Scientific evaluation of the status of the Northern Spotted Owl
Appendices

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Appendices include supplementary material, as well as commissioned papers by Monahan, Carey and Noon. These commissioned papers represent the work and opinions of the individual authors, not the SEI status review panel as a whole.

APPENDIX 1

MORPHOLOGICAL ANALYSIS METHODS

By W. Monahan
Commissioned by SEI for Northern Spotted Owl Status Review

Specimens were scored using eight characters; several others were examined but excluded from the final dataset due to lack of repeatability.

Character	Criteria
LTC & RTC	upper surface of middle claw, from tip to point of insertion with skin (left and right)
LWING & RWING	longest flattened primary, from tip to wrist joint (left and right)
LWBAR & RWBAR	dorsal surface of wing, mean number of bars on outter margins of exposed primaries 6,7,8,and 9 (left and right)
TAIL	longest flattened tail feather, from tip to point of insertion of middle two rectrices with skin
TBAR	dorsal surface of tail, mean number of bars on inner and outter margins of middle two rectrices

Characters considered but excluded because the criteria were either obscured (due to different methods of preparing specimens) or impossible to ascertain: 1) length of rictal bristles, 2) greatest diameter of facial disc, 3) total length with feathers, 4) length of exposed culmen, 5) length of bill from nostril, 6) height of bill at base, 7) height of bill at nostrils, 8) width of bill at base, 9) length of mandible to feathering on chin, 10) width of mandible at base, 11) length of gonys, 12) length of tarsus.

In testing for subspecific differences, characters were first analyzed using ANOVAs that also considered the effects of sex and season (molting vs. non-molting). Season criteria are detailed in Gutiérrez et al. (1995); we arbitrarily assigned “1” to April – September (molting) and “2” to October - March (non-molting). ANOVAs were conducted using all data for *occidentalis* and *caurina* and again after limiting the *caurina* records to “pure” northern individuals collected north of central Oregon (Haig et al. accepted).

Sexual differences were apparent in all characters except RWBAR (Table 3.2A). In the case of TBAR where the differences were especially pronounced, females possessed on average approximately two more bars than males. However, TBAR was not diagnostic for assigning sex in either subspecies. Seasonal or molt effects were only significant for RTC. While molt-status would certainly have an important effect on the plumage characters considered (e.g. LWING, TAIL, RWBAR), all specimens examined possessed the full complement of wing or tail feathers used to establish character criteria.

Principal components analysis (PCA) was subsequently conducted separately for each sex using unstandardized LWING, RWING, TAIL, TBAR, RTC, and RWBAR measurements. We selected these characters in order to maximize the limited number of *occidentalis* specimens included in the multivariate data matrix. PCAs were repeated using mensural characters only (LWING, RWING, and RTC) and again using the plumage pattern characters (TBAR and RWBAR). Results were qualitatively similar to the combined analysis and are hence not discussed further. Missing characters for many *caurina* specimens collected in Washington prevented us from performing PCAs with “pure” northern individuals. Hence, multivariate analyses included specimens that were potentially from mixed populations.

APPENDIX 2 ECOLOGICAL NICHE MODELING

By W. Monahan
Commissioned by SEI for Northern Spotted Owl Status Review

Here we consider the large-scale bioclimatic evidence on whether *occidentalis*, *lucida*, and *caurina* represent valid geographical subspecies. Analyses also consider subspecies status in light of new genetic data suggesting recent gene flow or introgression from *occidentalis* (Haig et al. accepted). In establishing the validity of a subspecies, we adopt the 75% rule as proposed by Amadon (1949). Using this definition, geographical and genetic subspecies are considered valid from a bioclimatic perspective if less than 25% of the modeled ecological niche of the focal subspecies intersects the modeled niche of the sister taxon.

Ecological niche models were developed using 1,075 spatially unique *S. occidentalis* point localities (obtained from Breeding Bird Surveys, USGS Bird Banding Lab data, and museum specimens) in conjunction with 19 climate variables summarizing global temperature, precipitation, and seasonality (methods in Appendix 3). We first tested for sampling biases in the occurrence dataset by comparing observed multivariate climate space against the breadth of climate conditions encompassed by each subspecies' geographic range. PCA results suggest that the current sample sizes are generally representative of each subspecies' potential niche (Fig. 3). Sampling is weakest for MSO, suggesting that model predictions for *lucida* will be conservative and likely tend to underestimate niche breadth. However, an alternative interpretation is that the *S. occidentalis* range map (accessed from NatureServe) is not representative of the true distribution of the species (Fig. 4). Congruence between the actual point occurrence data and range map is poor, particularly in the case of *lucida*. Future analyses will consider possible sampling biases relative to other and perhaps more accurate estimates of the *S. occidentalis* range.

Figure 3.4 shows a large point locality gap running through central Shasta County, California. This gap overlaps the purported geographical break separating CSO from NSO (Grinnell and Miller 1944) and coincides with transitions/breaks between populations of plants (Soltis et al. 1997) and other vertebrates, including *Ensatina eschscholtzii*, *Bufo boreas*, *Elgaria coerulea*, *Contina tenuis*, *Lampropeltis zonata*, and *Thamnophis atratus* (Stebbins 2003); *Sorex* (Shohfi and Patton unpub.), *Thomomys monticola*, *Clethrionomys californicus*, and *Zapus princeps* (Department of Fish and Game 1990a). The region also marks the northern/southern distributional limits for *Taricha granulosa*, *Batrachoceps attenuatus*, *Ascaphus truei*, *Rana cascadae*, *R. muscosa*, *R. pretiosa*, and *Masticophis lateralis* (Stebbins 2003); *Picoides nuttallii*, *Empidonax traillii* (summer), *Sayornis saya* (winter), *Pica nuttalli*, *Phainopepla nitens*, *Guiraca caerulea* (summer), *Spizella atrogularis* (summer), and *Carduelis lawrencei* (summer) (Department of Fish and Game 1990b). Such consistent distributional breaks, transitions, and limits across taxa legitimize the separation of point occurrence data as presented in Figure 3.4.

All models developed with the geographically assigned point locality data (Fig. 4) performed well relative to random expectations (Table 3.2). Additionally, the models generally yielded low errors of omission and commission. We selected the 1 km^2 WorldClim data and 2.5-97.5% bounding envelope for subsequent analyses because this combination provided the strongest overall performance while allowing us to retain point localities in Canada and Mexico (i.e. the geographic extremes). Geographic projections of these models revealed that predicted niche overlap only occurred between *caurina* and *occidentalis*, covering approximately 78,500 km^2 (Fig. 5). Because *occidentalis* and *lucida* are collectively sister to *caurina* (Barrowclough et al. 1999, Haig et al. accepted), we compared the predicted NSO niche relative to the predicted niche for CSO and MSO combined. However, since the *lucida* niche did not overlap with either *caurina* or *occidentalis*, this was effectively the same as comparing *caurina* against *occidentalis*. Percentage overlap totaled 22% for *caurina*, an estimate just shy of the 25% cutoff established by Amadon (1949).

We repeated the *occidentalis* models after re-assigning individuals to subspecies based on mitochondrial haplotype frequencies furnished by Haig et al. (accepted) (Fig. 6). Geographic projections of these new models (1 km^2 WorldClim data, 2.5-97.5% envelope) revealed a predominantly northwestward expansion of the *occidentalis* niche (Fig. 7). Niche overlap totaled 42% for *caurina* (148,200 km^2) and, as revealed in the previous models, no overlap occurred between *lucida* and either of the other two subspecies. Hence, MSO and CSO consistently fall out as valid subspecies according to the bioclimatic data. However, depending on how the *caurina* boundaries are delineated (geography vs. genetics), different results emerge regarding the validity of the NSO subspecies.

The aforementioned analyses fail to compare *caurina* relative to its sister taxon, the most recent common ancestor (MRCA) of *occidentalis* and *lucida*. Ideally, patterns of niche overlap for NSO should be examined using niche models reflecting climate conditions around the time of CSO/MSO divergence. Future research will incorporate these analyses. However, as an approximate method permissible with our current resources, we estimated the ecological niche of the MRCA while assuming that climate conditions around the time of divergence were roughly similar to the present day (see Appendix 3). While this assumption is biologically tenable, it nevertheless provides for a second method of considering subspecies validity relative to the 75% rule. According to these methods, percentage overlap totaled 19% (geographical) and 22% (genetic) of the predicted *caurina* niche and 13% (geographical) and 14% (genetic) for the MRCA.

In summary, the bioclimatic models suggest that *occidentalis* and *lucida* are valid subspecies because the predicted niches of the two taxa consistently exhibit less than 10% joint overlap. The validity of *caurina* from a niche perspective currently remains uncertain but a priority of future research. In addition to *occidentalis* extending up into the *caurina* range from the south, both subspecies potentially face additional challenges from invasion by the Barred Owl, *Strix varia* (Peterson and Robins 2003). Peterson and Robins (2003) show that the areas of greatest displacement by *S. varia*, given its current westward spread, will overlap most extensively with the *caurina* distribution. When coupled with the apparent northward expansion of CSO, these results suggest that *caurina* faces a unique set of ecological pressures relative to *occidentalis* and *lucida*. Hence, it is critical to fully evaluate the bioclimatic evidence addressing the possible uniqueness of *caurina*. Results from such analyses will also be important in interpreting the intra- and inter-subspecific patterns of genetic and morphological variation described in the literature.

FIGURE A.2-1. Spotted Owl point localities obtained from USGS Bird Banding Lab records, Breeding Bird Surveys, and several major museum collections. Localities separated by geographic subspecies: *caurina* (blue, *n* = 765), *occidentalis* (red, *n* = 178), and *lucida* (green, *n* = 132). Current range map (gray regions) provided by NatureServe.

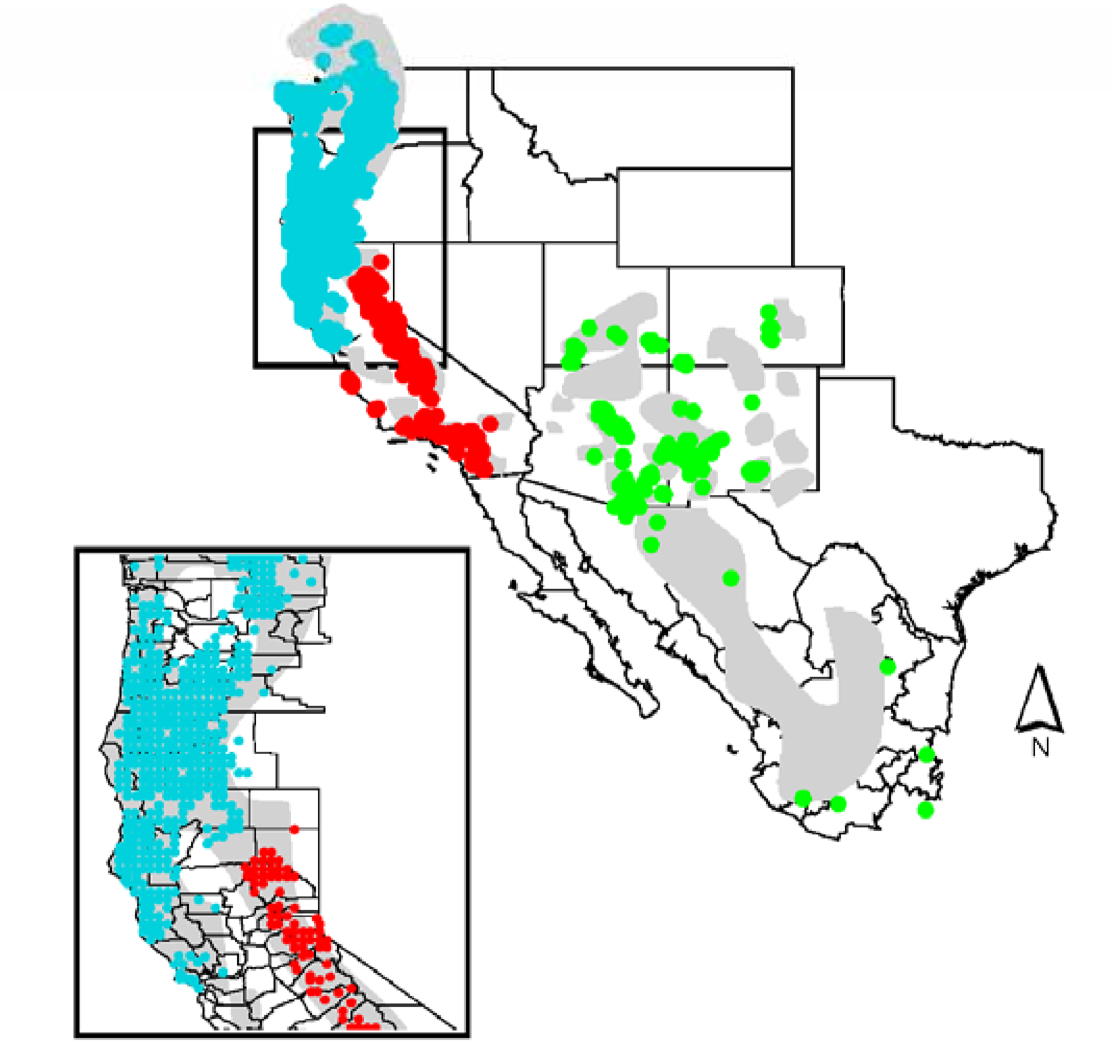


FIGURE A.2-2. Geographic projections of ecological niche models for *caurina* (blue), *occidentalis* (red), and *lucida* (green) obtained using point localities classified according to traditional subspecies criteria. Yellow areas identify regions of predicted niche overlap between *caurina* and *occidentalis* (78,500 km^2). Bold lines identifying subspecific "boundaries" were reconstructed from Grinnell and Miller (1944) and Gutiérrez et al. (1995).

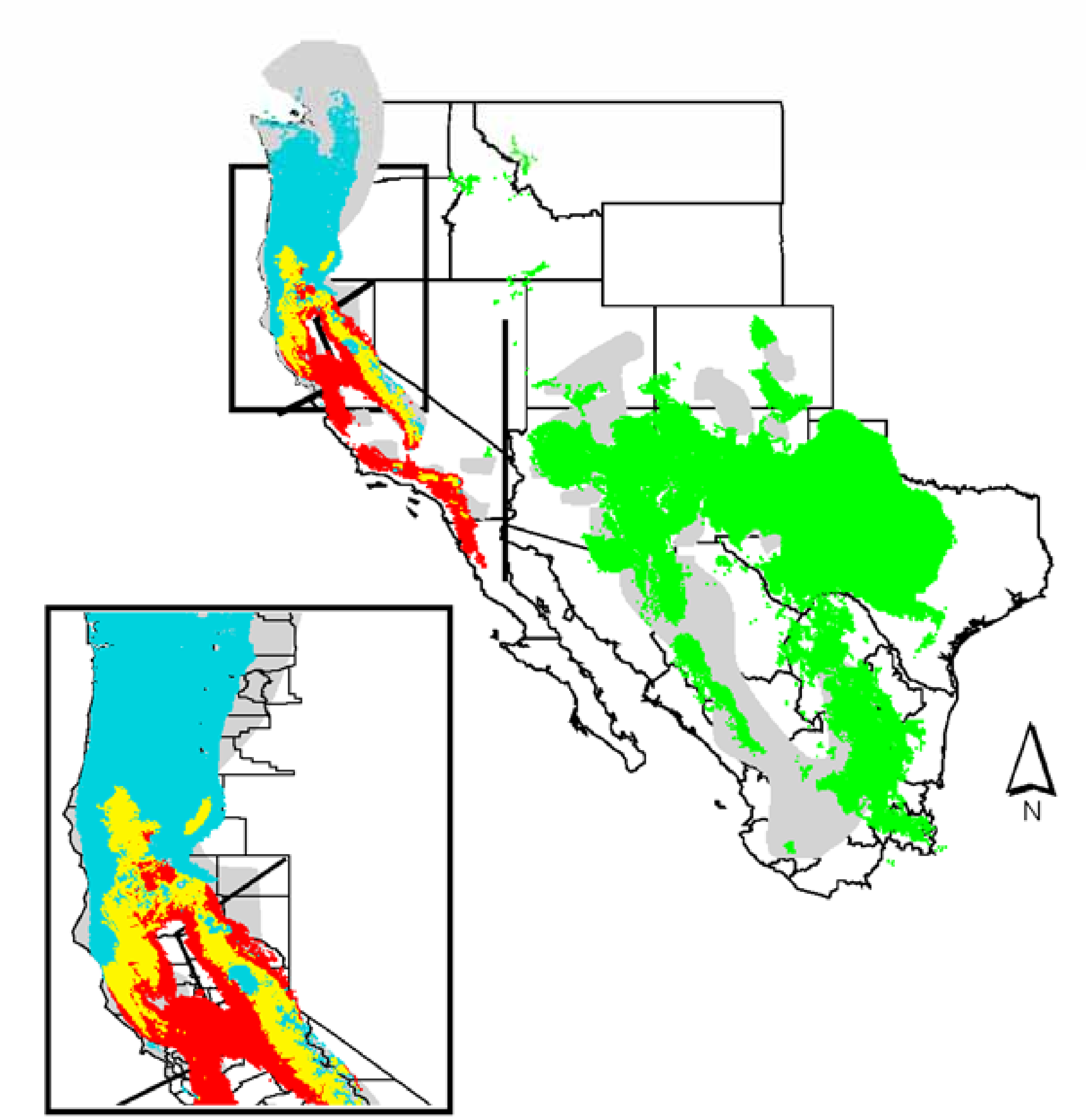


FIGURE A.2-3. Spotted Owl point localities separated according to mitochondrial haplotype frequencies: *caurina* (blue, *n* = 765), *occidentalis* (red, *n* = 178), and *lucida* (green, *n* = 132). Yellow points (*n* = 35) identify approximate locations of mixed NSO/CSO populations.

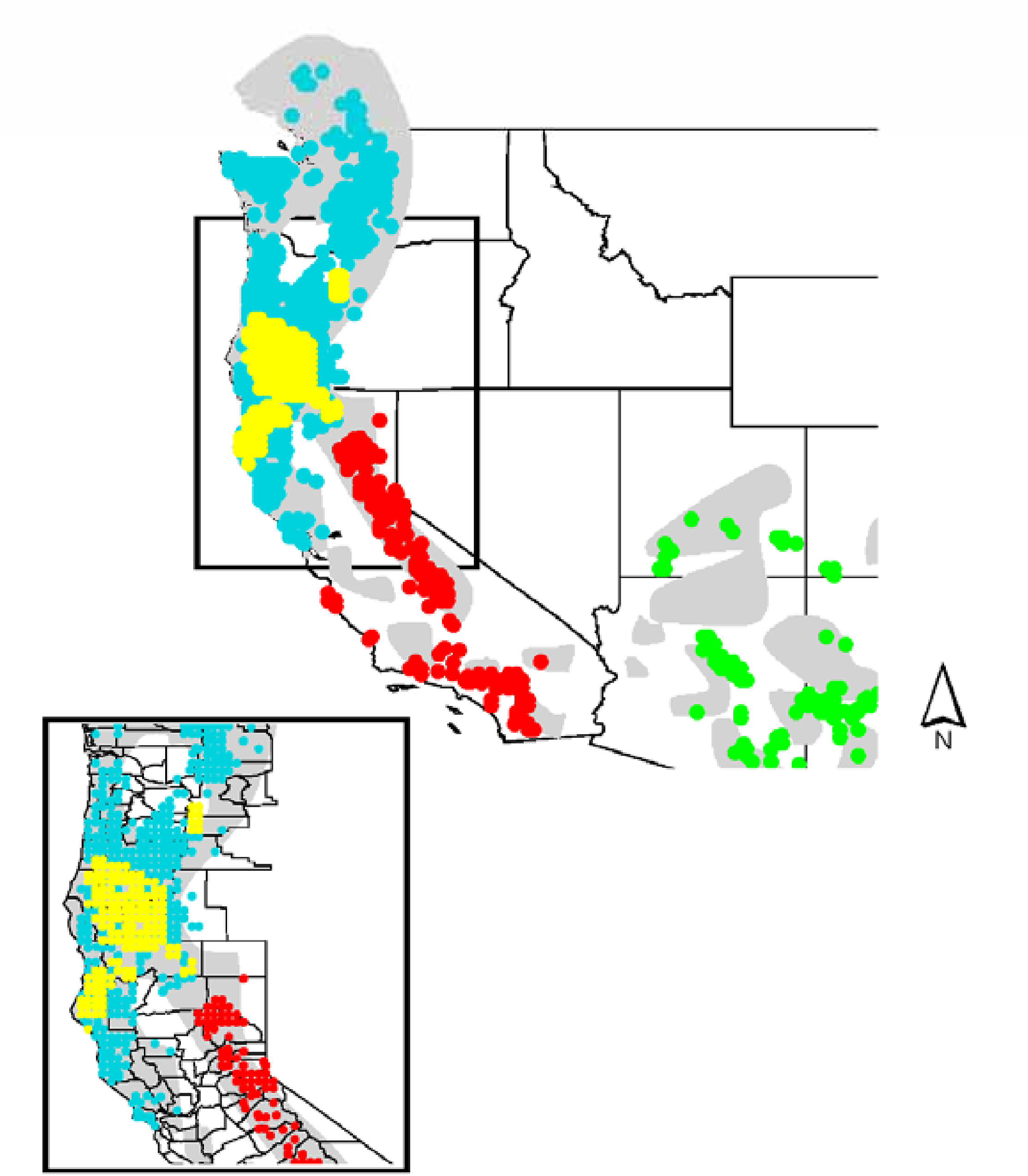
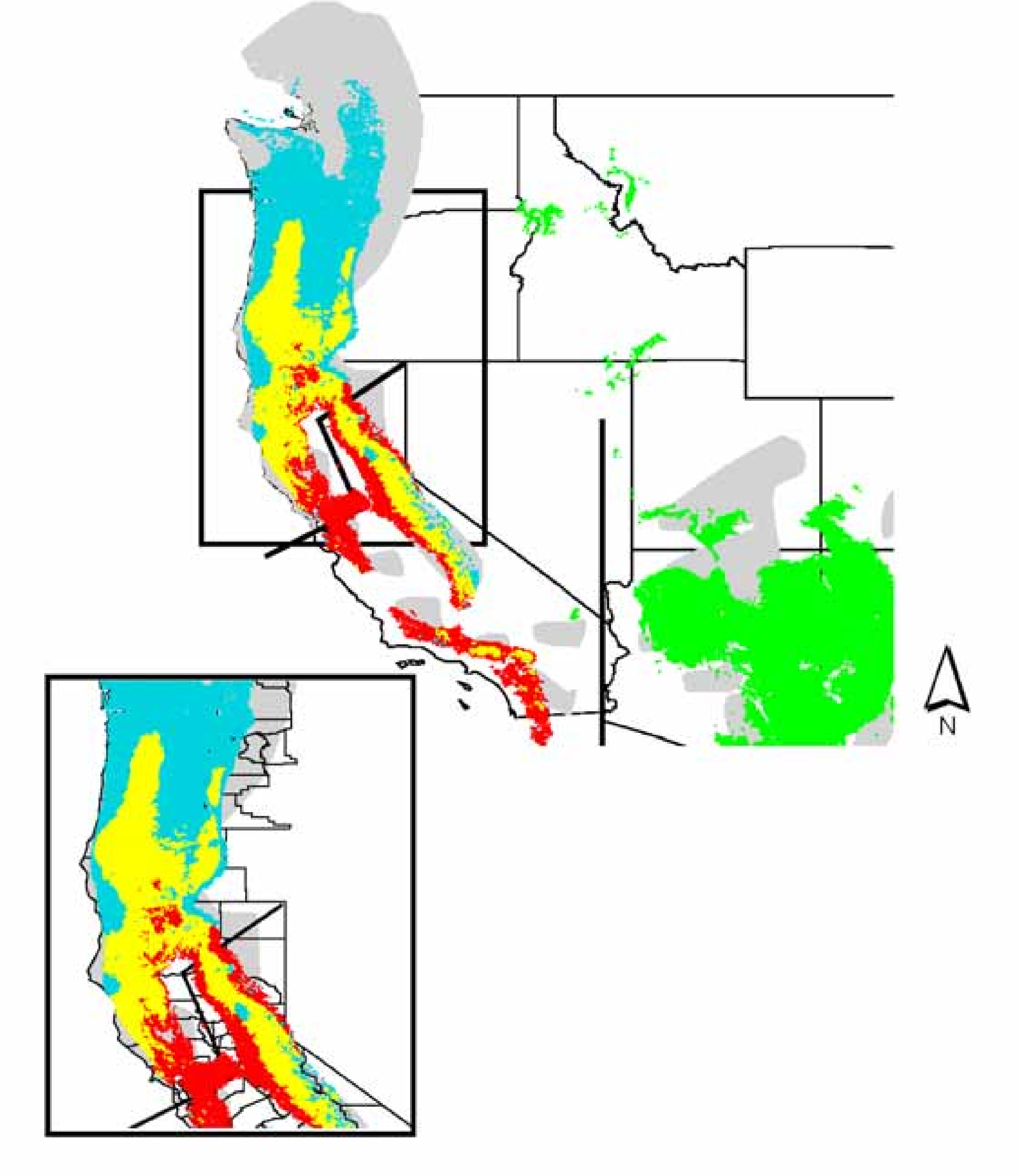


FIGURE A.2-4. Geographic projections of ecological niche models for *caurina* (blue), *occidentalis* (red), and *lucida* (green) obtained using point localities classified according to mitochondrial haplotype frequencies. Yellow areas identify regions of predicted niche overlap between *caurina* and *occidentalis* (148,200 km^2).



APPENDIX 3 ECOLOGICAL NICHE MODELING METHODS

By W. Monahan
Commissioned by SEI for Northern Spotted Owl Status Review

Point Occurrence Data: Spotted Owl point localities were obtained from Breeding Bird Surveys (n = 38) (*Sauer et al. 2003*), USGS Bird Banding Laboratory records (n = 20,162), and museum specimens (n = 183). Contributing museums included the National Museum of Natural History, Museum of Vertebrate Zoology, California Academy of Sciences, Burke Museum, Los Angeles County Museum, and the Mexican Atlas (Navarro-Sigüenza et al. in prep.), which included contributions from Louisiana State Museum of Natural Science, Western Foundation of Vertebrate Zoology, U.S. National Museum of Natural History, Texas Cooperative Wildlife Collections, Museum of Comparative Zoology, and Moore Laboratory of Zoology. This complete dataset reduced to 1,075 spatially unique occurrences (Geographic subspecies: n_{NSO} = 765, n_{CSO} = 178, n_{MSO} = 132). Given the new evidence documenting CSO mitochondrial haplotypes in habitats encompassed by the NSO range (Haig et al. accepted), we simulated mixed populations for use in assigning point localities to subspecies based on genetic (rather than purely geographic) criteria. This was achieved by applying the haplotype frequencies reported in Haig et al. (accepted) for southern Oregon and northern California to our original complete *S. occidentalis* point locality database consisting of 20,383 records. After eliminating duplicate coordinates, an additional 35 records were included to simulate mixed populations (Genetic subspecies: n_{NSO} = 765, n_{CSO} = 213, n_{MSO} = 132).

Climate Data: Models were developed using two sources of climate data. Daymet data, accessed from http://daymet.org/ (January 2004), provided coverage of the conterminous United States at 1 km^2 spatial resolution (18-year summaries, 1980-97, for 17 total variables). WorldClim bioclimatic data (Hijmans et al. 2004) provided global coverage for 19 variables at five minute (approximately 10 km^2) and 30 second (approximately 1 km^2) spatial resolutions (10- to 50-year summaries, 1950-2000). Daymet variables included temperature (max, min, and mean air temperature; day-to-day variability in max, min, and mean air temperature; number of frost days, growing degree-days, heating degree-days, and cooling degree-days), precipitation (mean daily rate and total), radiation (total and day-to-day variability in total shortwave radiation), and humidity (daily mean and day-to-day variability in water vapor pressure). WorldClim variables only summarized temperature (mean annual; mean diurnal range; isothermality; seasonality; max and min of warmest and coldest months; annual range; mean of wettest, driest, warmest, and coldest quarters) and precipitation (mean annual; mean annual of wettest and driest months; seasonality; mean annual of wettest, driest, warmest, and coldest quarters). See original references for additional information.

Sampling Biases: For each geographically defined subspecies, we used principal components analysis to screen for possible sampling biases by contrasting observed multivariate climate space against the multivariate climate space extracted from a range map. This was achieved by first intersecting the observed point localities with the climate variables of interest (10 km^2 WorldClim variables to limit computation time and match the spatial resolution of the USGS data). We then extracted all points at 10 km^2 from each subspecies' range and intersected these with the climate layers. Standardized variable values from the observed and range point datasets were reduced to two axes explaining the majority of the variation (PC1 and PC2). In the absence of sampling biases, a high degree of overlap is expected between the observed and range datasets when plotted in the same component space. With the possible exception of MSO, sampling appeared to be representative of the current range for each subspecies. As explained in the niche modeling section of the report, discrepancies between the observed MSO point localities and the MSO range map could mostly reflect inaccuracies in the range data provided by NatureServe (http://www.natureserve.org/).

Ecological Niche Models: Modeling procedures were carried out using BIOCLIM, a profile-matching algorithm that first computes the portion of multivariate climate space occupied by the original point localities and then extrapolates a bioclimatic envelope across the entire geographic area considered (Busby 1991). We ran BIOCLIM separately for each geographically defined subspecies using both Daymet (1 km^2 resolution) and WorldClim (1 and 10 km^2 resolution) climate data as input variables. Models were further separated according to different bounding envelopes of increasing stringency (0-100%, 2.5-97.5%, and 5-95%). After selecting an optimal model (see below), new bioclimatic envelopes were generated using the genetically assigned point localities for *occidentalis*. We then averaged the CSO-MSO minimum and CSO-MSO maximum climate values on a per variable basis and used these new climatic ranges to extrapolate hypothetical niches of their most recent common ancestor (according to both geographic and genetic criteria). Unfortunately, this averaging method assumes that climatic conditions around the time of divergence were similar to present day. Future analyses will circumvent this problem by utilizing climatic reconstructions c. 8,000-10,000 ybp.

Quantifying Model Performance and Patterns of Niche Overlap: Our goal was to select a single model that provided significant improvement over chance while minimizing errors of omission and commission. These criteria were assessed using three confusion matrix measures reviewed in Fielding and Bell (1997): Kappa, false positive rate (FP), and false negative rate (FN). To obtain the measures, we first randomly subsampled 50% of the geographically assigned point localities for each subspecies and used these in conjunction with the climate data to develop BIOCLIM models. The remaining point localities were then intersected with the BIOCLIM distributions and summarized according to true and false positives. Secondly, we combined the point localities of the other subspecies (NSO-CSO, NSO-MSO, and CSO-MSO) to generate absence data. These were intersected with the BIOCLIM models to yield true and false negatives. After calculating the three performance measures (Kappa, FN, and FP), we computed a single weighted score for purposes of prioritizing the models (Kappa-FN+FP*2). The weighting penalized false positives by a factor of two since our conclusions are especially sensitive to overprediction errors that potentially arise when reducing an *n*-dimensional niche to a handful of variables. Using these criteria, two optimal models emerged (Daymet 0-100% and 1 km^2 WorldClim 2.5-97.5%). We selected the WorldClim model because it allowed us to retain all point localities from Canada and Mexico (i.e. the geographic extremes). Lastly, we re-combined all point locality data by subspecies and used BIOCLIM in conjunction with the 1 km^2 WorldClim data to extrapolate new 2.5-97.5% environmental envelopes for use in estimating niche overlaps. In quantifying the degree of niche overlap among sister taxa, we intersected all pertinent combinations of the BIOCLIM outputs and summarized areas of predicted sympatry relative to each subspecies' total potential niche projected in geographic space.

APPENDIX 4 SUMMARY OF PREY BIOLOGY

Prepared by Lisa Sztukowski and Steven Courtney

This appendix provides a brief introduction to prey biology, including recent literature. Useful recent summaries of prey ecology are also provided by Aubry et al. (2003), Hallett et al. (2003) and Smith et al. (2003).

1. Northern Flying Squirrel (*Glaucomys sabrinus*)

The 25 recognized subspecies of Northern flying squirrel are nocturnal arboreal rodents that are active year-round in both coniferous and deciduous forest with a variety of stand conditions (Wells-Gosling and Heaney 1984, Rosenberg et al. 1996). Considered a keystone species, they disseminate the spores of etcomycorrhizal fungi, which enhance nutrient and water absorption in trees and are predated by a variety of mid-sized predators and owls (Carey et al. 2002, Carey 2000). Mycorrhizal and epigeous fungi are prominent in the diet of Northern Flying Squirrels; however, seeds, fruit, nuts, vegetative matter, insects, and lichens may also represent a significant proportion of their diet (Carey 1995a, Carey 2000, Carey et al. 1999, Thysell et al. 1998, Waters and Zabel 1995, Rosenberg et al. 1996). The overall diet is similarly between old and young forest types, but diversity and abundance of fungi vary between forest type, stand type and structure (Waters and Zabel 1995, Carey et al. 2002, North et al. 1997, Colgan et al. 1999, *Lehmkuhl et al. draft 2004b*).

Flying squirrel "den sites include: (1) cavities in live and dead old-growth trees, (2) cavities, stick nests and moss-lichen nests in small (10-50 cm dbh) second-growth trees, (3) cavities in branches of fallen trees, (4) nests in decayed stumps of felled old-growth trees and suppressed young trees [Carey et al. 1997] and…(5) witches broom formed by mistletoe infections" (Carey 2000:54).

Life history characteristics vary "from north to south, including adult body mass, rate of juvenile weight gain, age of sexual maturation for females, proportion of females that are sexually active, survivorship, population age structure, and population density. Some life-history attributes and predation seem density-dependant" (Carey 2000:45). Adults may weigh up to 194 g, but varies by physiographic province, age class, season, and occasionally between sexes (Villa et al. 1998, *Lehmkulh et al. draft 2004*, Carey 2000, Rosenberg and Anthony 1992, Witt 1991). Body mass is highest in winter and lowest in spring and summer, which corresponds to the fruiting cycles of Basidiomycetes and Ascmycetes, respectively (Carey 2000, Witt 1991). Seasonal abundance and diversity of hypogeous fungi may also influence reproductive chronology and population density (Forsman et al. 1994, Witt 1991, Carey 2000, Waters and Zabel 1995 Rosenberg et al. 1996, *Lehmkuhl et al. draft 2004b*).

Densities of Flying Squirrels generally tend to decrease toward the northern edge of the Spotted Owl's range (southern Coast Ranges and Western Cascades vs. Olympic Peninsula and North Cascades of Washington), with a few exceptions (Carey 1995a, 2000). In British Columbia, old spruce-fir forests of east of the Cascades had similar flying squirrel densities to northwestern Washington and high densities were found in coastal areas of western hemlock forests (Carey 2000). Density estimates may be influenced by region, methods, and analysis used (Martin and Anthony 1999). Densities tend to increase with stand age but results vary and are not always significant (Zabel and Waters 1995, Carey et al. 1992, Rosenburg and Anthony 1992, Witt 1992, Carey 1995, Rosenberg et al. 1996). Densities are generally influenced by forest type, legacy retention, management strategy, stand age and structure (Carey 1995a, 2000, *Lehmkuhl et al. draft 2004*). The three main factors may limit population densities: den structures, food, and predation limiting population size (Carey et al. 1992, Carey 2000, Carey et al. 2002, Witt 1991 and others).

2. Woodrats

Two species of woodrat occur within the range of the northern spotted owl. They include the bushy-tail woodrat (*Neotoma cinerea*), which has a broad patchy distribution throughout the Pacific Northwest, and the dusky-footed woodrat (*Neotoma fuscipes*), which is distributed through northern California, southwestern Oregon, and the Willamette Valley (Hall 1981, Carey et al 1999). Both species are "more common in the mixed-conifer forests of the Umpqua Valley margins… than in the southern Douglas-fir-western hemlock forests of the Coast Ranges or Western Cascades" (Carey et al 1999:73).

Dusky-footed woodrats are nocturnal, arboreal herbivores that are a major prey species for owls below 1,250 m (Barrows 1980, *Solis 1983*, Forsman et al. 1984, *Ward 1990*, Carey et al. 1992, Sakai and Noon 1993; Gander 1929, Linsdale and Tevis 1951, Sakai and Noon 1997). Dusky-footed woodrats are not found in Washington, and Douglas-fir-western hemlock forests in Oregon provide poor habitat as do some mixed-conifer and transition forest stands (Carey et al. 1999).

Generally, dusky-footed woodrat densities appear to follow stages influenced by habitat quality; the progression follows as: unsuitable habitat (recently burned clearcuts), to optimal habitat (sapling/bushy poletimber 15-40 years old and young redwood forest 5-20 years old) then a gradual decline to marginal habitat (small and large sawtimber stands/intermediated-aged forests) with a possible second peak in abundance in old forest as openings form in the canopy structure creating patches of stable, bushy understory (*Hamm 1995*, *Hamm and Diller 2002 draft*, Raphael 1988, Sakai and Noon 1993, Carey 1994, Carey et al. 1999). The gradual decline in abundance may reflect a change in habitat quality, including changes in the understory influencing food and nest site availability or reduced protection from predators. There is also a significant difference in abundance between thinned and unthinned mature stands, which may be also attributed to change in the understory (*Hamm and Diller 2002 draft*). Increases in woodrat abundance in redwood forests have been associated with increased vegetation density and increased amounts of redwood cover and decreased amounts of Pacific rhododendron and salal cover (*Hamm 1995*, *Hamm and Diller 2002 draft*).

Four subspecies of bushy-tailed woodrats exist in the Pacific Northwest and are active throughout the year (Carey 1991, Moses and Millar 1992). Bushy-tailed Woodrats use a variety of den/nest sites which appear to be climate dependant (Carey et al. 1999). Their reliance on patchy resources (such as rock outcrops, talus) increases the likelihood of intraspecific competition and may increase the home range size necessary to meet their needs (Moses and Millar 1992, Topper and Miller 1996).

Bushy-tailed woodrat abundance was low in upland Douglas-fir-western hemlock forests, and Douglas-fir transition forests. Bushy-tailed woodrats are generally absent from oak woodland, Douglas-fir forest-prairie habitats and some upland sites (Ryan and Carey 1995 in 655 Carey et al. 1999). Bushy-tailed woodrat densities increase in stream-side sites associated with bounders as well as old forests, valley-margins, and mixed conifer sites (Carey et al. 1999). Throughout Washington and Oregon, Bushy-tailed woodrat abundance varied in late-seral and stream-side stands but woodrats consistently occupied old, natural stands and were absent from 35-80 year old managed stands (Carey et al. 1999). "Overall, relative frequencies (percent of sites with woodrat captures…) suggest that optimum habitat for bushy-tailed woodrats was old, natural forests (>two-fold margin) with streams (almost a four-fold margin)" (Carey et al. 1999:73).

3. Red Tree Voles (*Arborimus or Phenacomys longicaudus*)

Red Tree Voles are highly specialized, colonial, arboreal species weighing approximately 27g found mostly in Douglas fir forests in the humid temperate region of Western Oregon, and northwestern California (Gillesberg and Carey 1991- Baily 1936 Hall 1981 Johnson 1973, Johnson and Maser 1982, Meiselman and Doyle 1996, Carey 1991, Johnson and George 1991, Maser 1998, Carey 1992). Due to difficultly in capturing and studying these specialized small mammals, their entire range may still be unknown as their known range has expanded as recently as 1995 (Corn and Bury 1986, 1988, Gillesberg and Carey 1991, Manning and Maguire 1995, Meiselman and Doyle, 1996).

Highest abundances occurred most frequently in old-growth with consistent patterns of variation in the Oregon Coast Range and Cascade Range provinces (supported by Carey 1989, Corn and Bury 1986, 1991, Aubry et al. 1991, Meiselman and Doyle 1996, Zentner 1977, Gillesberg and Carey 1991). However, many studies use nest abundance as the index of abundance, which may vary in nest detectability, and the accuracy of determining activity levels and type (*Swingle presentation*, Meiselman and Doyle 1996). Other studies use pitfall traps that may be biased as they are based on terrestrial activity which has not been accurately assessed and may be higher in older forest than in younger stands (*Forsman pers. comm.*). Nests are found in a patchy distribution with individual voles using multiple nests (*J. K. Swingle Conference 2003*).

Our knowledge of this species has increased primarily due to its importance as prey to the spotted owl and its use as an indicator species for old growth forests. However, much of its behavior, habitat, and microhabitat use remain elusive due to the expense and time-consuming nature of studying them. Radio tracking holds promise, expanding our knowledge of their behavior. Large expanses of habitat have yet to be surveyed and may result in future extension of their range. Felled tree surveys, in conjunction with current timber harvest practices, could be used as a cost-effective method of surveying large areas (Gillesberg and Carey1991). Unfortunately, this method destroys habitats and colonies. The red tree vole is listed as "closely associated" with old growth and therefore "the most vulnerable of the arboreal rodents to local extirpations resulting from the loss or fragmentation of old-growth Douglas-fir forests"

(Ruggiero et al. 1991, Huff et al 1992). Factors limiting red tree vole populations seem to "be the size of the old-growth stand, the length of time it has been colonized by red tree voles, and the noncatastrophic influences of fire, windstorms, and predation by owls" (Carey 1991). Dependence on large continuous tracks of old growth may limit population growth as the forest becomes more fragmented over time, increasing isolated populations and local extinction events.

4. Red-backed Voles

Two species of red-backed voles are prominent prey items of the northern spotted owl. These include the southern red-backed vole (*Clethrionomys gapperi*) and the western red-backed vole or California red-backed vole (*Clethrionomys californicus*).

Southern red-backed voles inhabit the Cascades, eastern Washington, northeastern Oregon (*Web-general description*). In the southern Washington Cascades, southern red-backed voles account for about 23% of the total captures (Aubry et al. 1991). Few studies in the Pacific Northwest have focused on this species, and most of those studies occurred prior to 1990.

Western red-backed voles occur in western Oregon and northwestern California (Alexander and Verts 1992, Rosenberg et al. 1994). Patterns of abundance associated with stand age have been inconsistent. Some studies indicate voles "occur more frequently in managed closed-canopy forests with little understory development, and select for habitat that has significant amounts of coarse woody debris (Tevis 1956, Gashwiler 1959, Maser 1981, Doyle 1987, Gomez 1992) or greater food resources (hypogeous fungi, Ure and Maser 1982)" (Rosenberg et al. 1994:266). Others have found no significant difference in the abundance of voles between young and older forests, but the stands selected in these studies were mostly naturally regenerated from wildlife (Corn and Bury 1991, Aubry et al. 1991, Gilbert and Allwine 1991, also listed in Rosenberg et al. 1994). Red-backed vole are "exceptionally rare in clearcuts" and their abundances were "strongly and negatively affected by clearcutting forests" which may have an effect for 10 to 60 years following clearcutting (Hooven and Black 1976, Taylor et al 1988, Raphael 1988, Rosenberg et al. 1994 in Mills 1995).

The presence of coarse woody debris, forest floor structure (i.e. organic soil depth) and food availability may influence vole abundance in old-growth forests and young fire-regenerated stands. In the Oregon Cascades, vole abundance was positively correlated with organic soil depth (Gomez 1992, Rosenberg et al. 1994). Inconsistent results indicate large amounts of coarse woody-debris may be critical habitat for voles. Numerous studies have found a relationship between coarse woody debris and vole abundance (Doyle 1987, Hayes and Cross 1987, Tallmon and Mills 1994). However Mills (1995) did not find that vole abundance corresponded with coarse woody debris. In this study there was a build up of woody-debris at the edge of remnant patches by fallen trees, blow downs and death that was not of advanced-decay class, which voles select (Tallmon and Mills 1994, Mills 1995).

Temporal fluctuations in trends of abundance were not consistent between areas; abundance would increase in one area, while decreasing in another (Rosenberg et al. 1994). Space-use trends by voles are consistent with the distribution of hypogeous sporocarps of mycorrhizal fungi, the primary component of their diet (Maser et al 1978, Ure and Maser 1982, Mills 1995).

5. Deer Mice

Two species of *Peromyscus* that are potential or actual Spotted Owl prey, *Peromyscus maniculatus* and *P. oreas*, have been studied in Washington and Oregon. The common deer mouse (*Peromyscus maniculatus*) occurred in relatively low numbers throughout Washington and Oregon with a strong association with clear-cuts and fragments; the forest deer mouse (*P. oreas*) was more abundant in old-growth, especially in forests of the Olympic Peninsula (Carey and Johnson 1995, Songer et al. 1997). *Peromyscus oreas* and *P. maniculatus* show an inverse correlation in relative densities at all sites and showed significant niche segregation across macroclimates (Songer et al. 1997). Competition between these species may limit *P. oreas* densities, as *P. oreas* reach “much higher” densities in fragment sites when *P. maniculatus* is absent (Songer et al. 1997). “As an arboreal species, *P. oreas* also is likely to be a more accessible prey for the spotted owl than *P. maniculatus*, which typically restricts its movements to the lowest stratum of the forest “ (Songer et al. 1997:1037).

In the West Cascades, the average density of deer mice was 7.3±0.9 mice/ha deer mice and represented a biomass of 161 g/ha, similar to the biomass of northern flying squirrels (173 g/ha) (*Rosenberg et al. 2001*). However temporal variability accounted for 67.6% of process variation among years with over a 20-fold fluctuation in abundance (*Rosenberg et al. 2001*). This may account for the high annual variability of each prey species in the diet.

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APPENDIX 5 RELATIONSHIP OF PREY AND FOREST MANAGEMENT

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Paper commissioned by SEI for Northern Spotted Owl Status Review

Spotted owls use diverse prey, ranging from insects to arboreal mammals (Forsman et al. 1984). While spotted owls may forage opportunistically, in any one region and in any one home range within a landscape, spotted owls (and other predators) tend to search for and prey upon a limited number of focal species in any given landscape or period (Forsman et al. 1984, Carey et al. 1992, Zabel et al. 1995). Other species may be taken opportunistically on a regular basis. For example, in western Washington, northern flying squirrels (Glaucomys sabrinus) constitute the predominant biomass in the diet. But spotted owls will regularly take the semi-arboreal Keen's mouse (Peromyscus keeni) and occasionally Douglas' squirrels (Tamiasciurus douglasii), despite the mouse's relatively small size and the squirrel's primarily daytime activity. Juvenile lagomorphs (hares and rabbits) will be taken in the summer when they are abundant, even though they seem to be at the upper size limit that a spotted owl can handle. And if rock outcrops are nearby and inhabited by bushy-tailed woodrats (Neotoma cinerea), the woodrats will be avidly hunted until reduced in abundance or extirpated. In the Pacific Northwest, the northern flying squirrel is the most universally sought after and consumed prey. However, in southwestern Oregon and northern California, dusky-footed woodrats (Neotoma fuscipes), bushy-tailed woodrats, or red tree voles (Arborimus longicaudus) may predominate in the owl's diet, even when and where flying squirrels are a mainstay. Regularly taken prey not only includes mice in the genus Peromyscus, but terrestrial rodents as well. Where the prey base consists of diverse and abundant arboreal and semi-arboreal small mammals >20 g and <400 g in body mass, spotted owl home ranges are smaller and more concentrated than when one or two prey species are in abundance or where prey biomass is low (Carey et al. 1992). Such conditions exist, for example, in valley margin old-growth forests in southwestern Oregon, where bushy-tailed woodrats, dusky-footed woodrats, flying squirrels, and red tree voles all may be abundant in the same 40-ha patch (Carey 1995, Carey et al. 1999a, c). In mosaics of forests of different seral stages and species composition, not all highly-valued prey may occur in each landscape unit (patch), but spotted owls will seek out and repeatedly use diverse patches, each containing an abundance of one or more prey (Carey and Peeler 1995, Zabel et al. 1995). Not all patches, by any means, will contain exploitable prey populations, and numerous patches of low foraging quality can have negative impacts on owl demography and behavior (Carey et al. 1992). Where prey populations are high, owls will forage in uncharacteristic environments—clearcuts near old growth with dusky-footed woodrats, deciduous riparian zones or rock outcrops with bushy-tailed woodrats, and dense stands of small-diameter trees containing dusky-footed woodrat colonies. But, even in the appropriate zoogeographic area, not all clearcuts, thickets, rock outcrops, or riparian areas will contain woodrats.

The owl's prey base not only differ in diversity spatially but the prey species themselves differ in their relative abundances among seral stages and forest types as animal and fungal diversity differs also and in their habitat relationships among biogeographic provinces (Gunther et al. 1983; Carey et al. 1992, 1999a, c; Rosenberg and Anthony 1992; Sakai and Noon 1993; Carey

1995, 2000a,b; Carey and Johnson 1995; Waters and Zabel 1995; Wilson and Carey 2000; Carey and Harrington 2001; Ransome and Sullivan, 2003; Ransome 2004; and many others). This phenomenon, once well understood (ecotypes, Odum 1971; populations in heterogeneous environments, Fretwell 1972), has often been overlooked in studies of the spotted owl prey base. Not all old growth is alike, not all second growth is alike, and flying squirrels in western British Columbia (Ransome 2004) may exhibit significantly different habitat relationships from flying squirrels in southwestern Oregon (Carey et al. 1999a) or northeastern California (Waters and Zabel 1995). In between, across the Western Hemlock Zone, flying squirrels (and other major prey species) exhibit large variation in population responses to seral stages and various habitat elements and habitat elements can differ markedly in their abundance within seral stage across a region (Carey 1995, 2002; Carey et al. 1997, 1999a, 2002; Carey and Harrington 2001). Rarely are animal abundances accurately predicted by the abundance of a single habitat element or a simple linear combination of >1 habitat element except for in geographically, seasonally, and developmentally limited samples. Liebig's law of the minimum seemed to work well for plants in homogeneous environmental conditions as individual soil elements were manipulated. But Shelford formulated his law of tolerance to emphasize the tremendous effects of interactions among biologically important variables as they jointly approached minima. And today, we recognize that floristic diversity as it influences dietary diversity and vegetation structural diversity as it influences availability of essential habitat elements mediate important effects of competition, competitive release, disease, and predation on small mammal abundances. Environmental conditions, floristics, and zoogeography condition the abundance and diversities of small mammals and their interspecific- and habitat relationships (Carey et al. 1999a, b; Carey and Harrington 2001, Johnson and O'Neil 2001). Competition among prey species for limited food or den resources and predation by various predators seem important. Even where prey may be abundant, however, vegetation structure may not be conducive to owl foraging—a prey species may be abundant, but unavailable. Such a condition may exist in second-growth forests with dense, low understory and large gaps between the understory and the nearest perches in the canopy. Spotted owls and other predators such as long-tailed weasels (Mustela frenata) may decimate prey populations in some patches and may not return to forage heavily in those patches for 1-2 years (Carey et al. 1992, Rosenberg and Anthony 1992, Carey and Peeler 1995, Wilson and Carey 1996). If owls are attracted to dense concentrations of prey that they are able to exploit, and if this exploitation can significantly reduce the density of the prey, then the history of foraging by owls (or weasels) must be accounted for before the value of any one or any one set of habitat elements to the prey species can be understood. Given all this complexity, the question is what do we know about the effects of forest management (positive and negative) on the diversity and biomass of prey available to the spotted owl?

Timber Harvest

Timber harvest (clearcutting, partial cutting, and variable retention harvest systems) is a catastrophic disturbance with both short- and long-term effects on prey. Surprisingly many forest-floor small mammals respond positively to clearcutting in the short-term (Gunther et al. 1983). This is simply because any disturbance entails release of certain resources that then become available to various life forms, including small mammals. Cone- and seed-laden branches come to the forest floor to be exploited by diverse small mammals. With site preparation, these are often destroyed but colonization by grasses, forbs, and shrubs benefits diverse prey species (dusky-footed woodrats, deer mice, Oregon creeping voles [Microtus oregoni]) but the site might well be uninhabitable for a considerable period by the most arboreal rodents—red tree voles, flying squirrels, and Douglas's squirrels. The degree to which legacies are retained during timber harvests is an important determinant of recolonization of the site by all life forms (Perry et al. 1989, Franklin et al. 2000), including the fungi that are the mainstay of the flying squirrel and California red-backed vole diets (Clethrionomys californicus) (Amaranthus et al. 1989). These legacies are diverse but include fungal mycelia (indeed intact forest floor microbial communities in patches of intact forest floor), coarse woody debris, intact vascular plants, and fungal and plant propagules. Intentional retention of legacies can accelerate the pace of ecosystem recovery (Franklin et al. 1997)—the rate of change in the new, self-organizing community will be rapid and prey species will be affected differentially. Dusky-footed woodrats are benefited by delayed recruitment of a dominant cohort of conifers and rapid recruitment by evergreen hardwoods; flying squirrels respond oppositely.

Perhaps the biggest consequence of conventional clearcutting comes not during the disturbance itself or the period of rapid reorganization, but later when the conifer canopy closes (the stem-exclusion or competitive exclusion stage, Oliver and Larson 1996, Carey et al. 1999c). Dense, closed-canopy second-growth without legacies can not only be devoid of exploitable prey populations (Carey 1995, Carey and Johnson 1995, Carey and Harrington 2001) but also poorly suited for owl roosting, foraging, or nesting (Carey et al. 1992). This period of low structural diversity can last >100 years (Carey et al. 1999c, Franklin et al. 2002) and can have profound effects on the capacity of the forest to develop biocomplexity in the future (Halpern et al. 1999, Carey 2003a). However, with legacy retention, patchy regeneration of multiple species including hardwoods, and natural disturbances during the periods following either a natural catastrophic disturbance by wind or fire or following partial cuts, the prey base can reach or exceed levels of diversity and abundance found in many old-growth stands and will be used for foraging and roosting by spotted owls (Carey et al. 1992, Rosenberg and Anthony 1992, Carey 1995, Glenn et al. 2004).

Thinning

Thinning can be done in many ways and for many purposes and has differing and diverse consequences on the ecosystem including effects on the prey themselves, the plants that provide them with food and cover, the fungi that provide them with food, and the health and resilience of the forest (Waters et al. 1994; Carey et al. 1996, Colgan et al. 1999; Graham et al. 1999; Carey 2000b, 2001; Thysell and Carey 2000, 2001; Wilson and Carey 2000a, b, 2002b; Carey and Wilson 2001; Sullivan et al. 2001; Muir et al. 2002). All thinning has short-term negative effects on understory plants (mechanical destruction) and below-ground fungi (death of host trees and mechanical destruction). Heavy thinning in the Mixed Conifer/Mixed Evergreen Zone may benefit woodrats and deer mice in the mid-term, but to the detriment of flying squirrels. Conventional thinning in the Western Hemlock Zone may result in very low flying squirrel populations through negative effects on truffle production and arboreal travelways (Colgan et al. 1999, Carey 2000b) and reduced foraging by spotted owls (Meiman et al. 2003) for a long time while increasing numbers of forest-floor rodents (Wilson and Carey 2000). Conventional thinning, however, may result in uniform dense understories unfavorable to both flying squirrels and owl foraging in the midterm. Variable-density thinning, however, hold promise for acceleration of the development of spotted owl habitat and dense prey populations (Carey 1995, 2001, 2003a. Carey et al. 1999a,b; Carey and Wilson 2001; Muir et al. 2002) especially when appropriate attention is paid to decadence (snags, cavity trees, and coarse woody debris) (Bunnell et al. 1999; Carey et al. 1999a, b; Carey 2002). There maybe a short-term impact on truffle production, flying squirrel abundance, and owl foraging, the ecosystem recovers more quickly and begins to develop more quickly and completely than following conventional thinning. Variable-density thinning has all the positive effects of conventional thinning, such and increased growth of trees, crown differentiation, development of understory, and increased flowering and fruiting of understory plants (Harrington et al. 2002, Wender et al. 2004) that provide important ancillary foods to spotted owl prey (Carey 2000a) without the same extent of negative mechanical impacts, loss of canopy connectivity, loss of spatial heterogeneity, loss of woody plant diversity (variable-density thinning stresses multipspecies management).

Fire Suppression

Fires play different roles in different ecosystems (Franklin et al. 2002). Some forests and their fauna are well-adapted to fire—understory may be highly flammable, but quick to recover, and overstory trees may be quite fire resistant. This is true of the mixed-conifer forest of southwestern Oregon and northern California, where the old-growth is even more patchy and coarse-grained than the forests to the north, with the forest incorporating various evergreen hardwoods and hard-leaved shrubs especially supportive of dense woodrat populations. Forest to the north in western Oregon and Washington have increasing fire return intervals up through British Columbia where millennia might pass without catastrophic fire on some sites. Wind can be an important catastrophic disturbance in coastal forest, but intermediate disturbances due to wind, ice, snow, and disease may prove to be more important in forest developmental processes. East of the Cascades, forest historically appeared to have shorter, but spatially highly variable fire return intervals, often with frequent fires of low to moderate intensity. There, fire suppression has altered the ecology of the forests with fire-adapted understories of grasses, forbs, and low shrubs being replaced by flammable ladder fuels that may threaten catastrophic destruction of the forest when fire does occur. But eastside forests are diverse and conditions in dry site ponderosa pine (*Pinus ponderosa*) are too often generalized to other types. Furthermore, grazing and silviculture has compounded the changes in eastside forests (Graham et al. 1999). Franklin et al. (2002) point out the patterns in eastside forest are often misunderstood, with patches within late-seral forests interpreted as independent stands instead of part of the forest mosaic. The traditional forestry view of stands as homogeneous units of vegetation and the human tendency to reduce variability to one or two dimensions portend many management mistake eastside. Researchers in interior forests have found that approaches to managing forest for diversity and support of top avian predators, like the goshawk (*Accipiter gentilis*) (Reynolds et al. 1992) entail much the same approach adopted by researchers seeking to solve the spotted owl/spotted owl prey base dilemma in Westside forests (Carey et al. 1992, 1999a, b, 2003a,b). The same will likely prove true in management of spotted owls and spotted owl prey eastside—spatial heterogeneity (patchiness) may prove to be the key to restoration of forest health and low intensity fire regimes while retaining patches of complex forests that benefit owls and their prey.

Forest Management & Owl Prey

The complexity associated with management of forests, spotted owls, and owl prey requires those who are interested in conservation of spotted owls to step back and take a decentered view. The focus of conservation might best be centered on the dynamics of ecosystems and landscapes, not individual species (Franklin 1993). Single-focus management, especially with a short-term view, has repeatedly had unintended consequences and produced big surprises. Any single forest management activity has the potential to have negative or positive consequences for one or more or all spotted owl prey depending on how it is implemented and what other measures are taken concomitantly. Thus, well thought out integrated management systems are necessary (Bunnell et al. 1999; Carey et al. 1999b; Lindenmayer and Franklin 2002; Loehle et al. 2002; Muir et al. 2002; Carey 2003 a,b,c). Not only must the life history of keystone species be considered (Holmes and Austad 1994, Stapp 1994, Carey 2000a) but also must various aspects of biodiversity, including that of soil organisms (Amaranthus et al. 1989, Perry 1989, Tilman 1999), complex ecological processes such a decadence resulting from tree death (Franklin et al. 1987, Bunnell et al. 1999), multiple processes involved in forest succession and development (Canham et al. 1990, Carey et al. 1999a, Franklin et al. 2002), and spatial scale both within ecosystems and among ecosystems within landscapes (Carey et al. 1999a, Carey 2003a,b). Key variables in management that will determine the effects of forest management include degree of (1) legacy retention and conservation, (2) multispecies management (conifers, hardwoods, and shrubs), (3) precommercial thinning (as it relates to precluding competitive exclusion and fostering species diversity and crown differentiation), (4) inducing heterogeneity at the proper scale with variable-density thinning while maintaining canopy connectivity in some places and interrupting it in other places, (5) conserving and augmenting natural decadence processes, (6) restoration of biodiversity lost to single-purpose management, (7) extended rotations; (8) consideration given to geotechnical analysis in providing a template for legacy patch retention; (9) conservation of biodiversity as it relates to ecosystem resilience and capacity to adapt to changing fire conditions; and (10) ability to grasp the complexity of highly altered ecosystems and determine which of multiple alternative relatively stable states might be achievable in the long terms and what mix of these to pursue and maintain on the landscape. There is no one-size-fits-all or any canned prescriptions; diagnosis must be done watershed by watershed and prescription should follow diagnosis. It must be recognized that past management has had diverse effects on spotted owls and their prey; some second growth has abundant prey, some second growth is depauperate in prey and other species. No one has yet demonstrated successful intentional acceleration of development of diverse and abundant spotted owl prey or spotted owl habitat—too little time has passed since attempts to do so were begun.

APPENDIX 6: TABLES FOR REFERENCE TO CHAPTER SIX (HABITAT TRENDS)

Prepared by Richard Bigley

Table 6.1. Estimates of Old Conifer (>150 Years of Age) Forest by World Wildlife Fund Ecozone.

Ecozone	Ecoregion Area (ac)	Historic Old Conifer Area (ac)	Percent Historic Old Conifer Area	Current Old Conifer Area (ac)	Percent Current Old Conifer Area
Northern Cascades Forests	3,158,076	1,894,846	60	1,263,650	40
Cascade Mountains Leeward Forests	3,954,697	2,372,818	60	1,038,794	26
Puget Lowland Forests	4,249,443	3,399,554	80	262,294	6
Central Pacific Coastal Forests	10,546,198	7,909,648	75	1,651,322	16
Willamette Valley Forests	3,676,277	735,255	20	86,468	2
Central and Southern Cascades Forests	11,073,240	8,304,930	75	3,283,455	30
Eastern Cascade Forests	13,338,801	5,335,520	40	1,699.643	13
Klamath-Siskiyou Forests	12,436,990	6,218,495	50	2,338,540	19
Totals	62,433,724	36,171,066	58	11,624,177	19

Source : *Jim Strittholt (Personal communication)* World Wildlife Fund Ecozones are approximately equal to the NSO habitat provinces see Jiang at al. (2004) for a description of the WWF Ecozones.

Table 6.2. Published Estimates of Annual Percent Change in Stand Replacement Disturbance (both human-caused and natural).

Location	Periods	Public Land	Private Land	Total Land	Reference
Klamath-Siskiyou	1972-1992	- 0.25	- 0.42	- 0.53	Staus et al. 2002
Rogue Basin	1972-1992	- 0.36	- 0.46		Staus et al. 2002
Klamath Basin	1972-1992	- 0.48	- 0.96		Staus et al. 2002
Central Cascades, OR	1972-1988	- 1.20	- 3.90		Spies et al. 1994
Tillamook Basin, OR	1972-1992			- 1.00	Strittholt and Frost 1995
Hoh River Basin, WA	1975-1991	- 1.47	- 3.45		Turner et al. 1996
Western Oregon	1972-1995			-0.9	Cohen et al. 2002

Table 6.3. Change in habitat from 1994 to 2003 resulting from Federal management actions and natural events by physiographic province. Habitat additions to the Federal land base through land transfers and exchanges are not included as they represent a change in ownership rather than a physical change to habitat across the landscape. Source *USDI 2004*

Physiographic Province	Forest Plan baseline	CAUSES OF HABITAT LOSS				TOTAL
		Mgmt[1]	Fire[2]	Wind	Insect/ Disease	
Olympic Peninsula	560,217	-87	-299	0	0	-386
WA East Cascades	706,849	-5,024	-5,754	0	0	-10,778
WA West Cascades	1,112,480	-11,139	0	0	-250	-11,389
Western Lowlands	0	0	0	0	0	0
OR Coast	516,577	-3,278	-66	0	0	-3,344
OR Klamath Mountains	786,298	-53,468	-117,622	0	0	-171,090
OR Cascades East	443,659	-13,867	-4,008	0	-55,000	-72,875
OR Cascades West	2,045,763	-51,122	-24,583	0	0	-75,705
Willamette Valley	5,658	0	0	0	0	0
CA coast	51,494	-250	-100	0	0	-350
CA Cascades	88,237	-5,091	0	0	0	-5,091
CA Klamath	1,079,866	-12,673	-15,869	-100	-390	-29,032
TOTAL	7,397,098	-155,999	-168,301	-100	-55,640	-380,040

[1] Includes all updates submitted by the Federal action agencies.

[2] Fires occurring in 2003 were not included here as the data were not yet available.

Table 6.4. The percentage of Federal Forest Plan baseline affected by habitat loss within each province and across the range of the Northern Spotted Owl. Source *USDI 2004*

Physiographic Province	Forest Plan baseline	Total	Within Province loss		% of Range-wide Loss[1]
			Total % change	Annual rate of change (%)	
Olympic Peninsula	560,217	-386	-0.07	-0.01	0.10
WA East Cascades	706,849	-10,778	-1.52	-0.17	2.84
WA West Cascades	1,112,480	-11,389	-1.02	-0.11	3.00
Western Lowlands	0	0	0	0	0
OR Coast	516,577	-3,344	-0.65	-2.42	0.88
OR Klamath Mountains	786,298	-171,090	-21.76	-1.83	45.02
OR Cascades East	443,659	-72,875	-16.43	-0.41	19.18
OR Cascades West	2,045,763	-75,705	-3.70	-0.41	19.92
Willamette Valley	5,658	0	0	0	0
CA Coast	51,494	-350	-0.68	-0.08	0.09
CA Cascades	88,237	-5,091	-5.77	-0.64	1.34
CA Klamath	1,079,866	-29,032	-2.69	-0.30	7.64
TOTAL	7,397,098	-380,040	-5.41	-0.57	100

[1] The contribution of habitat loss within each physiographic province to the range-wide total loss of habitat.

Table 6.5. Distribution of habitat effects on Federal lands by state from 1994 - 2003. Source USDI 2004

State	Forest Plan baseline[1]	CAUSES OF HABITAT LOSS			**% state baseline**	% total loss
		Mgmt loss	**Natural events**	Total		
WA	2,379,546 (32%)	-16,250	-6,303	-22,553	-0.95	5.95
OR	3,797,955 (51%)	-121,735	-201,279	-323,014	-8.50	84.99
CA	1,219,597 (17%)	-18,014	-16,459	-34,473	-2.83	9.07
All	7,397,098	-155,999	-224,041	-380,040		

[1] Percentages in parentheses is the percent of the Forest Plan baseline habitat.

Table 6.6. Comparison of Federal habitat trends presented in the listing document to recent trends of habitat change due to Federal management activities. Source *USDI 2004*

	Listing Document[1]		This report
Management agency and state	Pre-listing period (about 1981 to 1990)[2]	Anticipated rates (about 1991 to 2000)[3]	Calculated rates[4] (1994 to 2003)
FS in WA and OR	64,000 (1.33)	39,400 (0.82)	10,341 (0.21)
FS in CA	Not reported[5]	4,700 (0.41)	1,653 (0.14)
BLM in OR	22,000 (2.35)	23,400 (2.50)	4,911 (0.52)
Total		67,500 (0.98)	16,905 (0.24)

[1] Habitat change values were presented in the listing document in units of acres per year, rather than as a percentage of total available habitat per year. We converted these values to annual percentage rates by dividing by the habitat amount in the Forest Plan baseline for each management agency and geographic group and multiplying by 100 (annual percentage rates in parentheses, indicating negative changes).

[2] Reported in the listing document as observed trends from 1981-1990.

[3] Estimated in the listing document as trends expected in the next decade (1991-2001).

[4] Annual acreage totals calculated as the sum of effects from 1994 to 2003 divided by 9 years of record. Annual percentage rates calculated as described above.

[5] The listing document references a rate of 12,000 acres of habitat loss per year in California, but it was unclear what time period this rate represented. Consequently, we did not include it here.

The USFWS compared Forest Plan baseline acreages of suitable Northern Spotted Owl habitat for each administrative unit to their local habitat baselines . The purpose of this comparison was to assess the potential for bias in evaluation of project effects. Local habitat baselines were not available for all administrative units within the Forest Plan area.

Table 6.7 Comparison of habitat estimated by the local habitat baseline and the Forest Plan baseline. Source USDI 2004

Administrative Unit	Subset of Forest Plan Baseline	Local Baseline	Difference from Forest Plan in acres	Percent Difference from Forest Plan
Mount Baker-Snoqualmie NF	581,447	408,750	-172,697	-29.7
Olympic NF	250,714	246,175	-4,539	-1.8
Wenatchee NF	540,626	927,402	386,776	71.5
Gifford Pinchot NF	497,491	510,000	12,509	2.5
Mt. Hood NF	568,488	419,791	-148,697	-26.2
Deschutes NF	144,932	123,135	-21,797	-15.0
Siuslaw NF	234,257	270,343	36,086	15.4
Willamette NF	767,001	740,053	-26,948	-3.5
Umpqua NF	501,390	432,880	-68,510	13.7
Rogue Basin Riv. Sis/Rog NF and Med. BLM))	913,497	1,060,728	147,231	16.1
Klamath NF[1]	407,803	479,763	71,960	17.7
Shasta Trinity NF[1]	266,409	372,621	106,212	39.9
Six Rivers NF[1]	341,716	379,522	37,806	11.1
Mendocino NF[1]	101,168	133,432	32,264	31.9
Salem BLM	150,605	149,544	-1,061	-0.70
Eugene BLM	106,425	81,440	-24,985	-23.5
Coos Bay BLM	115,207	118,580	3,373	2.9
Roseburg BLM	196,039	205,330	9,291	4.7
Totals	6,685,215	7,059,489	374,274	5.60

Sources USDI 2004

[1] Local habitat baselines include only nesting/roosting habitat from the California Baseline.

USDI 2004 Appendix 10: Comparison of Northern Spotted Owl Location Data with Suitable Habitat Maps for Washington, Oregon, and California

A comparison of available (1994 vintage) spotted owl location and suitable habitat maps were made to help respond to questions over how representative the 1994 NW Forest Plan suitable habitat baseline map is of actual spotted owl habitat on Federal lands. An additional analysis was done for Northern California using a more current (and updated 1998) habitat map, which should provide habitat estimates more comparable to Oregon and Washington. Spotted owl sites, located using survey protocols, were mapped as points in the GIS database. Because of questions about the accuracy of recorded site locations[1], the points were buffered in this analysis to allow for recording discrepancies up to ¼ mile (400m) around each plotted location. Given the inherent biases in both the site location and habitat datasets (see earlier discussion about habitat map accuracy), these results provide only a relative sense of the utility of the suitable habitat map.

Table 1: Percent of Known Northern Spotted Owl Locations Found within Mapped Suitable Spotted Owl Habitat (data from 1994 NWFP/FEMAT databases)

	Washington		Oregon		California[a]		Rangewide Totals	
Size/Buffer	Owl Sites	Percent	Owl Sites	Percent	Owl Sites	Percent	Owl Sites	Percent
Point	558	67%	1873	69%	388	40%	2819	63%
100M	658	79%	2228	82%	521	54%	3407	76%
200M	702	84%	2360	87%	584	61%	3646	81%
400M	751	90%	2462	91%	683	71%	3896	86%
Total Sites	833		2716		961		4510	

[a] – habitat maps used for FEMAT in 1994 were considered to under-represent suitable spotted owl habitat on National Forest lands within the four Northern California Forests (Thomas et al. 1990, USDI 1992).

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[1] Metadata shows that known spotted owl location points (1988-1995) were derived by Federal and State agencies from several agency databases with varying levels of accuracy. For example, most sites were located by field survey crews on 1:24,000 Quad maps (accuracy estimated at +/-250 feet) and then transferred by database staff to GIS databases. Few, if any, sites were located using GPS technology, but were based on the locator's estimation of the site center using maps, photos, and other materials. Depending on the complexity of the topography and distance from known locations and the accuracy of the underlying maps relative to known points such as roads, these could vary widely in point accuracy.

Table 2: Percent of Northern Spotted Owl Locations Found within Updated Suitable Spotted Owl Habitat Map for Northern California (habitat data from 1998 interagency database)

Size/Buffer	California[b]		Adjusted Rangewide Totals	
	Owl Sites	Percent	Owl Sites	Percent
Point	554	58%	2985	66%
100M	723	75%	3609	80%
200M	773	80%	3835	85%
400M	790	82%	4003	89%

[b] – in 1995 the USDA Forest Service (Region 5) and the USFWS initiated a mapping effort to update suitable habitat maps for the four Northern California Forests (completed in 1998; Zabel et al. 2003).

APPENDIX 7 ALTERNATIVE BASELINES CONSIDERED

7.1.1 LOCAL HABITAT BASELINES

Many National Forests, BLM Districts, and National Parks within the Forest Plan area have local vegetation databases and habitat maps that have finer resolution and local accuracy than the Forest Plan baseline. Most local baselines were developed by individual administrative units and have not benefited from review outside of the land management agencies that developed the evaluation methods. These “local habitat baselines” are used to assess evaluating project-scale effects and to assist in analyses for Land and/or Resource Management Plan revisions (e.g., National Forests and BLM Districts).

7.1.2 THE CALIFORNIA BASELINE

Significant revisions to the California baseline have been proposed. The current baseline represents habitat largely defined using data collected in Oregon and thus excludes some vegetation types (e.g., smaller tree size classes) and does not consider some physical attributes (e.g., aspect) important for defining suitable Northern Spotted Owl habitat in northern California (Zabel et al. 2003). Northern Spotted Owls in the Klamath Province often utilize younger stands then in other provinces. Zabel et al. (2003) suggests that Northern Spotted Owl habitat selection differs from that observed in the northern portion of the range. Consequently, the Fish and Wildlife Service and the Pacific Southwest Region of the Forest Service recognized the need for a habitat baseline that more accurately reflected Northern Spotted Owl distribution in the Klamath Province for both management and regulatory purposes.

Zabel et al. (2002) refined the habitat baseline for the Northern Spotted Owl on Federal lands in California. Products of this effort included among other accomplishments, a new map of owl habitat for five National Forests in northern California. The California baseline effort applied a standardized approach across multiple administrative units in California, incorporated a high level of interagency participation, and underwent external peer review. The revised baseline has not been accepted as the Forest Plan baseline for that area by the Federal Agencies.

7.1.3 INTERAGENCY VEGETATION MAPPING PROJECT

The Interagency Vegetation Mapping Project is an effort by the Forest Service to develop maps of existing vegetation for the entire range of the Northern Spotted Owl using satellite imagery from Landsat Thematic Mapper. Field data from inventory plots and photo interpretation of plots, among other sources of ancillary data, are being used to develop regression models to predict vegetation characteristics from the Landsat data.

The expectation is that the resulting maps will provide a new baseline evaluation of Northern Spotted Owl habitat that was developed using a consistent approach across the entire range of the species. The Interagency Vegetation Mapping Project holds promise for providing improved information about Northern Spotted Owl habitat in the near future. It is anticipated that the project will be published within a year.

7.2 EVALUATION OF LOCAL BASELINES

The USFWS collected information on the relationship between the Forest Plan and local habitat baselines (USDI 2001; 2004). Local baselines are developed by individual administrative units and act as the basis for the evaluation of individual projects that are evaluated under section 7 consultations.

Across the range of the Northern Spotted Owl, these local habitat baselines contain approximately 374,000 acres more of Northern Spotted Owl habitat than estimated for the Forest Plan (7,059,489 acres versus 6,685,215 acres). Significant disparities exist between some local baselines and estimates of suitable habitat from the Forest Plan. The overall comparison of total habitat estimated by the local habitat baseline and the Forest Plan baseline obscures the wide variance in the deviation of each local habitat baseline from the Forest Plan baseline (Appendix 2 Table7). Concerning these difference the USFWS (USDI 2004:page number) stated:

> "Some administrative units have disproportionate influence on the outcome of the range-wide summary. For example, the local habitat baselines for eight administrative units show less habitat than estimated for the Forest Plan baseline. Taken together, lower baseline habitat estimates reported by these eight administrative units were lower than the Forest Plan baseline estimates by approximately 470,000 acres, with the Mount Baker Snoqualmie National Forest contributing about 172,000 acres (37 percent) to the overall difference. Twelve administrative units reported local baseline estimates that were greater that the Forest Plan baseline by a total of 844,000 acres, with the Wenatchee National Forest contributing 387,000 acres (46 percent) to the overall difference."

Since the USFWS depends on USFS for primary data, they cannot document the disparities

The USFWS offered a rationale for not using local baselines as a basis for regional comparisons. They make a logical argument that these local habitat baselines are unsuitable for broad-scale analyses due to several factors that limit the potential for aggregation in a range-wide summary. The procedures used to develop local habitat baselines varied with local information availability and needs, which in turn reflect administrative boundaries unrelated to biological differences. The same argument can be made for the Forest Plan baseline, however the Forest Plan baseline was designed to provide a regional perspective and with influence from the local baselines of the time. The specificity that is possible at the local scale may have little significance once effects are at the scale at which an understanding of a species' overall condition or status needs to be assessed because of averaging.

It is to be expected that local habitat definitions used by different administrative units also vary across the range of the owl (USDI 2004 Appendix 3). These differences may reflect biological differences in habitat use, however, methodological and subjective factors may also mask or exaggerate biological differences. Although some local baselines may have been subjected to rigorous validation, the procedures for validation have not been standardized. Thus, there is a large amount of variability among local baselines, which lowers our confidence in the information.

The Forest Plan baseline has remained constant while some local habitat baselines continue to evolve. As new technology has become available, local administrative units have continued to refine their estimates of suitable habitat. Unfortunately, changes in methods over time make it difficult to evaluate what habitat trends are due to differences in methods versus actual changes to habitat that have occurred. The USFWS notes (USDI 2004)

> "many of the local habitat baselines now in use across the range of the Northern Spotted Owl were developed after the Forest Plan baseline, shortening the time period that rates of change calculated using these baselines would represent. As seen with the revised California baseline, local habitat baselines generally have not been formally accepted at regional management levels of the land management agencies."

It is more effective to delineate suitable habitat at the local scale, because most local habitat baselines are developed using aerial photo interpretation. However, such photo interpretation may or may not have field verification. In theory, local baselines have the potential to increase site specificity and relation to the habitat that is actually being used by the owl (e.g., Klamath province habitat revision). However, in order for this to occur, a coordinated effort to understand the interface between local and provincial baselines must be undertaken. IVMP products have the potential to be validated on the local and provincial levels and will serve the need.

7.2.1 EVALUATION OF CALIFORNIA BASELINE

The USFWS recognized revisions to the California baseline, but cited several shortcomings on using the California Baseline as a reference condition. They determined that although the California baseline may be very useful for predicting owl presence or absence across the landscape (Zabel et al. 2003), it is not useful as a reference baseline condition against which to evaluate temporal and spatial changes in habitat range-wide.

The California baseline does not include suitable habitat on BLM lands in California and therefore does not provide a seamless habitat layer across all Forest Plan lands in California. Further, the California Baseline covers only a portion of the California range, which introduces discontinuity between similar habitat types in California and Southern Oregon. The California baseline was developed using different methods than those broadly applied across Oregon and Washington by not allowing a consistent reference point against which to evaluate changes in habitat conditions range-wide.

The California baseline was completed in 1999, and, therefore, only allows for examination of habitat trends over four years (1999-2003). Projects completed before 1999 may have been accounted for in the new baseline, but we were unable to discern if this were true. The difference in time frames between the California baseline and the Forest Plan baseline used in Oregon and Washington could make the California Baseline inconsistent given the nine years of forest change activities in Oregon and Washington. Lastly, the California baseline has yet to be adopted formally by the Forest Service as the revised habitat baseline for the Northern Spotted Owl in California.

APPENDIX 8 SUDDEN OAK DEATH

Prepared by J. F. Franklin

Sudden Oak Death (SOD) is a forest disease caused by the fungus-like pathogen, *Phytopthora ramorum* that was recently introduced from Europe. At the present time SOD is found in natural stands from Monterey to Humboldt Counties, California, and has reached epidemic proportions in oak and tanoak forests along approximately 300 km of the central and northern California coast (Rizzo et al. 2002a). It has also been found near Brookings, Oregon, killing tanoak and causing dieback of closely associated wild rhododendron and evergreen huckleberry (Goheen et al. 2002). It has been found in several different forest types and at elevations from sea level to over 800 m.

SOD is continuing to spread. Substantial transport of the pathogen within the Pacific Northwest and the North American continent has occurred as a result of the movement of infected nursery stock, the means by which it was originally introduced from Europe where it originated. Much of the following description of the organism and its effects comes from the web site of the California Oak Mortality Task Force (www.suddenoakdeath.org).

8.2 CHARACTERISTICS AND HOSTS OF SUDDEN OAK DEATH

SOD is currently known to infect a wide variety of herb, shrub, and tree species native to the Pacific Northwest in the form of trunk, twig and foliar infections (Rizzo et al. 2002b). Many species have exhibited only relatively benign foliar infections up to this point but tanoak (*Lithocarpus densiflorus*) and California black oak (*Quercus kelloggii*), among others, sustain lethal stem infections (Rizzo et al. 2002a). Species that are infected include: bigleaf maple (*Acer macrophylllum*), Pacific madrone (*Arbutus menziesii*), tanoak, Douglas-fir (Davidson 2002), Canyon live oak (*Quercus chrysolepis*), California black oak, Pacific rhododendron (*Rhododendron macrophyllum*) plus many other rhododendron and azalea species, wood rose (*Rosa gymnocarpa*), coast redwood (*Sequoia sempervirens*), western starflower (*Trientalis latifolia*), California bay laurel (*Umbellularia californica*), evergreen huckleberry (*Vaccinium ovatum*), grand fir (*Abies grandis*), California hazelnut (*Corylus cornuta*), salmonberry (*Rubus spectabilis*), cascara (*Rhamnus purshiana*), and poisonoak (*Rhus diversiloba*). Additional species are being added to the list nearly daily and may ultimately include many other plants native to forests occupied by Northern Spotted Owls since members of the Ericaceae, Rosaceae, Taxaceae, Taxodiaceae, and Pinaceae have all shown vulnerability.

SOD has caused widespread dieback of tanoak and several oak species in the central and northern coastal counties of California as a result of aggressive lethal bark infections (cankers) (Rizzo et al. 2002a). Tree death appears to occur when cankers expand in the trunk effectively girdling the tree and disrupting physiological function. Diseased trees are often attacked by other pest organisms, such as fungi that decay sapwood (*Hypoxylon thourasianum*) and bark beetles. In shrub species, symptoms can range from leaf spot to twig girdling, which do not necessarily result in the death of the plant.

Sudden Oak Death is so named because the whole crown of many affected trees appears to die rapidly with the foliage turning from a healthy green to brown over several weeks. The time from initiation infection to tree death may actually range from several months to several years. ***Tanoak appears to be the most susceptible species.*** All size classes from seedlings to large trees may be infected and killed (Rizzo et al. 2002a). A large number of opportunistic organisms are commonly observed on oak and tanoak trees and may hasten tree death. SOD infections also kill large trees of Canyon live oak and California black oak. SOD causes branch cankers and death of new shoots and small branches on Douglas-fir and coast redwood and death of sprouts of redwood; the long-term impacts of SOD on saplings and trees of Douglas-fir and coast redwood are unknown at this time. Death of Pacific madrone saplings has been observed and it is suspected that SOD can kill mature madrone trees.

Many of the species with foliar infections play a key role in spread of SOD by providing a reservoir of inoculum, which spreads aerially via wind-blown rain. Sporangia and chlamydospores are the most likely dispersal propagules and are generated on foliage. Two taxa known to provide massive foliar sources of inoculum are California bay laurel and rhododendron spp. (Davidson, Rizzo, and Garbelotto 2002).

APPENDIX 9 ANALYZING DATA ON BARRED OWL EFFECTS

Prepared by A. B. Franklin

Examples of Alternative Analyses evaluating the Effects of Barred Owl Presence on Trends in Territory Occupancy by Northern Spotted Owls in Redwood National Park (data from Appendix B in Schmidt 2003).

The purpose of this Appendix is to provide examples of 1) analyses that illustrate the problem in inferring that Barred Owls are replacing Spotted Owls (i.e., having a negative impact) when the inference is based solely on cumulative occupancy of sites, rather than annual occupancy, and 2) how different covariates can yield different results, and hence, provide different inferences.

We feel that more estimates (or inferences) should be based on *annual occupancy* (Figure 7A.1) because there may actually be a higher occupancy of the Spotted Owl territories than Barred Owl territories if Barred Owl territories are based almost solely on detections at night and not on roost and nest locations. We analyzed the data available to us from Schmidt (2003) where we had to assume that detectability was constant over time. This is a difficult assumption to meet so we present the analysis of these data as an example, rather than as a definitive analysis. Moreover, we do not know if these data are complete or if there are any other properties associated with the sampling design or field procedures that might also affect the results. A more appropriate analysis would use the occupancy estimators developed by MacKenzie et al (2003). ***A key point is that we are not trying to make inference from the data in Schmidt (2003) but are merely using it as an example***. Further, we consider the data in Appendix B of Schmidt (2003) to be proprietary to the biologists gathering this data, and thus feel it is their purview to fully explore the data they gathered post *Tanner (1999)* for future publication.

In our examples, there have been 36 Spotted Owl territories identified in Redwood National Park since 1993. Over the ten years from 1993-2002, 18 (50%) of these territories have had Barred Owl detected in them. The correlation of Barred Owl detections in Spotted Owl territories coupled with the apparent decline in the occupancy of these historic Spotted Owl territories could be improperly inferred as cause-and-effect. However, examination of the data on an annual basis (Table 7A.1), shows that only 6-20% of the territories have had Barred Owl detections in any given year, and some of these detections have been simultaneous with Spotted Owl detections in the same territory in the same year. In order to evaluate whether the decline in occupancy was related to Barred Owls we modeled the data from Appendix B of Schmidt (2003) using an information-theoretic approach.

Example 1. In the first example, we used annual number of Spotted Owl territories with Barred Owl detections as a covariate of Barred Owl presence (Table 9A.1). We then examined the trend in the proportion of Spotted Owl territories occupied each year using generalized linear models. We examined three time trends in the annual proportion of Spotted Owl territories occupied, a linear time trend (year), a log-linear time trend (lnyear), and a quadratic time trend (year*year). We also examined time trends using the Barred Owl covariate (BO) and no time trend (intercept). Thus, there were three hypotheses examined: 1) There was a time trend (either year, lnyear, or year*year models) in the annual proportion of territories occupied by Spotted Owls with no effects of Barred Owls, 2) the trend in annual proportion of territories occupied by Spotted Owls was due to Barred Owl presence in Spotted Owl territories (BO model), or 3) there was no discernible change in the annual proportion of territories occupied by Spotted Owls over time (intercept). Of the five models, the most parsimonious model was a log linear decline (Table 9A.2) which had the lowest AIC$_c$ and more than 60 percent of the Akaike weight was attributable to that model. This model indicated that the annual number of occupied Spotted Owl territories was declining ($\hat{\beta}$ = -0.302, 95% CI = -0.381, -0.222) and explained 84.8% of the variation in Spotted Owl occupancy (Table 9A.2). The Barred Owl effect model had essentially no Akaike weight (i.e., provided no explanatory power for the decline), which indicated that Barred Owl detections in Spotted Owl territories did not explain the negative trend in Spotted Owl occupancy. Although negative, the Barred Owl effect was not different from zero, based on 95% confidence intervals ($\hat{\beta}$ = -0.0134. 95% CI = -0.0374, 0.0105). This model explained only 5.1% of the variation in the Spotted Owl occupancy data. These results do not mean that another Barred Owl covariate, such as number of Barred Owls in the park (regardless of whether they were in Spotted Owl territories) would have had better explanatory power. However, evaluating cumulative numbers of Barred Owls occupying Spotted Owl territories suggests an impact on Spotted Owls, whereas this analysis shows that Barred Owls were not a plausible explanation for the decline in occupancy of territories by Northern Spotted Owls.

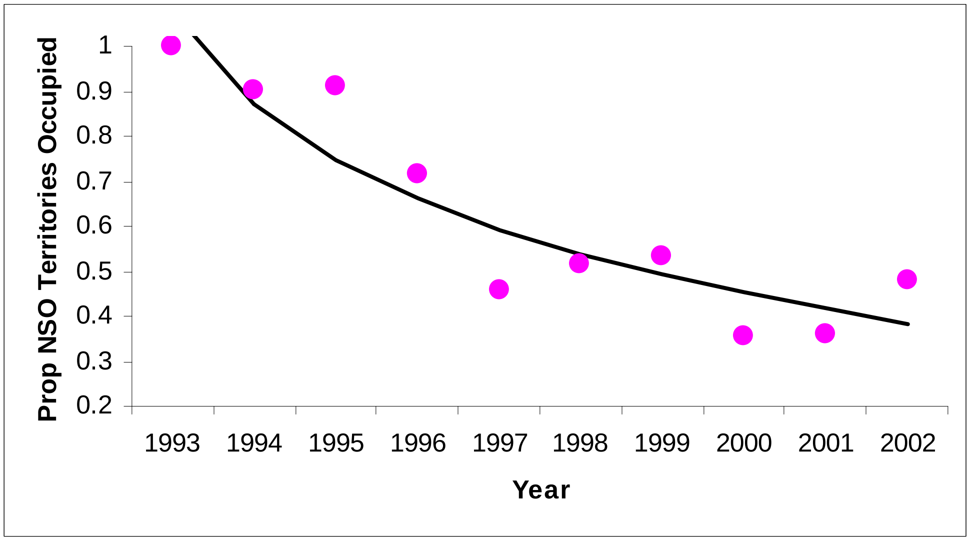

Figure 9A.1. Proportion of Northern Spotted Owl territories occupied annually. Solid line represents log-linear trend model {lnyear} selected based on minimum AICc.

Table 9A.1. Annual occupancy data from Redwood National Park used to analyze time trends and effects of Barred Owl detections on Spotted Owl occupancy (from Schmidt 2003 Appendix B).

Year	Prop. Territories occupied by Spotted Owls (OCC)	Prop. Spotted Owls with Barred Owl detections (BO)
1	1.000	0.222

2	0.903	0.065
3	0.909	0.121
4	0.714	0.057
5	0.457	0.086
6	0.514	0.114
7	0.533	0.133
8	0.355	0.194
9	0.361	0.194
10	0.481	0.222

Table 9A.2. Model selection results from analysis of proportion of territories occupied by Spotted Owls in Redwood National Park. The selected model {lnyear} based on minimum AICc is bolded

Model	-2log*L*	*K*	AIC	AICc	ΔAICc	Akaike Weight	R2
Year	-17.416	3	-11.416	-7.416	2.611	0.170	0.802
Lnyear	**-20.028**	**3**	**-14.028**	**-10.028**	**0.000**	**0.626**	**0.848**
Year^2	-23.780	4	-15.780	-7.780	2.248	0.204	0.895
BO	-1.738	3	4.262	8.262	18.290	0.000	0.051
Intercept	-1.211	2	2.790	4.503	14.531	0.000	0

Example 2. In evaluating Example 1, *S. Gremel* (*personal communication 2004*) suggested that a more appropriate covariate would be to categorize Spotted Owl sites as to whether they had ever had at least one Barred Owl detection during the period of data collection (BO+) and then to examine those sites separately from Spotted Owl territories that never had Barred Owl detections during the period of data collection (BO-). This response variable was the mean number of Spotted Owls surveyed per site for each year in the two different categories. Although Gremel also presented a re-analysis of these data through 2003, we present here a modified analysis that includes only 2002 to make it comparable with our Example 1. We also modified the analysis proposed by Gremel by using an analysis of covariance approach under the information-theoretic framework used in the first example. Although the covariate used cumulative information on Barred Owls in Spotted Owl sites, the analysis still examined annual rates rather than the cumulative number of sites affected over a given time period. The data used in this analysis are presented in Table 9A.3.

Table 9A.3. Mean number of Spotted Owls/surveyed site with (BO+) and without (BO-) Barred Owls on Redwood National Park from 1993-2002.

	Mean number of Spotted Owls/surveyed site	
Year	BO+	BO-
1993	2.00	1.57

1994	1.44	1.87
1995	1.40	1.85
1996	1.00	1.60
1997	0.60	1.00
1998	0.70	1.20
1999	0.75	0.86
2000	0.44	0.85
2001	0.43	0.80
2002	0.56	1.18

We examined five simple models, which were: 1) no time or Barred Owl effects (Intercept), 2) Barred Owl effects but no time trends (BO), 3) linear time effects but no Barred Owl effects (year), 4) a Barred Owl effect with a linear time effect that was the same for BO- and BO+ categories (BO+year; e.g., an additive effect of Barred Owls on time), and 5) a Barred Owl effect with a linear time effect that was different for BO- and BO+ categories (BO*year; e.g., and interaction between the Barred Owl effect and time). From this set of models, model {BO+year) was selected as the best model based on minimum AICc (Table 9A.4). This model had 76.3% of the Akaike weight, which was more than 3 times more likely than the next ranked model {year) with an Akaike weight of 21.9%. The selected model also explained 75.4% of the variation in the data (Table 7A.4).

Table 9A.4. Model selection results from analysis of Spotted Owl territories classified as having Barred Owl presences versus those where Barred Owls were absent in Redwood National Park. The selected model {year+BO} based on minimum AICc is bolded

Model	-2log*L*	*K*	AICc	ΔAICc	Akaike Weight	R2
Year	7.799	3	17.799	2.496	0.2189	0.624
BO+year	-0.697	4	15.303	0	0.763	0.754
BO*year	-2.247	5	22.753	7.45	0.018	0.772
BO	24.558	3	34.558	19.256	0	0.13
Intercept	27.349	2	33.063	17.76	0	0

Model {BO+year} had a negative time trend for both BO+ and BO- sites ($\hat{\beta}$ = -0.1318, 95% CI = -0.168, -0.096) and the detection of a Barred Owl in a Spotted Owl territory at least once during the study period appeared to have a negative effect on the mean number of Spotted Owls per site ($\hat{\beta}$ = -0.346, 95% CI = -0.554, -0.138) (Figure 7A.2).

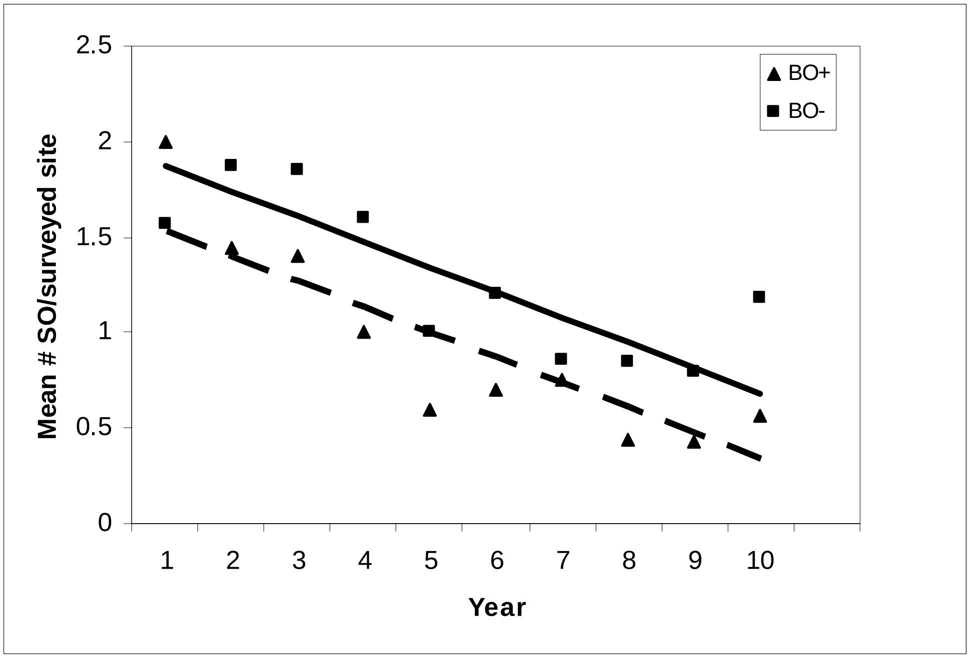

Figure 9A.2. Mean number of Northern Spotted Owls per site in sites where Barred Owls were detected at least once during the study period (BO+) and where Barred Owls were never detected (BO-). Solid and dashed lines represent trends from model {BO+year) for BO+ and BO- sites, respectively.

However, the presence of a Barred Owl effect in this analysis does not differentiate between the hypotheses that 1) Barred Owls are the effect or 2) BO+ sites were of lower habitat quality for Spotted Owls, which experienced inherently lower occupancy than BO- sites and that Barred Owls merely replaced Spotted Owls on the BO+ sites. A designed experiment would be needed to further differentiate between these two hypotheses (see Information Needs section). Thus, this example illustrates how a retrospective observational study could be used to set up a designed experiment, such as removing Barred Owls from the BO+ sites and seeing if the BO effect disappears.

Conclusions – In the two analyses presented here, we attempted to appropriately analyze the available data on an annual basis to illustrate why we think analyses of annual trends are more meaningful than cumulative trends. However, the major problem with these example analyses is that the data available to us were incomplete. At some future date, these data might be analyzed more completely by the owners of this data. Specific issues related to the basic problem include:

1 *Lack of information to quantify detectability* – Both analyses assume either complete detectability or constant detectability over time for both Barred and Spotted Owls. These assumptions are rarely met in wildlife population studies. The use of occupancy estimators that also account for detectability (e.g., MacKenzie et al. 2003) would be more appropriate for these data but require the within-year survey information, which was unavailable to us. Recent occupancy estimators allow for inclusion of two species (D. Mackenzie (personal communication), which would be ideal for examining the effects of Barred Owls on Spotted Owl territory occupancy.

2 *Lack of sampling variance* – Inclusion of annual sampling variances in the analyses would affect the estimates of effects and their standard errors. In the two examples, we ignored the presence of sampling variance, the inclusion of which will probably affect the ranking of the models and their estimates.

Although no inferences can be made from our two examples, two important lessons can be learned from this exercise:

1. We feel it is very important to be cautious about inferences derived from analysis using cumulative to assess population size and trends in Barred Owls and, more importantly, their effect on Spotted Owls. In retrospective analyses of the effects of Barred Owls on Spotted Owls, we believe more emphasis should be placed on annual trends rather than the cumulative numbers of sites where Barred Owls are detected.

2. Covariates used to estimate the effect of Barred Owls on Spotted Owls should be chosen with care and, ideally, should be developed with a consensus among scientists (e.g., see Anderson et al. 1999). In our two examples, different results may have been largely due to differences in the covariate used. A forum similar to the recent Spotted Owl meta-analysis would be an appropriate venue for determining the appropriate direction to take across a number of Spotted Owl studies that have relevant data on Barred Owls.

APPENDIX 10 DEVELOPING RECOVERY STRATEGIES FOR NORTHERN SPOTTED OWL POPULATIONS

By B. R. Noon

Commissioned by SEI for Northern Spotted Owl Status Review

Assessment of status and trends in population size, survival rates, and reproduction are succinctly summarized by time-specific estimates of λ_t, the finite rate of population change. Such assessments are typically retrospective—that is, they estimate how these parameters have changed from the initiation of a demographic study to the time of assessment. A description of what has occurred in the population is summarized by the time series of parameter estimates. Projections to the future status of the population are either not made or done very cautiously when evidence for temporal trends have been found.

Equally valuable are prospective analyses that project how various demographic rates are likely to change in the future under plausible land-use and environmental conditions. These analyses are inescapably less certain than retrospective analyses because the future is never known until it arrives. However, the endangered species evaluation and recovery process is inherently a type of risk assessment and thus requires prospective analyses (NRC 1995, Goodman 2002, Ralls et al. 2002). To develop a recovery plan strategy requires one to project the future consequences on the listed species of alternative management practices and conservation actions. Such evaluations logically fall until the broad category of population viability analysis (PVA) in that they project changes in population status given specific changes in one or more environmental variables. PVAs have traditionally focused on estimates of persistence likelihoods or times to extinction (e.g., Foley 1994). However, it is important to view PVA in a much broader context—that is, as an analytical tool to evaluate how resource management can change parameters influencing the probability of spotted owl persistence (Boyce 1992, Noon et al. 1999, Shaffer et al. 2002).

The critical parameter estimates required for informative PVAs have been thoroughly discussed by Boyce (1992), Noon et al. (1999), and White (2000). As discussed in the current assessment, reliable parameter estimates are available for most northern spotted owl populations and PVAs are justified given important caveats. Most important is that any population projections incorporate those factors that drive variation in birth and survival rates and include factors amenable to management intervention. Since causal relationships are still poorly known, future conservation actions should be conducted as large-scale manipulative experiments (Noon and Franklin 2002).

The history of demographic studies of northern spotted owl populations has been a combination of retrospective and prospective analyses. The emphasis in previous assessments has been on population dynamics with a particular focus on the estimation of λ. In all previous assessments, the owl researchers have taken great care to point out that estimates of λ are specific to the time and place in which they are estimated. Therefore, projections of λ (or its components) require an assumption of similar future conditions or a mechanistic understanding, or set of hypotheses, about future environmental conditions.

A Brief Historical Review

Initial assessments of the status of northern spotted owl populations used a hypothesis testing framework (Thomas et al. 1990, Murphy and Noon 1992). Three hypotheses were tested: 1) the finite rate of change (λ) is ≥ 1.0, 2) owls do not differentiate among habitats on the basis of forest age, structure, or composition, and 3) no decline has occurred in the areal extent of habitat types selected by owls.

Following an eigenanlysis of the stage projection matrix, the first null hypothesis was originally rejected based on the observation that λ was < 1.0 from two demographic study areas (Thomas et al. 1990). A 1995 reanalysis of demographic data from 11 study areas resulted in a more convincing rejection of this hypothesis (Burnham et al. 1996). At the time of this reanalysis, however, concerns were being expressed that estimates of λ may be biased low because of an underestimate of juvenile survival rate (Bart 1995).

The second null hypothesis addressed the question of whether the owl uses the forested landscape in the Pacific Northwest in a non-random fashion. At the time of listing, all of the northern spotted owl habitat studies concluded that owls select old forests, or younger forest that have retained characteristics of old forests, for nesting and roosting. Many studies published since the listing decision provide additional falsification of hypothesis 2. These studies were reviewed as part of the Northwest Forest Plan (NWFP) process (FEMAT 1993) and updated by Noon and McKelvey (1996).

The rejection of hypothesis 2 leads logically to a test of hypothesis 3. Based on data from National Forest lands in Oregon and Washington, Thomas et al. (1990) found significant declines since 1940 in the extent of owl habitat, a trend that was projected to continue into the future (Murphy and Noon 1992). Additional data since 1990 provided evidence of declines in California (McKelvey and Johnston 1992) and more regionally specific estimates of decline were reported in the draft Northern Spotted Owl Recovery Plan (USDI 1992).

Rejection of the three fundamental hypotheses listed above were fundamental to the listing of the northern spotted owl listing under the Endangered Species Act and of the initial development of the Northwest Forest Plan (FEMAT 1993). Landscape allocations adopted by the NWFP was based on an algorithm that focused on the location, size, shape, spacing, and context of current and regenerating late-successional forest patches planned for inclusion in a spotted owl reserve system. The goal of the design was to establish locally stable owl populations, widely distributed throughout their historic range. Even though suitable habitat was projected to decline outside of the reserves for several decades (FEMAT 1993), habitat loss within the reserves was projected to stop and the process of renewal to begin.

Role of Models. Models often serve as useful tools for prospective analyses because they are a means to project potential outcomes of alternative future states of the environment. In this context, models played a significant role in the development of the current conservation strategy for the northern spotted owl.

Model development and analyses progressed from simple to complex as more information became available. Initial analyses, focused on the Leslie projection matrix, explored life history sensitivities of spotted owls. Based on eigenanalysis methods, adult female survival was identified as the key demographic rate that most influences population growth (Lande 1988, Noon and Biles 1990). This insight contributed to the design of the current monitoring studies and led to an emphasis on obtaining precise and unbiased estimate of adult survival rates by using modern capture-recapture methods (Franklin et al. 1996).

An important area of uncertainty following rejection of the three hypotheses was the projected response of owl populations to continuing declines and fragmentation of suitable habitat. In this regard, a simple model developed by Lande (1987) for territorial species with obligate juvenile dispersal—the case for spotted owls—was deemed particularly relevant. This model predicted sharp, non-linear persistence thresholds as habitat was lost and fragmented. A variant of Lande's model, parameterized specifically for Northern Spotted Owls, suggested that an extinction threshold was being approached in the Pacific Northwest (Lamberson et al. 1992). The extinction threshold was attributable to two factors—the lost and fragmentation of habitat and the difficulty of a dispersing owl in finding suitable habitat and a mate.

Because the existing conservation literature and biogeographic principles were too broad for specific application, models were also used to refine the reserve design principles of the NWFP (Lamberson et al. 1994). These models suggested that persistence likelihood (as measured by the occupancy rate of territories) asymptotically increased as individual patch size increased to ~ 20 breeding pairs of owls. In addition, occupancy rates remained high if distances between patches were within 19 km of each other (intersecting the dispersal range of the majority of dispersing owls) and patch density was high. The models of Lamberson et al. (1992, 1994) could be considered a type of PVA since territory occupancy rates were a direct proxy variable for persistence likelihood. Collectively, these models suggested that long-term persistence required ~ 20% of the forested landscape to be maintained as suitable habitat with habitat arranged in patches of ≥ 20 pairs of owls connected by dispersal.

These initial models contributed significantly to the design of the NWFP. In retrospect, however, it was clear that these models were overly simplistic and based on several optimistic assumptions. These included no environmental stochasticity, optimal reserve shape (circular), no loss to sink habitats, and forest matrix conducive to dispersal, and 100% suitable habitat within reserves.

Even though the initial models of Lamberson et al. (1992, 1994) provided a plausible set of rules controlling the size and spacing of reserves, the actual landscape was highly constrained by geography, past land-use practices, and land ownership. Therefore, during the early stages of development of the NWFP work began on a new owl model designed to directly incorporate "real" habitat maps through a GIS interface. This model, a habitat-based population dynamics model, was spatially explicit, dynamic (it modeled landscape change and owl dispersal), and allowed investigation of the effects of individual heterogeneity (based on life stage and habitat quality) on owl population dynamics (McKelvey et al. 1993, Noon and McKelvey 1996).

The McKelvey model is initialized by intersecting the forested landscape with a hexagonal grid with cell size approximating the median size of an owl home range. Expected birth and survival rates at the scale of an individual cell are related to habitat attributes by a series of regression equations (e.g., Bart 1995). (Based on multiple studies, the amount of mature forest > 120 years old proved to be the strongest predictor variable). These functions provide initial estimates of the demographic rates for the current landscape. When the model is combined with timber harvest schedules, post-harvest recovery rates, and habitat quality functions it is possible to compare competing land management plans in terms of owl viability.

After the FEMAT team had defined the various land management options it was considering for adoption, the McKelvey model was used to evaluate several alternatives including one proposed by the FWS Recovery Plan. Given identical rules concerning initial habitat conditions and assuming no regrowth of owl habitat over the evaluation interval, the options diverged greatly in terms of both the expected number of owls and their distribution across the landscape (Noon and McKelvey 1996). In the end an option was selected that represented a compromise between maximizing owl viability, the viability of other species of concern, and competing economic interests.

Subsequent modeling efforts (Akcakaya and Raphael 1998, Hof and Raphael 1997) have not added greatly to our understanding of the factors putting spotted owls at risk or how to diminish those risks. In general, models of differing structure and invoking various assumptions have been consistent in recommending sizeable patches of habitat to support largely self-sustaining local populations connected by frequent dispersal events. In addition, there needs to be substantial redundancy (i.e., many large patches widely distributed throughout the range of the owl) because of strong spatial autocorrelation in the climatic events that affect northern spotted owl populations.

Perspectives on the Current Status and Trend

It is insightful to consider the current status and trend assessment in terms of the original three null hypotheses. Available data clearly indicate that hypotheses one and two would still be rejected. The current status review confirms that most owl populations are still in decline. In addition, habitat studies published since the review of Noon and McKelvey (1996) continue to demonstrate the association of owl nesting and roosting with late-successional forests (e.g., Franklin et al. 2000, Thome et al. 1999, Meyer et al. 1998, Ward et al. 1998). The decision on hypothesis three is less clear than in 1990. Since enactment of the NWFP, timber harvest rates on federal public lands have declined substantially with rates of harvest since 1994 averaging < 1% per year. Harvest rates on private and state lands within the range of the northern spotted owl are poorly known but it is probably safe to assume that they are greater than on federal public lands. In addition, suitable owl habitat has been loss since 1990 as a consequence of large, stand-replacing fire events (Chapter 6 of this review).

In summary, based on current population trends and habitat conditions it appears that the conditions that led the FWS to list the spotted owl as threatened in 1990 are still relevant today.

What are the expected population trends of spotted owls approximately a decade after enactment of the NWFP? Thomas et al. (1990) argued that population trend should stabilize at a lower equilibrium size sometime within the next 100 years. During the interim there was an expectation that the rate of decline would slowly decrease as habitat loss was arrested and new habitat regenerated in the habitat conservation areas. Two critical assumptions of Thomas et al. (1990) were that a case of no-net-loss of suitable habitat would be achieved prior to crossing an extinction threshold and that the conservation areas would eventually be fully occupied by owls (Murphy and Noon 1992). Current data on habitat trends suggest that the first assumption is approximately true on federal public lands. The second is probably false because of mixed ownership of many designated reserves and because of natural disturbance events.

It is possible that we are observing the transient dynamics of populations that are in the process of recovery but this is highly uncertain. Unfortunately, the most recent meta-analysis (Anthony et al. 2004) does not allow one to discriminate between the two key, opposing hypotheses—that is, 1) owl populations are slowly declining to a new, positive equilibrium, versus 2) owl populations have crossed a threshold and are slowly declining to extinction.

Future Strategy

As stated previously, recovery planning under the Endangered Species Act requires some sort of PVA to evaluate the likely outcomes associated with alternative conservation strategies (NRC 1995). To be beneficial, any viability modeling should be based on time horizons of a few decades (Goldwasser et al. 2000) and to the extent possible closely follow the guidelines proposed by White (2000): 1) be based on a realistic population model, or set of competing models, incorporating unbiased parameter estimates, 2) include spatial variation among local populations, 3) compute the distribution of persistence likelihoods based solely on estimates of the process variation (demographic and temporal) in demographic rates, and 4) incorporate individual heterogeneity in the demographic rates. In addition, a useful PVA for the purposes of recovery planning must include functions that relate the expected value of demographic rates (i.e., birth and survival) to key environmental drivers such as specific habitat elements, landscape patterns, and climatic variables (e.g., Franklin et al. 2000).

To initiate conservations action to accelerate the recovery of northern spotted owl populations requires a mechanistic understanding of the factors that affect λ. For the most part, these factors are poorly known (Noon and Franklin 2002). In addition, there is the strong possibility that the controlling factors vary among geographic locations. An appropriate framework for advancing understanding is to synthesize existing knowledge of plausible causal relationships in the form of predictive models. Inclusion of environmental drivers can be viewed as a multiple regression function in which the dependent variable is λ (or a given demographic rate) and the independent variables are various environmental factors. Independent variables can have positive or negative effects with effect size given by their regression coefficient. Disagreement over the factors to include in the model, or the size of the coefficients, can be viewed as competing models. In addition to the usual sources of uncertainty which accompany stochastic modeling, the inclusion of causal functions is accompanied by added uncertainty because: 1) the true relationship between environmental factors and the expected value of demographic rates may be poorly known (Noon and Franklin 2002), 2) the future value of the environmental factors is unknown, and 3) population outcomes associated with changes in multiple environmental factors are complex.

Because of numerous sources of uncertainty, the recovery planning process should be viewed in an active adaptive management context (sensu Walters 1986). The process is termed ‘active’ because the system is actively perturbed via experimentation (Walters and Holling 1990). Uncertainty or disagreement over what environmental variables are most relevant to future changes in spotted owl demographic rates would be addressed in the form of competing viability models. These models would make differing predictions over how owl populations would respond to changes in these variables. The degree of fit between prediction and observation would be used to discriminate among competing models and to update model structure and parameter estimates.

A test of competing recovery strategies could be implemented in the context of the current owl monitoring program. This would require the conduct of large-scale manipulative experiments across the different monitoring sites with a different set of variables changed at different sites in order to bracket the range of uncertainty or disagreement in different causal models. Also, it will be important to vary the types of conservation action taken because the factors limiting owl populations probably vary geographically. Continued monitoring of the local populations would be required in order to discriminate among competing models and to converge on what management actions are most likely to lead to owl recovery.

The scale of manipulation could focus on the individual territory or a subset of the study population. For reliable inference from the manipulations it is important that the essential elements of an experiment—randomization, replication, and control and treatment sites—be incorporated into the study design. Given the longevity of spotted owls and the possibility of lag effects, such experiments would need to be carried on over several years.

Examples of environmental factors under control of managers include manipulations of barred owl populations, use of small-diameter thinnings in late seral reserves to reduce fuel loads, closing and restoration of roads in areas of high owl density, supplemental feeding experiments, and total restriction on the harvest of large diameter trees. Based on existing understandings of plausible causal relationships, a priori predictions as to how these changes would affect the components of λ could be made and tested in the context of the existing monitoring program. To make progress in the recovery of owl populations, such large-scale manipulative experiments, conducted to reduce uncertainty over caused-effect relationships, need to be implemented.

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The science-based approach we have used in this review requires a commitment to cooperation, and to our processes of open discussion and debate. We not only thank the many participants, for respecting our processes, but congratulate them. The future of the Northern Spotted Owl and its northwest forest habitats is an important and emotive subject. If we have contributed in any way to future successful management, it will be through an objective evaluation of information and opinion – this is the achievement of all those who (regardless of their affiliations) worked with us in a cooperative spirit.