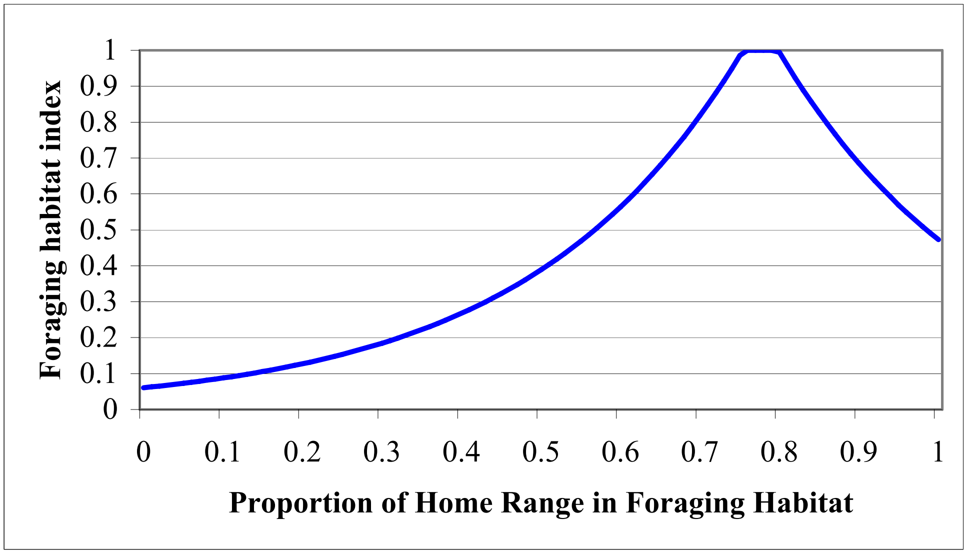

**FIG B4**. Relationship between foraging habitat index and the proportion of forested pixels with QMD of dominant conifers > 15.0 cm within a 1236-m radius analytical window.

$MinDen_f$: If $tph25_f < 25$, $\text{MinDen}_f = 0.1$, else $\text{MinDen}_f = 1.0$ (B.8)

Where:

$MinDen_f$ = The minimum tree density index
$tph25_f$ = All trees > 25cm dbh per hectare
$f$ = focal pixel

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# Appendix C. Habitat capability index for Warbling Vireo (*Vireo gilvus*)

**Developer: Brenda C. McComb**

**Reviewers: Joan Hagar, Robert Altman, Josh Lawler**

## *Background*

The warbling vireo occurs throughout the Oregon Coast Range physiographic province (Gardali and Ballard 2000). In the Oregon Coast Range, warbling vireos nest in mature hardwood stands from 1-37 m above ground (Gardali and Ballard 2000). Structural characteristics of the stand, with the exception of having tall hardwoods, are quite variable among nest sites (Gardali and Ballard 2000). Although most common and abundant in mature hardwood stands, they may also be found in young deciduous stands that develop following a clearcut (Ward and Smith 2000). There does not appear to be a strong effect of landscape pattern on abundance or productivity of this species (McGarigal and McComb 1995, Tewksbury et al. 1998, Gardali and Ballard 2000). However, this species does experience high levels of nest parasitism by brown-headed cowbirds, especially near agricultural areas (Ward and Smith 2000).

Warbling vireos forage on insects from just above the surface of the ground to the top of the canopy (Gardali and Ballard 2000). Caterpillars, bugs (Hemiptera) and beetles (Coleoptera) tend to dominate their diet (Gardali and Ballard 2000). Abundance of the species is positively correlated with the amount of hardwood forest in a landscape and the more mature the hardwood stand, the more detections that are noted (Stiles 1980).

## *Habitat capability index model*

The habitat capability model developed for the Coast Range is based primarily on nesting habitat requirements and secondarily on foraging requirements. A Habitat suitability index (HSI) model was developed by Banks et al. (1999) for use in Alberta, Canada. The Banks et al. (1999) model was informative and helped us structure the Coast Range model, but their model was not directly applicable to the Oregon Coast Range environment.

We assumed that minimum patch sizes must be greater than the territory size, which is 1-2 ha (Gardali and Ballard 2000). We included two sub-indices, one for nesting and another for foraging habitats. The nesting habitat capability score assigned to the focal pixel is dependent on hardwood canopy cover and density. The foraging habitat score is dependent on the dominance by hardwoods in a 2-ha window surrounding the focal pixel. The final capability score is the minimum of the two sub-index scores. That is, we hypothesize that overall habitat suitability can be no greater than the most limiting habitat component.

We assume that nesting habitat capability for warbling vireos improves with increasing canopy cover of hardwoods > 25 cm dbh per ha (Figure C1). These relationships are based on the findings of research on the species in Washington red alder (*Alnus rubra*) stands (Stiles 1980) and summary information provided by Gardali and Ballard (2000). Since the species is occasionally detected in mixed conifer-hardwood stands, we scaled this relationship to peak at 70% hardwood cover in stands with trees > 25 cm dbh.

We assume that optimum foraging areas occur in mature hardwood patches of a size sufficient to support a territorial nesting pair. The foraging capability sub-index in the HCI model is based upon the proportion of a 1-ha patch around the focal pixel that is dominated by hardwoods.

*Habitat capability index*

$$\textit{Warbling vireo } HCI_f = \textit{minimum}(NCI_f, LCI_f) \tag{C.1}$$

Where

$HCI$ = Habitat capability index
$NCI$ = Nesting capability index
$LCI$ = Landscape capability index
$f$ = focal patch

*Nesting capability index*

All metrics for this index are calculated for each pixel.

$$NCI_f = \frac{(HWCC_f * 2) + (DEN25_f * TreeSp_f)}{3} \tag{C.2}$$

Where

$NCI$ = Nesting capability index
$f$ = focal pixel
$HWCC$ = hardwood canopy closure index
$DEN25$ = Density of trees >25 cm index
$TreeSp$ = Tree species subindex

The HWCC index is based on predicted canopy closure in a pixel. The index is 0 from 0-40% canopy closure by hardwoods (Figure C1). The index increases linearly from 0 to 1 as hardwood canopy closure increases from 40-70%, then stays at 1 from 70-100%. This index is weighted twice as much as tree dbh since composition seems to be more important than structure for this species (Gardali and Ballard 2000).

$HWCC_f =$ If $HWCanopyClosure < 40\%$, then $HWCC_f = 0$ (C.3)
Else if $HWCanopyClosure \geq 40\%$ and $< 70\%$,
Then $HWCC_f = (HWCanopyClosure * 0.0333) - 1.3333$
Else $HWCC_f = 1$

Where
$HWCanopyClosure$ = Percent canopy closure * $TreeSp_f$ (tree species subindex).
$f$ = focal pixel

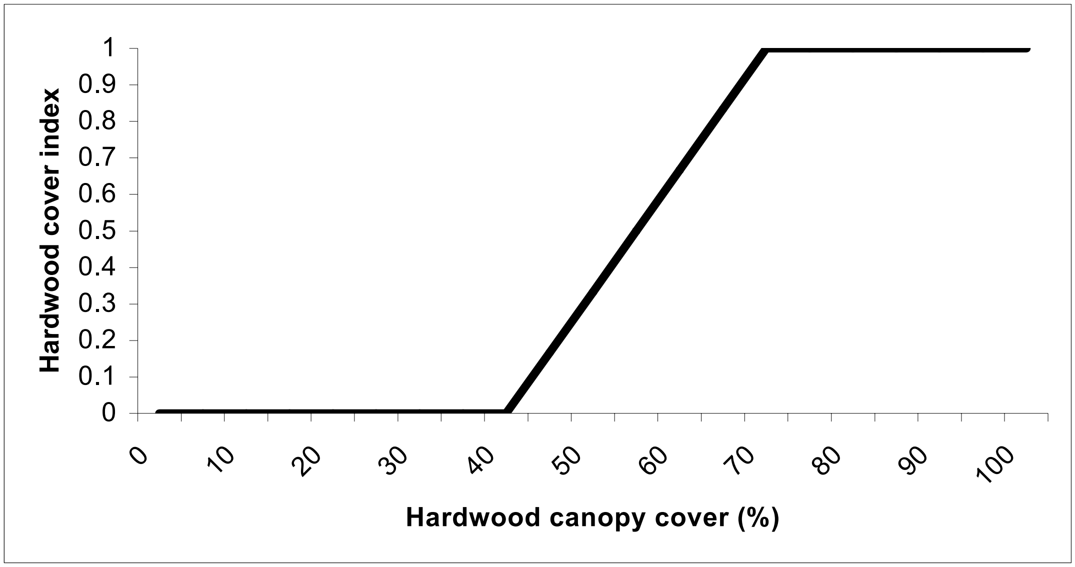

**FIG C1**. Relationship between hardwood canopy cover index and hardwood canopy cover.

Hardwood canopy closure is estimated based on the canopy closure of all trees is multiplied by the following tree species subindex. Breaks between species were designed by Dr. Janet Ohmann, PNW Research Lab, US Forest Service, based on examination of ground plot data. The 0.20 and 0.65 thresholds were chosen to approximate 30% and 70% thresholds based on canopy cover (Figure C2).

This tree species subindex is multiplied by total canopy cover to create hardwood canopy cover used in the previous index.

$TreeSp_f =$ If $BahProp_f < 0.2$, then $TreeSp_f = 0$ (C.4)
Else if $BahProp_f \geq 0.2$ and $< 0.65$,
Then $TreeSp_f = (BahProp_f * 2.2222) - 0.4444$
Else $TreeSp_f = 1$

Where

$BahProp_f$ = The proportion of hardwood basal area to total basal area
$f$ = focal pixel

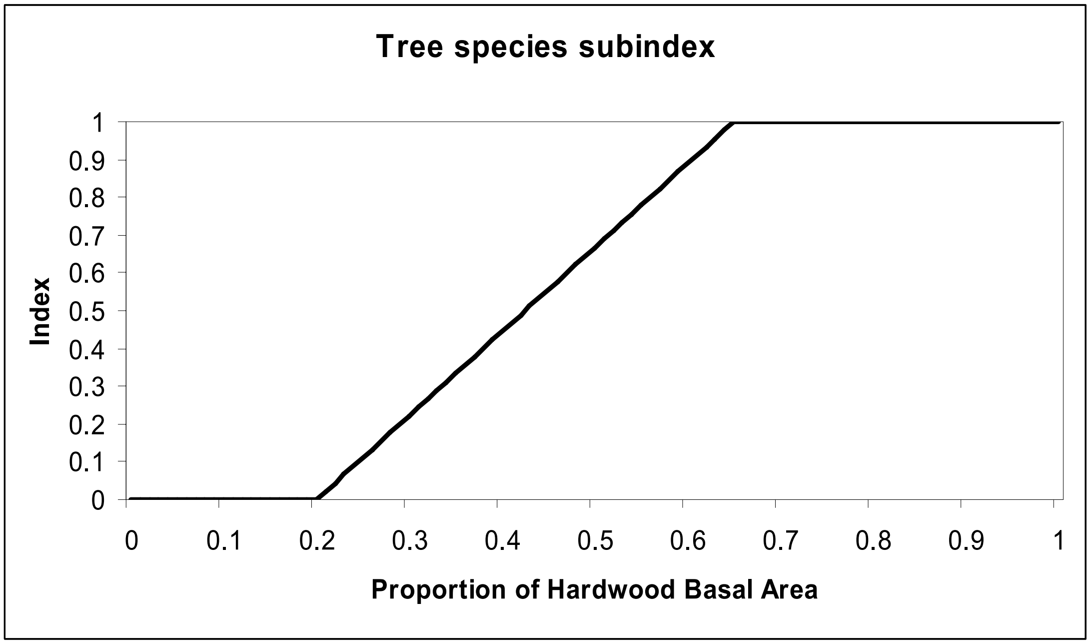

**FIG C2**. Relationship between tree species subindex and the proportion of hardwood basal area to total basal area.

$DEN25_f$: = If $Trees25_f < 25$ per ha Then $DEN25_f = 0$ (C.5)
If $Trees25_f \geq 25$ per ha and $Trees25_f < 123.55$ per ha
Then $DEN25_f = (0.01015 * Trees25_f) - 0.25368$
Else $DEN25_f = 1$

Where

$Trees25$ = Density of all trees >25 cm dbh per hectare
$f$ = focal pixel

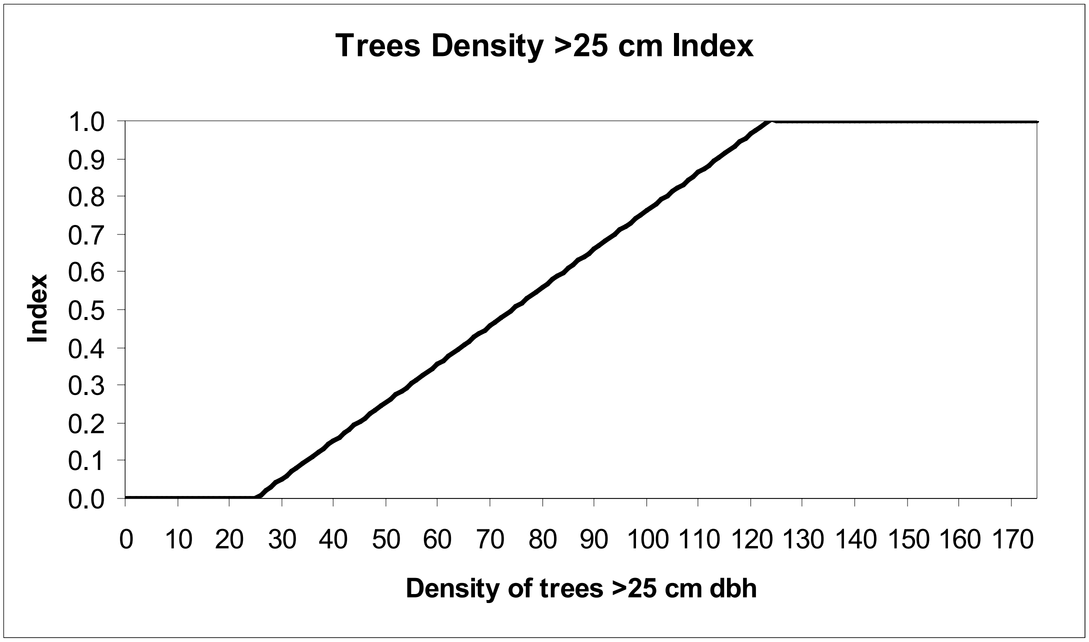

**Fig C3**. Relationship between Density of Trees >25 cm Index and the density of trees >25 cm dbh.

*Landscape capability index*

All metrics for this index are calculated within a 56-meter radius window and assigned to the focal pixel.

$$LCI_f = Average(HWCCw) \tag{C.6}$$

Where

*LCI* = Landscape capability index.

*f* = focal pixel

*HWCCw* = Hardwood canopy cover index averaged over the 56-m radius window

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