AN EVALUATION OF CANDIDATE MANAGEMENT INDICATOR SPECIES FOR NATIONAL FORESTS IN THE NORTHERN CALIFORNIA PROVINCE

ENVIRONMENTAL ASSESSMENT DIVISION
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prepared by

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EXECUTIVE SUMMARY

The National Forest Management Act of 1976 requires development and implementation of resource management plans for each unit of the National Forest (Forest) System. These Land and Resource Management Plans (Forest Plans) provide guidance to natural resource management activities on each Forest. In order to estimate the effects of management alternatives on fish and wildlife populations, 36 Code of Federal Regulations (CFR) 219.19 directs the U.S. Forest Service (USFS) to identify and select "certain vertebrate and/or invertebrate species present in the area" whose population changes are believed to reflect the effects of management activities. These species are referred to as management indicator species. At the Forest Plan scale, population trends of management indicator species are to be monitored and relationships to habitat changes are to be determined.

There are four Forests in the Northern California Province — the Klamath, Mendocino, Shasta-Trinity, and Six Rivers National Forests. Each of these Forests developed Forest Plans and associated environmental impact statements (EISs). These documents identified separate lists of management indicator species that were to be used by each Forest in implementing its Forest Plan. There are a combined total of 69 species that were selected by the Northern California Province Forests.

Biologists in the Northern California Province Forests determined that the existing list of management indicator species may not be appropriate to use for monitoring for several reasons, including the infeasibility of monitoring such a large number of species and inclusion of species whose population changes may not be closely related to forest-management practices. They used 10 criteria to select species for further consideration as management indicators. Using these criteria, seven vertebrate species were identified as candidate management indicator species: (1) northern spotted owl (*Strix occidentalis*); (2) white-headed woodpecker (*Picoides albolarvatus*); (3) pileated woodpecker (*Dryocopus pileatus*); (4) acorn woodpecker (*Melanerpes formicivorus*); (5) mountain chickadee (*Poecile gambeli*); (6) oak titmouse (*Baeolophus inornatus*); and pallid bat (*Antrozous pallidus*). Three plant species were also selected as candidate management indicator species: (1) Douglas-fir (*Pseudotsuga menziesii*); (2) ponderosa pine (*Pinus ponderosa*); and (3) black oak (*Quercus kelloggii*).

We evaluated these 10 species by gathering available literature and using this information to determine each candidate's suitability as a management indicator species for the Northern California Province. Information gathered in the literature review included the following: (1) life history and habitat associations; (2) sensitivity to environmental change; (3) population distribution, status, and trends; and (4) availability of appropriate monitoring protocols. To be considered a suitable management indicator species for the Northern California Province Forests, a candidate species must meet the requirements identified in 36 CFR 219.19, i.e., populations must be dependent on forest habitat conditions and any population changes therefore could be used as an indicator of the effects of forest-management activities.

We considered the following criteria when evaluating the suitability of each candidate to serve as a management indicator species:

- *Distribution.* A candidate must be reasonably well-distributed in all four Forests of the Northern California Province.
- *Population size.* A candidate must have populations that are sufficiently large to allow development of robust population-size estimates that could be used to detect the effects of forest-management practices.
- *Ability to be field identified and detected.* A candidate must have populations that could be effectively monitored on the basis of (a) ease of identification; (b) conspicuousness of individuals or sign; and (c) home range or territory characteristics.
- *Habitat requirements*. Populations of the candidate are known to track particular habitat types or characteristics.
- *Affected by forest-management practices.* A candidate must be sensitive to the effects of forest-management practices.
- *Monitoring methodology*. Methodologies must exist or could be developed to effectively monitor the species.

Douglas-Fir

Douglas-fir satisfies the criteria we established for suitable management indicator species for the Forests of the Northern California Province. The species occurs in high numbers in all four Forests of the Province. Its habitat requirements are well understood; populations are affected by forest-management practices; and a monitoring program exists that could be used to track population trends in the Province.

Douglas-fir occurs within a wide variety of forest types in northern California, and is distributed across nearly all of the Province. It is found on moist, well-drained soils, in extensive, often pure stands along the coast ranges, and is a common component in mixed conifer forests farther inland. Douglas-fir is a dominant or codominant species of five forest types covering large areas of much of the Province, except eastern Klamath and northeastern Shasta-Trinity National Forests. Seedling establishment, survival of seedlings and saplings, and overall growth of Douglas-fir can be strongly affected by forest-management activities, and over much of the Province, Douglas-fir is a preferred timber species. The distribution of Douglas-fir throughout the Province and its relative importance as a component of forest stands can be monitored by the evaluation of data currently collected under the USFS Forest Inventory and Analysis (FIA) program. The FIA program currently monitors vegetation on permanent plots across the four Forests of the Province at 5- to 10-yr intervals, and is designed to detect changes in forest parameters over time. Reports generated from the program can be customized for selected areas and tree species.

Ponderosa Pine

Ponderosa pine satisfies the criteria we established for suitable management indicator species for the Forests of the Northern California Province. The species occurs in high numbers in all four Forests of the Province. Its habitat requirements are well understood; populations are affected by forest-management practices; and a monitoring program exists that could be used to track population trends in the Province.

Ponderosa pine is distributed across nearly all of the Province, and occurs within a large number of forest types. It is a common component of mixed conifer forests, and is also found in pure stands, particularly in inland areas. Ponderosa pine is a dominant or codominant species of four forest types within the Province, primarily in the eastern portions of Klamath and Shasta-Trinity National Forests, southern portions of Six Rivers National Forest, and in Mendocino National Forest. Establishment, survival, and overall growth of ponderosa pine can be strongly affected by forest-management activities, and production of ponderosa pine is targeted by management practices over much of the Province. However, in some areas ponderosa pine forests have been replaced by other forest types, due to past fire suppression and selective logging practices. The distribution and relative importance of ponderosa pine throughout the Province could be monitored by the evaluation of data currently collected under the USFS FIA program.

Black Oak

Black oak satisfies the criteria we established for suitable management indicator species for the Forests of the Northern California Province. The species occurs in high numbers in all four Forests of the Province. Its habitat requirements are well understood; populations are affected by forest-management practices; and a monitoring program exists that could be used to track population trends in the Province.

Black oak occurs within a wide variety of forest types in northern California, and is distributed across most of the Province. It is found on dry soils as a common component in mixed conifer forests or hardwood stands, or as scattered pure stands, often within conifer forests. Black oak is a dominant or codominant species of two forest types, primarily in the eastern portion of Shasta-Trinity National Forest, and in scattered areas of the Klamath, Six Rivers, and Mendocino National Forests. Seedling establishment, survival, and overall growth of black oak can be strongly affected by forest-management activities. Although black oak numbers have declined in many areas, largely due to fire suppression, efforts to restore black oak are being undertaken over much of the Province. The distribution of black oak and its relative importance as a component of forest stands could be monitored by the evaluation of data currently collected under the USFS FIA program.

Northern Spotted Owl

The northern spotted owl satisfies the criteria we established for suitable management indicator species for the Northern California Province. It occurs throughout the year in all four Forests and does not migrate. About 25% of the Forest area in the Province is thought to be suitable for nesting, roosting, and foraging of the northern spotted owl.

Northern spotted owls use mature or old-growth conifer forests disproportionately more than they use other habitats. The owl nests in cavities in large-diameter trees with broken tops; cliff cavities; abandoned hawk or raven nests; and dwarf mistletoe. Roost sites are typically in areas of relatively dense vegetation (i.e., high canopy closure dominated by large-diameter trees and multiple-canopy layers). The spotted owl uses a wider variety of forest types for foraging than for nesting and roosting. While the northern spotted owl is considered a late-successional and old-growth forest species, the age of a forest is not as important for determining habitat suitability as the structure and composition of the forest (e.g., high-canopy closure; a multilayered, multispecies canopy dominated by large overstory trees; a high incidence of large trees with various deformities; numerous large snags; large accumulations of logs and other woody debris; and sufficient open space below the canopy for flying).

Sampling protocols are well established for the northern spotted owl. Generally, surveys first entail the use of nocturnal calling surveys to locate the occurrence of owls. Follow-up diurnal calling surveys and baiting are done to locate specific roosting and nesting sites, and to determine reproductive and nesting status. A two-year survey with three site visits per site per year is preferable.

White-Headed Woodpecker

The white-headed woodpecker satisfies most of the criteria we established for suitable management indicator species for the Forests of the Northern California Province. It occurs throughout the year in the Province and does not migrate. The range of the species includes at least some portions of all four Forests. It is easy to identify and cannot be confused with any other species. Monitoring programs exist for forest birds in general and woodpeckers in particular and could be adapted to monitor populations of white-headed woodpeckers. However, the species occurs in relatively low numbers in the Province, and these low numbers may make it difficult to develop a statistically robust program to detect the effects of forest-management practices on this species.

White-headed woodpeckers utilize a variety of conifer types and do not appear to rely on any one particular type or species. Old growth and the presence of adequate numbers of snags appear to be important habitat characteristics for the species and habitat capability models have been developed and used to evaluate the effects of management practices on populations of this species. However, quantitative relationships between habitat characteristics and white-headed woodpecker populations have not been developed.

Several established survey methodologies could be used to monitor white-headed woodpecker populations. Call-playback surveys can be used to determine relative abundance in different habitats and locations. Spot-mapping and distance techniques (line transects and point counts) are suitable for determining absolute density of birds. Spot-mapping is relatively more labor intensive and measures only the abundance of breeding birds. Distance techniques are less expensive and relatively easy to implement.

Pileated Woodpecker

The pileated woodpecker satisfies most of the criteria we established for a suitable management indicator species for the Northern California Province Forests. It occurs throughout the year in the Province and does not migrate. The range of the species includes portions of all four Forests. It is easy to identify and cannot be confused with any other species. The species occurs in low numbers in the Province, and these numbers may make it difficult to develop a statistically robust program to detect the effects of forest-management practices on this species.

Pileated woodpeckers utilize a variety of conifer forest types and do not appear to rely on any one particular type or species. The species appears to be dependent on old growth and habitat capability models have been developed and used to evaluate the effects of management practices on populations of this species. However, quantitative relationships between habitat characteristics and pileated woodpecker populations have not been developed. The same survey techniques appropriate for the white-headed woodpecker (discussed above) could be used to monitor the pileated woodpecker.

Acorn Woodpecker

The acorn woodpecker satisfies the criteria we established for a suitable management indicator species for the Forests of the Northern California Province. It occurs throughout the year in the Province and does not migrate. The range of the species includes portions of all four Forests. It is easy to identify and cannot be confused with any other species. The species is relatively common throughout the Province, and as a consequence a statistically robust program could be developed to detect the effects of forest-management practices on this species.

Acorn woodpecker populations are greatly affected by acorn production and therefore could be used to monitor the effects of forest-management practices on oak woodlands. The same survey techniques appropriate for the white-headed woodpecker (discussed above) could be used to monitor the acorn woodpecker. In addition, surveys of acorn woodpecker granary trees could be used to monitor relative abundance of the species in the Province.

Mountain Chickadee

The mountain chickadee satisfies most of the criteria we established for a suitable management indicator species for the Forests of the Northern California Province. It occurs

throughout the year in the Province and does not migrate. The range of the species includes at least some portions of all four Forests of the Province, although it may be more abundant in the eastern portions of the Province. The species is easy to identify and relatively conspicuous behaviorally.

The mountain chickadee is relatively common in the Province; thus, collection of sufficient data to monitor the effects of forest-management practices on this species should be possible. The densities of both breeding and nonbreeding populations of mountain chickadees appear to be high enough to allow statistical analysis of survey and monitoring data. The mountain chickadee is relatively easy to survey in a variety of habitats (disturbed and undisturbed) and terrains.

Mountain chickadees utilize a variety of conifer forest types and do not appear to rely on any one particular type or species. Relationships between habitat characteristics and chickadee populations have not been developed or quantified. Spot mapping, line transect, or point count techniques can be used to determine absolute or relative population density. Point counts may be the most cost effective and repeatable of these methods and would allow for collection of more samples.

Oak Titmouse

The oak titmouse satisfies the criteria we established for a suitable management indicator species for the Forests of the Northern California Province. It occurs throughout the year in the Province and does not migrate. The species is relatively common in northern California, and population size appears to be large enough to allow statistical analysis of survey and monitoring data. The range of the species includes at least some portions of all four Forests of the Province. The oak titmouse is easily recognized and has conspicuous, active behavioral patterns.

The oak titmouse appears to be dependent on oak woodland habitat, but quantitative relationships between habitat quality and population density have not been established. Spot mapping, encounter transect, or point count techniques can be used to determine absolute or relative population density. Point counts may be the most cost effective and repeatable of these methods and would allow for collection of more samples.

Pallid Bat

The pallid bat satisfies most of the criteria we established for a suitable management indicator species for the Forests of the Northern California Province. It occurs throughout the year in the Province and does not migrate. The range of the species includes at least some portions of all four Forests of the Province. However, there is currently inadequate information regarding the specific status, geographic distribution, reproductive ecology, and roosting and foraging habits for the pallid bat, particularly within the Northern California Province.

The pallid bat occurs throughout all four Forests of the Province. It occurs in low numbers in California, but density estimates for the Province have not been made. Over its range, the pallid bat uses a variety of habitat types including conifer and hardwood forests, shrublands, rocky canyons, grasslands, open farmland, and deserts. The availability of roosting sites is the most important habitat requirement for the pallid bat. Roosting sites include snags, rock crevices, stone piles, and man-made structures. Monitoring of bat populations is usually done with mist nets or ultrasonic detectors that can automatically record bat calls in an area. Obtaining precise counts of a nocturnal species like the pallid bat, which is also an overdispersed species, would be difficult. Therefore, any monitoring program for this species would have uncertainties associated with population estimates and trends.

Conclusions

Of the candidate species evaluated, Douglas-fir, ponderosa pine, black oak, northern spotted owl, acorn woodpecker, and oak titmouse satisfy all of the criteria we established for a suitable management indicator species. The white-headed woodpecker, pileated woodpecker, and pallid bat have populations that may be too small to support a statistically robust monitoring program. Small populations make it difficult to develop a program that can effectively and economically detect the effects of forest-management practices. Smaller populations result in an increase in the variability among sample counts and may require a substantial increase in sampling intensity to ensure that enough data are gathered to overcome the effects of this variability. The mountain chickadee and pallid bat utilize a wide variety of habitat types in the Province, and thus their populations may be less sensitive to the effects of forest-management practices than other candidate species.

Collocating sampling locations for vertebrate management indicator species with FIA plots would facilitate the gathering of data that could be used to perform simple or multiple regression analyses to test and develop specific quantitative relationships between populations of management indicator species and habitat parameters. This will be an important next step in development of any monitoring program.

Some savings could be realized by developing an integrated sampling program that surveys all species using the same sampling locations and consistent survey techniques. As suggested above, there would be advantages to collocating vertebrate sample locations with FIA plots used to survey trees. Similarly, standardization of a single inventory approach (e.g., point counts) for use with all birds (except for the northern spotted owl) would save time and money. Existing survey campaigns (e.g., North American Breeding Bird Survey) could be used to monitor the population status of management indicator species.

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NOTATION

The following is a list of the acronyms, initialisms, and abbreviations (including units of measure) used in this report.

Abbreviations

BLM	Bureau of Land Management
CalVeg	California vegetation classification system
CDFG	California Department of Fish and Game
CFR	Code of Federal Regulations
CPIF	California Partners in Flight
CWHRS	California Wildlife-Habitat Relationships System
dbh	diameter at breast height
drc	diameter at root collar
EIS	environmental impact statement
FEMAT	Forest Ecosystem Management Team
FIA	Forest Inventory and Analysis program
Forest	National Forest
Forest Plan	Land and Resource Management Plan
FWS	U.S. Fish and Wildlife Service
GIS	Geographic Information System
GPS	Global Positioning System
MELP	Ministry of Environment, Lands and Parks
MOF	Ministry of Forestry
MSL	mean sea level
MSRM	Ministry of Sustainable Resource Management
NPS	National Park Service
REO	Regional Ecosystem Office
RSL	Remote Sensing Laboratory
SAF	Society of American Foresters
USFS	U.S. Forest Service

Units of Measure

ac	acre(s)
cm	centimeter(s)
ha	hectare(s)
in.	inch(es)
ft	foot (feet)
g	gram(s)
km	kilometer(s)
m	meter(s)
mi	mile(s)

Units of Measure (Cont.)

min	minute(s)
mm	millimeter(s)
oz	ounce(s)
s	second(s)
yr	year(s)

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1 INTRODUCTION

The National Forest Management Act of 1976 requires development and implementation of resource management plans for each unit of the National Forest (Forest) System. These Land and Resource Management Plans (Forest Plans) provide guidance to natural resource management activities on each Forest and are normally updated every 10 to 15 years. In order to estimate the effects of management on fish and wildlife populations, 36 Code of Federal Regulations (CFR) 219.19 directs the U.S. Forest Service (USFS) to identify and select "certain vertebrate and/or invertebrate species present in the area" whose population changes are believed to reflect the effects of management activities. These species are referred to as management indicator species. In the selection of management indicator species, 36 CFR 219.19 directs the Forests to use the following categories of species where appropriate:

- Endangered and threatened plant and animal species identified on State and Federal lists for the planning area;
- Species with special habitat needs that may be influenced significantly by planned management programs;
- Species commonly hunted, fished or trapped;
- Nongame species of special interest; and
- Additional plant or animal species selected because their population changes are believed to indicate the effects of management activities on other species of selected major biological communities or on water quality.

Management indicator species are chosen by each Forest for inclusion in its Forest Plan. Each Forest Plan is accompanied by an Environmental Impact Statement (EIS) that evaluates the impacts of the proposed action and alternative management strategies on management indicator species. In addition to using management indicator species to evaluate alternative management strategies, 36 CFR 219.19 directs the USFS to monitor population trends of these species and determine the relationship of any population changes to habitat changes at the Forest Plan scale.

There are four Forests in the Northern California Province — the Klamath, Mendocino, Shasta-Trinity, and Six Rivers National Forests. Each of these Forests developed Forest Plans (USFS 1994a, 1994c, 1995a, 1995c) and associated EISs (USFS 1994b, 1994d, 1995b, 1995d) in the middle 1990s. These documents identified separate lists of management indicator species that

were to be used by each Forest in monitoring the effects of implementing its Forest Plan. Table 1 presents the management indicator species that were identified by each Forest. There is a combined total of 69 species that was selected by the Northern California Province Forests; the Klamath, Mendocino, Shasta-Trinity, and Six Rivers Forests individually selected 32, 13, 22, and 42 species, respectively.

Biologists in the Northern California Province Forests determined that the existing list of management indicator species may not be appropriate for monitoring for several reasons including the infeasibility of monitoring such a large number of species and inclusion of species whose population changes may not be closely tied to forest-management practices (e.g., migratory species, cyclical prey species). In addition, they eliminated fish from further consideration as management indicator species, because other current and on-going monitoring (e.g., water quality measurements) are more effective in evaluating the effects of management practices on aquatic systems. Consequently, the biologists reviewed 547 terrestrial vertebrate species and major invertebrate groups that are likely to occur in the four Forests for consideration as management indicator species. They used 10 criteria to select species for further consideration as management indicators. The following types of animal species were considered unsuitable as management indicator species:

- Species whose populations are likely to also be strongly affected by external confounding factors including migratory species, species that are hunted or harvested, species that have a strong affinity to human-modified landscapes, and species that exhibit strong population cycles.
- Species that utilize a wide range of habitat types and that can adapt quickly to landscape-level changes.
- Species whose range marginally includes the Northern California Province.
- Species whose distribution or habitat use is highly restricted within the Northern California Province.
- Species whose habitats are not significantly affected by forest-management activities.
- Rare and uncommon species.
- Species whose habitat requirements are poorly understood.
- Species whose population trends are highly variable, periodic, or short-cycled.
- Species that are difficult to identify because of strong similarity to other species, taxonomic uncertainties, or the need for specialized equipment.
- A tested monitoring methodology is not available for the species.

Species with these characteristics were thought to make poor indicators because their populations could vary in response to factors elsewhere or unrelated to management actions; they would not be sensitive to landscape-level changes brought about by management actions; or they could not be effectively monitored because of low population size or difficulty in identification. A similar screening process, but using somewhat different criteria, was used by the Ministry of Forests (MOF) to identify candidate management indicator species in British Columbia (MOF 1994).

Table 1 Management Indicator Species Identified in Forest Plans for Northern California Province Forests

Common Name	Scientific Name	Klamath[1]	Men docino[2]	Shasta-Trinity[3]	Six Rivers[4]
Fish					
Cutthroat trout	*Oncorhynchus clarkii*				x
Rainbow trout	*Oncorhynchus mykiss*	x			x
Steelhead	*Oncorhynchus mykiss*	x			x
Amphibians					
Arboreal salamander	*Aneides lugubris*				x
Black salamander	*Aneides flavipunctatus*			x	
Calif. red-legged frog	*Rana aurora draytoni*			x	x
Cascades frog	*Rana cascadae*	x			
Clouded salamander	*Aneidus ferreus*				x
Del Norte Salamander	*Plethodon elongatus*				x
Northern red-legged frog	*Rana aurora aurora*	x		x	
Olympic Salamander	*Rhyacotriton olympicus*				x
Tailed frog	*Ascaphus truei*	x			x
Reptiles					
Western pond turtle	*Clemmys marmorata*	x			x
Western fence lizard	*Sceloporus occidentalis*				x
Birds					
Acorn woodpecker	*Melanerpes formicivorus*	x	x	x	x
American dipper	*Cinclus mexicanus*	x			x
Bald eagle	*Haliaeetus leucocephalus*		x		
Black-backed woodpecker	*Picoides arcticus*	x			
Black-headed grosbeak	*Pheucticus melanocephalus*				x
Blue grouse	*Dendragapus obscurus*				x
Brown creeper	*Certhia americana*				x
Burrowing owl	*Athene cunicularia*	x			
California thrasher	*Toxostoma redivivum*		x		
Common merganser	*Mergus merganser*				x
Downy woodpecker	*Picoides pubescens*	x			x
Flammulated owl	*Otus flammeolus*	x			x
Green-tailed towhee	*Pipilo chlorurus*			x	
Hairy woodpecker	*Picoides villosus*	x			x
Hammond's flycatcher	*Empidonax hammondii*				x
Lazuli bunting	*Passerina amoena*				x
Loggerhead shrike	*Lanius ludovicianus*	x			
Northern goshawk	*Accipiter gentilis*	x	x	x	
Northern pygmy owl	*Glaucidium gnoma*			x	
Northern saw-whet owl	*Aegolius acadicus*			x	
Northern spotted owl	*Strix occidentalis*	x	x	x	x
Peregrine falcon	*Falco peregrinus*		x		
Pileated woodpecker	*Dryocopus pileatus*	x	x	x	x
Pinyon jay	*Gymnorhinus cyanocephalus*	x			

Table 1 (Cont.)

Common Name	Scientific Name	Klamath[1]	Men docino[2]	Shasta-Trinity[3]	Six Rivers[4]
Red-breasted sapsucker	*Sphyrapicus ruber*	x			x
Ruffed grouse	*Bonasa umbellus*				x
Sage thrasher	*Oreoscoptes montanus*	x			
Western scrub-jay	*Aphelocoma californica*				x
Song sparrow	*Melospiza melodia*			x	
Swainson's hawk	*Buteo swainsoni*	x			
Tree swallow	*Tachycineta bicolor*			x	
Turkey	*Meleagris gallopavo*			x	
Vaux's swift	*Chaetura vauxi*	x			x
Western bluebird	*Sialia mexicana*				x
Western screech owl	*Otus kennicottii*			x	x
Western tanager	*Piranga ludoviciana*				x
White-headed woodpecker	*Picoides albolarvatus*	x			x
Willow flycatcher	*Empidonax traillii*			x	
Winter wren	*Troglodytes troglodytes*				x
Wood duck	*Aix sponsa*				x
Yellow-breasted chat	*Icteria virens*				x
Mammals					
American marten	*Martes americana*	x	x	x	x
Black bear	*Ursus americanus*	x		x	x
California vole	*Microtis californicus*			x	
Douglas' squirrel	*Tamiasciurus douglasii*		x		x
Dusky-footed woodrat	*Neotoma fuscipes*				x
Fisher	*Martes pennanti*	x	x	x	x
Long-tailed vole	*Microtus longicaudus*	x			
Montane vole	*Microtus montanus*	x			
Mule deer	*Odocoileus hemionus*	x	x	x	x
Northern water shrew	*Sorex palustris*	x			
Pronghorn	*Antilocapra americana*	x			
Elk	*Cervus elaphus*		x	x	
Western gray squirrel	*Sciurus griseus*	x	x		x
Western harvest mouse	*Reithrodontomys megalotis*			x	
Total Number of Species		**32**	**13**	**22**	**42**

[1] Source: USFS (1994b)
[2] Source: USFS (1995b)
[3] Source: USFS (1994d)
[4] Source: USFS (1995d)

x = Species identified as a management indicator species for the Forest.

Using these criteria, Forest biologists identified seven vertebrate species with the highest potential to serve as management indicator species: the northern spotted owl, white-headed woodpecker, pileated woodpecker, acorn woodpecker, mountain chickadee, oak titmouse, and pallid bat. Three plant species—Douglas-fir, ponderosa pine, and black oak—were also considered potential management indicator species because they are widespread across the Province, are affected directly by management practices, and are important components of wildlife habitat. These 10 species are considered candidate management indicator species and were the subject of further evaluation to determine their suitability as such. This evaluation is the subject of this report.

Candidate Management Indicator Species for Northern California Province Forests

Northern spotted owl	*Strix occidentalis*
White-headed woodpecker	*Picoides albolarvatus*
Pileated woodpecker	*Dryocopus pileatus*
Acorn woodpecker	*Melanerpes formicivorus*
Mountain chickadee	*Poecile gambeli*
Oak titmouse	*Baeolophus inornatus*
Pallid bat	*Antrozous pallidus*
Douglas-fir	*Pseudotsuga menziesii*
Ponderosa pine	*Pinus ponderosa*
Black oak	*Quercus kelloggii*

Our evaluation consisted of gathering available literature and using this information to determine each candidate's suitability as a management indicator species for the Northern California Province. Information gathered in the literature review included the following: (1) life history and habitat associations; (2) sensitivity to environmental change; (3) population distribution, status, and trends; and (4) availability of appropriate monitoring protocols. To be considered a suitable management indicator species for the Northern California Province Forests, a candidate species must meet the requirements identified in 36 CFR 219.19, i.e., populations must be dependent on forest habitat conditions and any population changes therefore could be used as an indicator of the effects of forest-management activities.

We considered the following criteria when evaluating the suitability of each candidate to serve as a management indicator species:

- *Distribution.* A candidate must be reasonably well-distributed in all four Forests of the Northern California Province.
- *Population size.* A candidate must have populations that are sufficiently large to allow development of robust population-size estimates that could be used to detect the effects of forest-management practices.
- *Ability to be field identified and detected.* A candidate must have populations that could be effectively monitored on the basis of (a) ease of identification; (b) conspicuousness of individuals or sign; and (c) home range or territory characteristics.
- *Habitat requirements*. Populations of the candidate are known to track particular habitat types or characteristics.
- *Affected by forest-management practices.* A candidate must be sensitive to the effects of forest-management practices.
- *Monitoring methodology*. Methodologies must exist or could be developed to effectively monitor the species.

These criteria are consistent with those presented by Dale and Beyeler (2001) for effective ecological indicators. They used the following criteria: (1) easily measured; (2) sensitive to stresses on the system; (3) respond to stress in a predictable manner; (4) changes signify an impending change in the system; (5) predict changes that can be averted by management actions; (6) integrative, i.e., the full suite of indicators measures the key components of the system; (7) have a known response to disturbance, stress, and changes over time; and (7) have low variability of response.

In addition to literature review, the California Wildlife–Habitat Relationship System (CWHRS) was used to define the dependence of vertebrate candidate management indicator species' on habitats in California and to determine the approximate range of each vertebrate species in the Northern California Province. CWHRS Version 8.0 software (CDFG 2002) was used in this portion of the evaluation.

Geographic Information System (GIS) data were downloaded for analysis from sites maintained by USFS including the Remote Sensing Laboratory's (RSL) GIS Clearinghouse (RSL 2003), and the Regional Ecosystem Office (REO 2003). GIS data were processed and maps of potentially suitable habitat generated using ESRI ArcInfo 7.1.2. Suitable habitat for each species was determined from CDFG (2002), and the location of these habitats in the four Forests of the Province was mapped. These maps should be considered as presentations of *potentially* suitable habitats because they are based only on the presence of a particular CWHRS vegetation type without consideration of other characteristics of the habitat that are important to species (e.g., seral stage, canopy closure). In addition, they assume that the species are found throughout the Province; however, some species (e.g., white-headed woodpecker, mountain chickadee, and oak titmouse) do not appear to be distributed across the entire Province.

2 BACKGROUND

The Klamath, Mendocino, Shasta-Trinity, and Six Rivers Forests are collectively referred to here as the Northern California Province Forests. In total, these Forests occupy about 2,285,100 ha (5,646,700 ac) in northwestern California (Colusa, Del Norte, Glenn, Humboldt, Lake, Mendocino, Modoc, Shasta, Siskiyou, Tehama, and Trinity Counties; Figure 1). The Northern California Province encompasses several mountain ranges including the Cascade Range, Coast Range, Klamath Mountains, Salmon Mountains, and Trinity Mountains.

Land Areas of Northern California Forests

Klamath National Forest — 679,900 ha (1,680,000 ac)

Mendocino National Forest — 360,100 ha (889,900 ac)

Shasta-Trinity National Forest — 849,500 ha (2,099,200 ac)

Six Rivers National Forest — 387,500 ha (957,600 ac)

2.1 DESCRIPTION OF THE NORTHERN CALIFORNIA PROVINCE FORESTS

The Province is dominated by mixed-species conifer and conifer–hardwood forests (FEMAT 1993; USFS and Bureau of Land Management [BLM] 1994). In general, these forests are fragmented because of poor soils, climate, wildfires, timber harvest, and roads (USFS and BLM 1994). Historically, timber harvest in the Province has been a combination of selective harvest and clear-cutting, and this practice has resulted in forests that now contain a mixture of old trees left after harvest and younger trees that have regenerated. Fire is an important component of the Northern California Province Forests, but fire suppression has resulted in significant accumulation of fuel in some areas, and changes from historical conditions in species composition and forest stand structure (USFS and BLM 1994).

The Klamath National Forest, approximately 687,870 ha (1,680,000 ac) in size, supports a diverse mixture of vegetation types ranging from open, relatively dry ponderosa pine forests in the southern Cascade Range to high-elevation mixed conifers near Russian Peak to grasslands on the eastern side of the Forest (USFS 1994b). Vegetative groups identified in the Forest's timber inventory include westside mixed conifer (43%); Douglas-fir (20%); nonforest types including chaparral, shrublands, grasslands, and wetlands (16%); eastside mixed conifer (6%); ponderosa pine (6%); westside true fir (*Abies* spp.; 5%); eastside true fir (2%); lodgepole pine (*Pinus contorta*; 1%); and hardwoods (1%) (USFS 1994b). Dominant plant communities (using the CalVeg community type classification[1]; USFS 1981), in decreasing order of coverage, include Pacific Douglas-fir (20%), mixed conifer–pine (13%), white fir (*Abies concolor*; 9%), red fir (*Abies magnifica*; 6%), and upper montane mixed chaparral (5%).

[1] The CalVeg system is a standard vegetation classification system for the state of California that is based on remote-sensing information (USFS 1981). CalVeg community types are presented throughout this report for reference. Community type descriptions including species composition, community structure, and physiographic setting can be found online at http://www.fs.fed.us/r5/rsl/projects/mapping/zone-map.shtml.

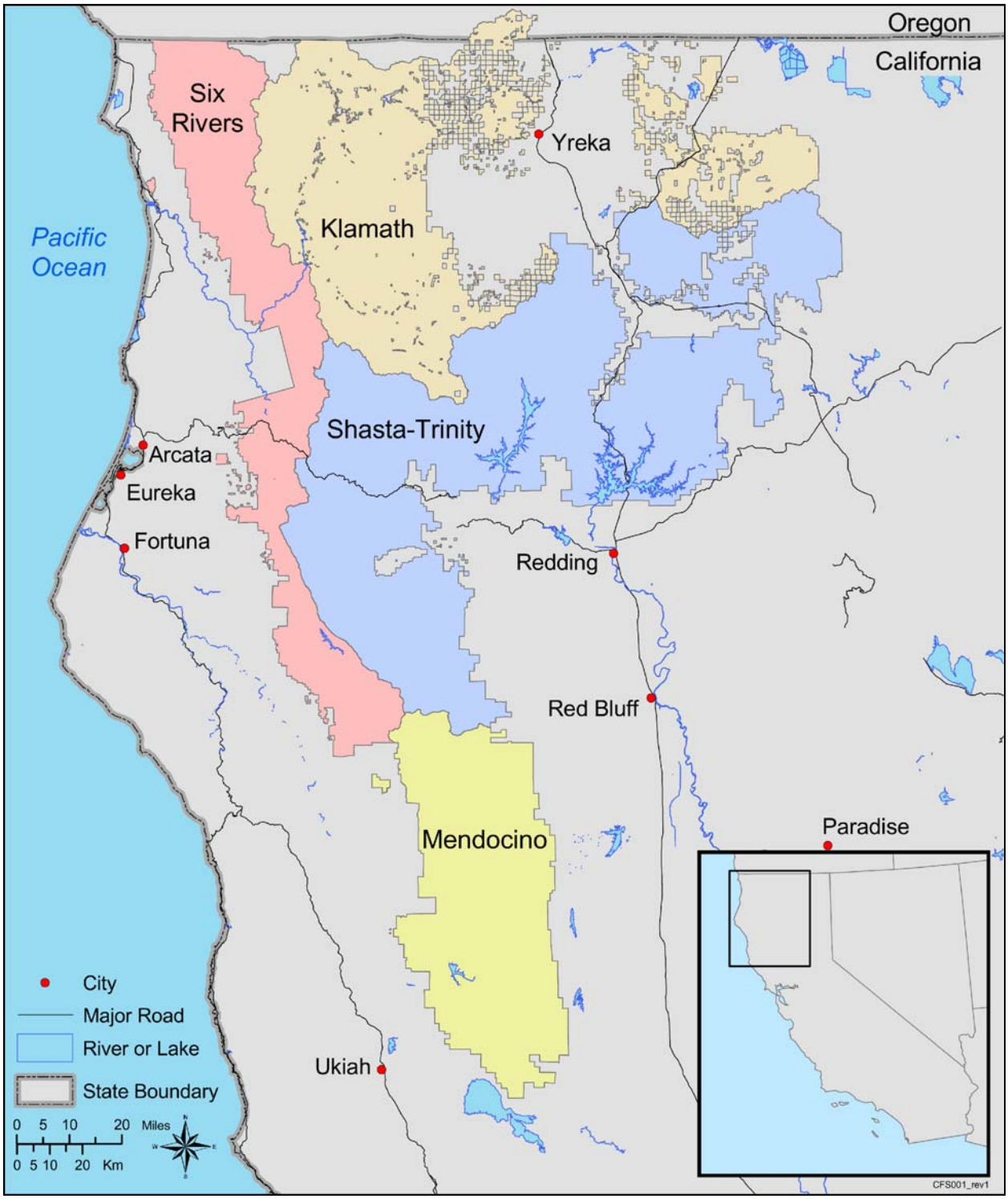


Figure 1 Forests of the Northern California Province

The Mendocino National Forest is approximately 360,100 ha (889,900 ac) and supports the following vegetation types (1) mixed conifer (39%), (2) chaparral (24%), (3) conifer–hardwood (14%), (4) hardwoods (10%), (5) knobcone pine (*Pinus attenuata*; 4%), (6) grass (3%), (7) red fir (2%), and (8) hardwood savannah (2%) (USFS 1995b). Dominant plant species include Douglas-fir, ponderosa pine, sugar pine (*Pinus lambertiana*), white fir, red fir, incense cedar (*Libocedrus decurrens*), black oak, blue oak (*Quercus douglasii*), canyon live oak (*Quercus chrysolepis*), madrone (*Arbutus menziesii*), manzanita (*Arctostaphylos* spp.), chamise (*Adenostoma* spp.), and *Ceanothus* spp. Dominant plant communities (using CalVeg community types; USFS 1981) in decreasing order of coverage include Douglas-fir–pine (19%), lower montane mixed chaparral (14%), Pacific Douglas-fir (8%), annual grass–forb (7%), and mixed conifer–pine (7%).

The Shasta-Trinity National Forest encompasses an area of approximately 849,500 ha (2,099,200 ac). Major vegetation types on the Forest include conifers (mixed conifer, Douglas-fir, ponderosa pine–Jeffrey pine (*Pinus jeffreyi*), red fir–white fir types; 79%), hardwoods (black oak, live oak, and riparian woodlands; 9%), chaparral (dominated by a variety of hardwood shrubs especially *Ceanothus* spp. and *Arctostaphylos* spp.; 7%), and grasslands (1%) (USFS 1994d). Dominant plant communities (using CalVeg community types; USFS 1981) in decreasing order of coverage include mixed conifer-pine (20%), Pacific Douglas-fir (15%), Douglas-fir–pine (9%), white fir (6%), and upper montane mixed chaparral (5%).

The Six Rivers National Forest is 387,500 ha (957,600 ac) in size. Vegetation series in the Forest include Douglas-fir (37%); tanoak (*Lithocarpus densiflorus*; 35%); white fir (9%); lodgepole pine and white oak (*Quercus garryana*; 5% each); red fir, Jeffrey pine, and grasslands (2% each); Port Orford cedar (*Chamaecyparis lawsoniana*; 1%); and knobcone pine, western white pine (*Pinus monticola*), black oak, chaparral, and redwood (*Sequoia sempervirens*; < 1% each)(USFS 1995d). Dominant plant communities (using CalVeg community types; USFS 1981) in decreasing order of coverage include Pacific Douglas-fir (50%), ultramafic mixed conifer (6%), and tanoak–madrone (5%).

2.2 MANAGEMENT PRACTICES IN THE NORTHERN CALIFORNIA PROVINCE FORESTS

Land within the four Northern California Province Forests are allocated as (1) administratively withdrawn, (2) wilderness, (3) late-successional reserves, (4) riparian reserves, (5) matrix, and (6) adaptive management areas (USFS 1995c). Administratively withdrawn lands are those lands where management emphasis precludes scheduled timber harvest. Included in this category are research natural areas, special habitat-management areas, experimental forests, and special interest areas. Wilderness areas have

Land Allocations in Northern California Forests

Administratively withdrawn — 233,500 ha (577,000 ac)

Wilderness — 510,300 ha (1,261,000 ac)

Late-successional reserves — 593,300 ha (1,466,000 ac)

Riparian reserves — 312,000 ha (771,000 ac)

Matrix — 510,000 ha (1,260,000 ac)

Adaptive management areas — 166,000 ha (409,000 ac)

been Congressionally withdrawn and forest-management practices in these areas are prohibited. Late-successional reserves are managed to protect and enhance conditions of late-successional and old-growth forest ecosystems, which serve as habitat for dependent species including the northern spotted owl. Limited stand management is permitted in late-successional reserves. Riparian reserves are areas along all streams, wetlands, ponds, lakes, and unstable and potentially unstable areas where riparian-dependent species receive primary emphasis. Adaptive management areas are set aside to develop and test new management approaches that integrate and achieve ecological and economic health, and other social objectives. Matrix lands are those lands not in the above categories where most timber harvest takes place.

The Forest Plans and their EISs for the Northern California Province identify an ecosystem-management approach to forest-management that is intended to maintain biodiversity and that is consistent with the Northwest Forest Plan (USFS and BLM 1994). With this approach, the Forests set aside reserves, according to the land allocations identified above, where timber harvest and other resource development or land-disturbing activities are prohibited or restricted. A mix of even-aged and selective harvest is intended to provide the variety and distribution of habitats that are typical of natural systems (USFS 1995c). Other activities on the Forests that can affect ecosystems include prescribed burning and other fire-management activities; pest management; livestock grazing; road construction and maintenance; dispersed and developed recreation, natural and artificial regeneration, and mineral extraction. These activities can have a variety of ecosystem effects, described below, that would be the subject of management indicator species monitoring.

Prescribed burning and wildfire can have important effects on ecosystems including shifts in species composition towards fire-adapted species, reductions in the encroachment of woody or nonnative vegetation in grasslands, changes in age-structure or seral state, reductions in understory species, reductions in the density of overstory species, creation of openings, and habitat fragmentation. Depending on their intensity, frequency, seasonal timing, areal extent, and distribution across a landscape, prescribed burning and wildfire can restore and maintain ecosystems in a more natural state or produce conditions that are atypical of the natural ecosystem. When appropriately used, prescribed fire and wildfire-management are important tools for maintaining biodiversity in the Forests.

Timber-management activities can increase biodiversity by mimicking the effects of fire and other natural disturbance factors, but can also reduce biodiversity by altering the species composition of forests, removing snags, creating large even-age mono-specific stands, and fragmenting habitats. Newer timber-management practices (e.g., green-tree retention, overstory removal, uneven-aged timber management, and regeneration) attempt to reduce adverse effects while increasing biodiversity (USFS 1995b). Current Forest Plans adopt these newer practices and utilize an ecosystem-management approach to timber management that includes a reduction in the rate of timber harvest and restriction of harvest to 10% or less of the total Forest area (USFS 1994a, 1994c, 1995a, 1995c).

An integrated pest-management approach is used in the Forests that includes detection, prevention, evaluation, suppression, and monitoring (USFS 1994b, 1995b, 1995d). Pest management can result in the removal of dead, diseased, or infested trees, which can affect

habitat conditions for a variety of wildlife species. An integrated pest management program includes the use of chemical pesticides (herbicides, fungicides, and insecticides), which can have adverse effects on wildlife. Overall, pest-management activities under the current Forest plans have been reduced from previous levels largely as a result of a reduction in the area of each Forest that is subject to timber-management activities (USFS 1995b).

Livestock grazing can affect ecological conditions depending on its intensity and distribution across the landscape. Heavy grazing tends to decrease species and structural diversity while light to moderate grazing can increase these characteristics (USFS 1995b). Grazing tends to result in a shift in species composition towards more unpalatable species and a reduction in the density of vegetation; grazing of saplings can result in poor regeneration of overstory trees and more even-aged stands. Depending on the intensity of grazing, trampling can affect soil compaction, erosion, plant growth, and regeneration rates. Riparian areas may be particularly susceptible to this effect.

Road construction, recreational development, and mineral extraction can affect Forest ecosystems by creating openings devoid of vegetation and increasing human activities and associated disturbances. Developments result in fragmented habitats, increases in human access and disturbance, and higher rates of erosion and sedimentation. The intensity of the effect is a function of the location and nature of the area affected, the areal and temporal extent (duration) of the disturbance, the timing of the activity or development, and mitigation (e.g., erosion control, reclamation activities) associated with development.

3 EVALUATION OF CANDIDATE MANAGEMENT INDICATOR SPECIES

This section presents information on the ten candidate management indicator species evaluated in this report. The following topics are presented for each species (1) distribution, habitat, and life history; (2) status, population trends, and factors affecting populations; (3) the apparent suitability of the species to be used as a management indicator species by the Forests of the Northern California Province based on the criteria discussed in Section 1; and (4) monitoring protocols that could be used to monitor populations of the management indicator species. Of particular interest is how well the populations of each species might track ecosystem-level changes caused by forest-management practices and whether or not monitoring protocols could be used that would provide statistically robust population estimates or indices at reasonable cost.

3.1 DOUGLAS-FIR

The Douglas-fir is a large tree that typically reaches 61–76 m (200–250 ft) in height and 1.5–2.4 m (5–8 ft) in diameter. Two distinct varieties of Douglas-fir are distinguished, and are separated geographically. The typical variety, *P. menziesii* var. *menziesii*, occurs in the Pacific coast region, west of the Sierra Nevada in California, while *P. menziesii* var. *glauca*, generally smaller in size, occurs in the Rocky Mountain region (USFS 1990a).

3.1.1 Distribution, Habitat, and Life History of Douglas-Fir

Douglas-fir occurs over a wide range that extends from central British Columbia to central California, and in the Rocky Mountains from British Columbia to central Mexico (USFS 1990a). In California, Douglas-fir is found in the Klamath and Coast Ranges, down through the Santa Cruz Mountains, as well as in the Sierra Nevada south through Yosemite National Park. In the Cascades and Sierra Nevada, it is generally found from 610 to 1,830 m (2,000 to 6,000 ft) in elevation, and at somewhat lower elevations in river valleys and canyon bottoms (USFS 1990a). The occurrence and relative abundance[2] of Douglas-fir within the Klamath, Mendocino, Shasta-Trinity, and Six Rivers Forests is shown in Figure 2.

Most Douglas-fir occur on moist, well-drained soils. Soil conditions for best growth are deep, well-aerated, acidic soils (pH 5 to 6); poorly drained or compacted soils are not suitable for Douglas-fir (USFS 1990a). Soils that support Douglas-fir range in texture from clays to gravelly sands, have moderate to high organic matter content, and are moderately acid. The mean annual precipitation in coastal portions of the range of Douglas-fir varies from 76–340 cm (34–134 in.), and 61–305 cm (24–120 in.) in the Cascades and Sierra Nevada (USFS 1990a).

[2] Relative abundance is presented as dominant (one of the most abundant species in community), associate (consistent and common species in community, but not dominant), uncommon (present in low numbers).

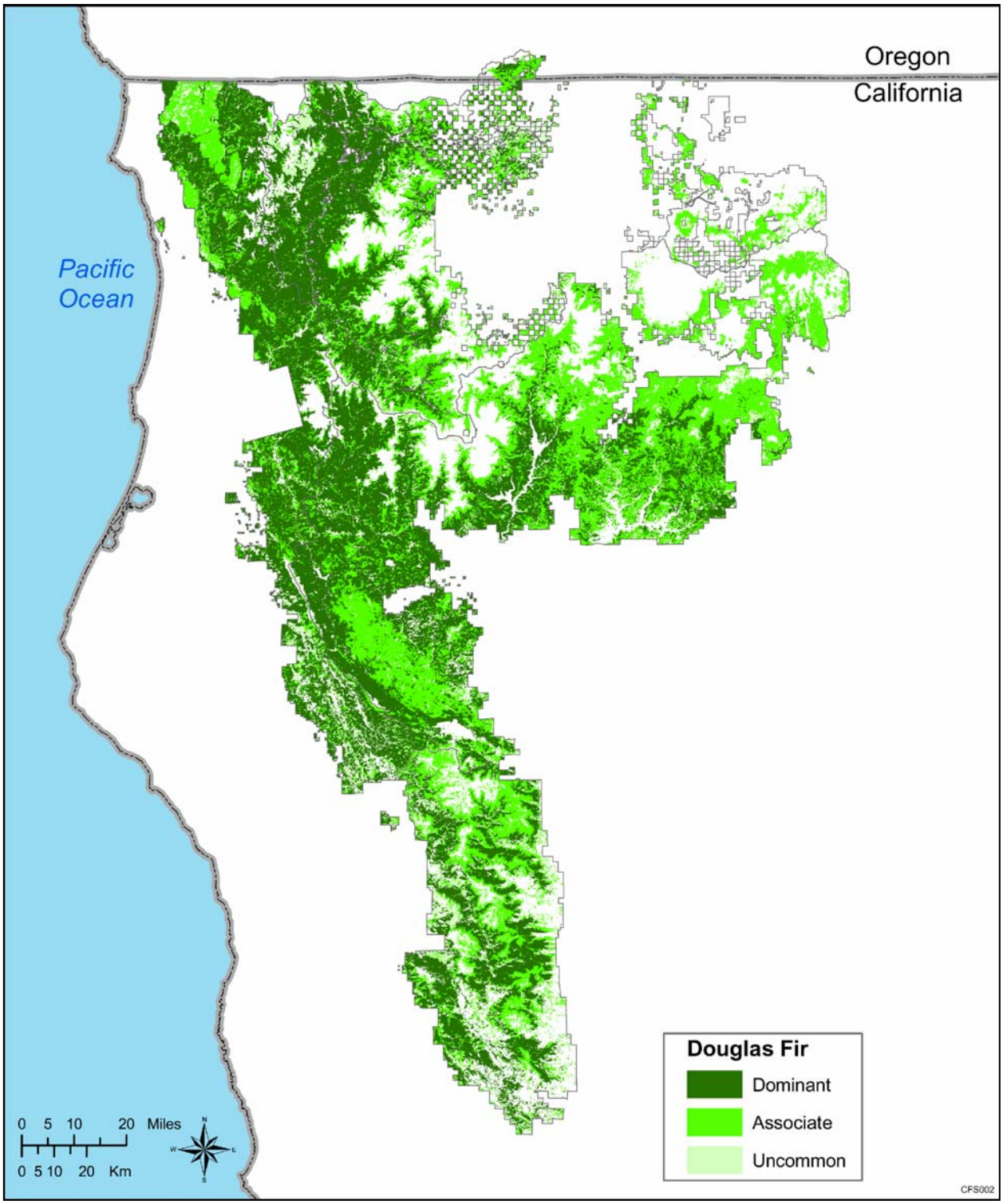


Figure 2 The Occurrence of Douglas-Fir in the Four Forests of the Northern California Province

Douglas-fir is a major species component of the following Society of American Foresters (SAF)[3] cover types: Pacific Douglas-fir (229), Douglas-fir–western hemlock (230), Port Orford cedar (231), and Pacific ponderosa pine–Douglas-fir (244), and is a minor component of ten others (USFS 1990a). Vegetation types in California have also been classified using the CalVeg Classification System developed by the USFS (USFS 1981). Douglas-fir is a dominant or codominant species in the following CalVeg vegetation types: Pacific Douglas-fir, Douglas-fir–grand-fir, Douglas-fir–pine, Douglas-fir–white-fir, and redwood–Douglas-fir.

The proportion of Douglas-fir within a mixed stand varies in relation to aspect, elevation, and soil, due in large part to the effects of temperature and moisture (USFS 1990a). For example, in northern areas, Douglas-fir grows mostly on south slopes while in the southern portion of its range it grows predominantly on north slopes. It is generally found at elevations up to 760 m (2,500 ft) in the northern portion of its range and up to 2,900 m (9,500 ft) in the south. Site history, particularly fire history, also affects stand composition, and almost pure stands of Douglas-fir have become established in some areas because of the historic occurrence of periodic fire (USFS 1990a).

The root system of the Douglas-fir generally consists of a tap root with lateral branches. Douglas-fir typically reach 500 yr in age, and some exceed 1,000 yr. Douglas-fir begins to produce cones at about 12 to 15 yr of age, and produce the greatest number of cones at 200 to 300 yr of age (USFS 1990a). Old-growth stands produce considerably more cones per unit area than younger stands. Cone production within a stand may be heavy only once in seven years, and may be high on only 25% of trees within a stand in a good production year (USFS 1990a; Uchytil 1991). Douglas-fir seeds are consumed by a wide variety of wildlife, such as mice, voles, chipmunks, shrews, song sparrow, and dark-eyed junco (USFS 1990a, Uchytil 1991).

Douglas-fir seedling growth is slow during the first 5 yr, but gradually accelerates and reaches 1 m (3.3 ft) per yr in 8 to 10 yr on good sites. The species achieves maximum growth rate at 20 to 30 yr of age (USFS 1990a). A tap root is rapidly formed during early growth, and grows 90% of its length in six to eight years in deep soils (USFS 1990a). Seedling establishment is best on moist mineral soil, and heavy accumulations of litter result in poor seedling survival (USFS 1990a).

3.1.2 Status, Population Trends, and Factors Affecting Populations of Douglas-Fir

The Douglas-fir is a major component of western North American forests and one of the most important timber trees in the U.S. In northern California, Douglas-fir develops extensive, often pure stands along the coast and is a common component of mixed conifer forests in the Sierra Nevada. Regeneration of Douglas-fir may be highly variable after harvest. Natural regeneration of Douglas-fir is limited by a relatively small seed supply (only up to 40% of which are viable), seed consumption by insects and wildlife, competition from other plant species, and unfavorable conditions where seeds fall, and may result in highly uneven spacing (USFS 1990a). Establishment of nearly pure stands after fire or clearing by natural seeding can occur in 10 or

[3] Forest cover types of the United States have been described by the Society of American Foresters (Eyre 1980). Numbers in parentheses represent codes used to identify SAF forest cover types.

15 yr on mesic to moist sites but may require 100 yr on drier sites (USFS 1990a). Direct seeding results are often highly variable and uneven, and require subsequent interplanting or thinning (USFS 1990a).

Douglas-fir has been historically maintained as a major forest component primarily through historic fire regimes, and is currently managed as a significant timber resource. Current regeneration practices require planting of Douglas-fir seedlings following harvest of this species (USFS 1990a, Uchytil 1991). Other forest-management practices in the Province also favor Douglas-fir over noncommercial forest tree species.

Browsing of Douglas-fir is generally avoided by livestock, although domestic sheep may lightly browse new spring growth of seedlings or saplings (Uchytil 1991). Browsing of seedlings by wildlife can be severe and result in mortality or reduced growth (Uchytil 1991). Survival of seedlings can be enhanced by stand-management activities such as the use of control measures implemented within the first four years of planting, which may include repellents, physical barriers, or planting palatable forbs (Uchytil 1991).

Overall tolerance of shade by Douglas-fir is intermediate. Young Douglas-fir are relatively shade-tolerant and light shade promotes survival of first-year seedlings (USFS 1990a; Uchytil 1991). The successful establishment of Douglas-fir seedlings is often promoted under the canopies of large black oaks, however, black oak root crown sprouts following fire or harvest may out-compete conifer seedlings (Howard 1992). Competition from both herbaceous and woody species can hinder regeneration by creating shade and reducing soil moisture (USFS 1990a; McDonald and Abbott 1994). Control of plant species associated with Douglas-fir is often necessary for successful Douglas-fir regeneration, because of the far greater growth rate of competing species which can result in over-topping and suppression of young Douglas-fir (USFS 1990a; McDonald and Fiddler 1996).

Rapidly growing stands of Douglas-fir are successfully established by clear cutting followed by planting of nursery seedlings. Shelterwood cuts followed by planting are also successful because of the shade-tolerance exhibited by Douglas-fir seedlings (USFS 1990a). Douglas-fir growing in competition with shrubs or trees respond well to release by thinning while at the pole and small sawtimber size, and good growth may be enhanced by a nitrogen fertilizer (USFS 1990a). Trees remaining after heavy traditional thinning of a closed stand, however, are susceptible to damage. Douglas-fir that remain in the understory after overstory harvest show a strong growth response, and result in multi-aged stands that have greater ecological value than clearcuts (Uzoh et al. 1998). Variable-density thinning can result in an increase in both structural complexity and native understory species richness (Thysell and Carey 2001).

Mature Douglas-fir is resistant to fire because of its relatively thick bark, and fire has historically promoted and maintained extensive forests of Douglas-fir. During the twentieth century, the fire regime changed to less frequent, but more severe fires in portions of the Pacific Northwest (McKelvey et al. 1996; Everett et al. 2000; Oliver 2000). Over the long term, fire suppression would result in the replacement of Douglas-fir with less fire-resistant tree species (USFS 1990a; Uchytil 1991).

3.1.3 Suitability of Douglas-Fir as a Management Indicator Species

Douglas-fir satisfies the criteria we established for a suitable management indicator species for Forests of the Northern California Province (see Section 1). Douglas-fir is an important component of a large number of vegetation types within the Northern California Province Forests. The forest types that include Douglas-fir are extensive, and cover much the Province (Figure 2). Douglas-fir is therefore important for the establishment of habitat characteristics for many species that prefer these vegetation types. Changes in the Douglas-fir population across the Province that result from forest-management activities would thus be expected to result in a change in habitat characteristics across a wide range of vegetation types and a wide area of the Province. Those species that utilize the Douglas-fir forest types would likely be affected by such changes in habitat characteristics, and would thus be affected by the management activities. Effects would likely be greatest on those species that are directly dependent on Douglas-fir, for example, as a food source or nesting site.

The extensive forests of Douglas-fir across Northern California and other areas of the Pacific Northwest became naturally established over time as a result of natural processes (USFS 1990a; Uchytil 1991). However, Douglas-fir establishment and survival can be strongly affected by forest-management activities. The habitat conditions conducive to growth and regeneration of Douglas-fir are affected by a number of factors including stand density, competition with other tree species (which can increase shading and decrease moisture availability), and fire regime (Uchytil 1991; McDonald and Abbott 1994; McDonald and Fiddler 1996; Everett et al. 2000).

Forest-management activities can influence Douglas-fir population patterns and trends by affecting seedling establishment and altering habitat conditions. For example, establishment success of Douglas-fir stands following harvest or fire is greatly increased by planting nursery-grown seedlings, and seedling survival may be increased by controlling wildlife browsing. Planting with species other than Douglas-fir would directly influence Douglas-fir presence within a stand. Removal of competing tree species from a young stand can increase the survival of Douglas-fir. The use of prescribed fire as a management tool can be conducive to establishing and maintaining a high level of Douglas-fir presence within a stand, whereas fire suppression may result in Douglas-fir decline.

3.1.4 Monitoring Protocols for Douglas-Fir

The USFS maintains a Forest Inventory and Analysis (FIA) program that is designed to collect information on forest composition and quality and detect changes in forest parameters over time (USFS 2001). The status of Douglas-fir as a component or dominant species of forest types within the Northern California Province can be effectively monitored by evaluation of data currently collected under this program. Changes in the Douglas-fir population that result from forest-management activities including, for example, changes in distribution across the Province, within vegetation types, or age distribution could be detected using the FIA program.

The FIA program monitors vegetation on over 900 permanent plots in the four Northern California Province Forests. The FIA program in northern California is a research program area of the Pacific Northwest Research Station, Portland Forestry Sciences Laboratory. FIA data are collected annually from 10% to 20% of all plots, resulting in a 5- to 10-yr monitoring interval for each plot.

To monitor forest areas, the FIA program uses aerial photography and satellite imagery for Phase 1 data points. Phase 1 points are characterized as forest or non-forest, and variables such as fragmentation and distance are measured. A subset of these points is designated as Phase 2 plots and used for field-data collection. The distribution of Phase 2 plots results in an average of approximately one plot for every 2,430 ha (6,000 ac). Forest-health monitoring data are collected on Phase 3 plots, which are a subset of the Phase 2 plots. One of every 16 Phase 2 plots is designated as a Phase 3 plot, resulting in a distribution of approximately one plot for every 96,000 acres. In the Pacific Northwest Unit, there are 867,000 Phase 1 data points, 34,665 Phase 2 field plots, and 2,189 Phase 3 field plots.

The procedures for data collection on Phase 2 field plots are described in USFS (2003a), and additional field instructions for California are included in USFS (2003b). Within each Phase 2 plot, circular subplots are clustered in groups of four, with three subplots arranged around a central subplot, at 120°, 240°, and 360°, with 36.6 m (120 ft) between subplot centers. Each subplot has a radius of 7.3 m (24 ft) and an area of approximately 0.02 ha (0.04 ac). In subplots, data are collected on all trees at least 12.7 cm (5 in.) diameter at breast height (dbh) or diameter at the root collar (drc). Each subplot contains a microplot, set at 90° from the subplot center and 3.7 m (12 ft) between microplot center and subplot center. Microplots have a radius of 2.1 m (6.8 ft) and an area of 0.0001 ha (0.003 ac). In microplots, data are collected on all seedlings (dbh or drc less than 2.5 cm [1.0 in.], and length at least 0.15 m [0.5 ft] in conifers and 0.3 m [1.0 ft] in hardwoods) and saplings (dbh or drc 2.5 cm [1.0 in.] up to 12.7 cm [5.0 in.]).

Phase 2 field plots also include larger annular plots for large trees. Annular plots are centered on each of the subplots. The radius of annular plots is 18 m (58.9 ft), resulting in an area of 0.1 ha (0.25 ac). Phase 3 plots also include quadrats, one m^2 (10.8 ft^2) in size. Each Phase 3 subplot contains three permanent quadrats, arranged at 30°, 150°, and 270° from the subplot center. Within each quadrat, all vascular plants are identified and quantified, as well as litter, moss, bare ground, and rocks. Additional species are identified in the subplot and estimated cover and vertical layer of greatest foliage density is recorded for each species in the subplot.

Qualitative and quantitative data are collected on sample trees and other plants, including condition and physical parameters, as well as vegetation community data and site physical characteristics. Information collected related to plot condition includes land use, forest type, stand age, stand size class, regeneration status, and tree density. Data collection on trees in FIA plots includes species, diameter, height, tree class, status, crown class, crown ratio, damage, tree/crown quality, tree grade, volume, and growth. Sampling on FIA plots also includes identification and counts of tree seedlings by species. For each plot, the density of trees by species, basal area by species, and timber volume by species can be determined, as well as changes in stand characteristics. Factors recorded for Phase 2 plots also include stand history, growth, mortality, removals, site index, stocking, slope, elevation, aspect, and physiographic

class. In addition to the Phase 2 data collected, Phase 3 plot data collection includes vegetation composition, abundance, and spatial arrangement, disturbance, crown conditions, soil conditions, lichen community composition, downed woody debris, insects, diseases, and ozone damage. Vegetation species diversity and structure, which allows for a thorough assessment of tree diversity and community type classification (including overstory and understory), and exotic species presence and abundance, are determined for Phase 3 plots.

FIA reports currently include tree height, age, growth, percent cover by species, and other information for tree-stand evaluation. Reports generated from the FIA database can be customized for a selected area and for selected tree species. Evaluation of existing and future FIA data would provide a cost-effective monitoring tool for evaluating the Douglas-fir population within the Forests of the Northern California Province. Use of these data would allow the Forests to monitor Douglas-fir populations without developing a new protocol.

3.2 PONDEROSA PINE

Ponderosa pine is a large tree and typically reaches 39–55 m (130–180 ft) in height and 1.2 m (4 ft) in diameter. Three distinct varieties of ponderosa pine are distinguished, and separated geographically. The typical variety, *P. ponderosa* var. *ponderosa*, occurs in the Pacific coast region. *P. ponderosa* var. *scopulorum* occurs in the Rocky Mountain region, while *P. ponderosa* var. *arizonica* occurs primarily in southeastern Arizona. The latter is also recognized as a separate species, *P. arizonica*.

3.2.1 Distribution, Habitat, and Life History of Ponderosa Pine

Ponderosa pine occurs over a wide range, extending from southern British Columbia to northern Mexico, and from the Pacific coast to central Nebraska. Ponderosa pine occurs primarily in mountains, with the most extensive stands at 1,219 to 2,438 m (4,000 to 8,000 ft) in elevation; however, it may range from sea level in the north to 3,050 m (10,000 ft) in the southern portion of its range. In California, ponderosa pine extends south along the coastal ranges and through the Cascades and Sierra Nevada to near San Diego, ranging in elevation from 150 to 1,070 m (500 to 3,500 ft) in the northern part of the state and 1,610 to 2,230 m (5,300 to 7,300 ft) in the southern portion (USFS 1990a). The occurrence of ponderosa pine within Northern California Province Forests is shown in Figure 3.

Ponderosa pine is a major component of the Interior ponderosa pine forest SAF cover type (SAF type 237), Pacific ponderosa pine–Douglas-fir type (244), and Pacific ponderosa pine type (245). It is present in 65% of all forest cover types in the western continental U.S. (USFS 1990a). Ponderosa pine is a dominant or codominant species in the following CalVeg vegetation alliances: Douglas-fir–pine, eastside pine, ponderosa pine, and ponderosa pine–white fir. Ponderosa pine is also an associate in the mixed conifer-fir, Klamath mixed conifer, mixed conifer–pine, white fir, black oak, bitterbrush (*Purshia tridentata*), and Brewer oak (*Quercus garryana* var. *breweri*) vegetation types. In California, the tree species primarily associated with

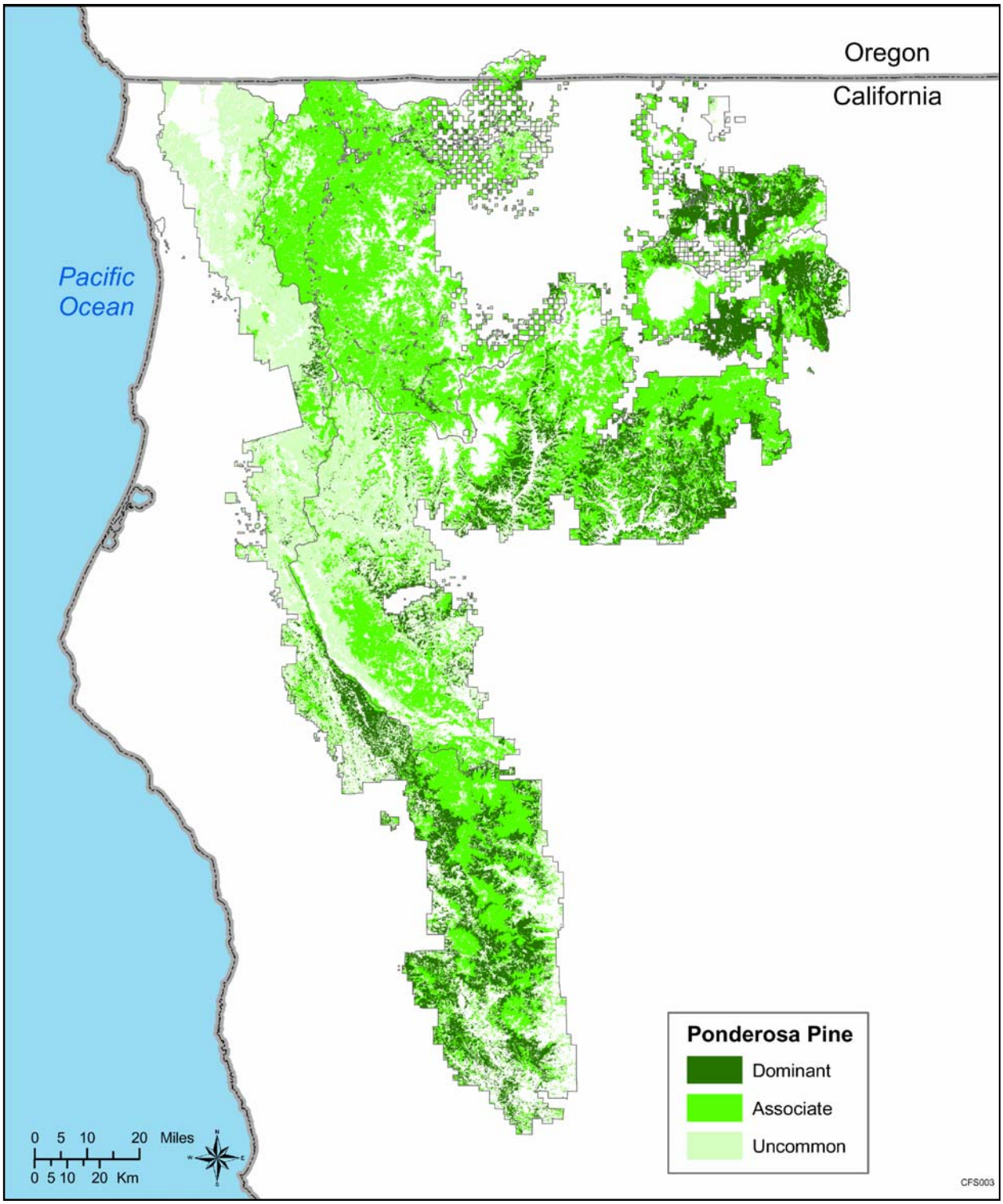


Figure 3 The Occurrence of Ponderosa Pine in the Four Forests of the Northern California Province

ponderosa pine are white fir, incense cedar, Jeffrey pine, sugar pine, Douglas-fir, black oak, and western juniper (USFS 1990a).

The root system of ponderosa pine generally consists of a tap root with lateral branches. Ponderosa pines can reach 600 yr in age. Cone production begins at about seven years and production continues to at least 350 yr, although the highest seed viability is from trees 60 to 160 yr old (USFS 1990a). Heavy cone production occurs every eight years, on average, and medium seed crops are produced every two or three years in California (USFS 1990a; Habeck 1992). The cones open at maturity and seed dispersal may be assisted by rodents, such as chipmunks (*Tamias* spp.), through seed caching. A large proportion of cones and seeds may be consumed by insects and wildlife.

Ponderosa pine has relatively low nutrient requirements and is able to grow on soils too low in nitrogen and phosphorus for other tree species (USFS 1990a). Seed germination, as well as initial seedling survival and growth, is reduced by stress resulting from low soil moisture (USFS 1990a). Older seedlings can better tolerate low soil moisture by rapid root growth and reduced transpiration.

3.2.2 Status, Population Trends, and Factors Affecting Populations of Ponderosa Pine

The ponderosa pine is a common species in the western U.S., and is a commercially important timber tree. This species is often found in large pure stands, but also occurs in mixed coniferous forest. In higher-elevation mesic forests, ponderosa pine is a seral species among other more competitive conifers (USFS 1990a). In some areas, ponderosa pine forests have been replaced by other forest types. Understory species became dominant in these forests due to fire suppression and harvest of ponderosa pine, resulting in a change of forest cover type on about 2 million ha (5 million ac) in the Pacific Northwest between 1965 and 1990 (USFS 1990a). Overall stand density in ponderosa pine forests in portions of the north Cascade Mountains has increased 81% over the past 100 yr, with a maximum from about 1950 to 1960, primarily due to fire suppression (Ohlson and Schellhaas 2000). Increased density creates conditions conducive to catastrophic fire or outbreaks of insects or disease. The proportion of Douglas-fir relative to ponderosa pine in these areas has also increased.

Low soil moisture is often a limiting factor in establishment and growth of ponderosa pine. Summer rainfall is low in many areas supporting ponderosa pine, and may be 2.5 cm (1 in.) or less for July and August (USFS 1990a). Available soil moisture influences the distribution of ponderosa pine in some areas, and can be related to soil texture and depth (USFS 1990a). Ponderosa pine tends to have higher survival and growth rates on coarse-textured sandy soils than on fine-textured clay soils, and less root development in fine-textured soils (USFS 1990a).

Ponderosa pine is browsed by a variety of wildlife, and seedlings, saplings, and pole-sized trees can be heavily damaged or killed; sheep and cattle may also cause damage through trampling, bedding, and browsing (USFS 1990a). Ponderosa pine mortality may result from insect attacks, such as from the western pine beetle (*Dendroctonus brevicomis*), either directly or through introduction of fungal pathogens (USFS 1990a; Habeck 1992; Haverty et al. 1996). Pine

beetles are attracted to slash, resulting from management activities, and fire-injured trees (Mitchell and Sartwell 1974). The presence of slash may be conducive to the increase of bark beetles, and may result in high beetle populations (USFS 1990a). Defoliation of ponderosa pine by pandora moth (*Coloradia pandora*) affects growth and can be related to tree spacing (Cochran 1998). Dwarf mistletoe (*Arceuthobium campylopodium*) may also cause mortality or reduction in growth in some ponderosa pine. Root diseases, resulting from several pathogens, cause tree mortality. Some pathogens, such as *Armillaria mellea* and *Heterobasidion annosum*, can increase in dead roots and can be spread by airborne spores to freshly cut stumps (Nelson and Harvey 1974; USFS 1990a). Mortality may also result from needle cast and rust diseases. Trees infected by the needle cast *Elytroderma deformans* are attacked by bark beetles (USFS 1990a).

The successful establishment of ponderosa pine seedlings is often promoted under the canopies of large black oaks; however, black oak root crown sprouts may outcompete conifer seedlings (Howard 1992). Large numbers of ponderosa pine seedlings can become established following group-selection logging (McDonald and Abbott 1994), although the abundance of ponderosa pine may decline after several decades (McDonald and Reynolds 1999). The survival and growth of young ponderosa pine seedlings can be adversely affected by competing vegetation, such as trees, shrubs, or grasses, which can decrease soil moisture (USFS 1990a; McDonald and Abbott 1994; McDonald and Reynolds 1999; McDonald 1999; McDonald and Fiddler 1999a). Ponderosa pine is generally shade intolerant, and shading from competing conifers can reduce growth of seedlings and saplings considerably (USFS 1990a; McDonald and Abbott 1994). Because of this, ponderosa pine tends to grow in even-aged stands. Thinning of dense stands can result in a good growth response (USFS 1990a; Cochran and Barrett 1998; Cochran and Barrett 1999). High stand densities may result in bark beetle attacks because of weakened trees (USFS 1990a).

Seedlings of ponderosa pine are killed by fire. However, older trees are protected from fire damage by thick bark, and are more tolerant of fire than other associated conifers, such as Douglas-fir, especially sapling and pole size trees (USFS 1990a). Fire effects on survival and growth are minor when up to 50% of the crown is scorched (USFS 1990a). In general, fire-related mortality from crown scorching is less from late-season fires. With periodic fires, ponderosa pine is maintained as a dominant species in mid-elevation stands, whereas understories of other conifers develop as a result of fire suppression. In portions of the Pacific Northwest, the fire regime changed to less frequent, but more severe fires during the twentieth century (McKelvey et al. 1996, Everett et al. 2000, Oliver 2000). In the absence of fire, the composition of stands may change as ponderosa pine becomes reduced and loses dominance, especially following their harvest, resulting in changes of forest cover types to those dominated by other conifer species (USFS 1990a).

3.2.3 Suitability of Ponderosa Pine as a Management Indicator Species

The ponderosa pine satisfies the criteria we established for a suitable management indicator species for Forests of the Northern California Province (see Section 1). Ponderosa pine is an important component of several vegetation types within the Northern California Province

Forests. The vegetation types that support ponderosa pine cover much of the Province, although the species is most dominant in the eastern portion of the Shasta-Trinity National Forest, the southern portion of the Six Rivers National Forest and in Mendocino National Forest (Figure 3). In general, ponderosa pine is less abundant and dispersed across the Province than is Douglas-fir. Ponderosa pine is, however, important for the establishment of habitat characteristics for many species that utilize these vegetation types. Changes in the ponderosa pine population across the Province that result from forest-management activities would thus be expected to result in a change in habitat characteristics across a wide range of vegetation types and a wide area of the Province. Those species that utilize ponderosa pine forest types would likely be affected by such changes in habitat characteristics. Effects would likely be greatest on those species that are dependent on ponderosa pine as a primary habitat component.

The establishment and survival of ponderosa pine, and the persistence of forest types dominated by ponderosa pine, can be strongly affected by forest-management activities (USFS 1990a). The habitat conditions conducive to growth and regeneration of ponderosa pine are affected by a number of factors including stand density, increased shading and decreased moisture availability resulting from competition with other species, and fire regime (USFS 1990a; McDonald and Abbott 1994; McDonald 1999). Forest-management activities can influence ponderosa pine population patterns and trends by affecting seedling establishment and altering habitat conditions. For example, removal of competing tree and shrub species from a stand can increase the survival and growth of seedling and sapling ponderosa pine (USFS 1990a; McDonald and Fiddler 1995; McDonald and Abbott 1997; McDonald and Everest 1996; Fiddler and McDonald 1999; McDonald and Fiddler 1999b). Planting with species other than ponderosa pine would directly influence the representation of ponderosa pine within a stand. The use of prescribed fire as a management tool can be conducive to establishing and maintaining a high level of ponderosa pine presence within a stand (USFS 1990a). Fire suppression, especially when combined with selective harvesting of the ponderosa pine overstory, may result in a reduction in ponderosa pine and loss of ponderosa pine-dominated forest types. Following overstory removal, ponderosa pine can have a low growth response compared to competing understory species (Uzoh et al. 1998). Factors affecting ponderosa pine populations that are not directly related to management activities include disease and insect outbreaks that can result in tree mortality, and weather patterns such as prolonged droughts. Survival of ponderosa pine can be affected by harvesting practices that may result in an increase in fungal infections or insect infestations in remaining mature or seedling trees (Nelson and Harvey 1974; Mitchell and Sartwell 1974; USFS 1990a). For example, slash remaining after tree harvest can result in high populations of bark beetles.

3.2.4 Monitoring Protocols for Ponderosa Pine

The status of ponderosa pine as a component or dominant species of forest types within the Northern California Province can be effectively monitored by evaluation of data currently collected under the FIA program, which is a research program of the USFS Pacific Northwest Research Station (USFS 2001). The FIA monitoring program is designed to collect information on forest composition and quality and detect changes in forest parameters over time. Changes in the ponderosa pine population that result from forest-management activities including, for

example, changes in distribution across the Province, within vegetation types, or age distribution could be detected using the monitoring protocols of the FIA program.

The FIA program currently monitors vegetation on permanent plots across the National Forest lands in California, including the four Northern California Province Forests (USFS 2002). The FIA monitoring protocols are discussed in Section 3.1.4. Data collection on FIA plots includes tree species present, density of trees by species, basal area by species, and timber volume by species. Herbaceous layer sampling on FIA plots includes identification of tree seedlings. FIA data are collected at 5- to 10-yr intervals. FIA reports currently include tree height, age, growth, percent cover by species, and other information for stand evaluation. Reports generated from the FIA database can be customized for a selected area and for selected species. Evaluation of FIA data would provide a monitoring tool for evaluating the ponderosa pine population within the Forests of the Province. A more complete description of the FIA program is provided in Section 3.1.5.

3.3 BLACK OAK

The black oak, also called the California black oak, is a large deciduous oak generally reaching 24 m (80 ft) in height and 1.4 m (4.5 ft) in diameter, although large individuals may reach a height of 36 m (120 ft) and a diameter of 1.6 m (5 ft). A scrub form of black oak may occur on poor sites (USFS 1990b).

3.3.1 Distribution, Habitat, and Life History of Black Oak

The range of black oak is somewhat limited, extending from southwest Oregon to southern California and into Baja California (USFS 1990b; Howard 1992). It occurs in valleys, foothills, and lower mountains of the Coast Ranges and Sierra Nevada, generally from 60 to 2,438 m (200 to 8,000 ft) in elevation. In north-central California, black oaks below 305 m (1,000 ft) elevation are found primarily on north slopes or in sheltered draws, while those above 1,067 m (3,500 ft) are generally on south and west slopes (USFS 1990b). The most extensive development of the black oak forest cover type is in the southern Cascades, Klamath Mountains, northern portions of the Sierra Nevada and Coast Ranges, from 455 to 915 m (1,500 to 3,000 ft) (Eyre 1980). Black oak is found in California in the northern Coast Range south to Marin County, intermittently in the Santa Cruz and Santa Lucia Mountains, and more commonly in the San Bernardino, San Jacinto, and Agua Tibia Mountains (USFS 1990b). Black oak is abundant on the west side of the Sierra Nevada in California, and becomes intermittent south of Kings Canyon, and increases in the Tehachapi Mountains (USFS 1990b). Black oak currently exceeds 5% of stand basal area on 25% (1,748,685 ha [4,321,000 ac]) of Forest Service lands in California (Gaman and Casey 2002). The occurrence of black oak within Northern California Province Forests is shown in Figure 4.

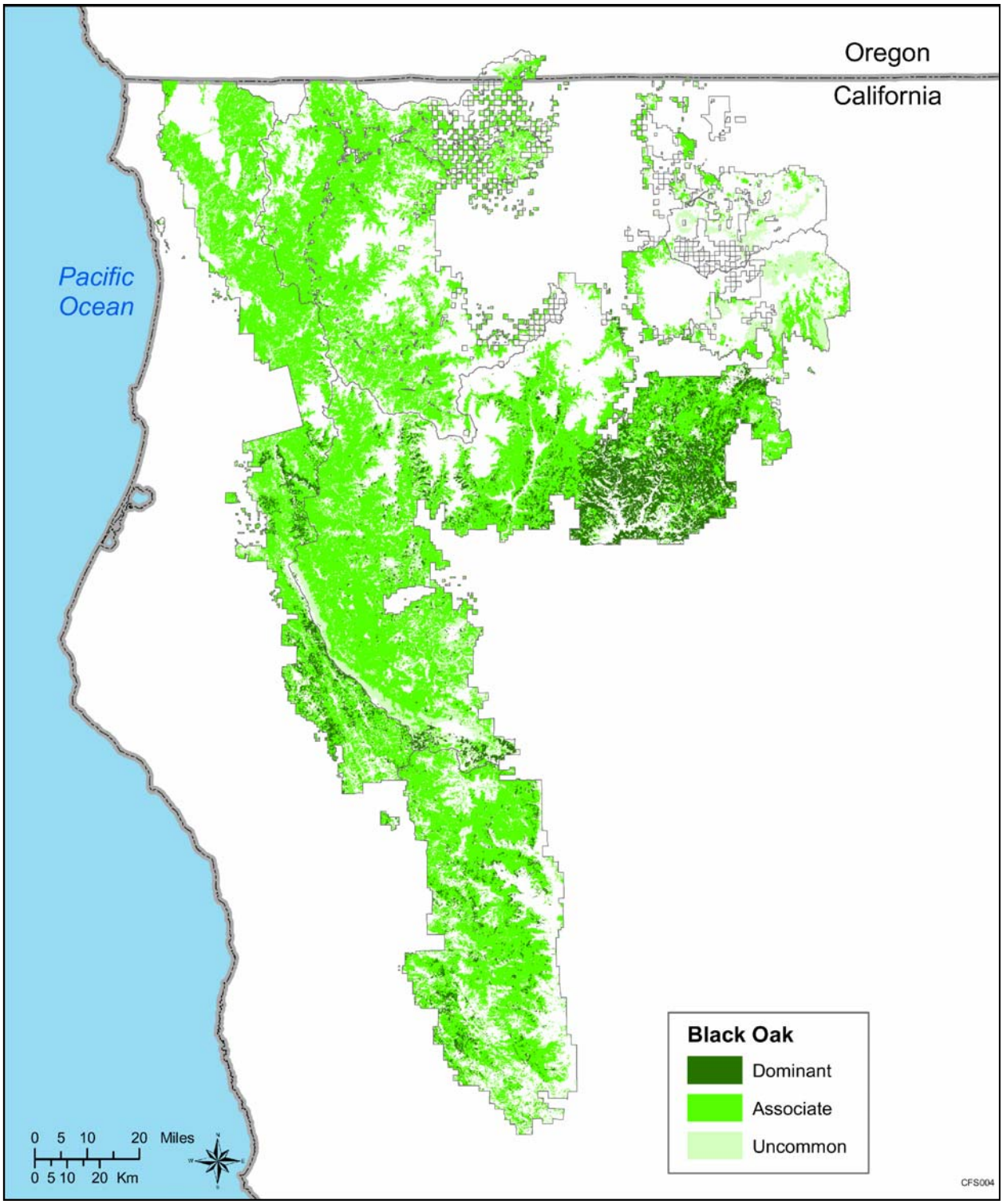


Figure 4 The Occurrence of Black Oak in the Four Forests of the Northern California Province

Black oak typically grows on dry, well-drained, sandy, gravelly, and rocky soils, ranging from medium-textured loams and clay-loams to sandy-loams and gravelly-clay loams, with lower occurrence on clay soils and best growth on deep, medium- and coarse-textured, slightly acid loams (USFS 1990b, Howard 1992). It often occurs in mixed conifer forests or hardwood stands, or may occur in nearly pure stands. Scattered pure stands may occur within coniferous forests, and sites supporting pure stands of black oak are usually unfavorable for the growth of conifers, or are subject to recurring disturbance such as fire or logging (Howard 1992). Stands are usually a single age class, as a result of resprouting and growth after fire (USFS 1990b).

At locations supporting black oak in California, annual rainfall ranges from 30 to 38 cm (12 to 15 in.), in northwest California, to 76 to 254 cm (30 to 100 in.) in northeast California, but may exceed 292 cm (115 in.) locally (USFS 1990b). The greatest size and abundance of black oak are found where 10 to 50% of annual precipitation is in snowfall (USFS 1990b).

Black oak is the primary component of the California black oak SAF cover type (246), and a major component of the Douglas-fir–tanoak–Pacific madrone (234) and the Pacific ponderosa pine–Douglas-fir (244) cover types. It also occurs in the Pacific Douglas-fir (229), Port Orford cedar (231), redwood (232), Oregon white oak (233), Sierra Nevada mixed conifer (243), Pacific ponderosa pine (245), Jeffrey pine (247), knobcone pine (248), canyon live oak (249), blue oak–gray pine (250), and California coast live oak (255) cover types (USFS 1990b; Howard 1992).

Black oak is a dominant or codominant species in the black oak and productive mixed hardwood CalVeg vegetation types, and is typically an associate in the Pacific Douglas-fir, Douglas-fir–pine, knobcone pine, mixed conifer–pine, ponderosa pine, blue oak, Oregon white oak, madrone, valley oak, bitterbrush, and low sagebrush vegetation types. Additional commonly associated overstory species are incense cedar, tanoak, interior live oak (*Quercus wislizenii*), Pacific dogwood (*Cornus nuttallii*), and bigleaf maple (*Acer macrophyllum*) (USFS 1990b).

Black oak may live up to 500 yr. Although acorn production begins at about 30 yr of age, heavy production generally begins at 80 to 100 yr and may average 6,500 acorns per year for a 150 to 200-yr-old tree (USFS 1990b). The root system of black oak consists of one to several tap roots, with large associated lateral roots, which may reach to bedrock, and a number of surface roots (USFS 1990b; Howard 1992).

3.3.2 Status, Population Trends, and Factors Affecting Populations of Black Oak

The black oak is one of the predominant species of hardwoods in California. The range-wide population of black oak has greatly decreased as a result of human activities, including logging practices, fuelwood harvest, and fire suppression, as well as drought and animal foraging (Howard 1992; Fritzke 1997). The overall decline in the black oak population has not been quantified.

Black oak is browsed by mule deer (*Odocoileus hemionus*) and livestock, and the acorns are consumed by a wide variety of wildlife, such as the acorn woodpecker, as well as livestock

(Howard 1992). Black oak is highly susceptible to fungal pathogens, such as those that cause heart rot (*Inonotus dryophilus* and *Laetiporus sulphereus*), root rot, or butt rot (Howard 1992). Root rot and butt rot can be caused by *Armillaria mellea*, which attacks older and fire-damaged trees (USFS 1990b; Howard 1992). Several leaf diseases, including oak leaf fungus (*Septonia quercicola*) and oak anthracnose (*Gnomonia veneta*) attack black oak, and infestations of mistletoe (*Phoradendron villosum* ssp. *villosum*) are frequent (USFS 1990b). Sudden oak death, caused by the pathogen *Phytophthora ramorum*, has recently caused high levels of mortality in black oak in some areas of California (Kelly 2002).

Survival and growth of black oak seedlings has generally been poor on planted sites (McDonald and Tappeiner 2002). Where black oak is planted for habitat restoration or hardwood production, fall-planted acorns result in the best seedling establishment, but replanting is generally necessary due to high levels of herbivory and acorn predation, unless wire caging is used (Howard 1992; McDonald and Tappeiner 2002). Some acorns may require overwinter stratification before germination, while others may germinate immediately (Howard 1992). Acorns are highly susceptible to fungal attack, and many are consumed by insects, wildlife, and livestock. Percent germination varies considerably, and ranges from 20 to 93%, for example, in San Bernadino National Forest (Howard 1992). Burial of acorns by seed-caching rodents (e.g., California ground squirrel, *Citellus beecheyi*) or birds (e.g., Steller's jay, *Cyanocitta stelleri*) increases germination rates and seedling-establishment rates (Howard 1992).

Undisturbed leaf litter, a light duff, or moist, well-aerated mineral soil are conducive to seedling germination and establishment; heavy clay soils or compacted soils (such as from logging) prevent seedling establishment (USFS 1990b; Howard 1992; McDonald and Tappeiner 2002). Emerging seedlings are often eaten by deer or killed by drought (Howard 1992). The presence of coarse woody debris may provide some protection of oak seedlings, as well as maintain moisture and nutrient levels (Tietje et al. 2002). Initial growth during the first six to seven years is primarily in the vertical roots, while shoot and horizontal root development are slow (USFS 1990b; Howard 1992).

Black oaks grow well in full sunlight and are relatively intolerant of shade, although it can persist in shade at early stages (USFS 1990b). Black oak saplings are relatively intolerant of dense shade, growing tall and thin to reach sunlight. Intolerance increases with age, and overtopping by other trees generally results in mortality (USFS 1990b). Where fire suppression allows encroachment by shade-tolerant species, black oak establishment has decreased (Fritzke 1997). The ability of black oak seedlings to compete with conifer seedlings is poor, and growth generally does not over-top young ponderosa pine, however black oak seedlings are able to grow through chaparral brush (USFS 1990b; Howard 1992). Competing vegetation may result in seedling mortality (USFS 1990b).

Black oak stands can be established from plantings in clearcut areas, and stand productivity is promoted by thinning which is recommended when trees are 9 to 15 m (30 to 50 ft) tall or when tree density exceeds 29 m^2 of basal area per ha (125 ft^2 per ac)(Howard 1992). Thinning has been shown to greatly increase diameter growth of black oak (USFS 1990b).

Crown fires usually top-kill black oak trees in a stand and can completely kill pole-size trees (with a crown diameter of 4.6 to 9.1 m [15 to 30 ft]) and smaller (Howard 1992). The thin bark of young trees makes them highly vulnerable to fire damage, while moderately thick (2 to 5 cm [1 to 2 in.]) bark on larger trunks provides some fire resistance. The presence of brush surrounding black oak often contributes to complete kill (Howard 1992). Effects of surface fires on black oak are related to fire intensity and tree size (Howard 1992). Severe surface fires result in complete kill of many black oaks, while moderate-severity surface fires result in localized damage on larger trunks, completely kill roughly half of young trees, and top-kill of most of those remaining. Low-severity fires result in localized damage to trees that are pole-size and under. Greater damage to trees occurs from fires during the spring active growing season.

Following cutting or burning, black oak re-sprouts extensively from the stump or root collar, and larger trees produce a larger number of sprouts and more vigorous sprouts; very old trees may not produce new sprouts (USFS 1990b; Howard 1992). Short-cut stumps produce more sprouts than high-cut stumps, which in larger trees may produce stool sprouts (stems grown from stump sprouts above the root collar) which are prone to heart rot at an early age (USFS 1990b). A large root system allows for vigorous re-sprouting even during the dry season (Howard 1992). Root-crown sprouts grow rapidly in full sunlight. Sprout growth in shelterwood cuts has been observed to be lower than that in clearcuts, partially due to effects of cynipid gall wasps (USFS 1990b). Sprouting black oaks grow faster than other woody species, including conifers, and are dominant for many years; however, aggressive shrub species may be strong competitors with re-sprouting oaks after a fire or cutting (USFS 1990b).

Light- or moderate-intensity fire produces soil conditions suitable for seed germination and seedling establishment by reducing the amount of litter, creating a more open canopy which promotes seedling and new shoot growth (USFS 1990b; Howard 1992). In addition to litter removal, light fires promote seedling establishment by reducing damaging molds and insects in litter (Howard 1992; McDonald and Tappeiner 2002). Following fire, decreases in insect predation of developing acorns, increases in seedling density, and decreases in sapling mortality due to root rot have been observed (USFS 1990b; Howard 1992). Historic fire regimes which supported black oak were generally light to moderate-intensity surface fires at 3.5-yr intervals (Howard 1992).

In the absence of fire, shade-tolerant woody species tend to increase. Lacking disturbance, black oaks may eventually become only scattered remnants in a mixed conifer forest (USFS 1990b). Black oaks in Forests in California often occur among fast-growing conifers, gradually lose access to direct sunlight, and may have difficulty retaining their vigor (Gaman and Casey 2002). Large black oaks may function as nurse trees, and provide conditions for the establishment and growth of conifer seedlings, which eventually may attain dominance or codominance in the stand (USFS 1990b). Shade-tolerant conifers appear to out-compete black oaks, as a result of fire suppression and possibly climate change (Gaman and Casey 2002). Fuel buildup of downed woody debris, along with an increased density of understory brush and sapling conifers, results in an increase in crown fires. Where fire suppression has resulted in dense forests, clearing of the understory prior to prescribed fire is used to promote black oak recruitment (Howard 1992). Late fall, low-intensity prescribed fires are likely to have the most

favorable effect on black oaks because they result in lower levels of tree loss and better seedling establishment (Howard 1992).

3.3.3 Suitability of Black Oak as a Management Indicator Species

The black oak satisfies the criteria we established for a suitable management indicator species for the Forests of the Northern California Province (see Section 1). The black oak is a dominant species in two vegetation types, and a frequent associate in many other types covering extensive areas of the four Northern California Province Forests. The vegetation types that support black oak cover much of the Province, but it is dominant only in the eastern portion of the Shasta-Trinity National Forest, and scattered areas of the Six Rivers and Mendocino National Forests (Figure 4). The black oak is, however, an important habitat component for many species that utilize these vegetation types. Changes in the black oak population across the Province, as a result of forest-management activities, would be expected to result in a change in habitat characteristics across a wide range of vegetation types and a wide area of the Province. Those species that utilize the black oak forest types would likely be affected by such changes in habitat characteristics, and would thus be affected by the management activities. Effects would likely be greatest on those species that are dependent on the black oak as a food source.

The establishment and survival of black oak can be affected by forest-management activities. The habitat conditions conducive to growth and regeneration of black oak are affected by a number of factors including stand density, competition with other species, and fire regime. Forest-management activities can influence black oak population patterns and trends by affecting seedling establishment and altering habitat conditions. For example, soil compaction from logging equipment can reduce or prevent seedling establishment.

Management activities may include re-establishing black oak, through planting, in areas formerly occupied by black oak stands or planting within old nonregenerating stands. Removal of competing tree species from a stand can increase the survival and growth of seedling and sapling black oak. The use of prescribed fire as a management tool can be conducive to establishing and maintaining black oak within a stand. Frequent, light-intensity ground fires can reduce competition and create suitable conditions for seedling establishment; however, severe fires can result in black-oak mortality. Fire suppression may result in a reduction of black oak in mixed stands and losses of the black oak forest type. Some harvesting techniques, such as leaving high-cut stumps, can result in a decrease in the health and survival of regrown trees in a black oak stand. Factors affecting black oak populations that are not directly related to management activities include disease outbreaks, such as sudden oak death, that can result in a high level of tree mortality, and weather patterns such as prolonged droughts.

3.3.4 Monitoring Protocols for Black Oak

The status of black oak as a component or dominant species of forest types within the Province can be effectively monitored by evaluation of data currently collected under the FIA program, which is a research program of the USFS Pacific Northwest Research Station. The FIA

monitoring program is designed to collect information on forest composition and condition and detect changes in forest parameters over time. Changes in the black oak population that result from forest-management activities including, for example, changes in distribution across the Province, within vegetation types, or age distribution could be detected using the monitoring protocols of the FIA program.

The FIA program currently monitors vegetation on permanent plots across the Forest lands in California, including the four Forests in the Northern California Province. The monitoring protocols used in the FIA program are described in Section 3.1.4. Data collected on FIA plots include tree species present, density of trees by species, basal area by species, and timber volume by species. Herbaceous layer sampling on FIA plots includes identification of tree seedlings. FIA data are collected at 5- to 10-yr intervals. FIA reports currently include tree height, age, growth, percent cover by species, and other information for stand evaluation. Reports generated from the FIA database can be customized for a selected area and for selected species. Evaluation of FIA data would provide a monitoring tool for evaluating the black oak population within the Province.

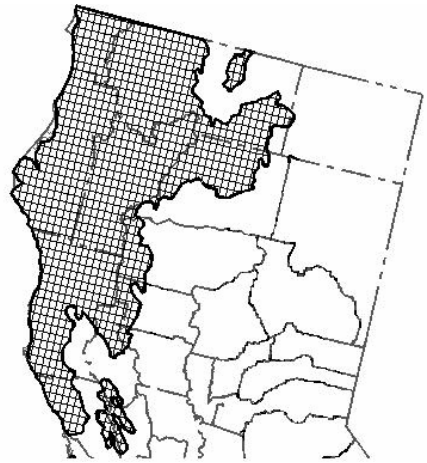
3.4 NORTHERN SPOTTED OWL

Three subspecies of spotted owl are recognized: California spotted owl (*Strix occidentalis occidentalis*), Mexican spotted owl (*Strix occidentalis lucida*), and northern spotted owl (*Strix occidentalis caurina*). The northern spotted owl occurs throughout the four Forests of the Northern California Province. It is a medium-sized owl that is darker brown with smaller white spots and markings than the other spotted owl subspecies. Males are 18.6 to 33.5 cm (7.3 to 13.2 in.) long and weigh 460 to 690 g (16.2 to 24.3 oz), while females are 22.8 to 39.9 cm (9.0 to 15.7 in.) long and weigh 540 to 850 g (19.0 to 30.0 oz) (Gutierrez et al. 1995).

3.4.1 Distribution, Habitat, and Life History of the Northern Spotted Owl

The northern spotted owl occurs from southwestern British Columbia, Canada, through the Coast and Cascade Ranges (both west and east sides) of Washington and Oregon, south into southwestern Oregon and northwestern California north of San Francisco (Tesky 1992). Even though there have been large decreases in the abundance of the spotted owl since the late 1800s, the current geographic distribution of all three subspecies appears to be similar to their historic distribution (Noon and Franklin 2002).

Range of the Northern Spotted Owl in Northern California (CDFG 2002)



The distribution of the northern spotted owl is strongly tied to habitat conditions. Spotted owls use mature or old-growth conifer forests

significantly more than expected relative to their availability within their home ranges in northwestern California (Solis and Gutierrez 1990). In the southern portion of the Province, especially on the Mendocino National Forest, spotted owls and their suitable habitat are more scattered than in the northern part of the Province due to both natural conditions and recent timber harvest. Nevertheless, the spotted owl populations remain well distributed throughout most of the Province (U.S. Fish and Wildlife Service [FWS] 1995). Confirmed sightings of spotted owls and spotted owl habitat based on data developed by the Regional Ecosystem Office (REO 2003) are plotted in Figure 5.

Habitat of the Northern Spotted Owl

Spotted owl habitat consists of four components: (1) nesting, (2) roosting, (3) foraging, and (4) dispersal. While the northern spotted owl is considered a late-successional and old-growth forest species, the age of a forest is not as important for determining habitat suitability as the structure and composition of the forest. Components of preferred habitat for northern spotted owl are: 60 to 80% canopy closure; a multilayered, multispecies canopy dominated by large overstory trees (> 76 cm [30 in.] dbh); a high incidence of large trees with various deformities; numerous large snags; large accumulations of logs and other woody debris; and sufficient open space below the canopy for flying (Tesky 1992; FWS 1995). These characteristics tend to become more common as the forest ages (LaHaye and Gutierrez 1999). The specific characteristics of each attribute (e.g., multilayered and multispecies canopy, canopy closure, mistletoe infestation, and the presence of remnant large trees and snags) vary across the spotted owl's range (Everett et al. 1997).

The median home range size for the northern spotted owl is as large as 2,000 ha (5,000 ac), depending in part on the amount of suitable foraging habitat and geographic location (Tesky 1992). The mean summer home-range size in northwestern California was 413 ha (1,020 ac), with males having smaller mean home-range size than females (338 ha and 538 ha, respectively) (Solis and Gutierrez 1990). Spotted owls may incorporate large tracts ($\geq$ 400 ha [990 ac]) of old-growth forest into their home ranges (Franklin 1992).

Most spotted owls share some portion of their home range with other spotted owls. On average, adjacent males and females shared 52.1% of their home ranges. In contrast, adjacent females shared only 10.4% of their home ranges while adjacent males shared only 2.3% of their home ranges. Overlap between members of a pair was greatest at 62.4% of their home ranges (Solis and Gutierrez 1990). The shared areas between territories are generally the periphery of home ranges where owls spent a relatively small proportion of their time (Forsman et al. 1984). Territory core areas are about 1.6 km^2 (0.6 mi^2) relative to the expected home range size in northwestern California of about 4.2 to 5.9 km^2 (1.6 to 2.3 mi^2; Franklin et al. 2000). Habitat fragmentation tends to result in an increase in home range size (NatureServe 2003). Winter home range size is generally larger than summer, but the habitats used are structurally similar (Gutierrez 1985).

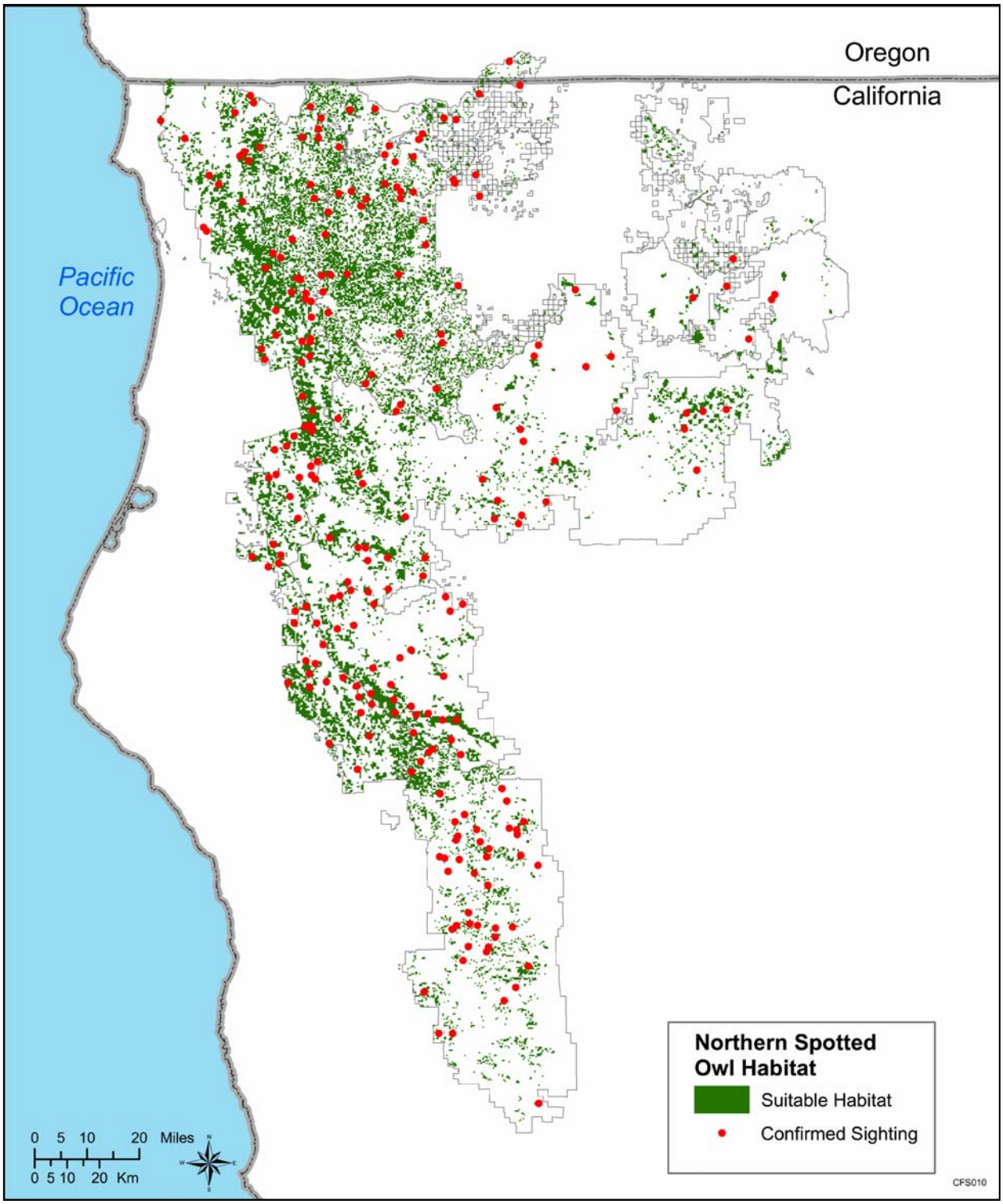


Figure 5 Northern Spotted Owl Habitat and Confirmed Sightings in the Forests of the Northern California Province. Source: REO (2003)

In California, the northern spotted owl most commonly occurs in Douglas-fir and mixed-conifer forests, but also occurs in forests dominated by other species including redwood and ponderosa pine. Understory layers in spotted owl habitats are often dominated by shade-tolerant trees such as western hemlock (*Tsuga heterophylla*), western redcedar (*Thuja plicata*), grand fir (*Abies grandis*), white fir, Pacific yew (*Taxus brevifolia*), vine maple (*Acer circinatum*), canyon live oak, California bay (*Myrica californica*), and tanoak (Tesky 1992). The stands used in California have high canopy closure (about 80 to 90%) and are old (> 150 yr) with a high degree of stand decadence (Gutierrez 1985).

Although the northern spotted owl generally occurs in old-growth conifers (> 200 yr old) or mixtures of mature (100 to 200 yr old) and old-growth conifers (Tesky 1992), it also may use areas with extensive cover of young trees (< 100 yr). Population density is generally lower in younger forests compared to older forests (Herter et al. 2002). Most of these younger forests have significant remnants of earlier stands that were influenced by fire, wind storms, or inefficient logging operations. The owls tend to nest and roost in the older components of these stands. No northern spotted owls have been located in forests younger than 36 yr old (Forsman et al. 1984).

Habitat use by spotted owls in the Northern California Province is similar to that reported for comparable studies in Oregon and Washington (Hunter et al. 1995). Spotted owl nest and roost sites in northwestern California were characterized by lower amounts of nonvegetated and herbaceous land cover, and greater amounts of mature and old-growth conifer forest which was less fragmented than random sites. Mean amounts of mature and old-growth forest within 200 ha (494 ac) radius plots were 94.1 ha (232.5 ac), 92.0 ha (227.3 ac), and 71.8 ha (177.4 ac) for nest, roost, and random sites, respectively (Hunter et al. 1995). Spotted owls in the Six Rivers National Forest used complex, structured forests that had a dominant canopy of large (> 52.5 cm [20.7 in.] dbh) conifer trees for roosting and foraging (Solis and Gutierrez 1990).

About 25% of the area of the Northern California Province Forests is believed to be suitable for nesting, roosting, and foraging of the spotted owl. Generally, this habitat consists of conifer or mixed conifer–hardwood forests with multiple canopy layers; multiple overstory conifers > 40.6 cm (16 in.) dbh; and total canopy closure among dominant, codominant, and understory trees of > 60%. Some nest sites may occur in stands of smaller trees or with a lower canopy closure; however, such sites are not typical (FWS 1995). Owl site centers on non-Federal lands are usually found in remnant stands of older forest or in younger forests that have had time to regenerate after harvest. In addition, adjacent non-Federal lands can provide foraging and dispersal habitat for owls whose site centers are on Federal lands (FWS 1995).

The northern spotted owl does not build a nest. Instead, it uses cavities in large-diameter trees with broken tops, cliff cavities, and abandoned hawk or raven nests (Tesky 1992). The brooms caused by dwarf mistletoes (*Arceuthobium* spp.) that form platforms are also used by the northern spotted owl for nest sites. Those used by the northern spotted owl are generally the larger and denser brooms located away from the stand edge (Bull et al. 1997). A specific type of nest structure is not required; rather it must be a structure large and stable enough to support a female and her offspring (LaHaye and Gutierrez 1999). Cavity nests provide a more secure environment for the young (e.g., prefledged young are less likely to fall or jump from the nest)

(Forsman et al. 1984). Eighty-one percent of all nests in northwestern Oregon were in cavities compared to only 50% in the Klamath Mountains and on the east slope of the Cascades. These differences appear to reflect regional differences in the availability of different nest types (e.g., mistletoe brooms did not occur in western Oregon) (Forsman et al. 1984).

Within northwestern California, northern spotted owls were found to nest in eight different tree species, although 83% of nests were located in Douglas-firs. Sixty percent of nests were located in broken-top trees, whereas cavity and platform nests each accounted for 20% of nests. Nests are usually located 9 to 55 m (30 to 180 ft) above the ground (CDFG 2003a). Minimum nest tree age averaged 288 yr (range 75 to 688). Nest sites were located at elevations of 36 to 1,507 m (118 to 4,944 ft). Over 70% of nests were at elevations between 500 and 1,200 m (1,640 and 3,937 ft). The upper elevation limit corresponds to the ecotone separating midelevation forests from subalpine forests. This boundary is probably related to the availability of suitable habitat and prey rather than to physiological limits (LaHaye and Gutierrez 1999). Most nests are located on the lower portions of slopes. These areas are generally more productive, fire may be less frequent, and surface water is more likely to be available. The mean distance to water from nest trees was 117 m (385 ft) (LaHaye and Gutierrez 1999).

On managed timberlands in northwestern California, spotted owl nest sites occurred primarily on young (31 to 60 yr old) forests of redwood and Douglas-fir (Folliard et al. 2000). Landscapes near nests had a greater amount of total edge. In addition, nest site locations were located lower on slopes. In this particular study, dusky-footed woodrats (*Neotoma fuscipes*) were found to be the primary prey species. Edges may provide the opportunities for owls to prey on woodrats that are abundant in early seral habitats (Folliard et al. 2000).

The attrition rate of nest trees can be high (e.g., 17% of nests examined from 1970 through 1978 were no longer usable by the end of 1978) (Forsman et al. 1984). Individual spotted owl pairs in the nine-year study used as many as five different nest trees (within a 10-ha [25-ac] area) (Swindle et al. 1999).

Roost sites are typically areas of relatively dense vegetation (high canopy closure dominated by large-diameter trees and multiple-canopy layers). During the summer these sites are usually cool, shady spots near streams or are on the lower third of slopes (Gutierrez et al. 1995). Of the 645 owls observed in day roosts, 640 were perched on limbs in trees or large woody shrubs and five were perched on limbs or logs on the forest floor. All roosts were in forests. Over 90% of the roosts were in old-growth forests (Forsman et al. 1984).

Spotted owls respond to variation in temperature and exposure by moving within the canopy to find favorable microclimate. The multilayered stand structure of roost sites facilitates this movement. Therefore, spotted owls may require old-growth forests to avoid heat stress (Gutierrez et al. 1995). Owls roosted significantly higher in the forest canopy during cold, wet weather than during warm or hot weather (Forsman et al. 1984). Use of areas located far from nest sites is energetically more feasible during winter when owls do not have to return to the nest one or more times each night. Use of larger home ranges in winter may also be a response to gradually declining prey populations (Forsman et al. 1984). During warm or hot weather, owls usually roosted low in the forest understory in small trees or shrubs, thereby reducing their

exposure to solar radiation and high temperatures. When it was cold, raining or snowing, owls roosted significantly higher in the forest overstory (Forsman et al. 1984). Spotted owls need adequate thermal cover in all seasons. Many areas of potential spotted owl habitat may not provide relief from heat without old growth (Gutierrez 1985).

The northern spotted owl may move from foraging areas to well-shaded sites in or below the canopy to meet thermoregulatory needs, avoid predation, and avoid harassment by corvids and other passerines. Roosting sites may not reflect the same habitat qualities necessary to meet forage or reproductive needs (Herter et al. 2002). Owls tend to roost in small trees in the forest understory during warm weather and high up in large trees during cold or wet weather; both types of roosts are provided by a layered canopy structure (Tesky 1992). Daytime roosts on average are 165 m (550 ft) from water.

During the nonbreeding season, roost sites selected by the northern spotted owl occurred in older forest more often than young forest. They selected sites lower in elevation, with larger tree diameter, fewer trees per ha, greater canopy cover, less cover of low shrubs, and less downed wood compared to random locations. Females used old-growth and mature forest to a greater degree than males. The northern spotted owl may select these roost sites for characteristics that facilitate the capture of prey (e.g., ease of movement and unimpeded access to the forest floor). However, there was some use of younger forests during the nonbreeding season, particularly those with characteristics in common with old forests (e.g., widely spaced trees, dense canopies, and several shrub layers) (Herter et al. 2002).

The spotted owl uses a wider variety of forest types for foraging than for nesting and roosting. While habitat that meets nesting and roosting can also provide foraging habitat, some foraging habitat may be inadequate for nesting and roosting. This is especially true in the southern, lower elevations of the northern spotted owl's range where it often forages along edges of dense forests and in more open forests (FWS 1995). Although variable, foraging habitat is generally characterized by high canopy closure and complex structure. The smaller males appear to use stands that have a higher tree density than stands used by the larger females (Gutierrez et al. 1995).

Spotted owls showed a strong preference to forage in unlogged, old-growth forests. Areas that had been clearcut or burned within the previous 20 yr were rarely used for foraging. Although second-growth forests were not preferred for foraging, some owls did spend as much as 36% of their time foraging in such forests. This suggests that young forests provide at least marginal foraging habitat after reaching 25 to 35 yr of age (Forsman et al. 1984). Snag volume and tree height class diversity (a measure of canopy layering) were the stand structures significantly associated with owl foraging intensity (North et al. 1999).

Dispersal habitat consists of forest stands with adequate tree size and canopy closure to provide some degree of protection from avian predators and to allow owls to forage at least occasionally (FWS 1995). In western Oregon, older forest (old-growth and mature stands) was the vegetation type most frequently used during dispersal (35%) and colonization (61%), and spotted owls selected closed-canopy forests over open-canopy forest for both activities (Miller et al. 1997). Habitat that allows spotted owls to disperse may be unsuitable for nesting,

roosting or foraging, but provides an important linkage among blocks of nesting habitat both locally and over the range of the spotted owl (FWS 1995). Dispersal habitat in the Northern California Province generally consists of conifer or mixed conifer–hardwood forests, with smaller dominant trees or lower canopy closure than for nesting, roosting, and foraging habitat; multiple canopy layers, with multiple large overstory conifers greater than 25 cm (10 in.) dbh; and a canopy closure among dominant, codominant; and understory trees greater than 40% (FWS 1995).

Life History of the Northern Spotted Owl

Northern spotted owls usually do not breed before the age of three, although they can breed as early as one year of age. In a sample of radio-tagged owls, 22% of males and 44% of females were paired at one year of age, but only 1.5% of males and 1.6% of females were actually breeding at one year of age. At two years of age, 68% of males and 77% of females were paired, but only 5.4% of males and 2.6% of females were breeding (Forsman et al. 2002). Nesting may not occur every year; some owls do not breed over periods of five to six years (Gutierrez et al. 1995).

Clutch size of northern spotted owls is typically two eggs, but ranges from one to three eggs. The incubation period is 30 ± 2 days (Forsman et al. 1984). The female incubates the eggs for about 30 days and broods the young, while the male provides most of the food for the young. Northern spotted owls fledge in May or June, three to five weeks after hatching. Fledglings are fed by both parents until August or September. Young owls may leave the nest before they are fully capable of flight to escape increasing numbers of parasites in the nest (Forsman et al. 1984).

After leaving the nest, siblings stay together for the rest of the summer, and usually remain near the nest (Forsman et al. 1984). By October, the young become independent and disperse from the nest area. Of 386 juveniles that were radio-tagged, at least 188 (49%) died before or during dispersal. For those that cause of death could be inferred, 68% died from predation, 26% from starvation, and 6% from accidents. Eighty-one percent of predations were caused by birds and 19% by mammals or unknown predators. However, some of the mortality attributed to predators may have actually been scavenging of owls that died from starvation (Forsman et al. 2002).

Many juvenile spotted owls move long distances during their first fall and spring and frequently leave the area under study (Bart 1995). The most common pattern observed for dispersing juveniles was for them to move rapidly away from the natal site during September through November. After the initial surge of movement, most individuals settled in one or more temporary home ranges before eventually acquiring territories (Forsman et al. 2002). Dispersal distances greater than 100 km (60 mi) are rare for the northern spotted owl (Forsman et al. 2002).

A minimum of 6% of the nonjuvenile owls in Oregon and Washington changed territories each year (breeding dispersal). The likelihood of breeding dispersal was higher for females, young owls, owls that did not have a mate in the previous year, and owls that lost their mate from the previous year. Median distances dispersed by adults (3.5 km [2.2 mi], no significant

difference between sexes) were shorter than for juveniles (13.5 km [8.4 mi] for males and 22.9 km [14.2] for females) (Forsman et al. 2002). They do not acquire their own territories until they are about two or three years old. Average life span is 15 to 20 yr (Tesky 1992; Gutierrez et al. 1995).

The northern spotted owl generally exhibits high, relatively constant adult survival rates, and low variable recruitment rates. Survival of territory holders and recruitment into the territorial populations are negatively correlated. The long reproductive life-span of the northern spotted owl allows for some recruitment of offspring by an owl pair even if that recruitment does not occur every year (Franklin et al. 2000).

The northern spotted owl is nonmigratory and adults generally occupy the same home range year-round (Forsman et al. 1984; Gutierrez et al. 1995). Following the breeding season, mating pairs spend increasing amounts of time apart and generally expand their foraging area (Herter et al. 2002). In February or early March (33 to 68 days before eggs are laid), the resident male and female on each territory begin to roost together near the eventual nest site (Forsman et al. 1984).

The northern spotted owl generally remains inactive during the day, although it will occasionally attempt to capture prey near the roost site, retrieve cached prey, change roost trees, or drink and bathe at streams near the roost (Forsman et al. 1984). The spotted owl is a sit-and-wait predator, and when not actively nesting or raising young, will remain in a stand used for foraging for more than a day (Herter et al. 2002). The species forages primarily at night. It will move to several different sites to forage within a single night; the rate of movement is highest during spring and summer (Gutierrez et al. 1995).

Prey species include the dusky-footed woodrat, red tree vole (*Phenacomy*s *longicaudus*), northern flying squirrel (*Glaucomys sabrinus*), terrestrial voles (*Microtus californicus*, *M. oregoni*, and *Clethrionomys californicus*), and white-footed mice (*Peromyscus maniculatus*, *P. boylii*, and *P. truei*; Ward et al. 1998). The northern spotted owl also eats small birds, bats, and large arthropods. It may cache excess food (CDFG 2003a). The mixture of arboreal and terrestrial species in the diet of spotted owls indicates that owls forage at all levels of the forest, from the ground to the upper canopy (Forsman et al. 1984). In general, the diet is dominated by flying squirrels and red tree voles in Douglas-fir and western hemlock forests and by dusky-footed woodrats in mixed conifer forests. The home range of the owl is smaller where the diet is dominated by woodrats, such as in northwestern California (Gutierrez et al. 1995).

3.4.2 Status, Population Trends, and Factors Affecting Populations of the Northern Spotted Owl

The northern spotted owl exists in relatively low numbers in California and elsewhere and is therefore given special or protected status by a number of agencies and conservation groups. The CDFG (2003b) lists the following conservation status rankings for the northern spotted owl:

- *Federal Status*—listed as threatened within its entire range in 1990.

- *State Status*—not listed.
- *World Conservation Union Red List*—listed as Lower Risk near threatened (LRnt) for full species (does not qualify for conservation dependent, but is close to qualifying for vulnerable, i.e., facing a high risk of extinction in the medium-term future).
- *California Department of Forestry and Fire Protection*—listed as sensitive (warrants special protection during timber operations).
- *California Natural Diversity Database*—listed as S2S3 within California (S2S3: the subspecies within California has a restricted range, and is rare or endangered).
- *United States Bird Conservation Watch List*—listed for full species.
- *Audubon Watch List*—listed for full species (species faces population declines and/or threats such as habitat loss on their breeding and wintering grounds, or with limited geographic ranges).

The northern spotted owl is not listed by the State of California.

Population Trends of the Northern Spotted Owl

In the 1980s, it was suspected that there were 3,000 to 4,000 pairs of northern spotted owls. In the late 1980s and early 1990s, a total of 3,605 pairs were located rangewide (incomplete survey), with 954 pairs occurring in California. Most studies conducted in the early 1990s indicated that northern spotted owl habitat was being reduced by as much as 5% per year, and that similar rates of population decline were occurring. However, in northwestern California, a 60% to 83% reduction in population was estimated for the Six Rivers National Forest (Franklin et al. 1990).

In 1994, there were 5,431 known locations, or site centers, of northern spotted owl pairs or resident single owls in Washington, Oregon, and California. Thirty-one percent (1,687) were in California with 837 of them on non-Federal lands. Seventy-five percent (631) of the sites on non-Federal lands were located on the California Coast Province, where owls are fairly common in second-growth timber stands. Two hundred and four of the California site centers on Federal lands depend on some portion of suitable owl habitats on adjacent non-Federal lands to support the owls (FWS 1995).

An annual decline of 3.9% in the northern spotted owl population was observed in the U.S. between 1985 and 1998. This rate of decline is an improvement over the previously estimated 4.5% per year rate of decline. This slowing in the rate of decline is attributed to habitat protection measures implemented in the early 1990s (Blackburn and Harestad 2002).

Population density estimates for northwest California are 0.24 per km^2 (0.09 per mi^2) or about 0.6 per km^2 (0.2 per mi^2) of suitable habitat. Density estimates for the western Oregon and northwestern California area are up to 0.27 adults and subadults per km^2 (0.7 per mi^2; NatureServe 2003). Spotted owl densities are likely to vary substantially with habitat quality and quantity. For example, in one study area there were 22 owls within a 10,000-ha (24,710-ac) area,

while a study area that had undergone extensive logging had only 12 owls within a 20,000-ha (49,420-ac) area (Gutierrez 1985).

Population trends of northern spotted owls should be interpreted with caution. Franklin (1992) hypothesized that the northern spotted owl population may be regulated by territorial behavior. Population estimates are primarily based on hearing calls and subsequent observation of territorial owls and their young. The observed number of territorial birds may remain relatively stable as non-territorial birds replace any territorial birds that were lost (Franklin 1992). A decline in non-territorial birds may go undetected because these birds are not usually detected during surveys.

Factors Affecting Populations of the Northern Spotted Owl

Studies have suggested that there are several factors that influence reproduction of northern spotted owls including weather; prey abundance; habitat quality and fragmentation; and nest-site characteristics. Almost all owl pairs produce young in occasional good years, but these years are interspersed with poor years in which few owls nest or produce young. In long-lived birds like spotted owls, which typically do not breed until they are several years old, dispersal distance has little influence on lifetime reproduction compared to other factors such as annual or local variation in weather or prey abundance (Forsman et al. 2002).

Populations of spotted owls have declined substantially in Canada and the United States in the past decade. Habitat loss is likely the primary factor responsible for population declines (Blackburn et al. 2002). Five threats that may cause the spotted owl population to decline, and become extirpated within portions of Canada, include reduced habitat availability and connectivity, random environmental events, low genetic variability, mortality from predators, and the effects of competition with the closely related barred owl (*Strix varia*) (Blackburn et al. 2002).

Forest decadence plays an important role in spotted owl habitat. First, suitable cavity nest sites are formed as a result of the decay and aging process. Secondly, snags, decadent and rotting trees, and downed woody debris may provide cover and habitat for flying squirrels and wood rats, the spotted owl's major prey (Gutierrez 1985). Larger owl home ranges occur in areas of low prey density (Carey et al. 1990). When foraging conditions are poor and females are forced to leave the nest to search for food, the clutch may be lost. Poor food resources probably do not improve soon enough to allow a female to renest (Gutierrez 1985). Ward et al. (1998) found that the northern spotted owl's preferred prey, the moderately sized dusky-footed woodrat, provides a potential energetic benefit over other smaller prey. Owls may selectively forage in the ecotone between late and early seral mixed-conifer forests because prey is more accessible in the open, younger habitats (Ward et al. 1998). Selection of late seral edge sites for foraging implies that some degree of fragmentation may provide an energetic benefit to the northern spotted owls that occur in areas where young forests produce an abundance of primary prey. However, loss of late seral forest around nest and roost sites is correlated with lower reproduction and survival (Ward et al. 1998). Thus, there is evidently a trade-off in the need for interior habitat (for nesting and

protection from predators and inclement weather) and ecotones (for maximizing foraging opportunities; Franklin et al. 2000).

The larger and more aggressive barred owl is invading the Pacific Northwest and has the potential to overrun much of the range of the northern spotted owl (Peterson and Robins 2003). It can also hybridize with the northern spotted owl (Tesky 1992). The barred owl has a greater tolerance for broken and open forest conditions and does not seem to be in similar peril as the northern spotted owl (Marcot 1995). In Olympic National Park, Washington, more than 10 of 53 monitored northern spotted owl sites were either unoccupied or the northern spotted owls were displaced 750 m (2,500 ft) or more due to barred owls. The ratio of new barred owl sites found per new spotted owl territory has increased from one per 50 in the 1980s to one per 10 to 20 in the middle 1990s. This suggests that the barred owl population in California is increasing. In addition, several barred owl × spotted owl hybrids have been reported in California. Because of the potential for hybridization, competition for food and habitat, and predation, it appears that the barred owl could negatively influence spotted owl populations (Dark et al. 1998). This biological invasion may prove to be the primary threat to the northern spotted owl in otherwise protected landscapes such as National Parks (National Park Service [NPS] 2000). The combination of habitat fragmentation and competition from the barred owl may present a bleak forecast for the northern spotted owl (Peterson and Robins 2003).

Common predators of the northern spotted owl include the great horned owl (*Bubo virginianus*), northern goshawk, Cooper's hawk (*Accipiter cooperii*), and red-tailed hawk (*Buteo jamaicensis*). The common raven (*Corvus corax*) also preys on juvenile spotted owls (Tesky 1992). Predation of spotted owls probably occurs opportunistically at low levels for adult spotted owls and at higher levels among juveniles; but this predation pressure is probably not strong enough to explain the spotted owl's association with old-growth forest. However, from a management point of view, predation could become a serious factor. For example, if great horned owls increase in response to forest fragmentation, then the opportunity for their preying on spotted owls will increase (Gutierrez 1985). Parasites or disease may be a predisposing factor in some cases of starvation or predation (Forsman et al. 2002).

The annual cycle of the northern spotted owl can be divided into five critical periods based on weather conditions and specific life history stages: (1) winter stress period (November through February); (2) early nesting period (March through April); (3) late nesting period (May); (4) heat stress period (July through August); and (5) dispersal period (September through October) (Franklin et al. 2000). Temporal variations in populations of northern spotted owls may be primarily driven by annual variation in climate, especially in spring (the owl's highest daily energy expenditures occur during the breeding season; Franklin et al. 2000). The negative effect of winter precipitation occurs through impacts on survival of young experiencing their first year of independence (Franklin et al. 2000).

Prior to Federal listing of the northern spotted owl, habitat alteration due to clear-cutting was the primary threat to the species within the Northern California Province. Currently the major threat is wildfire. From 1989 through 1995, large fires destroyed or degraded substantial areas of owl habitat on the Klamath, Shasta-Trinity, and Mendocino National Forests (FWS 1995). In many areas of the Province, spotted owl habitat is naturally fragmented by

chaparral, stands of deciduous hardwoods, and low-elevation vegetation types. Fire suppression may have encouraged the development of mixed conifer habitat suitable for spotted owls, while timber harvest removed substantial amounts of suitable habitat (FWS 1995).

Fires can be stand-replacing events that result in the loss of suitable spotted owl habitat for a significant period. Following a fire, new seedlings may continue to become established for as long as 60 yr. This extended regeneration period may be responsible for some of the structural diversity associated with suitable spotted owl habitat. In the long term, fire probably benefits the spotted owl, but the short-term loss of habitat to fire could increase the risk of extirpation within some areas (Holthausen et al. 1995). No specific information on the direct effect of fires on northern spotted owls is available. However, direct fire-related mortality and nest destruction probably occurs (Tesky 1992).

Suppression of a natural regime of frequent, low-intensity ground fires has increased the probability of large-scale or stand-replacing fires because fuel levels and structural continuity have increased. Thus, while the amount of suitable habitat for the spotted owl may have increased, there is also a greater risk of habitat loss due to a catastrophic fire. The current management dilemma is to strike a balance between conserving short- and long-term habitat for owls while developing a more stable forest landscape that is less likely to be adversely affected by pathogens and fire (Buchanan et al. 1995).

Bond et al. (2002) hypothesize that wildfires may have little short-term impact on survival, site fidelity, and reproductive success of spotted owls. Furthermore, prescribed burning could be an effective tool in restoring habitat to natural conditions with minimal short-term impact on resident spotted owls. However, large-scale experiments on the effects of prescribed burning would need to be done to corroborate these observations and to establish cause-and-effect relationships (Bond et al. 2002). Prescribed burns that produce low- to moderate-intensity fires may benefit spotted owls by reducing fuels, thus reducing the chance of stand-replacing, high-intensity fires. However, activities in preparation for prescribed burns may have some negative impacts on spotted owls (e.g., disturbing owl reproductive efforts or reducing canopy cover in some roost sites) unless preburn activities and prescribed burns near nests are conducted in the fall and winter (BLM 2003).

Northwestern California has been subjected to locally heavy timber harvest for about 100 yr. Little old growth forest is left in coastal areas that are primarily in private ownership. Forests of varying ages, including late successional reserves, occur on USFS land particularly at mid to higher elevations (Pierson and Rainey 1998). Forests suitable for northern spotted owls in the Pacific Northwest have declined by 61% since the 18th century because of logging, with most of this decline occurring since the 1930s. In addition, once contiguous blocks of mature and old-growth forests have become increasingly fragmented into mosaics of different seral stages (Franklin et al. 2000). About 31% of the four Forests of the Northern California Province are now covered by mature and old-growth conifer forests (Franklin et al. 2000).

In addition to a decline in the amount of older forest, the landscape in the Pacific Northwest has been altered by the practice of dispersed clearcut logging. As a result, stands of mature and old-growth coniferous forest have become increasingly patchy in their distribution.

The resulting fragmentation may jeopardize the long-term survival of the northern spotted owl through a reduction in the dispersal success of both juveniles searching for their own home range and older owls displaced by timber harvest (Lamberson et al. 1992). The gradual elimination of old-growth and mature forests is the most serious threat to the spotted owl (Forsman et al. 1984). Two factors may contribute to increased mortality in clearcuts: a greater vulnerability to avian predators (e.g., great horned owl and northern goshawk); and decreased foraging activity, resulting in starvation, following the termination of dispersal at an abrupt habitat edge (Miller et al. 1997).

Spotted owls exhibit high affinity and tenacity to home ranges and favored nest locations. They may continue to reside and nest in specific locations in spite of adjacent habitat alterations subsequent to original selection of the site. Thus, spotted owl pairs may select a particular nest site prior to habitat alteration and then continue using it even if conditions that affect survival and reproduction have changed (Swindle et al. 1999). However, Forsman et al. (1984) observed that spotted owls frequently disappear once the area surrounding a nest has been harvested. Since the northern spotted owl is strongly territorial, the likelihood of juveniles successfully dispersing to suitable areas will decrease as suitable habitat becomes patchier in its distribution. Depending on their size, clearcuts may prevent dispersal or force owls to disperse greater distances to locate older forest patches (Miller et al. 1997). Failure to disperse successfully would increase mortality, especially for juveniles (Lamberson et al. 1992; Miller et al. 1997).

While the primary loss of spotted owl habitat is due to timber harvest (Gutierrez et al. 1995), habitat losses also occur as a result of urban and suburban expansion, water development, agricultural development, and mining (Gutierrez et al. 1995). The numbers of spotted owl killed by shooting is unknown, but is expected to be low. Owls seemed undisturbed by occasional passage of vehicles on narrow secondary gravel roads. They are sometimes killed by automobiles and by flying into tree limbs (Gutierrez et al. 1995).

3.4.3 Suitability of the Northern Spotted Owl as a Management Indicator Species

The northern spotted owl satisfies the criteria we established for a suitable management indicator species for the Forests of the Northern California Province (see Section 1). It occurs throughout the year in the Province and does not migrate. The range of the species includes all four Forests of the Province. The species is easily identifiable and it occurs in sufficient numbers to support a monitoring program. Monitoring programs currently exist for the spotted owl in the Province.

A great deal is known about the life history and ecology of the northern spotted owl due to its association with late-successional forests that are of high commercial value (Gutierrez et al. 1995). Also, common field methods and methods of analysis established for the northern spotted owl are seldom achieved in studies of other bird species (Noon and Franklin 2002). Among all avian-research programs, results from those on the spotted owl may have the greatest effect on land-use policy because all three subspecies require large areas associated with large, old trees that have high economic value (Noon and Franklin 2002). Additionally, Federally listed species are managed for recovery, thus their populations are

monitored independent of being designated as a management indicator species. Therefore, this can provide an advantage in the selection of the northern spotted owl if interagency funding and cooperation occurs to maximize monitoring efforts.

However, there are some concerns or considerations regarding use of the northern spotted owl as a management indicator species. Unambiguous studies regarding spotted owl habitat selections are difficult to execute because of the scale and expense required for a proper experimental design (LaHaye and Gutierrez 1999). With its very high adult survival rate, longevity, and site-tenacity, northern spotted owl populations may be able to persist over the short term even in the face of extensive reductions in the amount or quality of available habitat. Thus, there may be a significant time lag in the response of spotted owl population size to decreases in carrying capacity (Noon and Biles 1990; Hunter et al. 1995). When favorable habitat is limited, surplus adults may accumulate in moderate densities in poor habitat where breeding success is greatly reduced. In such case, estimates of species abundance alone may be misleading (Van Horne 1983).

3.4.4 Monitoring Protocols for the Northern Spotted Owl

Documents that provide protocols that can be used for conducting surveys of spotted owls include Forsman (1983), CDFG (1992), and Ministry of Sustainable Resource Management (MSRM 2001). Lint et al. (1999) describes options for effective monitoring of long-term status and trend of the northern spotted owl.

Forsman (1983) describes the techniques that have been found to be most effective for locating and surveying spotted owls, and is the primary guidance used for northern spotted owl studies. The CDFG (1992) protocol was developed for survey areas where Federal and non-Federal activities may remove or modify northern spotted owl habitat. The protocol has been endorsed by the FWS, and is used for obtaining information on spotted owl occupancy. The CDFG (1992) protocol serves the following two purposes: (1) it provides adequate coverage and assessment of the area to determine the presence of spotted owls; and (2) it ensures a high probability of locating resident spotted owls and identifying owl territories that may be affected by a proposed management activity. It defines and describes survey areas, survey duration, habitat to be surveyed, survey methods (generally using the methodology provided in Forsman [1983]), and recommendations for spotted owl surveyor credentials and qualifications (CDFG 1992). MSRM (2001) provides standard protocols for raptor censuses (including owls) at different levels of intensity (e.g., presence-not detected, relative abundance, and absolute abundance) at various times of the years. It is a more generalized protocol, dealing with a variety of raptors. However, it does provide guidance for the northern spotted owl.

To conduct a thorough search for roosts using the method described by Forsman (1983), the investigator systematically walks and broadcasts spotted owl calls to be heard in all potential roost areas. To locate as many roosts as possible using this approach, the investigator looks for signs of roosting (e.g., fecal material, owl pellets, or molted feathers) under trees in each area where owls are found roosting (Forsman 1983). Nocturnal calling surveys are the most effective and most frequently used technique for locating spotted owls. Roosts and general nest locations

may be located during the day by calling in suspected roost or nest areas. Specific nest trees are located by either baiting with a live mouse to induce the owl to visit the nest, calling in suspected nest areas to stimulate the female to call or fly from the nest, or by observing adults during prenesting displays (Forsman 1983).

Many owl surveys are conducted during the breeding season, when owls are territorial and active nests can be located. Spotted owls lay eggs between March 1 and April 10, incubation lasts 28 to 32 days, and the young remain in the nest for about 34 days. Therefore, attempts to locate nests should be conducted between mid-March and early June (Forsman 1983). However, care must be taken not to disturb raptors during courtship, egg laying or incubation, or when young are still in the nest. If disturbed, adults may abandon nests or move to a new location in succeeding years. Other consequences of disturbance can include exposure of eggs or young to hypothermia or overheating or to predators (MSRM 2001).

The territorial status of spotted owls can be determined by one of the following criteria: (1) multiple responses by owls within the breeding period, (2) multiple responses by owls over two or more years in the same general area, (3) presence of an active nest, (4) observations of an adult owl with young, or (5) presence of a pair of owls roosting together (Blackburn et al. 2002). For an accurate estimate of the number and distribution of spotted owl pairs, it is usually necessary to survey an area several times (e.g., two or three times in one summer and repeated at least two summers in a row) (Forsman 1983). If time does not permit a complete survey, several methods can be used to determine the relative abundance of owls in different areas or cover types. For example, one can compare the average number of owls responding per unit length of road transect, calling point, or unit time (Forsman 1983).

Radio-telemetry has been used as part of a wildlife-inventory program, including many on the spotted owl, to address the following objectives: (1) provide information on use and selection of landscapes and habitats by a species; (2) provide locations of key habitat elements; (3) provide descriptions of home ranges; and (4) provide input on population size or dynamics. However, radio-telemetry studies are expensive, labor intensive, and potentially stressful to animals (Ministry of Environment, Lands, and Parks [MELP] 1998a). Radio-tagged northern spotted owls had lower survival (females), fledging of young, and nesting rates than color-banded owls. Therefore, it was suggested that spotted owl researchers should avoid using backpack-mounted radio tags (Paton et al. 1991).

Several environmental factors may affect monitoring results. For example, poor weather such as high winds, rain, and fog can influence both the bird's behavior and the observer's ability to detect and identify individuals. High winds and rain are potentially a greater problem in forests due to increased noise in the canopy than in open areas (Forsman 1983; MSRM 2001). Sometimes owls may not respond to playback in years when prey is not abundant. It is also important to avoid playing calls during courtship, egg-laying, or incubation periods as this may disrupt successful breeding (MSRM 2001). Detection of young owls is poor for the spotted owl because many young birds either do not respond to call surveys, or respond in a very tenuous fashion that is difficult to detect (Forsman et al. 2002).

Both spotted owls and barred owls respond to tape recordings and vocal imitations of spotted owl calls (Herter and Hicks 2000). Because barred and spotted owls are similar in appearance and sound somewhat alike, investigators must be familiar with the calls and appearance of both species.

3.5 WHITE-HEADED WOODPECKER

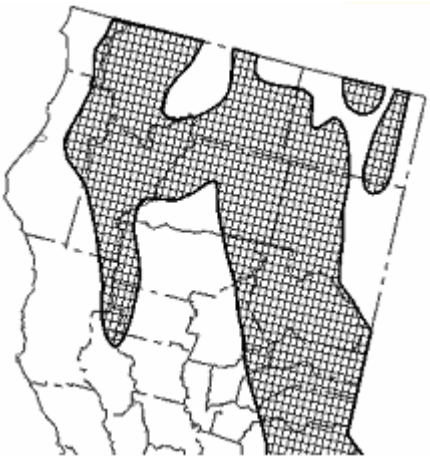
The white-headed woodpecker is about 23.5 cm (9.25 in.) long, with a wing span of about 40.5 cm (16 in.), and body weight of about 61 g (2.1 oz) (Sibley 2000). White feathers completely cover the head, throat, and upper breast, but the rest of the body is almost totally black. The large, white wing patch at the base of the primary feathers of each wing is conspicuous in flight. Males have a red patch on the back of the head and juveniles have a pale red patch on the crown.

3.5.1 Distribution, Habitat, and Life History of the White-Headed Woodpecker

The white-headed woodpecker is a permanent resident of montane coniferous forest ranging from the Okanagan Valley of southern British Columbia southward through north-central Washington, northern Idaho, Oregon (east of the Cascade Mountains) to San Diego County in southern California (Cornell Laboratory of Ornithology 2003). Although mainly a species of montane forests at elevations of 1,100 to 2,550 m (3,600 to 8,400 ft) MSL, the white-headed woodpecker occasionally moves to lower elevations during the nonbreeding season. Small (1994) reports white-headed woodpecker habitat as being strongly associated with pine forests of the Transition and Lower Canadian Life Zones at elevations ranging from 1,200 to 2,300 m (4,000 to 7,500 ft).

Small (1994) provides the following account of the white-headed woodpecker's distribution in California. The species occurs in California from the Oregon border southward through the inner mountain ranges of the Klamath Mountains as far south as northern Lake County. It also occurs in the mountains enclosing the northern end of the Sacramento Valley but is not found in the Shasta Valley to the Cascade Mountains. The white-headed woodpecker inhabits the Warner Mountains and areas southward through the Cascades–Sierra boundary to northern Kern County. Its range is discontinuous throughout portions of the Southern Coast Range and elsewhere in southern California. It occurs in all four Forests of the Northern California Province (CDFG 2002).

Range of the White-Headed Woodpecker in Northern California (CDFG 2002)



The white-headed woodpecker is relatively uncommon throughout its range. It is so rare in Washington that it is designated a candidate species for state-listing as threatened or endangered. In British Columbia, the population was fewer than 100 birds when surveyed in 1990 and was listed as endangered in 2000 (Environment Canada 2003). Although now endangered, the white-headed woodpecker was relatively common in the mid 1960s through the mid 1970s. The reason for its decline in British Columbia is unknown.

CDFG (2003c) describes the habitat type of the white-headed woodpecker as montane coniferous forest at elevations that include red fir and lodgepole pine forests of the Sierra Nevada, Cascade, Klamath, Transverse, and Peninsular Ranges and the Warner Mountains. The species prefers semiopen forest stands with large coniferous trees with a canopy cover of 40% to 70% in California populations (CDFG 2003c). Raphael and White (1984) and Milne and Hejl (1989) found white-headed woodpecker in mixed conifer forest in the Sierra Nevada Mountains. Common tree species in the Sierras where white-headed woodpeckers were observed included Jeffrey pine, ponderosa pine, white fir, red fir, Douglas-fir, and black oak (Garrett et al. 1996).

CWHRS habitat types that are most suitable for the white-headed woodpecker include Klamath mixed conifer, Sierran mixed conifer, white fir, Jeffrey pine, ponderosa pine, and eastside pine. CalVeg community types that correspond with these habitat types are presented in CDFG (1998). The distribution of white-headed woodpecker habitats on the four Northern California Province Forests is mapped in Figure 6.

White-Headed Woodpecker Habitats in Forests of the Northern California Province (CDFG 2002)

Habitat Type	Suitability[1]
Klamath mixed conifer	0.64
Sierran mixed conifer	0.64
White fir	0.64
Jeffrey pine	0.62
Ponderosa pine	0.62
Eastside pine	0.60
Red fir	0.48
Lodgepole pine	0.44
Montane hardwood conifer	0.42
Douglas-fir	0.23
Montane riparian	0.23
Montane hardwood	0.22
Subalpine conifer	0.21

[1] Suitability varies from 0 to 1 with 1 being optimum.

Old-growth forests are believed to be critical to the survival of the white-headed woodpecker throughout its range. The species requires high densities of snags for roosting and nesting. Snag densities of 45 large (> 58 cm [23 in.] dbh) snags per 40 ha (100 ac) is recommended to maintain a goal density of five pairs of white-headed woodpeckers per 40 ha (100 ac) (Garrett et al. 1996). In Washington, the white-headed woodpecker was associated mainly with open-canopy mature and old-growth ponderosa pine forests where they rely on decaying snags for nesting and roosting (Lewis and Rodrick 2002).

The EIS for the Klamath Forest Plan (USFS 1994b) presented a habitat capability model[4] for snag-dependent wildlife species that included the white-headed woodpecker. The model

[4] Habitat capability models have been developed by the USFS and are descriptions of the capability of habitat to support a species or groups of species on the basis of physical and biological habitat variables. These models are based on the best available information including qualitative and quantitative information and professional judgment.

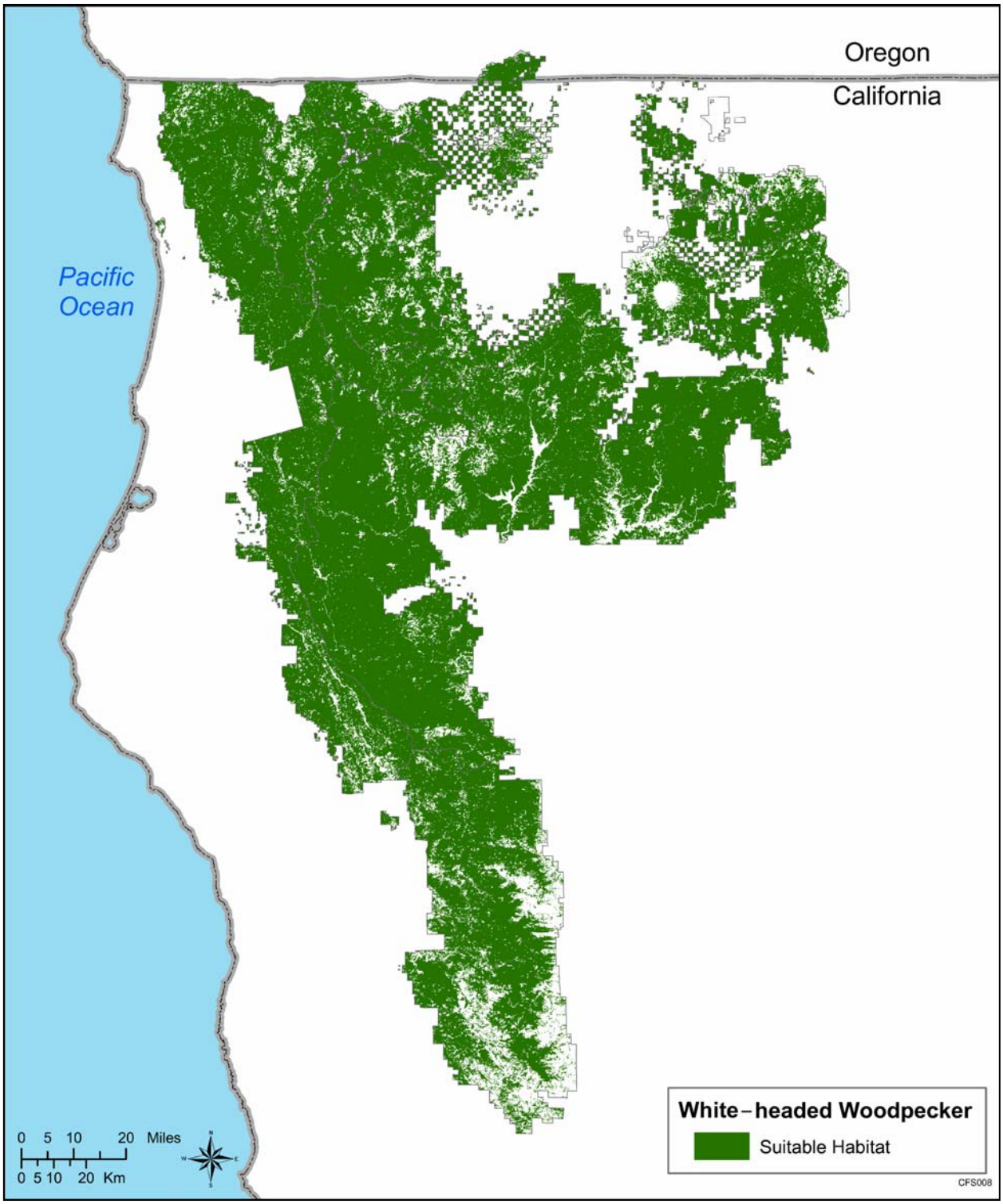


Figure 6 White-Headed Woodpecker Habitats on the Four Forests of the Northern California Province

identified high, moderate, and low capability habitats based on the density, characteristics, and location of snags in the following types: (1) Douglas-fir; (2) westside mixed conifer, (3) eastside and westside true fir, (4) ponderosa pine, and (5) lodgepole pine. High-capability habitats had relatively larger snags, higher snag densities, and more hard snags than soft snags. Snags in high capability habitats were located near the edges of meadows, shrublands, and riparian areas as opposed to the interior of forests or in open areas.

The white-headed woodpecker breeds from mid-April to late August (CDFG 2003c). Initiation of breeding is marked by increased drumming, calling, and courtship. Nest-building occurs in April and May (Garrett et al. 1996). The white-headed woodpecker normally produces one brood per year with egg-laying in May and early June (Garrett et al. 1996). Clutch size of four to five white eggs is typical (Terres 1991). Incubation takes about 14 days and the young leave the nest after a 26-day fledging period (Garrett et al. 1996). At two study sites in central and south central Oregon, nest success (i.e., fledging one or more young) was 83 to 87% based on observations of 57 nests.

Large snags or stumps greater than 61 cm (12.5 in.) in diameter with hard bark and soft interior are preferred nest sites (CDFG 2003c). Leaning or downed logs and stumps are also used as nest sites. Raphael and White (1984) observed the white-headed woodpecker nesting in snags of burned pine and fir trees. Milne and Hejl (1989) found a high reliance on dead snags as nest sites by this species in western Sierra Nevada populations. All but one of 53 nests were found in dead snags with a mean height of 8 m (26 ft) and mean dbh of 80 cm (31 in.). The average height of the nest was about 3 m (10 ft) above the ground surface. Ponderosa pine and red fir were the most common nest trees in this study. Ponderosa pine is also used predominantly for nest sites in southern and central Oregon (Garrett et al. 1996).

Large mature pine trees are important as food sources for the white-headed woodpecker (CDFG 2003c). Pine seeds and a variety of adult insects comprise the white-headed woodpecker diet. A strong dependence on ponderosa pine seeds is typical throughout much of the year, particularly in fall and winter (Garrett et al. 1996). Beetles, scale insects, and ants were the insects most consumed by the white-headed woodpecker in a Sierra Nevada population (Otvos and Stark 1985). Based on stomach-content analyses of the Sierra Nevada population, ponderosa pine seeds comprised 14% and 8% of the diet of males and females, respectively, with invertebrates representing 75% of male total food volume compared to 85% of female food volume.

The white-headed woodpecker pecks, flakes, and chips the bark of conifers in search of insects. They often attach upside down to unopened large cones. Seeds are excavated and extracted and taken to an “anvil site” on the same or nearby tree for cracking prior to seed ingestion (Garrett et al. 1996). Some sexual differences exist in foraging behavior. Male white-headed woodpeckers spend more time foraging on large pine cones than do females. Also, males tend to spend more time foraging in the upper third of the tree canopy compared with females who feed more frequently on the trunk and outer portions of the lower lateral branches (Garrett et al. 1996). Both sexes are known to forage on different trees depending on season. Garrett et al. (1996) noted that females spent more time foraging on incense cedar in winter than

did males, and speculated that females were foraging on scale insects (*Xylococculus macrocarpae*) that occur under the cedar's loose bark.

In central Oregon, the mean home range size was 104 ha (260 ac) and ranged from 67 to 163 ha (165 to 400 ac) in old-growth coniferous forests. Home range size tended to be larger in fragmented forests where mean home range size was 321 ha (790 ac) and ranged from 57 to 445 ha (140 to 1,100 ac) (Dixon 1995). South-central Oregon populations had similar home range sizes in old-growth and fragmented sites with mean sizes of 212 ha (525 ac; range 172 to 324 ha [425 to 800 ac]) and 342 ha (845 ac; range 171 to 704 ha [420 to 1,700 ac]), respectively. No information is available on home range size in northern California populations of the white-headed woodpecker.

3.5.2 Status, Population Trends, and Factors Affecting Populations of the White-Headed Woodpecker

The white-headed woodpecker is not listed as threatened or endangered by either the State of California or the Federal government. It does not have special protection status in the Northern California Province.

Garrett et al. (1966) summarized studies of population density of the white-headed woodpecker in California and Oregon (see Table 2). Population density during the breeding season ranged from 0 to 5 pairs per 40 ha (100 ac).

North American Breeding Bird Survey results for the northern California region indicate that white-headed woodpeckers are encountered regularly, but in low numbers (Sauer et al. 2003). Less than one individual observed per survey route per year is typical for the region. Over the 1966 to 1996 period, breeding bird survey data suggest that the population of white-headed woodpeckers has been increasing in the region (about 2% per year).

Audubon Christmas Bird Count data (Audubon 2003) for white-headed woodpeckers in count circles[5] within the Northern California Province are shown in Figure 7. White-headed woodpeckers were observed in very low numbers[6], and not in all years of survey. The species was observed only on the Mt. Shasta, Willow Creek, and Yreka count circles. No population trend is apparent when the data from 1980 to 2003 for all count circles in the Province are plotted (Figure 8).

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[5] Audubon Christmas Counts occur from December 14 to January 5 and are conducted in a 24-km (15-mi) diameter circle in which all birds seen or heard are counted in a single day. Count circles in the Northern California Province include Del Norte (Del Norte County), Mendocino Coast (Mendocino County), Mt. Shasta (Shasta County), Redding (Shasta County), Willow Creek (Humboldt County), and Yreka (Siskyou County).

[6] Audubon Christmas Bird Count data are presented as total number of individuals observed (seen or heard) and the total number of birds observed per party-hr (the number of observers × number of hours those observers spent on count).

Table 2 Breeding Population Density of White-Headed Woodpecker in California and Oregon

Location	Habitat	Survey Period	Breeding Density (No. per 40 ha [100 ac])
California			
Western Sierra Nevada	Old growth; mixed conifer, red fir	1988–1992	3.1–5.0 pairs
Sierra Nevada	Mixed conifer forest	1948–1978	Mean 2.3 pairs; range 1-5
Sierra Nevada, Sagehen Creek Field Station	Mixed conifer forest	1976–1978	1.2 pairs
	Second growth mixed pine forest	1966–1985	0.5 pairs on burned plots
			0 –0.2 pairs on unburned plots
Oregon			
Northeastern Oregon; Wallowa–Whitman National Forest	Managed mixed forest	1978–1980	0–0.5
	Old-growth forest	1978–1980	1.0

Source: Garrett et al. (1996)

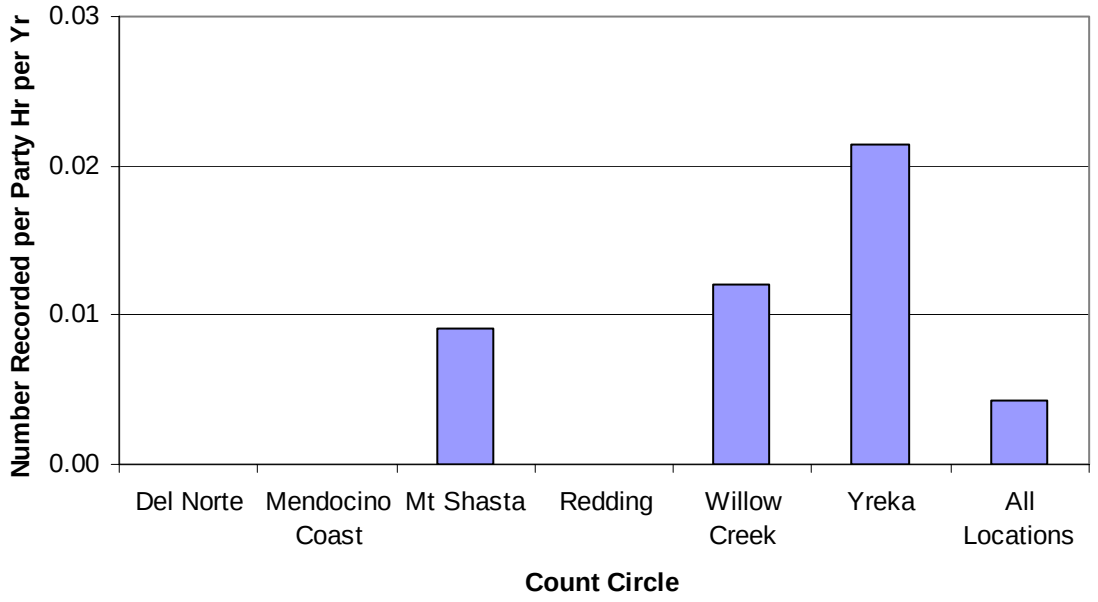


Figure 7 Mean Number of White-Headed Woodpeckers Counted per Year per Party-Hr in Audubon Christmas Bird Counts on Count Circles within the Northern California Province (1980-2003)

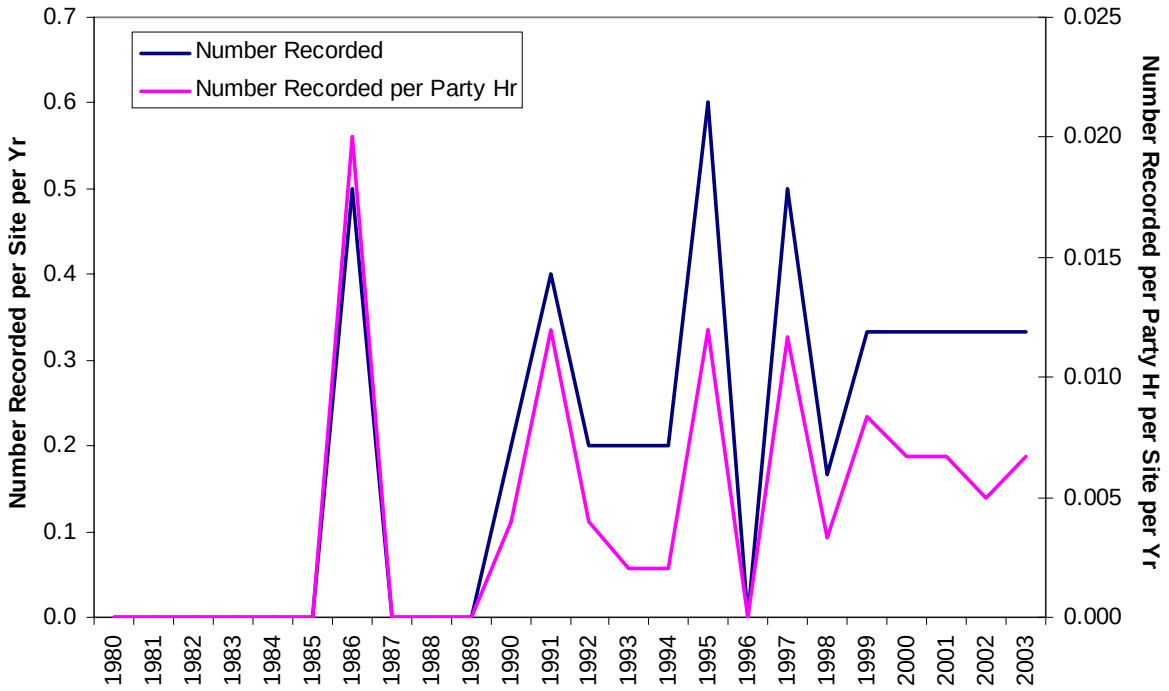


Figure 8 Mean Number of White-Headed Woodpeckers Counted per Year in Audubon Christmas Bird Counts within the Northern California Province (1980-2003)

Habitat loss and degradation is a major cause of population declines in some portions of the white-headed woodpecker's range. Logging practices in the Blue Mountains of Oregon led to population declines there (Garrett et al. 1996). The reduction in number of snags for nest and foraging sites is believed to limit population numbers on non-Federal lands in Oregon and Washington (Ohmann et al. 1994). Removal of stumps and leaning or fallen snags is also thought to reduce available nest sites. Garrett et al. (1996) reported that the loss of large ponderosa pine trees poses a threat to white-headed woodpecker populations in Oregon. Raphael et al. (1987) reported the use of burned areas that had snags remaining after the fire, and suggested that the white-headed woodpecker is more tolerant of disturbance than other cavity-nesting species of closed-canopy forest.

Human activity near nest and roost sites can cause the white-headed woodpecker to abandon available habitat. However, Garrett et al. (1996) found that the white-headed woodpecker population of the Angeles National Forest, an area of high recreation use, has not declined. Nest abandonment usually does not occur unless human activity directly affects a nest or roost tree.

Nothing is known of the effects of parasites on the white-headed woodpecker. Predation on the white-headed woodpecker has not been quantified, although Garrett et al. (1996) indicate that chipmunks have been observed consuming white-headed woodpecker eggs, and great-horned owls are known to prey on adult white-headed woodpeckers in Oregon.

3.5.3 Suitability of the White-Headed Woodpecker a Management Indicator Species

The white-headed woodpecker satisfies the criteria we established for a suitable management indicator species for the Forests of the Northern California Province (see Section 1). It occurs throughout the year in the Province and does not migrate. The range of the species includes at least some portions of all four Forests of the Province. It is easy to identify and cannot be confused with any other species. Monitoring programs exist for forest birds in general and woodpeckers in particular and could be used to monitor populations of white-headed woodpeckers. However, the species apparently exists in relatively low numbers in the Province, and these low numbers may make it difficult to develop a statistically robust program to detect the effects of forest-management practices on this species.

White-headed woodpeckers utilize a variety of conifer types and do not appear to rely on any one particular type or species. Old-growth and the presence of adequate numbers of snags appear to be important habitat characteristics for the species and habitat capability models have been developed and used to evaluate the effects of management practices on populations of this species. However, quantitative relationships between habitat characteristics and white-headed woodpecker populations have not been developed.

3.5.4 Monitoring Protocols for the White-Headed Woodpecker

Relatively few studies have been conducted specifically to determine white-headed woodpecker population densities. Spot mapping of permanent plots has been used to determine white-headed woodpecker breeding densities and use of old-growth forest and burned areas (Bock and Lynch 1970; Raphael et al. 1987; Raphael and White 1984). Raphael and White (1984) censused breeding birds on eight, 8.5 ha (21 ac) plots and one 6.7 ha (17 ac) plot in a mixed conifer forest in the Sierra Nevada Mountains 10 to 16 times per yr from May to early July. Each census took two to four hours per plot to conduct.

British Columbia (MELP 1999a) developed inventory methods for woodpeckers. In evaluating various survey methods used for woodpeckers, MELP (1999a) addressed the advantages and disadvantages of call-playback surveys, woodpecker-sign surveys, spot mapping, and distance methods in an attempt to develop standardized survey protocols. Many of the conclusions in MELP (1999a) are based on studies conducted in the Pacific Northwest, and thus are applicable to the Northern California Province. Advantages and disadvantages of each survey methodology are summarized in Table 3.

MELP (1999a) provides guidance on what factors researchers should consider for each of the above survey methods including sampling design, sampling effort, and detailed field methods and equipment needed for surveying woodpeckers. The protocol also includes detailed inventory data forms for woodpeckers and an evaluation of data analysis techniques for the each survey method category. The following is a summary of information provided in MELP (1999a) for each of the following survey categories: (1) presence-absence, (2) relative abundance, and (3) absolute abundance.

Table 3 A Comparison of Methods used to Survey Woodpeckers

Method	Advantages	Disadvantages
Call Playback/Drumming Imitation	▪ Good for secretive birds ▪ Useful for species during breeding season or for species that defend territories during winter ▪ Allows determination of relative abundance ▪ Can be conducted quickly along existing routes; advantageous for species with large home ranges ▪ Generates good results for presence/not detected surveys	▪ Statistical problem in specifying probability of species absence ▪ Weather conditions, observer experience affect species numbers detected ▪ Optimal sample effort must be determined on a study-specific basis ▪ Time required to stratify habitats before surveying ▪ Drumming imitations only good if birds approach closely enough for detection ▪ More costly than transects or point counts because of equipment needs ▪ Annual variations in environmental conditions can affect responsiveness and results
Sign Survey	▪ Cavities of some species can be identified on basis of size and shape without climbing trees or snags ▪ Evidence of foraging on snags based on excavations observed ▪ Allows direct observation of woodpeckers on snags	▪ Recent excavations can be detected for only about 1 yr ▪ Nest excavations are often incomplete so active nesting area or precise nest location is difficult to determine ▪ Difficult to walk along transects in rugged terrain
Spot Mapping	▪ Allows territory mapping ▪ Preferred method for determining absolute abundance of breeding birds ▪ Record species, sex, number, and behavior	▪ Difficult to survey in rugged terrain ▪ Good method only if all birds can be counted ▪ Spot mapping requires extensive effort; sites visited 6-10 times at 1-week to 10-day intervals
Distance Methods: Point Counts, Line Transects	▪ Counts and distance estimates can be made in 10 min for point counts ▪ Transects less labor-intensive than spot mapping ▪ Both breeding and nonbreeding birds detected along transects potentially allowing for a better population density estimate than spot mapping ▪ Decrease in sightability with distance increase from observer can be accounted for statistically	▪ Distance estimation difficult for multiple-species surveys in dense forest habitat ▪ Observer awareness of lateral distance from transect line difficult in dense habitat ▪ 60 to 80 woodpecker observations within a given study area are needed to allow abundance analysis

Source: MELP (1999a).

Presence–Absence Surveys

Call-playback and drumming-imitation surveys are recommended in MELP (1999a) as the best and most efficient method to determine the presence of woodpeckers during the breeding season. Call playback surveys are generally more conclusive than drumming surveys because, whereas call responses can be easily identified to species, drumming responses cannot. During the nonbreeding season, when birds are less conspicuous, surveys for woodpecker sign, including feeding, roosting, and nesting excavations, can be used to monitor for woodpecker presence. These indirect methods, however, may be inconclusive regarding the species of woodpecker that made the sign. For all presence-absence surveys, the interpretation of negative information (i.e., species not detected) can be problematic because nonresponse can result from low population density, inadequate sampling intensity, or conditions during the survey.

Detailed guidance is presented in MELP (1999a) on how to conduct call playback surveys for woodpeckers. In general, call-playback surveys are conducted by walking or driving along a transect "line" and stopping every 100 to 600 m (328 to 1,968 ft) to play a call. The density of the habitat and size of the woodpecker should be considered in determining the distance between sampling points, with closer intervals being more suitable for denser habitats and smaller species. Appropriate distances for spacing call stations have not been determined specifically for the white-headed woodpecker or other species.

Some disadvantages of call-playback surveys include (1) relatively greater effort and expense required (associated with the cost of equipment and the time required to operate the equipment during the survey) relative to some other survey approaches; (2) annual variation in environmental conditions that affect the timing of breeding activities and peak responsiveness; (3) dominance interactions among species that might affect detectability of less aggressive species; and (4) statistical limitations resulting from negative information (no responses detected).

MELP (1999a) also presents guidance on conducting sign surveys for woodpeckers. The report provides descriptions of feeding and nesting excavations for some species, but not for the white-headed woodpecker. The general approach used to conduct a sign survey is to walk slowly along a line transect, recording information about woodpecker sign and species encountered along the route. Information also is gathered on terrain, habitat conditions, and tree characteristics.

Relative Abundance Surveys

Call-playback, drumming-imitation, and woodpecker-sign surveys can also be used to collect data on relative abundance of woodpeckers (MELP 1999a). It is applicable for woodpecker species with relatively large home ranges such as the white-headed woodpecker and species that are otherwise difficult to detect. The method provides only an index of relative abundance, not a population estimate. Surveys of woodpecker sign (excavations) can provide relative abundance information during the nonbreeding season. Measures of relative abundance are presented as the number of detections per unit of sampling effort (e.g., sampling hours or

transect distance). Considerations and limitations presented in the description of the use of call playback, drumming imitation, and woodpecker sign for presence-absence surveys also apply to these relative abundance surveys.

To be useful, relative abundance surveys in different study areas should be as consistent as possible (MELP 1999a). It is important that sampling effort is equal among sampling areas (e.g., same amount of time, same number of samples, same lengths of transects, same spacing among sample points). In addition, observers should move at a constant rate. To ensure statistical rigor, a sufficient number of samples should be collected, and minimum sample size should be based on the variability among samples using power analysis techniques (MELP 1999a). Adequate sample sizes are particularly important for monitoring trends in populations.

Absolute Abundance

Absolute abundance can be determined by territory mapping (spot mapping) or by using encounter transect distance methods, which are potentially less labor intensive (MELP 1999a). Spot mapping is essentially a count of all breeding individuals in a survey area; nonbreeding birds without territories can rarely be counted using this approach. Birds in dense vegetation or in a nest cavity may not be observed during the spot-mapping survey.

Spot mapping is a widely accepted method for obtaining absolute abundance of territorial species such as woodpeckers during the breeding season (Ryder 1986; Ralph et al. 1993). In addition, the data gathered can be used to determine distribution, population density, and territory size. Spot mapping requires repeated surveys of established plots. During each survey, the location and characteristics of each woodpecker observed is recorded on a base map. Clusters of observations are assumed to define the spatial characteristics of a territory, and a breeding pair is assumed to occupy each territory (even though the female may not be recorded during each survey). Assumptions of the spot-mapping approach are presented in MELP (1999a).

Spot-mapping plots are usually 20 ha (50 ac) in size, and rectangular or square in shape. All plots should be the same size, sufficiently distant from adjacent plots to assure independence and should be placed away from the edges of habitats. Flagged grid lines are placed inside each plot and walked at a constant pace to facilitate mapping. Each plot requires about three hours to complete.

Distance methods (e.g., line transects, point counts) allow for estimates of absolute density and may be a viable alternative to the more labor intensive spot-mapping approach described above (MELP 1999a). With this approach, a line transect is walked slowly or an observer stands at a point location for some specific amount of time, and all individual birds encountered and their distance from the observer are recorded. Advantages of distance methods are (1) they are less labor intensive than spot mapping, and larger sample sizes can be obtained; (2) breeding and nonbreeding birds are counted; and (3) the decrease in observability at increasing distance from the observer can be accounted for statistically. Time to conduct individual point counts is typically 10 min. Time to conduct a line transect will depend on the

length of the transect. Distances from birds to the observer will be more difficult to determine in denser habitat.

Huff et al. (2000) described a habitat-based point-count protocol for terrestrial birds in Washington and Oregon. Their approach is similar to other point-count protocols, but sampling is stratified according to habitat. Such an approach could provide valuable data when attempting to determine relationships between population size and habitat conditions. They recommend that point-count stations be established for sampling repeatedly and that each be surveyed three times per year during the breeding season. Survey duration is 5 min per site. The protocol requires five or more point-count stations within each monitoring site (habitat units). This replication is used to reduce variability among sites that are a result of sampling error.

Of the approaches discussed above, spot mapping, line transects, and point counts would be the most reliable for providing the information needed to determine trends in population size across the Province. Relative abundance techniques may not provide the information needed to monitor population trends and relationships between populations and habitat characteristics. The ability of any monitoring program to detect such trends and the effects of forest-management practices will depend on the design of the program including, among other things, the dispersion of survey sites, the range of habitats surveyed, and the number of samples. Guidance for developing appropriate study designs are provided in a wide variety of sources including MELP (1998b, 1999a), Huff et al. (2000), and Hayek and Buzas (1997).

3.6 PILEATED WOODPECKER

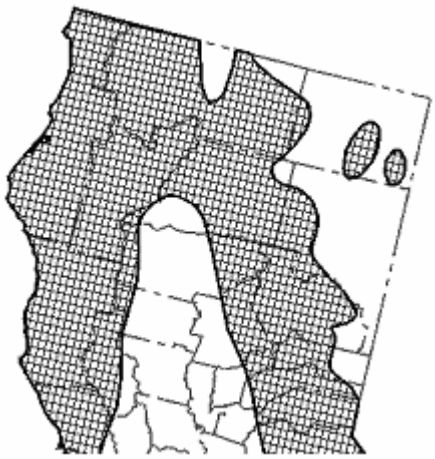
The pileated woodpecker is the largest woodpecker found in North America and has a mean body length of 42 cm (16.5 in.), wing span of 74 cm (29 in.), and weight of 290 g (10 oz) (Sibley 2000). It has a black body with white wing patches, long neck, is broad winged with a relatively long tail, and a prominent red crest. Both sexes possess the red crest although it is more pronounced in males. A distinctive white stripe on the side of the neck, white chin, and black bill are also characteristic of the pileated woodpecker.

3.6.1 Distribution, Habitat, and Life History of the Pileated Woodpecker

The current distribution of the pileated woodpecker is described by Bull and Jackson (1995) as extending from Nova Scotia and New Brunswick westward through southern Quebec and Ontario, west through southern Manitoba, central Saskatchewan, and central Alberta, southward through montane regions of Washington, western Montana and Idaho, Oregon (west of the Cascades) to central California. In the eastern U.S., its distribution is limited to forested regions of states east of the Mississippi River. The pileated woodpecker's range extends southward from central Ontario to Florida and west to extreme eastern Texas, Oklahoma, Kansas, and eastern Iowa into western Minnesota and North Dakota.

In California, the pileated woodpecker occurs from the Oregon border, south in the Coast Range forests and east through the interior Coast Range forests bordering Sacramento Valley. Its distribution in these forests is intermittent. From the San Francisco Bay area, where it is mostly absent, it occurs south of Santa Cruz County and to western Santa Clara, Alameda, and Contra Costa Counties. The pileated woodpecker also occurs in the Klamath Mountains region from the Oregon border south through the Cascades-Sierra axis on the western slopes of the Sierra Nevada Mountains to the Greenhorn Mountains (Small 1994). The species is found in all four Forests of the Northern California Province (CDFG 2002).

Range of the Pileated Woodpecker in Northern California (CDFG 2002)



A variety of forested habitats are used by the pileated woodpecker. They prefer forests that have a relatively tall closed canopy. Schroeder (1982) assumed that optimum pileated woodpecker habitat would have a canopy closure of 75% or greater. In the western portions of its geographic range, it prefers old-growth conifer forests but also occurs in open woodlands and areas of second-growth (California Partners in Flight [CPIF] 2003a).

The pileated woodpecker is adept at chiseling dead snags to excavate nest cavities. Hollow trees are typically used as roost sites during the night and in foul weather. Roost cavities are found in both live and dead trees. Bull and Jackson (1995) examined 60 roost trees in northeastern Oregon and found that 95% had a hollow interior resulting from decay. In the Blue Mountains of northeast Oregon, differences were found between roost and nest trees with respect to elevation based on a study of radio-tagged adults and nestlings (Bull et al. 1992). Roost trees were located lower on slopes than were nest trees. Roost trees also were smaller in diameter with more holes than nest trees.

In a study of summer home range and habitat use by pileated woodpeckers in western Oregon, Mellen et al. (1992) found that trees used for nesting were not large enough in forest stands that were less than 70 yr old. However, conifer forest and deciduous riparian stands greater than 40 yr old were used for foraging and roosting. Nest sites in Oregon and Washington varied from 15 to 38m (49 to 125 ft) above ground (Bull and Jackson 1995). For nesting, pileated woodpeckers depend on habitats with high densities of downed, standing logs; trees with dead snags and stumps; and large live trees. Bull and Jackson (1995) found an inverse relationship between home range size, log volume, stump volume, and canopy closure.

CWHRS habitat types that are most suitable for the pileated woodpecker include Klamath mixed conifer, montane hardwood conifer, Sierran mixed conifer, white fir, Douglas-fir, ponderosa pine, and Jeffrey pine. Older seral stages of these habitats have the highest suitability. CalVeg community types that correspond to these habitat types are presented in CDFG (1998).

The distribution of pileated woodpecker habitats on the four Northern California Province Forests is mapped in Figure 9.

Pileated Woodpecker Habitats in Forests of the Northern California Province (CDFG 2002)

Habitat Type	Suitability[1]
Klamath mixed conifer	0.38
Montane hardwood conifer	0.38
Sierran mixed conifer	0.38
White fir	0.38
Douglas-fir	0.37
Ponderosa pine	0.34
Jeffrey pine	0.33
Lodgepole pine	0.23
Red fir	0.21
Blue oak foothill pine	0.20
Montane hardwood	0.20
Montane riparian	0.13
Redwood	0.13
Eastside pine	0.12

[1] Suitability varies from 0 to 1 with 1 being optimum. Mean suitability values for all seral stages of each habitat type are presented. Older seral stages have higher suitability scores.

Many estimates of optimal habitat patch size have been proposed for pileated woodpeckers in the northwestern United States. The habitat suitability model developed by Schroeder (1982) assumed a minimum of 129.5 ha (320 ac) was needed to meet pileated woodpecker needs and up to 243 ha (600 ac) in less suitable habitat. Bull and Jackson (1995) estimated the average home range size of pileated woodpecker pairs in northeastern and western Oregon varied from 407 to 478 ha (1,006 to 1,181 ac), respectively.

The EIS for the Six Rivers Forest Plan (USFS 1995d) presented a habitat capability model for the pileated woodpecker that identified high, moderate, and low capability habitats based on vegetation type, seral stage, habitat (nest territory) size, nest type characteristics, snag density, and downed log density. High-capability habitat (preferred) consists of mature (111 to 200 yr) or old growth (greater than 200 yr) habitats of the following types: (1) mixed evergreen with chinquapin or rhododendron; (2) Klamath montane with Douglas-fir; and (3) Coast Range montane with Douglas-fir. Preferred habitats also are within 0.4 km (0.25 mi) of water; have relatively high snag densities, including large snags capable of being used as a nest tree; and a large number of downed logs. Moderate-capability habitats (required) are younger (early mature, 70 to 111 yr), further from water, and had lower densities of snags and downed logs.

Bull and Jackson (1995) summarize information on breeding phenology of the pileated woodpecker. In California, the time of nesting is a function of elevation with young fledging as early as mid-May in coastal areas and from late May to early July in montane locations. Birds excavate nest cavities by striking the tree with the bill and chipping away wood, an activity that can take three to six weeks. Nest cavities are used only once as nest sites, but are frequently used as roost cavities later. The pileated woodpecker raises only one brood per year unless the eggs are destroyed. If the first brood is destroyed, the female lays again, and sometimes uses the same nest cavity. A clutch size of four eggs is typical with a range of one to six. The incubation period is about 18 days with both sexes involved in attending the nest. Fledging occurs 26 to 31 days after hatching in California.

The information presented here on food habits is primarily from Bull and Jackson (1995) unless otherwise noted. The pileated woodpecker consumes mainly insects, but is known to also eat wild fruits and nuts when available. Carpenter ants found in decaying snags are the single

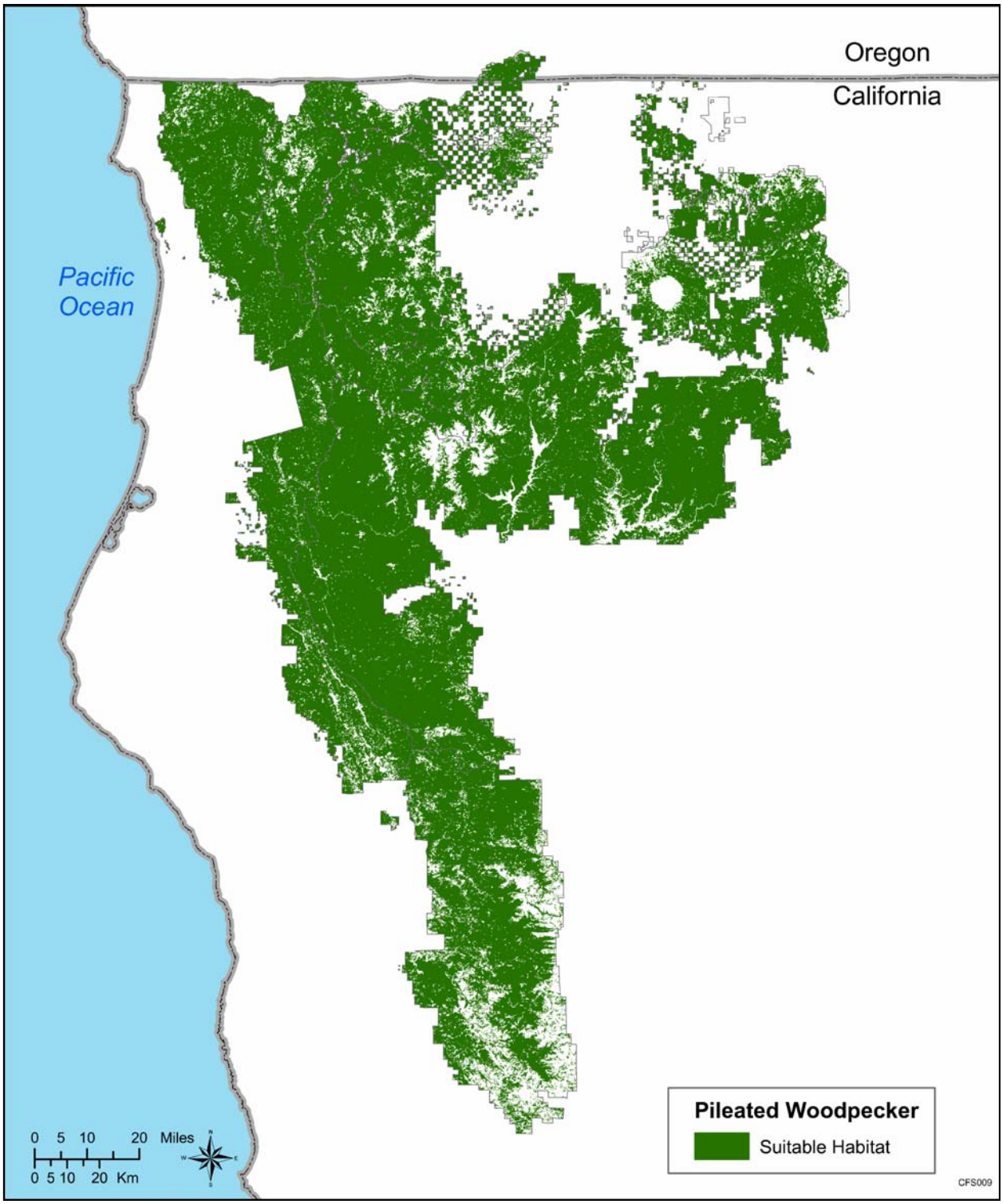


Figure 9 Pileated Woodpecker Habitats on the Four Forests of the Northern California Province

most important food item taken throughout the year although wood-boring beetle larvae are also important. Both ants and beetle larvae are consumed all year long. The dependence on downed logs and snags as foraging sites has been documented in Oregon populations of pileated woodpeckers. Feeding occurred on logs 38% of the time, 38% on dead trees, 18% on live trees and 6% on stumps (Bull and Holthausen 1993). The pileated woodpecker also gleans branches, trunks and bark of trees in search of food. It picks western spruce budworm (*Choristoneura occidentalis*) from Douglas-fir and grand fir branches. In Oregon, adults fed on ponderosa pine cones during early fall. During the winter season, the pileated woodpecker feeds primarily by excavating insects from soft or decaying wood. In northeastern Oregon, diet composition based on 330 scat samples included 68% carpenter ants, 29% thatching ants (*Formica* spp.), 0.4% beetles, and 2% other arthropods.

The pileated woodpecker has been described as a keystone species in forests of the Pacific Northwest (Aubry and Raley 2002). A keystone species is critical to ecosystem function and many associated species because its influence on ecosystem characteristics is disproportionately large compared to its abundance (Duncan 2003). The pileated woodpecker is the only species that excavates extensively into sapwood and heartwood for food (Duncan 2003). These excavations are used by other birds to access prey that would normally be unavailable to them. Pileated woodpecker excavations also expose wood in live trees, snags, and logs to insect attack and fungal infection and therefore increase the rate of nutrient cycling (Duncan 2003). Excavation in live wood exposes healthy trees to heart-rot fungi and increases the future availability of nesting, roosting, and foraging habitat (Duncan 2003). The health of pileated woodpecker populations may be critical to that of a wide variety of other forest wildlife species. Monitoring pileated woodpecker populations would provide information relevant to these other species.

3.6.2 Status, Population Trends, and Factors Affecting Populations of the Pileated Woodpecker

The pileated woodpecker is not listed as threatened or endangered by either the State of California or the Federal government. It does not have special protection status in the Northern California Province.

Population density estimates for pileated woodpeckers are quite variable, and depend on the census method and location of populations within its geographical range. Bull and Jackson (1995) summarized population density information based on data from various studies and showed that density varies from one breeding pair per 160 to 220 ha (395 to 543 ac) in California to one pair per 356 ha (879 ac) in northeast Oregon. They did not reference any studies for the Northern California Province.

North American Breeding Bird Survey results for the northern California region indicate that pileated woodpeckers are encountered regularly, but in low numbers (Sauer et al. 2003). Less than one individual observed per survey route per year is typical for the region. Over the 1966 to 1996 period, breeding bird survey data suggest that the population of pileated woodpeckers has been declining in the region (about 2% per year).

Audubon Christmas Bird Count data (Audubon 2003) for pileated woodpeckers in count circles within the Northern California Province are shown in Figure 10. Pileated woodpeckers were observed in relatively low numbers, but in most years of the survey. The species was observed in all six of the count circles in the Northern California Province, but highest numbers were observed in the Willow Creek and Yreka circles. No population trend is apparent when the data from 1980 to 2003 for all count circles in the Province are plotted (Figure 11).

Timber harvesting is probably the single most important factor that has reduced habitat for the pileated woodpecker. The removal of large diameter trees and standing dead trees, and loss of canopy closure has reduced available habitat (Bull and Jackson 1995). The loss of canopy cover and forest fragmentation are likely factors that increase vulnerability of pileated woodpeckers to predation from raptors as they move between habitat patches. CPIF (2003a) reported that the most important threats to the pileated woodpecker are conversion of forest habitats to nonforested habitats, short-rotation of forest stands, even-aged forest-management practices, monoculture forestry, fragmentation, and removal of timber harvest residue and downed logs from the forest floor.

Predation may be a factor in pileated woodpecker nestling mortality. Raccoons (*Procyon lotor*), bobcats (*Felis rufus*), American martens, long-tailed weasels (*Mustela frenata*), tree squirrels (*Sciurus* spp. and *Tamiasciurus* spp.), American crows (*Corvus brachyrhynchos*), and jays are known to prey on nestlings (CDFG 2003d). Adults are preyed on by the northern goshawk, Cooper's hawk, red-tailed hawk, great horned owl, and American marten (Bull and Jackson 1995).

3.6.3 Suitability of the Pileated Woodpecker as a Management Indicator Species

The pileated woodpecker satisfies most of the criteria we established for a suitable management indicator species for the Forests of the Northern California Province (see Section 1). It occurs throughout the year in the Province and does not migrate. The range of the species includes portions of all four Forests of the Province. It is easy to identify and cannot be confused with any other species. Monitoring programs exist for forest birds in general and woodpeckers in particular and could be used to monitor populations of pileated woodpeckers. However, the species exists in low numbers in the Province, and these low numbers may make it difficult to develop a statistically robust program to detect the effects of forest-management practices on this species.

Pileated woodpeckers utilize a variety of conifer forest types and do not appear to rely on any one particular type or species. The species appears to be dependent on old-growth forests, and habitat capability models have been developed and used to evaluate the effects of management practices on populations of this species. However, quantitative relationships between habitat characteristics and pileated woodpecker populations have not been developed.

Mellen et al. (1992) noted that pileated woodpeckers are known to use immature (less than 70 yr old) forest stands and suggested that they may not be good indicators for other species

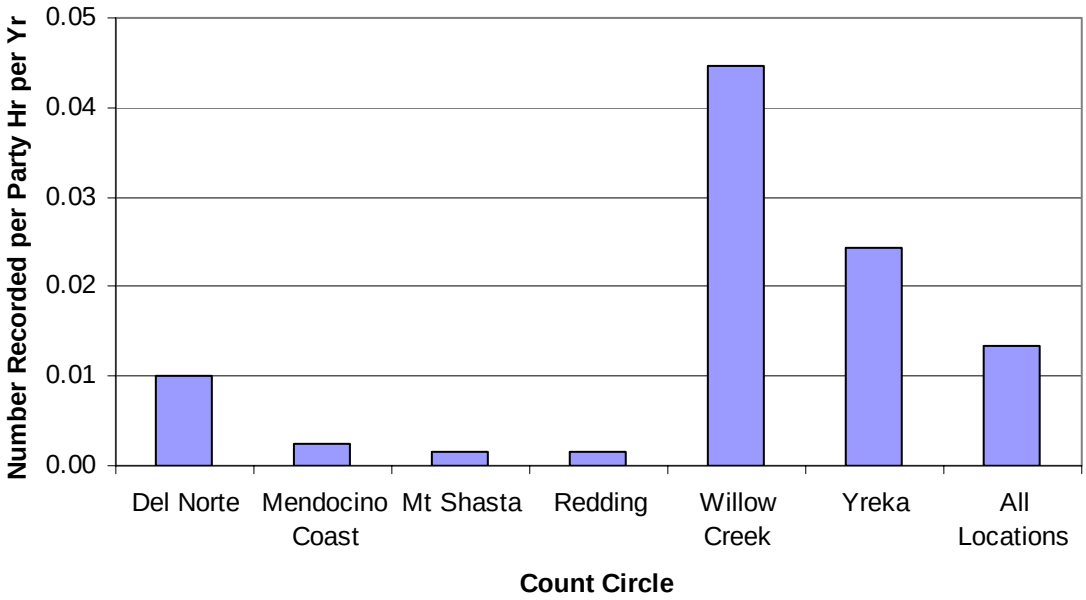


Figure 10 Mean Number of Pileated Woodpeckers Counted per Year per Party-Hr in Audubon Christmas Bird Counts on Count Circles within the Northern California Province (1980-2003)

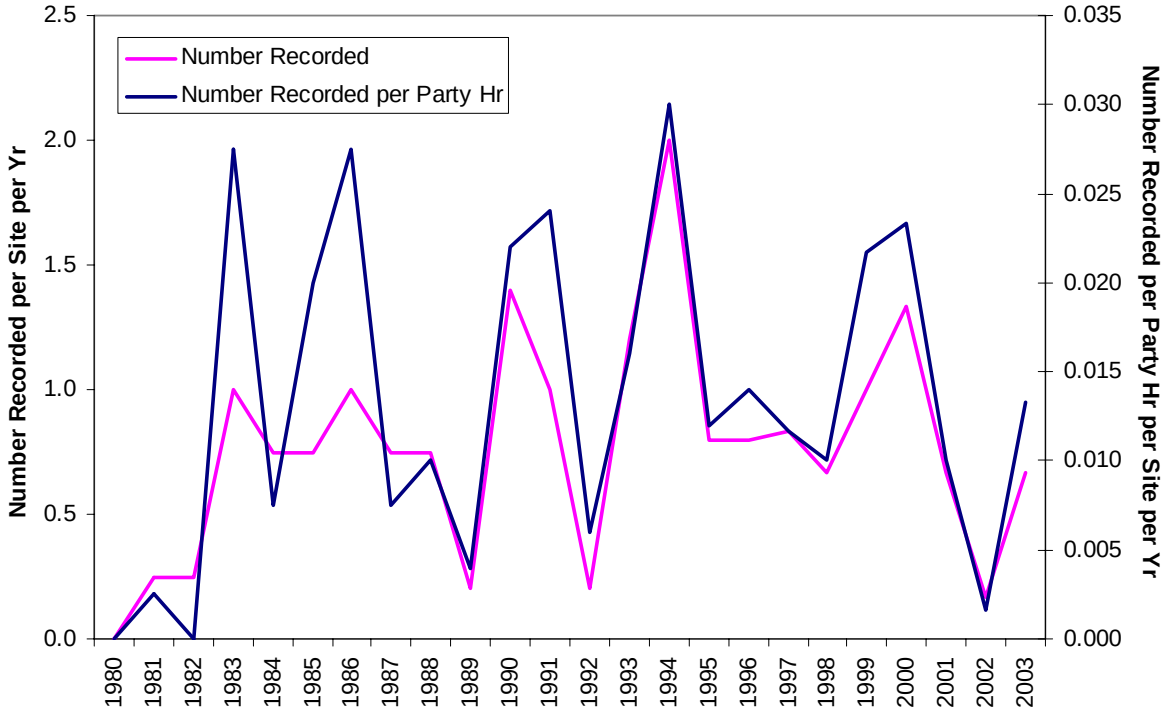


Figure 11 Mean Number of Pileated Woodpeckers Counted per Year in Audubon Christmas Bird Counts within the Northern California Province (1980-2003)

associated with mature forests. Aubry and Raley (2002) suggest that the pileated woodpecker is a keystone species in Pacific Northwest forests and that the health of pileated woodpecker populations may be critical to that of a wide variety of other forest wildlife species. Detailed studies in Oregon (Bull 1987; Bull and Holthausen 1993; Aubry and Raley 1994) have evaluated the density and size of snags needed for foraging, nesting, and roosting sites and suggest that similar monitoring could be used in the Northern California Province to monitor relative abundance of pileated woodpeckers in response to forest-management activities.

3.6.4 Monitoring Protocols for the Pileated Woodpecker

Several methods for monitoring pileated woodpecker populations have been used in the western portions of the species range including radio-tracking, point counts, call-back surveys, line transects, and spot-mapping methods.

Line-transect surveys have been used to survey pileated woodpeckers on the Wallowa–Whitman and Umatilla Forests in northeastern Oregon (Bull and Holthausen 1993). Surveys were taken on nine study areas along variable-width corridors established at about 1 km (0.6 mi) intervals. There were two to three transect lines in each study area with an average length of about 8 km (13 mi), which allowed researchers to come within 0.5 km (0.3 mi) of all locations in each study area. Surveys were conducted for a period of three to five hours, one hour after sunrise from late March to early May in three consecutive years. One person walked along each transect and stopped every 0.5 km (0.3 mi) for five minutes to listen for and observe pileated woodpeckers. If no detections were made, the observer vocalized an imitated, pileated woodpecker call every 30 s to elicit a response. All transects within a study area were sampled on the same day. A relative index of pileated woodpecker abundance was calculated from the line transect data in each study area by dividing the total number of birds detected by the transect length. Density could not be directly calculated because birds often flew toward the observer before detection.

Radio-tagging was used in a study to determine roosting habitat of pileated woodpeckers in Union, Baker, and Umatilla Counties of northeastern Oregon (Bull et al. 1992). Twenty-two adults were captured at nest sites on five study areas during June and July and five adults were captured at roost sites from July to December. Birds were equipped with an 11-g (0.4-oz), two-stage transmitter mounted on a backpack harness. Birds were located every two weeks by following them to a roost tree or by locating the bird after dark and marking the roost tree. The survey occurred over a nine-month period from June 1989 to March 1990.

Mellen et al. (1992) monitored summer home range and habitat use of pileated woodpeckers in western Oregon using radio-tagging. Radio transmitters were attached to 24 adults captured at nests after egg hatching or at a point in time after incubation was assumed to be complete. Problems occurred when radio antennas on some birds were broken or bent, which limited the range of the radio signal, making it impossible to continue monitoring bird movements. Adequate data were collected, however, to evaluate the home ranges of 11 birds and the habitat use of 14 birds.

Although relatively more costly than monitoring birds using line transects or point counts, radio-tracking can be used to measure home range, movements, and habitat use of individual birds. Radio tracking can be used to monitor individual birds in rugged terrain unless data must be collected by observers on foot.

British Columbia (MELP 1999a) has developed inventory methods for woodpeckers including the pileated woodpecker. In evaluating various survey methods used for woodpeckers, MELP (1999a) addressed the advantages and disadvantages of call-playback surveys, woodpecker-sign surveys, spot mapping, and distance surveys (line transects and point counts) in an attempt to develop standardized survey protocols. Much of their report is based on woodpecker studies conducted in the Pacific Northwest, and thus is applicable to pileated woodpecker populations in the Northern California Province. See Section 3.5.4 for a discussion of these and other survey techniques, and the advantages and disadvantages of each.

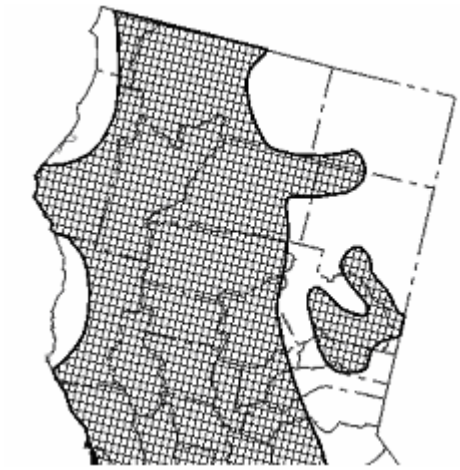
3.7 ACORN WOODPECKER

The acorn woodpecker is a medium-sized woodpecker that weighs 65 to 90 g (2.3 to 3.2 oz) (Koenig et al. 1995), has a typical body length of 23 cm (9 in.), and wing span of about 43 cm (17 in.) (Sibley 2000). Adults have a black and white face, a white rump patch, white wing patches, and a glossy black back. A red crown and yellowish color on the throat is typical adult coloration in both sexes. No other woodpecker species is similarly patterned.

3.7.1 Distribution, Habitat, and Life History of the Acorn Woodpecker

The geographic range of the acorn woodpecker in northern and central California includes the Klamath Mountains and Cascade Mountains east to southwest Modoc County and the western slopes of the Sierra Nevada to mountains around the southern end of the San Joaquin Valley (Small 1994). The acorn woodpecker does not occur in the low coastal woodlands of Del Norte and Mendocino Counties. Further south, the acorn woodpecker has a spotty distribution and occurs in only a few coastal lowland areas (Koenig et al. 1995). The species' geographic range also includes western Oregon, southeastern Arizona, the mountains of New Mexico, the Trans Pecos region of west Texas, the mountains of Mexico and Costa Rica, and the highland area of Panama. The acorn woodpecker is found in all four Forests of the Northern California Province (CDFG 2002).

Range of the Acorn Woodpecker in Northern California (CDFG 2002)



Acorn woodpeckers are very dependent on forests where oak trees are present (Koenig et al. 1995). In California, they are typically found in oak woodlands, Douglas-fir, and redwood forests in mountainous areas at altitudes that extend to the upper limit of oak trees which can vary from about 900 to 2,400 m (3,000 to 8,000 ft) in northern California. Acorn woodpeckers are disappearing from the Central Valley and other interior valleys because of the disappearance of large oak trees (Koenig et al. 1995). Habitat commonly used by the acorn woodpecker is a mixture of oak woodland growing on alluvial terraces adjacent to riparian groves of sycamore and mixed pine forest (Small 1994). Desirable habitat for the acorn woodpecker in the western Sierra Nevada of California is characterized as open oak forest stands and oak-conifer forest that are at least 6 ha (15 ac) in size. Oak and oak-conifer stands should contain at least four species of oaks and contain snags (CDFG 2003e).

CWHRS habitat types that are most suitable for the acorn woodpecker include blue oak foothill pine, blue oak woodland, coastal oak woodland, Valley oak woodland, montane hardwood conifer, montane hardwood, montane riparian, and ponderosa pine. CalVeg community types that correspond to these habitat types are presented in CDFG (1998). The distribution of acorn woodpecker habitats on the four Northern California Province Forests is mapped in Figure 12.

Acorn Woodpecker Habitats in Forests of the Northern California Province (CDFG 2002)

Habitat Type	Suitability[1]
Blue oak foothill pine	0.68
Blue oak woodland	0.68
Coastal oak woodland	0.68
Valley oak woodland	0.68
Montane hardwood conifer	0.67
Montane hardwood	0.63
Montane riparian	0.57
Ponderosa pine	0.54
Redwood	0.27
Sierran mixed conifer	0.25
Klamath mixed conifer	0.18
White fir	0.18
Eastside pine	0.16
Jeffrey pine	0.16
Douglas-fir	0.05

[1] Suitability varies from 0 to 1 with 1 being optimum.

The EIS for the Klamath Forest Plan (USFS 1994b) presented a habitat capability model for hardwood-dependent wildlife species that included the acorn woodpecker. The model identified high, moderate, and low capability habitats based on the frequency of low intensity fires (less than every five years considered preferred), stand age (80 to 200 yr preferred), high snag density, large patch size, and a high percentage of oak (greater than 50% preferred).

The following discussion of acorn woodpecker reproduction is based on information from CDFG (2003e) and summary information from Koenig et al. (1995). Adult acorn woodpeckers excavate their own nest holes, and nest cavities vary from 1.8 to 30 m (6 to 100 ft) above the ground. Soft, decaying wood of live trees or snags of oaks, sycamores, and various conifers typically are used as nest sites. Acorn woodpeckers breed from early April through late July and produce one or two broods per year. Second broods tend to be produced in years that follow a large acorn crop and mild winter weather. Peak breeding activity occurs in May and June. The average clutch size is four to five eggs with large clutches attributed to more than one female laying in a single nest. Incubation occurs over an 11 to 14 day period, and the young fledge in about 30 to 32 days.

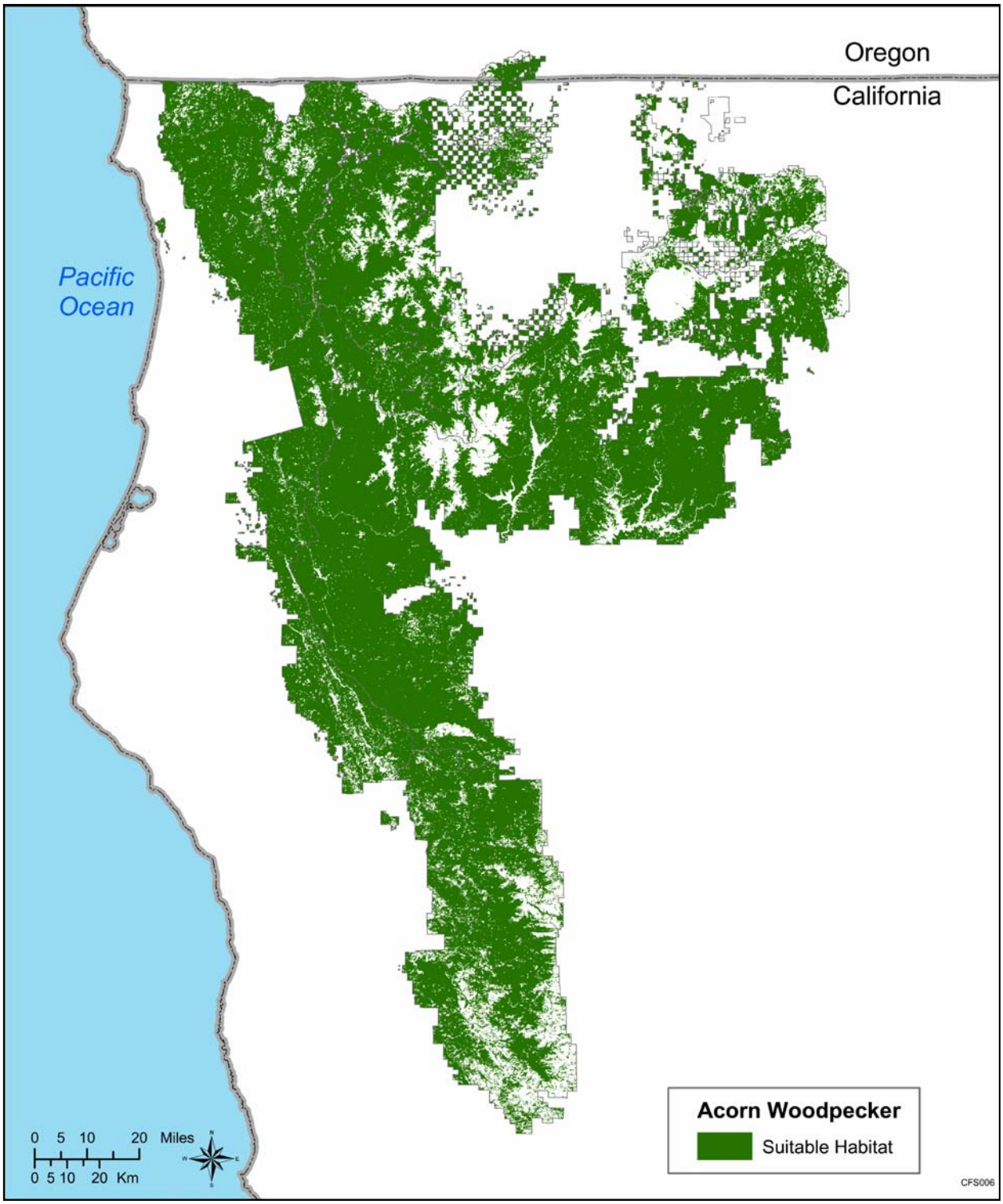


Figure 12 Acorn Woodpecker Habitats on the Four Forests of the Northern California Province

The acorn woodpecker lives in communal groups that contain up to 16 individuals and contain at least one breeding pair. Nonbreeding members help in parental care at the nest, but the breeding pair contributes the most time in nesting activities and feeding nestlings (Mumme et al. 1990). Communal groups defend territories containing large, widely spaced trees that are used for acorn storage. Many territories are permanently occupied during the year and over several years. Average territory size in the Central Valley of California was 4.7 ha (11.5 ac) with a range of 1.5 to 8.1 ha (3.8 to 20 ac). Territories are abandoned when a failure of the acorn crop occurs.

The diet of acorn woodpeckers is mostly acorns and insects (Koenig et al. 1995). Small (1994) reports that "granaries" of stored acorns are created in oaks, sycamores, pine trees, fence posts and in telephone or power poles. Unripe acorns and insects comprise the primary diet of the acorn woodpecker in summer and fall. Acorns are gathered from trees and the ground and either consumed or stored in the granaries. The abundance of stored acorns is critically important to winter survival. During late winter when acorn stores are depleted, the acorn woodpecker drills holes in trees to feed on sap. Insects are important in the diet during spring months.

Individual granary trees may have up to 50,000 holes that are used to store individual acorns (Koenig et al. 1995). Each group of acorn woodpeckers has one primary and one or more secondary granary trees. Holes are not drilled deep enough to damage tree health since they normally do not penetrate the cambium layer. Large granary trees are the result of several years of drilling by many generations of acorn woodpeckers. Holes are reused from year to year as storage sites.

3.7.2 Status, Population Trends, and Factors Affecting Populations of the Acorn Woodpecker

The acorn woodpecker is not listed as threatened or endangered by either the State of California or the Federal government. It does not have special protection status in the Northern California Province. Information on population density was not found for acorn woodpecker populations in the Northern California Province. Koenig et al. (1995) reported a maximum density of 8 individuals per ha (3.2 per ac) elsewhere in California based on data from Root (1988).

North American Breeding Bird Survey results for the northern California region indicate that acorn woodpeckers are encountered regularly, and in fairly high numbers in certain areas (Sauer et al. 2003). The number of individuals observed per survey route per year ranged from less than one to as many as 30 individuals. Over the 1966 to 1996 period, breeding bird survey data suggest that the population of acorn woodpeckers has been relatively stable in the region.

Audubon Christmas Bird Count data (Audubon 2003) for acorn woodpeckers in count circles within the Northern California Province are shown in Figure 13. Acorn woodpeckers were observed in relatively high numbers, and in all years of survey. The species was observed in all of the count circles in the Northern California Province, except for Del Norte; highest

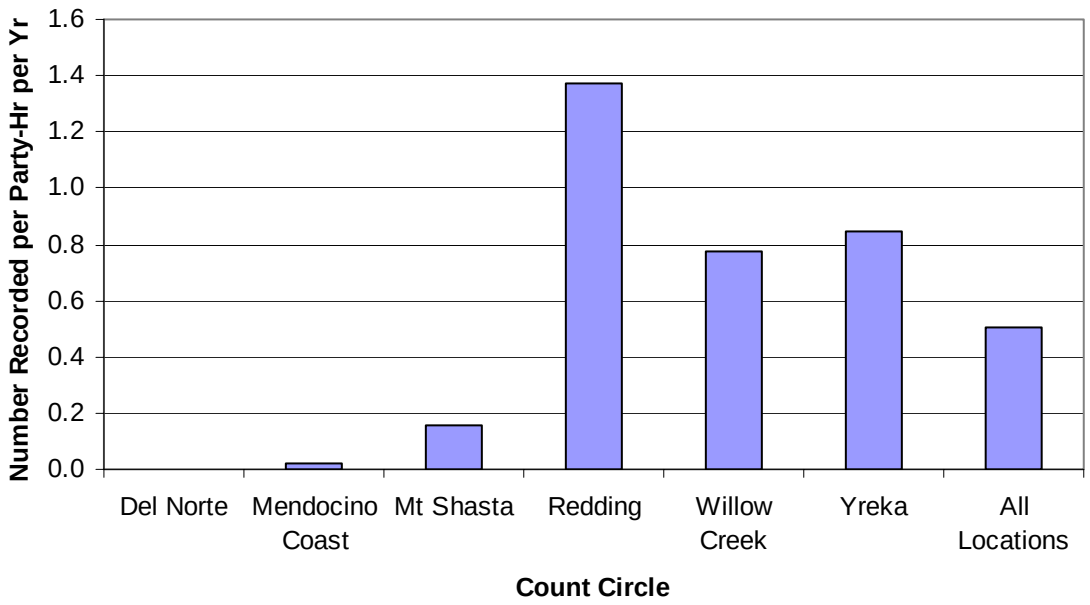


Figure 13 Mean Number of Acorn Woodpeckers Counted per Year per Party-Hr in Audubon Christmas Bird Counts on Count Circles within the Northern California Province (1980-2003)

numbers were observed in the Redding, Willow Creek, and Yreka circles. No population trend is apparent when the data from 1980 to 2003 for all count circles in the Province are plotted (Figure 14).

The most serious threats to the acorn woodpecker are habitat loss and degradation (Koenig et al. 1995). Removal of large mast-producing species such as oaks and pines that provide food, granaries, nest, and roost sites can adversely affect the acorn woodpecker. In many parts of the acorn woodpecker's range, large areas of oak and other important woodlands have been cleared for firewood or development. Reduction in the acorn crop has caused temporary, localized abandonment of acorn woodpecker habitat (Koenig et al. 1995). Disturbance of montane riparian forest stands important to acorn woodpeckers has resulted from overgrazing (Koenig et al. 1995). An additional potential threat is competition from hole-nesting species such as the European starling (*Sturnus vulgaris*). Despite these threats, acorn woodpecker population declines have not been apparent in California.

3.7.3 Suitability of the Acorn Woodpecker as a Management Indicator Species

The acorn woodpecker satisfies the criteria we established for a suitable management indicator species for the Forests of the Northern California Province (see Section 1). It occurs throughout the year in the Province and does not migrate. The range of the species includes portions of all four Forests of the Province. It is easy to identify and cannot be confused with any other species. Monitoring programs exist for forest birds in general and woodpeckers in

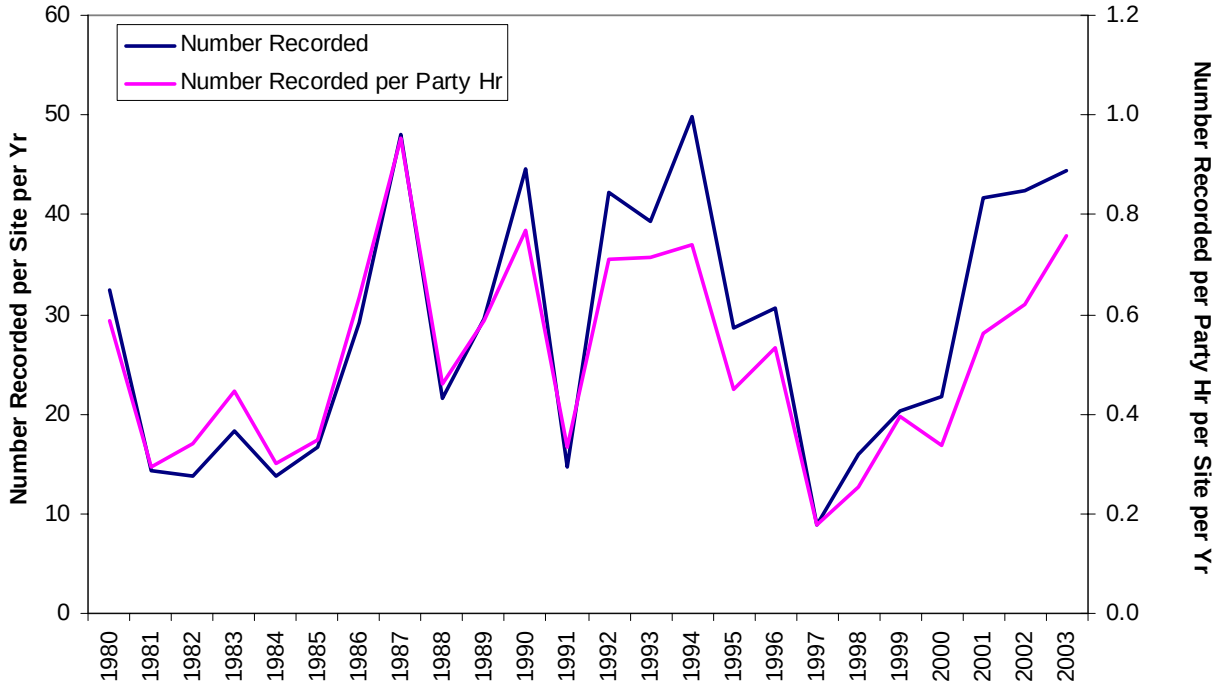


Figure 14 Mean Number of Acorn Woodpeckers Counted per Year in Audubon Christmas Bird Counts within the Northern California Province (1980-2003)

particular and could be used to monitor populations of acorn woodpeckers. The species is relatively common throughout the Province, and as a consequence it should not be difficult to establish a statistically robust program to detect the effects of forest-management practices on this species.

Behavioral characteristics of the acorn woodpecker facilitate monitoring. It forms communal groups that are conspicuous. Its use of granary trees where acorns are stored facilitates documentation and monitoring of populations.

Acorn woodpeckers utilize a variety of habitat types, but are particularly dependent on oaks. Habitat capability models that evaluate the effects of management practices on oak woodlands, and subsequently, acorn woodpeckers, have been developed and used to evaluate the effects of management practices on populations of this species. However, specific quantitative relationships between habitat characteristics and acorn woodpecker populations have not been developed. So much research has been conducted on behavior, dispersal, nesting, and foraging behavior elsewhere in California (e.g., the Hastings Reservation in central coastal California) that an extensive literature database is available to provide the basis of a monitoring program.

3.7.4 Monitoring Protocols for the Acorn Woodpecker

No published information was found on monitoring protocols used for northern California populations of the acorn woodpecker. Koenig et al. (2000) discussed the use of radio-tracking to study dispersal of acorn woodpeckers from the communal group location. This technique provided information on birds dispersing short distances. Radio-tracking could be used to monitor acorn woodpecker response to habitat fragmentation or modification. However, radio-tracking would be an expensive and time-consuming method for use as a general protocol for monitoring acorn woodpecker populations.

British Columbia (MELP 1999a) has developed inventory methods for woodpeckers. In evaluating various survey methods used for woodpeckers, MELP (1999a) addressed the advantages and disadvantages of call-playback surveys, woodpecker-sign surveys, spot mapping, and distance surveys (line transects and point counts) in an attempt to develop standardized survey protocols. See Section 3.5.4 for a discussion of these and other survey techniques and the advantages and disadvantages of each.

The species' use of conspicuous granary trees could form the basis of a monitoring program that examined the relative abundance of these features at various sampling locations across the Province. Care would need to be taken to ensure that granary trees were active during the survey. See MELP (1999a) for a full discussion of the application of woodpecker- sign surveys for monitoring relative abundance.

In general, caution should be taken in interpreting annual monitoring data on relative abundance or density estimates of acorn woodpeckers. Koenig et al. (2000) indicate that acorn woodpeckers abandon oak forests in years when acorn production is low. Thus, observations or measurements of acorn production should be conducted concurrently with the monitoring of acorn woodpecker populations.

3.8 MOUNTAIN CHICKADEE

The mountain chickadee is a small songbird that is about 13.3 cm (5.25 in.) long with a wing span of 21.6 cm (8.5 in.) and weighs approximately 11 g (0.4 oz) (Sibley 2000). The mountain chickadee has a black cap and throat; a black stripe through the eye; a white stripe above the eye; plain gray back, wings, and tail; and white underparts (Sibley 2000). The white stripe above the eye distinguishes this species from all other chickadees in North America. There is no difference in appearance between the sexes.

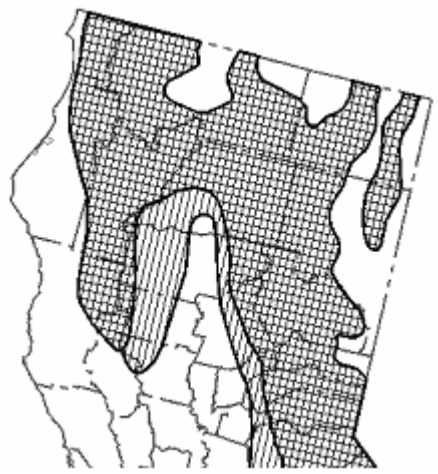
3.8.1 Distribution, Habitat, and Life History of the Mountain Chickadee

Unless otherwise noted, the following information on distribution, habitat, and life history is taken from the mountain chickadee species account of McCallum et al. (1999). The mountain chickadee's geographic range extends from southern Arizona and New Mexico northwestward through Colorado, Utah, Nevada, central California, mountainous regions of

western Wyoming, Idaho, Montana, Oregon, Washington, western Alberta, British Columbia and the Yukon Territory in Canada. The range in Oregon and Washington is predominantly in the Cascade Mountains, and Blue and Wallowa Mountains of northeastern Oregon.

In east central and northern California, it is a permanent resident of montane coniferous forests. The mountain chickadee occurs from the Oregon border southward in the Klamath Mountains, the Siskiyou Mountains and the inner northern Coast Range to northeastern Lake County (Small 1994). It also is found in the Cascades-Sierra region south to Kern County. CDFG (2003f) describes the mountain chickadee's distribution in northern California as the northern interior mountain ranges, and the Sierra Nevada Mountains. The species is found in all four Forests of the Northern California Province.

Range of the Mountain Chickadee in Northern California (CDFG 2002)



The geographic ranges of the mountain and black-capped chickadee (*Poecile atricapillus*) overlap in some parts of the western U.S. and western Canada, although black-capped chickadees typically inhabit deciduous or broadleaf vegetation and mountain chickadees normally occur in coniferous vegetation. Some overlap exists in breeding territories in transitional and mixed habitat types.

Terres (1991) described mountain chickadee habitat as mixed conifer forests that consists of lodgepole pine, spruces, firs, junipers, and hemlocks in the inner coastal ranges of the Pacific Northwest, Sierra Nevada Mountains, and the Rocky Mountains. In Modoc County, California, mountain chickadees occur in forests dominated by white fir and ponderosa pine (Kleintjes and Dahlsten 1995). Other tree species in their study area included Jeffrey pine, incense cedar, and western juniper. Grundel (1987) studied feeding habits of mountain chickadees in the Modoc National Forest northeast of Adin, California, on three sites dominated by ponderosa pine, sugar pine, and white fir. Forest sites in their study varied in elevation from 1,600 to 2,000 m (5,250 to 6,560 ft).

McCallum et al. (1999) reported that seasonal use of different trees by mountain chickadees in the western Sierra Nevada followed a certain pattern: (1) white fir trees were favored during nesting; (2) there was a strong reliance on black oak during fall; (3) incense cedar was favored during winter, and (4) sugar pine was used heavily during late summer. In winter, mountain chickadees often moved down-slope from coniferous forest at higher elevations to riparian and hardwood habitats, and occasionally moved as far as the humid outer coastal ranges. However, some individuals continued to occupy subalpine coniferous forest during the winter months.

CWHRS habitat types that are most suitable for the mountain chickadee include a variety of conifer types. Hardwoods and other habitats have relatively lower suitability for this species.

CalVeg community types that correspond to these habitat types are presented in CDFG (1998). The distribution of mountain chickadee habitats on the four Northern California Province Forests are mapped in Figure 15.

Mountain Chickadee Habitats in Forests of the Northern California Province (CDFG 2002)

Habitat Type	Suitability[1]
Sierran mixed conifer	0.82
Klamath mixed conifer	0.70
White fir	0.70
Eastside pine	0.68
Jeffrey pine	0.68
Lodgepole pine	0.68
Ponderosa pine	0.68
Red fir	0.67
Montane hardwood conifer	0.52
Douglas-fir	0.47
Subalpine conifer	0.46
Juniper	0.44
Pinyon juniper	0.44
Aspen	0.43
Montane riparian	0.43
Montane hardwood	0.21
Blue oak foothill pine	0.16
Blue oak woodland	0.16
Coastal oak woodland	0.16
Closed cone pine cypress	0.16
Valley oak woodland	0.16
Montane chaparral	0.10
Mixed chaparral	0.05

[1] Suitability varies from 0 to 1 with 1 being optimum.

Mountain chickadees are monogamous and territorial during the breeding season, which extends from May to July; adults raise one or two broods each season. Nests are made in abandoned woodpecker holes in dead snags of deciduous or coniferous trees (CDFG 2003f), and are lined with soft plant material and fur (Terres 1991). Several studies have documented the use of nest boxes by mountain chickadees in northern California (Dahlsten and Copper 1979; Grundel 1987; Kleintjes and Dahlsten 1995). Clutch size varies from six to eight eggs, and incubation takes about 14 days. Young usually fledge 21 days after hatching.

Mountain chickadees form groups after the breeding season (McCallum et al. 1999). These groups consist of multiple pairs and juveniles. Group membership can be relatively stable from year to year and can consist of as many as three pairs and associated juveniles. Juveniles typically disperse after fledging and join groups outside their natal territory by August and September (McCallum et al. 1999).

In a study of mountain chickadees in a pinyon (*Pinus monophylla*)–juniper (*Juniperus osteosperma*) woodland in the White and Inyo Mountains of California, Hall and Morrison (2003) found that (1) nest construction occurred in early May; (2) mean clutch size was 6.1; (3) fledging occurred in late June; and (4) proximity to the nearest conspecific's nest was 74 m. Egg-laying occurred in early May during two years of a three-year study of breeding phenology in a white fir–mixed pine study site in Modoc County, California (Kleintjes and Dahlsten 1995). Dahlsten et al. (1992) reported mean clutch sizes of 6.9 to 7.2 and 5.5 to 5.9 for first and second broods, respectively, based on 24 yr of nest-box data from Modoc County. Mean clutch size for first broods was significantly lower at a higher elevation site (mean 6.9; 1,800 m [5,900 ft]) than at two lower sites (mean 7.1; 1,500 to 1,650 m [4,900 to 5,400 ft];) (Dahlsten et al. 1992).

McCallum et al. (1999) summarized the results of numerous studies on feeding behavior and diet of mountain chickadees. Several researchers documented foraging behavior and diet of the mountain chickadee in northern California based on observations at nest boxes (Dahlsten and

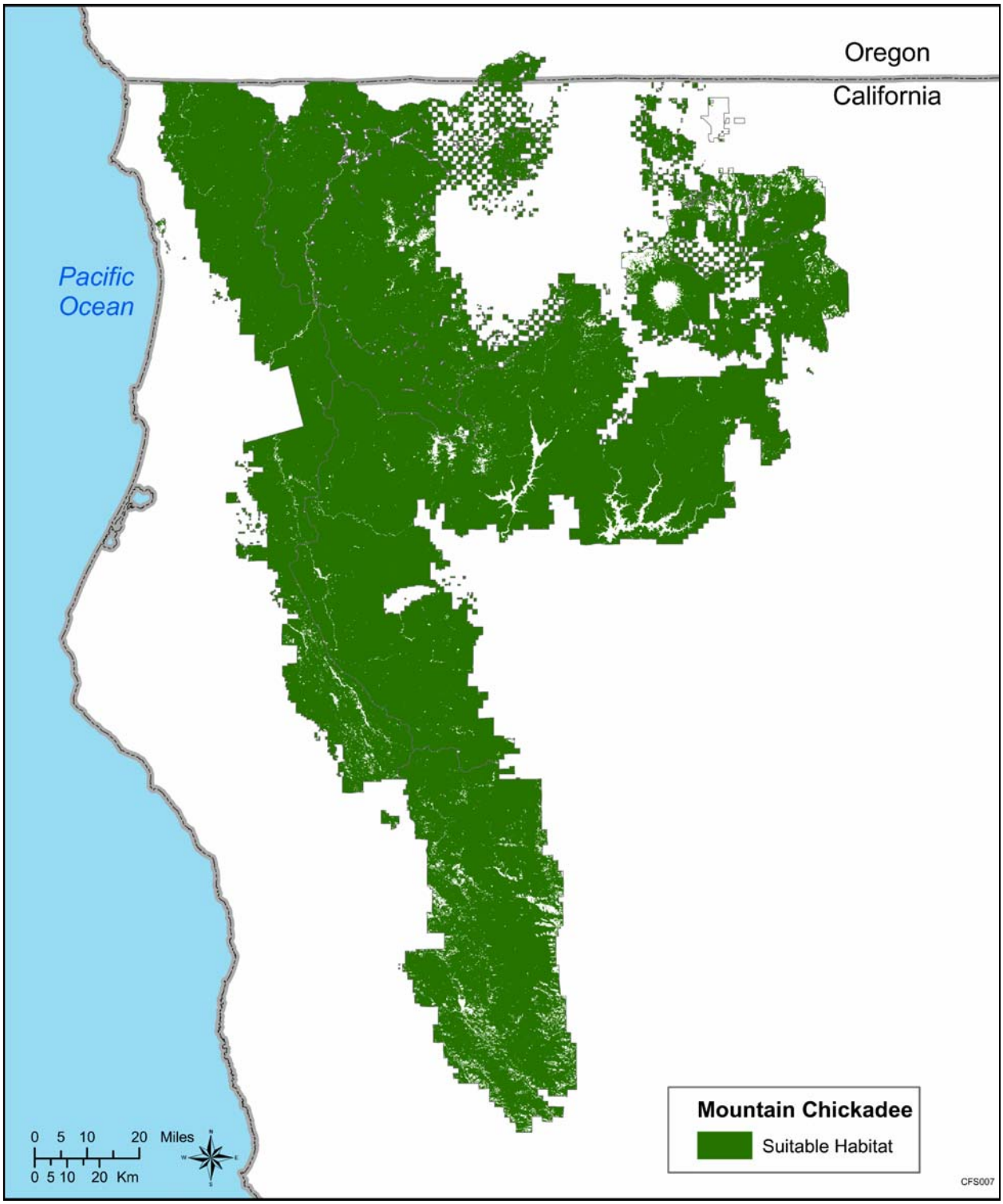


Figure 15 Mountain Chickadee Habitats on the Four Forests of the Northern California Province

Copper 1979; Grundel 1987; Grundel and Dahlsten 1991; Kleintjes and Dahlsten 1995). Little information was found that related feeding to forest structure or specific habitat variables. Kleintjes and Dahlsten (1995) showed a strong correlation between the timing of breeding and the presence of new shoots of white fir in the Modoc National Forest. Each year of their study, white fir bud expansion occurred approximately one month after initiation of egg-laying. This coincided with a period of greater availability of arthropods feeding on new fir needles.

Adult mountain chickadees forage mainly by gleaning insects from the outer periphery of coniferous trees (Kleintjes and Dahlsten 1995). Once fledging occurred, adults spent more time foraging within the canopy closer to the main trunk and main branches presumably to stay closer to young birds during the first few days after fledging.

Dietary studies conducted at nest boxes fitted with cameras showed that larval hymenopterans and lepidopterans, and adult beetles comprised most of the nestling diet. Adults showed a tendency to be opportunistic feeders and the variety of food items brought to the nests decreased with nestling age. Dahlsten and Copper (1979) analyzed stomach contents of adult mountain chickadees during a six-year study in northeastern California. Stomachs contained a variety of arthropods, some seeds, and unidentified vegetable material. Mean percentages of arthropods were highest in spring and fall (73% and 75% respectively) and lowest in winter (53%). Caching of seeds by mountain chickadees during the fall months has been documented in lodgepole pine and blue spruce (*Picea pungens*) forests of Colorado (McCallum et al. 1999). Caching behavior has also been reported in mountain chickadee populations in New Mexico and California.

3.8.2 Status, Population Trends, and Factors Affecting Populations of the Mountain Chickadee

The mountain chickadee is not listed as threatened or endangered by either the State of California or the Federal government. It does not have special protection status in the Northern California Province.

CDFG (2003f) summarized the literature on population densities of mountain chickadees in various parts of their geographic range (Table 4), but did not include data for the Northern California Province. McCallum et al. (1999) reported breeding territories in Alberta of 6.5 ha (16.1 ac), and 16 groups occupied a group territory of 260 ha (640 ac) area over a five-year period in New Mexico.

North American Breeding Bird Survey results for the northern California region indicate that mountain chickadees are encountered regularly, and in moderately high numbers in certain areas (Sauer et al. 2003). The number of individuals observed per survey route per year ranged from less than one to as many as 12 individuals. Over the 1966 to 1996 period, breeding bird survey data suggest that the population of mountain chickadees has declined (about 1%) in the southern portion of the area but increased (greater than 1.5%) in the northern portion. Regression analysis of the breeding bird survey data from 1966-1996 showed a significant 1.4% per yr decline in the overall U.S. population of mountain chickadees (Sauer et al. 1997).

Table 4 Population Density of the Mountain Chickadee in Arizona and California

Location	Habitat	Population Density (No. per 40 ha [100 ac])
Arizona		
White Mountains	Mixed coniferous forest	44.7 – 58.9
Northern	Juniper-pinyon–ponderosa pine ecotone	8
California		
Sierra Nevada	Mixed coniferous forest	31
Modoc National Forest	Mixed coniferous forest	48

Source: CDFG (2003f)

Data from the Audubon Christmas Bird Count show no change in numbers recorded from 1945 to 1985 in Washington, Oregon, and northern California, and a 1.9% increase per year over the entire species range (McCallum et al. 1999). Data for mountain chickadees in count circles within the Northern California Province are shown in Figure 16 (Audubon 2003). Mountain chickadees were observed in relatively high numbers, and in all years of survey. The species was observed in all of the count circles in the Northern California Province, except for Mendocino Coast; highest numbers were observed in the Mt. Shasta, Willow Creek, and Yreka circles. The data indicate a possible slight increase in numbers of mountain chickadees observed over the 1980 to 2003 period (Figure 17).

McCallum et al. (1999) reported on factors that limit populations and reproductive success in mountain chickadees. In some breeding populations, population size was a function of the size of the pinyon seed crop in the previous fall and winter. Weather also may influence population size. Hejl et al. (1988) documented an increase in breeding bird density in a western Sierra Nevada population from 1983 to 1985 following a record snowfall. The authors did not attribute any single factor to the increase in bird density, but did document an increase in arthropod biomass during the following breeding season.

Laboratory studies of the seasonal energetics of mountain chickadees suggest that cold winter temperatures may limit populations when food caches are scarce (Cooper 2000). In a time-activity study, mountain chickadees did not increase the percentage of the daylight hours spent foraging in winter even though the daily energy expenditure by an individual bird increased by 36% compared to requirements during the summer months. This suggests more efficient foraging during cold temperatures because of the use of readily available seed caches.

Parasite infestation of nestlings, nest predation, and disease are factors known to effect nesting success of mountain chickadees. Avian pox is found in California, Colorado, and New Mexico mountain chickadee populations. In northern California, over 90% of nests were infested with blood-feeding flies (*Protocalliphora sp.*). Weasels (*Mustela* spp.), rubber boa (*Charina bottae*), and gopher snake (*Pituophis melanoleucus*) are known to prey on chickadee eggs and nestlings in nest boxes (Dahlsten et al. 1992).

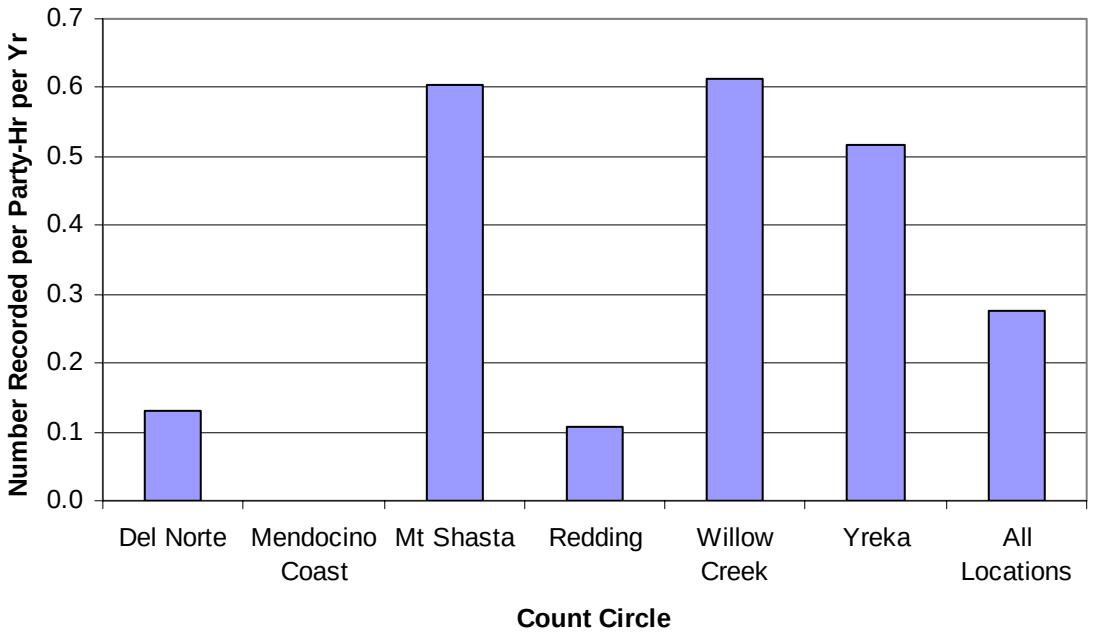


Figure 16 Mean Number of Mountain Chickadees Counted per Year per Party-Hr in Audubon Christmas Bird Counts on Count Circles within the Northern California Province (1980-2003)

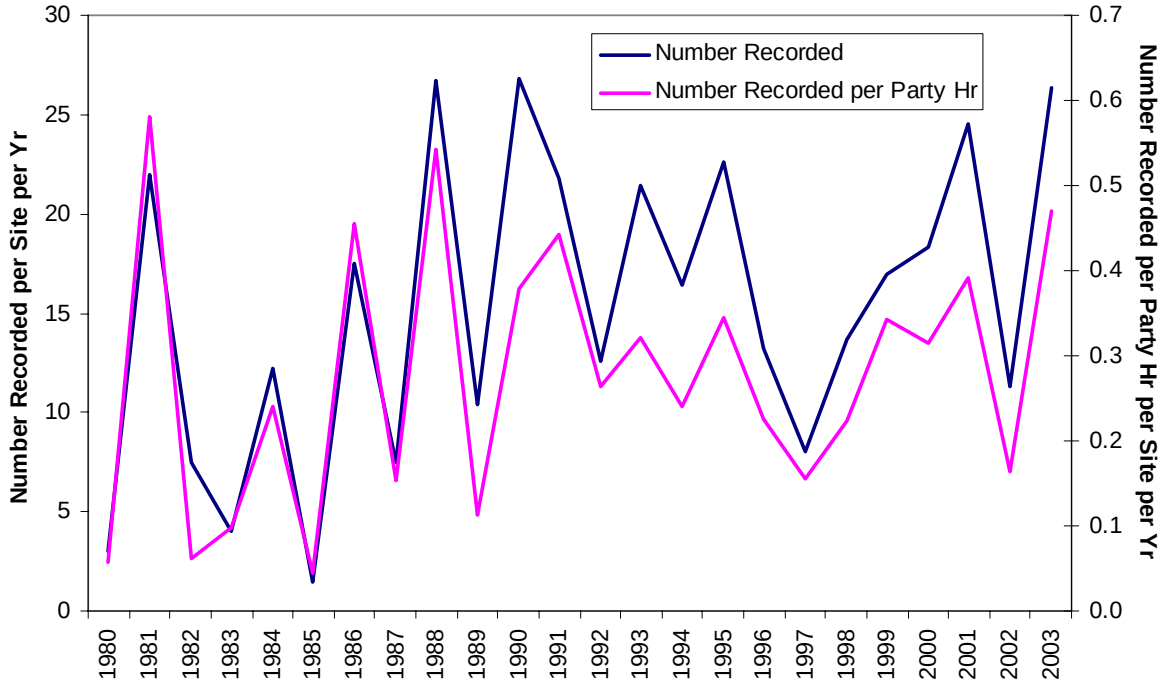


Figure 17 Mean Number of Mountain Chickadees Counted per Year in Audubon Christmas Bird Counts within the Northern California Province (1980-2003)

Fire was shown to affect mountain chickadee abundance at a site on Donner Ridge at the University of California Sagehen Creek Field Station in Nevada County, California (Bock and Lynch 1970). Surveys on burned and unburned sites conducted six, seven, and eight years after the burn found mean densities of 15.6 pairs per 40 ha (100 ac) on unburned sites and 5.2 pairs per 40 ha (100 ac) on burned sites. Burned areas were generally devoid of mature trees with the exception of a few mature Jeffrey pine and white fir in small clumps. Several fire-tolerant shrubs survived in the burned area. The unburned area consisted of a mixed conifer overstory of mature Jeffrey pine and white fir with some lodgepole pine, sugar pine, red fir, and incense cedar. Several shrubs occurred in the lower strata along with young conifers on the unburned plot.

McCallum et al. (1999) provided insight on the relationship between current and past timber-management practices on mountain chickadee populations. In the Apache Sitgreaves National Forest in Arizona, population density was 58.9 pairs per 40 ha (100 ac) on unlogged areas and 30.8 pairs per 40 ha (100 ac) in logged areas one year after logging. Franzreb (1978) noted that mountain chickadees were more generalized foragers (in terms of tree species foraged in) in logged areas than in unlogged areas. In the western Sierra Nevada of California, mountain chickadees typically used forested areas dominated by one of six tree species: Douglas-fir, ponderosa pine, white fir, incense cedar, black oak, and sugar pine. Past management activities have favored Douglas-fir and ponderosa pine, but removed the other four species. The dependence of mountain chickadees on insect prey found in incense cedar trees is believed to be a key factor to winter survival of mountain chickadees. McCallum et al. (1999) suggested that mountain chickadees in the Sierra Nevada avoid population cycles caused by the cyclical seed crops that occur in other portions of the species' range. Consumption of insect prey found in incense cedars in winter obviates the need to rely solely on seeds for survival in winter. Other researchers have suggested that maintaining 150 small incense cedars (< 20 cm dbh) per ha (2.5 ac) is adequate to support mountain chickadee populations in northern California (Morrison et al. 1989).

3.8.3 Suitability of the Mountain Chickadee as a Management Indicator Species

The mountain chickadee satisfies most of the criteria we established, identified in Section 1, for a suitable management indicator species for the Forests of the Northern California Province. It occurs throughout the year in the Province and does not migrate. The range of the species includes at least some portions of all four Forests of the Province, although it may be more abundant in the eastern portions of the Province.

The song and diagnostic characters of the mountain chickadee make it easy to detect and identify in the field. The white superciliary eye stripe is easy to detect and readily distinguishes the mountain chickadee from the black-capped chickadee. However, the white color is on only the tips of the feathers and can disappear as the feathers wear (Cicero 2000; Sibley 2000).

The species is relatively common in the Province, thus, collection of sufficient data to monitor the effects of forest-management practices on this species should be possible. The densities of both breeding and nonbreeding populations of mountain chickadees appear to be high enough to allow statistical analysis of survey and monitoring data. Based on the studies by

Franzreb (1978) and Bock and Lynch (1970) it seems that the mountain chickadee is relatively easy to survey in a variety of habitats (disturbed and undisturbed) and terrains.

The mountain chickadee has been the subject of much research, but, to our knowledge, monitoring of populations has not been used to determine the effects of management activities, and specific quantitative relationships between habitat characteristics and mountain chickadee populations have not been developed. The mountain chickadee uses a wide variety of conifer forest types, and, as a consequence, monitoring populations of this species may indicate the condition of conifer habitats in general rather than any one type of conifer habitat. A variety of survey techniques is routinely used for bird populations and could be applied to a monitoring program for the mountain chickadee. These are discussed in the following section.

3.8.4 Monitoring Protocols for the Mountain Chickadee

Several survey techniques have been used to study mountain chickadees. Spot mapping was used in the Willow Creek watershed of the Apache-Sitgreaves National Forest in Arizona to estimate population density of mountain chickadees (Franzreb and Omhart 1978). The study area was a mixed conifer forest dominated by Douglas-fir and ponderosa pine. Censuses were conducted in June, July, and August on two, 15.5 ha (38.2 ac) plots along parallel lines located about 50 m (165 ft) apart and 390 m (1,280 ft) long in logged and unlogged areas. Each study plot was censused six times each month for a total of 18 censuses during June, July, and August. Each census began one-half hour after sunrise and continued for two to three hours. Assuming the maximum of three hours per census-day per plot, the total effort per three month survey period for both plots would be about 110 hrs.

Bock and Lynch (1970) also used a spot-mapping method to determine mountain chickadee density on burned and unburned mixed conifer forests in the Sierra Nevada Mountains of California. The purpose of their study was to compare breeding bird populations on two permanent plots established five years after a forest fire that encompassed 15,800 ha (39,000 ac) of Donner Ridge. The areas surveyed were within two rectangular 8.5-ha (21-ac) study plots. Grid lines established in each plot, with marked posts along the lines, allowed observers to record the location of birds within the plot on photocopied maps. A total of 113 censuses of birds was conducted on the burned and unburned plots during May, June, and July in three consecutive years. Each survey lasted one to three hours. Assuming the maximum of three hours per survey, the amount of field time in this study totaled about 340 hr.

Relative abundance of mountain chickadees was determined using point counts in a study on five nest-box grids established in the White and Inyo Mountains of California at elevations of 1,515 to 4,245 m (4,970 to 13,925 ft) in pinyon–juniper woodland (Hall and Morrison 2003). Each grid consisted of nest boxes spaced at 50-m (165-ft) intervals. No information was given on time needed for the point-count surveys in this study.

Monitoring of nest boxes on a rectangular grid was used by Dahlsten et al. (1992) to estimate population density of mountain chickadees in Modoc County at elevations ranging from 1,500 to 1,800 m (4,920 to 5,900 ft) on a site dominated by white fir where logging had removed

most of the ponderosa and Jeffrey pine. Banding of adults and young trapped at the nest boxes enabled researchers to determine survival between years. Nest boxes were checked weekly during the nesting season between May and August.

Each of these techniques has advantages and disadvantages for use in monitoring mountain chickadee populations. Spot mapping is most effective in more open-canopy habitats. Spot mapping has limited use in dense closed-canopy areas where observers have difficulty locating individual birds or identifying trees used for nesting or foraging. Nest-box monitoring can provide information on presence and absence between years and changes in reproductive success based on nesting data. Distance methods, including line transects and point counts, may be used to determine absolute abundance in various habitats. These methods have the advantage of being relatively fast to conduct particularly when points or transects are along forest roads.

British Columbia (MELP 1999b) developed inventory methods for forest songbirds including chickadees. In evaluating various survey methods used for songbirds, MELP (1999b) addressed the advantages and disadvantages of encounter transects, point counts, and spot mapping in an attempt to develop standardized survey protocols. Spot-mapping was considered the most labor-intensive of the techniques considered and would be appropriate only for relatively small areas when absolute abundance data of breeding birds was required. Encounter transects and point counts, on the other hand, can be used to cover much greater areas with far less effort. Point counts were considered preferable to encounter transects, because they were more easily standardized and they were a more efficient method for obtaining large sample sizes. The conclusions in MELP (1999b) are expected to apply to the Northern California Province.

Huff et al. (2000) described a habitat-based point-count protocol for terrestrial birds in Washington and Oregon. Their approach is similar to other point-count protocols, but sampling is stratified according to habitat (see Section 3.5.4).

3.9 OAK TITMOUSE

The oak titmouse is a drab gray-colored bird with a medium gray or grayish-white breast and a small crest. Adult weight varies from 11 to 21 g (0.4 to 0.7 oz) in females and 12 to 20 g (0.4 to 0.7 oz) in males (Cicero 2000). Adult body length is about 14.6 cm (5.75 in.) and wing span is about 23 cm (9 in.) (Sibley 2000).

The oak titmouse and juniper titmouse were previously considered one species, the plain titmouse (*Parus inornatus)*. Geographic variation within the plain titmouse together with genetic differences within the family Paridae led to a reclassification of the plain titmouse as two sibling species in the genus *Baeolophus* (Cicero 2000; American Ornithologists' Union 1997). Variation in voice, morphology, coloration, and ecology also led to the reclassification of the plain titmouse into two separate species.

3.9.1 Distribution, Habitat, and Life History of the Oak Titmouse

The following account of the oak titmouse is taken largely from Cicero (2000) and is based on her review of the literature. The geographic range of the oak titmouse extends from southern Oregon through portions of central California and coastal regions southward to northern Baja California, Mexico. The sibling species, the juniper titmouse (*Baeolophus ridgwayi)* occurs from south central Oregon through most of Nevada, all of Utah, portions of Arizona, New Mexico, western Colorado, and a portion of southeastern Idaho. The oak titmouse is found mainly in oak or oak-pine woodlands and the juniper titmouse occurs in pinyon-juniper woodlands primarily in the Great Basin. Both species occur at elevations that range from 686 to 2,438 m (2,250 to 8,000 ft) (Cicero 2000).

In California, the oak titmouse is absent from the northwestern coastal region and the San Joaquin Valley. Breeding populations occur in northern California in Siskiyou County (Shasta Valley) and Trinity County (south fork of the Trinity River). The species' range extends eastward in Siskiyou County to the border with Modoc County where it has some contact with the juniper titmouse in Lava Beds National Monument. Small (1994) reported that the distribution of the oak titmouse in California is discontinuous and complex, with a northern California distribution in the Siskiyou Mountains in Humboldt and Siskiyou Counties; the Marble and Scott Bar mountains in Siskiyou County; and the Warner Mountains of the Modoc Plateau and the low mountains of central Lassen County. The oak titmouse is found in at least some portions of all four Forests of the Northern California Province, but its distribution in the forests is patchy and not well known.

Range of the Oak Titmouse in Northern California (CDFG 2002)

CPIF (2003b) program presented a map for the oak titmouse in California that shows confirmed and possible breeding locations within the current and historic range overlaid on the current distribution of oak woodland habitats. Potential breeding locations are located in the Six Rivers and Mendocino National Forests.

There is a clear affiliation of the oak titmouse with oak woodland habitat in California (CDFG 2003g). Typical habitat in Shasta County is oak or oak–pine dominated woodlands (Small 1994). The extreme northern California population occurs in woodlands dominated by Garry oak (*Quercus garryi*) and *Ceanothus* spp., and mixed oak woodlands (Small 1994). The CPIF Bird Conservation Plan (CPIF 2003c) described the ecology and breeding habitat of the oak titmouse in California based on the available literature. Breeding habitat was described as woodland habitat dominated by oaks, chaparral, and oak riparian woodlands. Cicero (2000) reported that dry, warm habitats without oaks are also inhabited by the oak titmouse. This habitat is comprised mostly of western juniper (*Juniperus occidentalis*).

CWHRS habitat types that are most suitable for the oak titmouse consist of a variety of oak woodland types as well as montane hardwood conifer and montane hardwood habitats. CalVeg community types that correspond to these habitat types are presented in CDFG (1998). The distribution of oak titmouse habitats on the four Northern California Province Forests are mapped in Figure 18.

Oak Titmouse Habitats in Forests of the Northern California Province (CDFG 2002)

Habitat Type	Suitability[1]
Blue oak foothill pine	0.79
Blue oak woodland	0.79
Coastal oak woodland	0.79
Valley oak woodland	0.72
Montane hardwood conifer	0.62
Montane hardwood	0.61
Closed cone pine cypress	0.50
Mixed chaparral	0.43

[1] Suitability varies from 0 to 1 with 1 being optimum.

The oak titmouse lays eggs in late March and April in northern California populations; one brood per year is typical (Cicero 2000). Clutch size is six to eight eggs per clutch and eggs are incubated for 14 to 16 days. Young fledge about 16 to 21 days after hatching (Terres 1991).

The oak titmouse is territorial and defends territories all year. Territory size for northern California populations has not been determined. In the San Francisco Bay area, territories were 1.7 to 2.6 ha (3.5 to 6.4 ac; Cicero 2000). The oak titmouse typically nests in natural tree holes or old woodpecker-excavated holes. An examination of data from museum collections of 114 natural nests in California revealed that 74 were from cavities and 40 were in woodpecker holes (Cicero 2000). At the San Joaquin Experimental Range nest cavities were found most frequently in blue oak, and live oak trees (CPIF 2003c). A total of 60% of nests occurred in excavated cavities of blue oaks. Natural cavity nests were evenly divided between blue oak (47%) and live oak (48%).

Nest characteristics were described by Cicero (2000). Data on 49 nests from Madera County, California had the following average characteristics: (1) nest height above ground of 4.6 m (15 ft); (2) cavity entrance diameter of 4.3 cm (1.7 in.); and (3) nest depth of 25.7 cm (10.1 in.). Nest material can include grass, moss, hair, and feathers.

The diet of the oak titmouse is comprised of both plant and animal matter (CDFG 2003g). Seeds of various oaks and thistles, willow catkins, and other species are important food items (Bent 1946, Martin et al. 1951). A variety of arthropods is consumed and includes leafhoppers, scales, aphids, ants, beetles, wasps, and spiders (Cicero 2000). Caterpillars and grasshoppers were the main food items in the diet of nestlings in a study of the oak titmouse in northern California (Kleintjes and Dahlsten 1992).

Foraging behavior varies depending on tree species and microhabitat characteristics. Oak titmice forage frequently in the subcanopy on the bark of trees. When foraging occurs in foliage, titmice forage more on the outer leaves of the tree rather than in the inner leaves or the upper foliage. The primary mode of capturing insects is by gleaning on bark and foliage. The oak titmouse uses the bill to excavate insects from bark and to hammer acorns or nuts against a branch to open them (Cicero 2000).

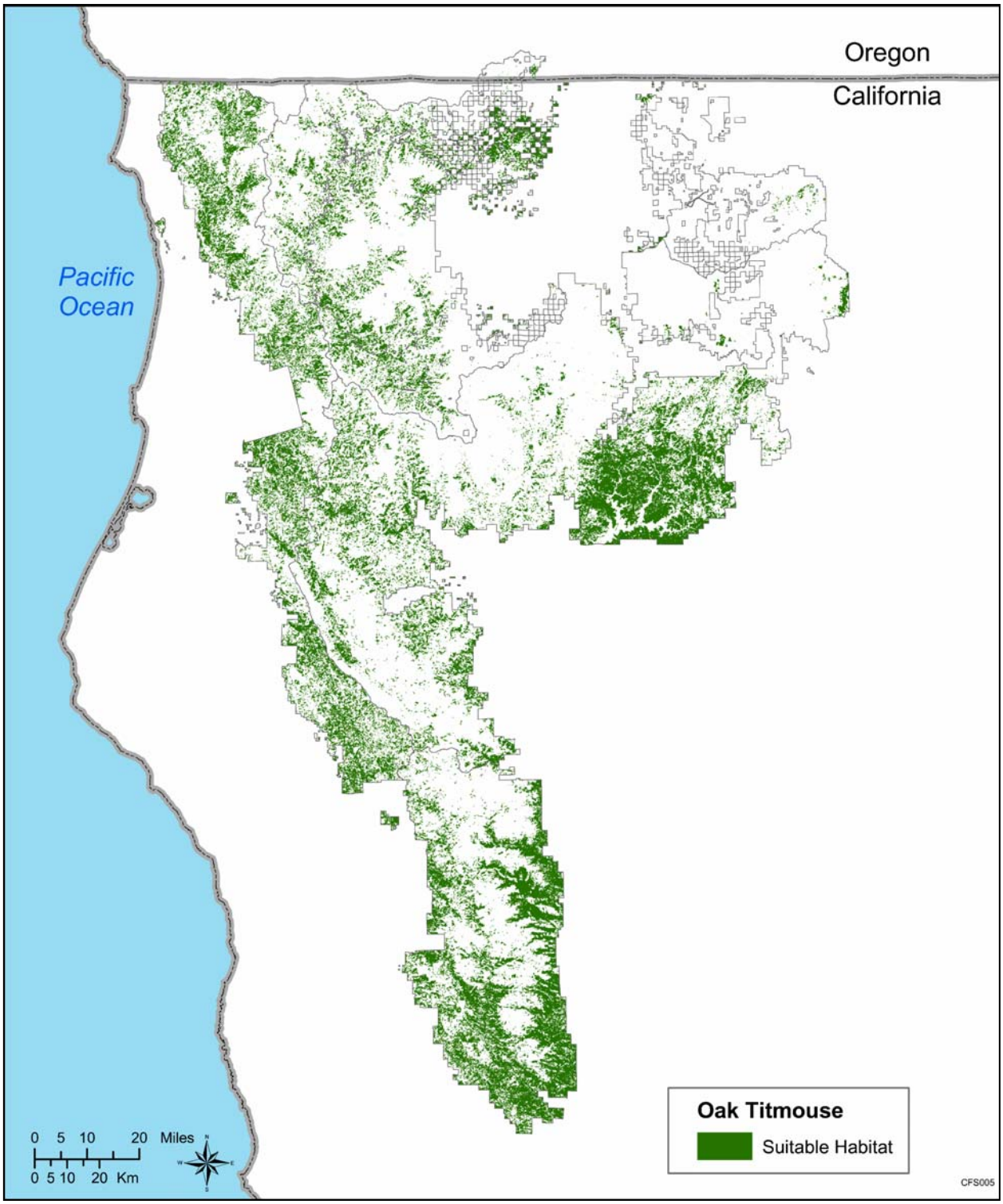


Figure 18 Oak Titmouse Habitats on the Four Forests of the Northern California Province

3.9.2 Status, Population Trends, and Factors Affecting Populations of the Oak Titmouse

The oak titmouse is not listed as threatened or endangered by either the State of California or the Federal government. It does not have special protection status in the Northern California Province.

Only one study was found that reported oak titmouse population density in northern California. Wilson (1992) reported mean densities of 48 per 40 ha (100 ac) during 1986 to 1987 in blue oak woodland habitat of Mendocino County. Cicero (2000) summarized density data (i.e., number of pairs per ha [2.5 ac]) from various surveys conducted elsewhere in California as follows: Alameda County, 0.3 to 0.5; Contra Costa County, 1.8; Madera County, 0.5 to 0.7; San Luis Obispo County, 1.2; San Mateo County, 1.1.

North American Breeding Bird Survey results for the northern California region indicate that oak titmice are encountered regularly, and in moderately high numbers (Sauer et al. 2003). The number of individuals observed per survey route per year ranged from less than one to as many as 20 or more. From 1966 to 1996, data suggest that the oak titmouse population has increased (greater than 1.5% per year) in the northern California region, but has decreased statewide. Population declines have been attributed to the loss of oak woodlands (Cicero 2000).

Audubon Christmas Bird Count data (Audubon 2003) for oak titmice in count circles within the Province are shown in Figure 19. Oak titmice were observed in relatively high numbers, and in all years of survey. The species was observed in only the Redding and Mt. Shasta count circles, however, with the highest numbers being recorded in the Redding count circle. No population trend is apparent when the data from 1980 to 2003 for all count circles in the Province are plotted (Figure 20).

Sources of mortality for the oak titmouse include predation and parasitism. Cicero (2000) assumed that predation is a major cause of mortality in the oak titmouse although no quantitative data are available to support her assertion. Nest failures have been attributed to predation by western scrub-jays, and gopher snakes. Nest parasites and interspecific competition for next cavities may cause mortality where cavities are limiting. Wilson (1992) reported that nestlings died because of drowning due to upward orientation of cavities in a study in Mendocino County, California. However, the excess of cavities at this site probably reduced the potential for nestling mortality caused by nest parasites or competition for nest cavities.

The single most important factor that impacts oak titmouse populations is removal of oak trees from woodlands (Cicero 2000). Oak woodland habitat is disappearing in California as a result of conversion to agriculture, use as rangeland, urbanization, and cutting for home heating. The availability of oak trees for nesting is critical to continued maintenance of oak titmouse populations at current levels. Especially important is protection of older, larger oak trees typically used for nesting, and maintaining forests with high plant diversity. The CPIF Bird Conservation Plan for the oak titmouse (CPIF 2003c) recommends the following actions be taken to protect and enhance habitat: (1) increase the number of dead standing oak trees, especially trees with dead limbs and heartwood decay, and (2) maintenance of a 40% to 70%

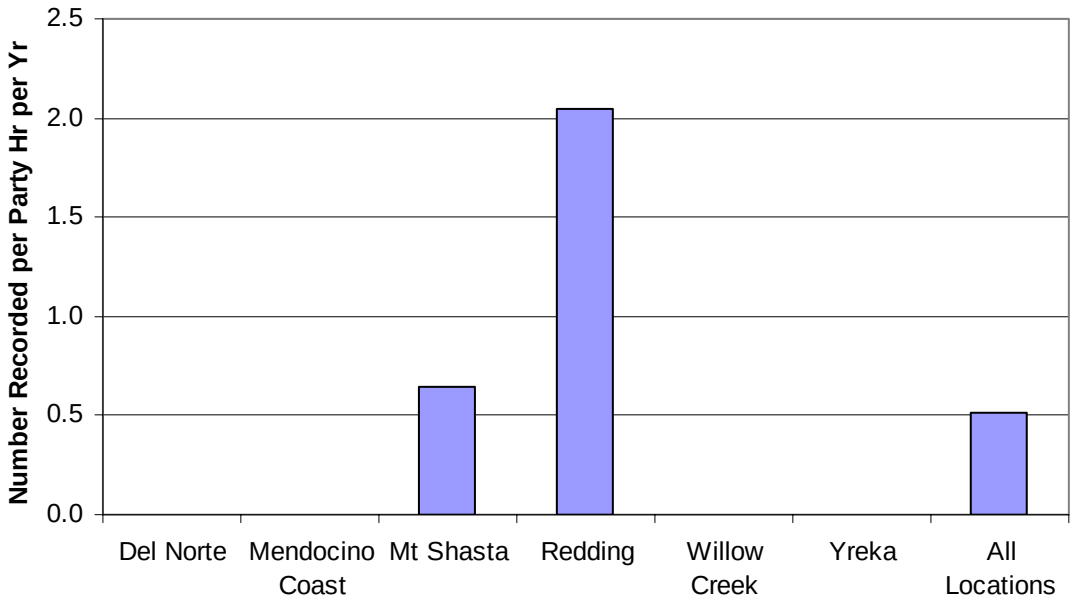


Figure 19 Mean Number of Oak Titmice Counted per Year per Party-Hr in Audubon Christmas Bird Counts on Count Circles within the Northern California Province (1980-2003)

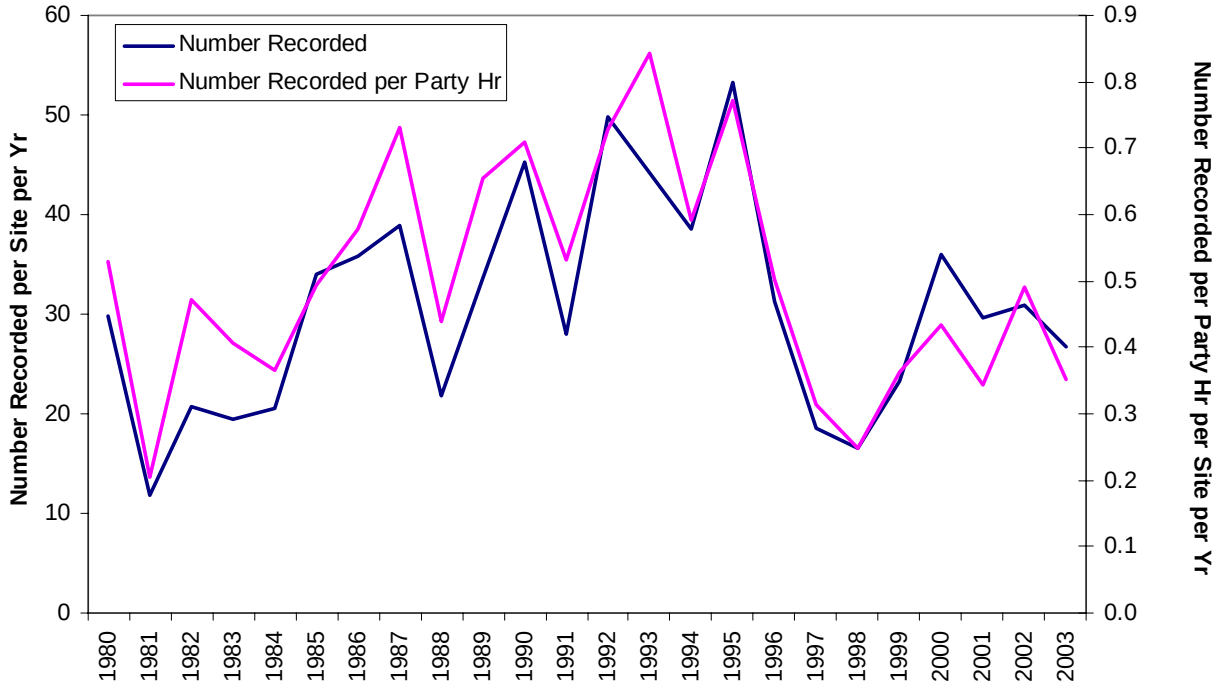


Figure 20 Mean Number of Oak Titmice Counted per Year in Audubon Christmas Bird Counts within the Northern California Province (1980-2003)

canopy cover when thinning oak woodlands. Cicero (2000) cites studies that emphasize oak titmouse habitat protection can be accomplished through maintenance of 50 to 100 ha (150 to 250 ac) management units and considering differences in ownership patterns across the landscape since more than 80% of oak woodlands in California are privately owned.

3.9.3 Suitability of the Oak Titmouse as a Management Indicator Species

The oak titmouse satisfies the criteria we established for a suitable management indicator species for the Forests of the Northern California Province (see Section 1). It occurs throughout the year in the Province and does not migrate. The species is relatively common in northern California. The range of the species includes at least some portions of all four Forests of the Province. The oak titmouse is easily recognized by its crest, gray-white breast, overall gray coloration, and conspicuous active behavioral patterns. A distinctive song and tendency to call while perched on the tops of trees should allow observers to easily detect individuals during monitoring.

The oak titmouse is strongly associated with oak woodland. On the basis of data collected during annual breeding bird surveys for the California foothills, declining numbers per route surveyed have occurred since 1987. Cicero (2000) suggests this decline could be the result of oak woodland habitat loss in California. Monitoring of oak titmouse abundance in oak woodlands of the Northern California Province through annual breeding bird surveys could be used to detect population trends.

A variety of survey techniques are routinely used for bird populations and could be applied to a monitoring program that used the oak titmouse as an indicator species. These are discussed in the following section.

3.9.4 Monitoring Protocols for the Oak Titmouse

Relatively few quantitative studies have been conducted of the oak titmouse that would enable a comparison of survey techniques. Wilson (1992) used spot mapping to census use of blue oak woodland by oak titmice at the University of California Hopland Field Station in Mendocino County. He evaluated the use of oak titmouse nesting success as an indicator of habitat quality. Past management activities and grazing had created an oak-savanna type habitat in the plots with oaks interspersed by grassland. Wilson censused 23, 5 ha (12.5 ac) plots in relatively open-canopy oak woodland habitat. Surveys of each plot were conducted once per week between mid-March and June by walking grid lines at a pace of 50 m (165 ft) per 6 min and recording birds observed on plot maps. Each census was conducted in the early morning and required about one hour to complete. All birds observed were recorded on plot maps and detection was assumed to be 100% for individuals within 50 m (164 ft) of either side of the survey route since the habitat was relatively open. A total of 10 censuses per plot per year were conducted.

British Columbia (MELP 1999b) reviewed inventory methods (encounter transects, point counts, and spot mapping) for forest songbirds including titmice. They suggested that point counts were preferable to encounter transects, because they were more easily standardized and they were a more efficient method for obtaining large sample sizes. Both of these methods were considered preferable to spot mapping. Huff et al. (2000) described a habitat-based point count protocol for terrestrial birds in Washington and Oregon specifically for use in monitoring population trends. These methods are discussed in Section 3.8.4.

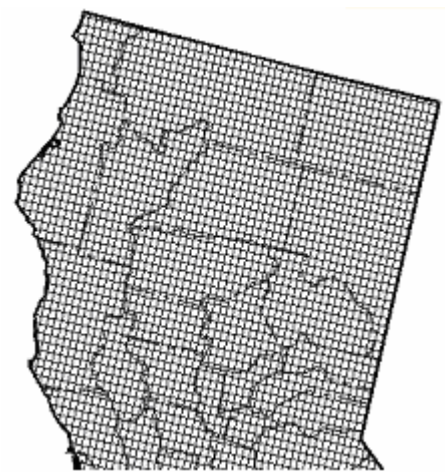
3.10 PALLID BAT

The pallid bat is a relatively large bat with a total length of 9.2 to 13.5 cm (3.6 to 5.3 in.) and a wing span of 36 to 39 cm (14 to 15 in.). Females are slightly larger than males, weigh 13.9 to 28.9 g (0.49 to 1.02 oz) and 13.6 to 24.1 g (0.48 to 0.85 oz), respectively. The pallid bat has slate gray wings and a pale abdomen. It is characterized by large tan ears. It is unique among bat species in that it mostly catches ground-dwelling prey by passive listening for their rustling sounds (5 to 25 kHz) and pouncing on them from the air. Echolocation is reserved for general orientation, using cues greater than 35 kHz (Arizona Game and Fish Department 2003; Cohen 2001; Fuzessery et al. 1993; Hermanson and O'Shea 1983).

3.10.1 Distribution, Habitat, and Life History of the Pallid Bat

The pallid bat ranges over most of western North America from southern British Columbia, east as far as Oklahoma and Kansas, and south to the Mexican states of Jalisco and Queretaro (Cohen 2001). It is broadly distributed throughout California, and occurs in a number of habitats from sea level to over 2,000 m (6,500 ft). The pallid bat occurs east of the Cascade Range in Washington and Oregon, in the southern Coast Range in Oregon, and throughout the spotted owl range in California. The pallid bat is a locally common species of low elevations in California. It occurs statewide except for the high Sierra Nevada from Shasta and Kern counties and the northwestern corner of the state from Del Norte and western Siskiyou counties to northern Mendocino County (Harris 2003). The pallid bat is found in all four Forests of the Northern California Province.

Range of the Pallid Bat in Northern California (CDFG 2002)



The pallid bat both roosts and forages in oak woodlands. The distribution of the pallid bat in the Trinity Mountains appears to be somewhat patchy, and perhaps linked to areas with substantial concentrations of oak (Pierson and Rainey 1998). However, it also roosts in large

ponderosa pine snags, redwoods, and giant sequoias (*Sequoiadendron giganteum*; Pierson and Rainey 1998). The number of known sites for the pallid bat in the Northwest Forest Plan area (USFS and BLM 1994) is low to moderate compared to other bat species. However, there are enough sites and populations that loss of a small number would not affect species persistence in the area (USFS and BLM. 2002).

Over their entire range, the pallid bat uses a variety of habitat types including coniferous and hardwood forests, brushy terrain, rocky canyons, grasslands, open farmland, and deserts where suitable roost sites exist (Arizona Game and Fish Department 2002). It is most common in open, dry habitats with rocky areas for roosting (Harris 2003).

The pallid bat occurs throughout California in a variety of habitats including low desert, oak woodland, and coastal redwood forests up to about 3,000 m (9,840 ft; BLM 1999). The pallid bat is generally not considered a forest bat, and throughout its entire range it is typically associated with xeric sites. In the area of the Northwest Forest Plan, it uses large snags in oak woodlands and xeric forest types (USFS and BLM 2002). The pallid bat is often associated with oak savannah habitat at lower elevations (Stillwater Sciences 2003). Pallid bats observed in a ponderosa pine forest of Arizona demonstrated a preference for roosting in oaks (New Mexico Game and Fish 2002). The pallid bat also uses caves, buildings, bridges, and other man-made structures.

CWHRS habitat types that are most suitable for the pallid bat include a variety of habitat types oak woodlands, coastal scrub, and chaparral. Conifer forests and other habitat types generally have lower suitability for this species. CalVeg community types that correspond to these habitat types are presented in CDFG (1998). The distribution of pallid bat habitats on the four Northern California Province Forests is mapped in Figure 21.

Pallid Bat Habitats in Forests of the Northern California Province (CDFG 2002)

Habitat Type	Suitability[1]
Blue oak woodland	0.76
Coastal oak woodland	0.76
Valley oak woodland	0.76
Barren	0.67
Blue oak foothill pine	0.66
Chamise-redshank chaparral	0.66
Coastal scrub	0.66
Mixed chaparral	0.66
Annual grass	0.33
Pasture	0.33
Klamath mixed conifer	0.33
Sierran mixed conifer	0.33
White fir	0.33
Bitterbrush	0.22
Alkali desert scrub	0.11
Agricultural crops	0.11
Douglas-fir	0.11
Eastside pine	0.11
Jeffrey pine	0.11
Juniper	0.11
Lodgepole pine	0.11
Low sagebrush	0.11
Montane chaparral	0.11
Montane hardwood conifer	0.11
Montane hardwood	0.11
Montane riparian	0.11
Pinyon juniper	0.11
Ponderosa pine	0.11
Redwood	0.11
Red fir	0.11
Wet meadow	0.11

[1] Suitability varies from 0 to 1 with 1 being optimum.

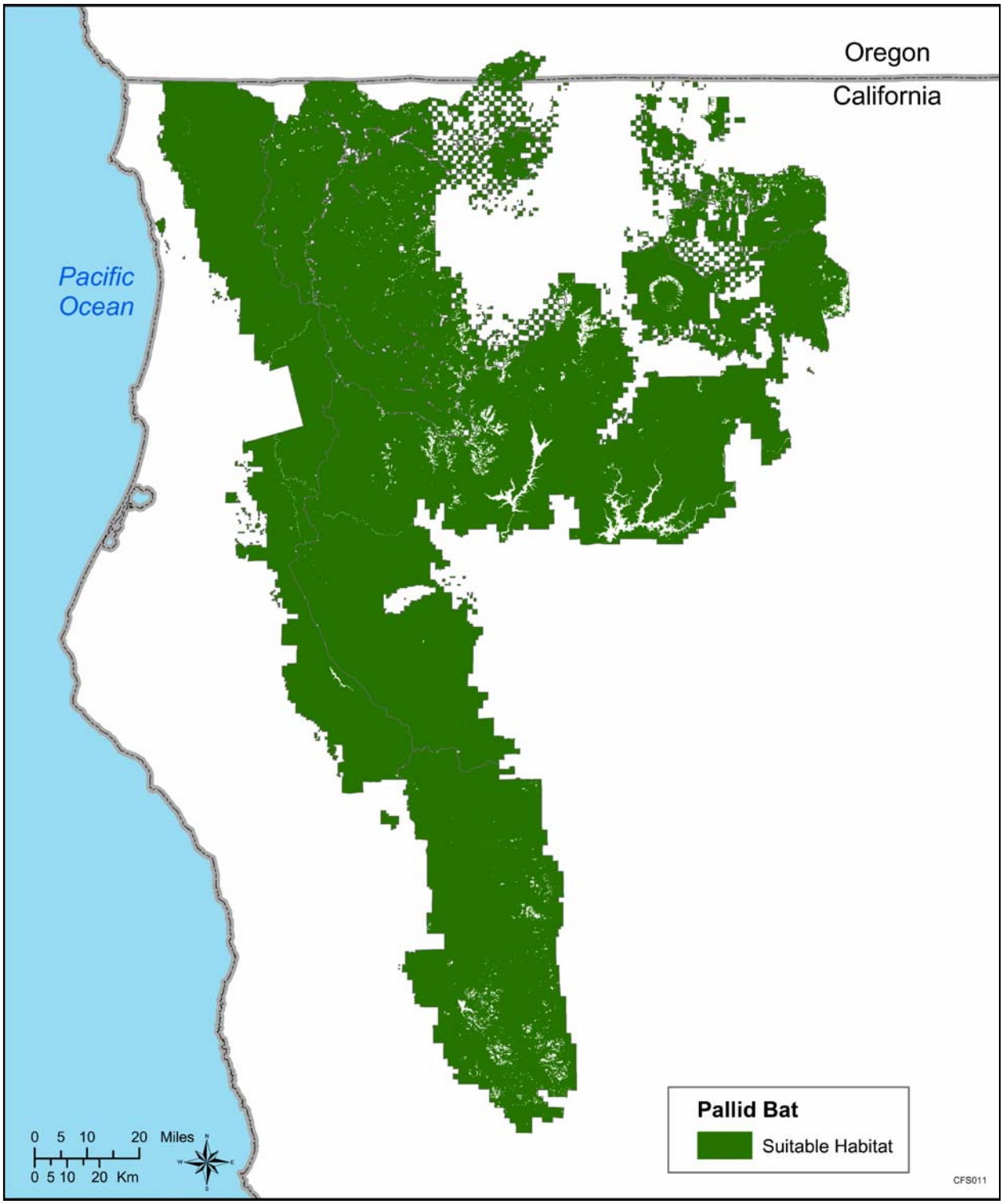


Figure 21 Pallid Bat Habitats on the Four Forests of the Northern California Province

The most important habitat requirement for the pallid bat is roosting sites. Stands with abundant roost sites and foraging opportunities are probably the highest quality habitat for bats (Humes et al. 1999). Roosts provide protection from predators, serve as sites for social interactions and the rearing of young, and provide a stable thermal environment and protection from the elements (Vonhof and Barklay 1996). The pallid bat requires day roosts (including nursery roosts, summer male roosts, and transient roosts), night roosts, and winter roosts. In general, day roosts provide more secure cover and stable conditions than night roosts. Night roosts provide areas for rest after feeding sessions and areas for social interactions (South Dakota Bat Working Group 2003). Nursery and winter roosts are particularly important to bat survival. The former provides protection from predators and microclimates required for pregnant or nursing females and developing young, while winter roosts offer a stable environment characterized by slight wind movement, humid conditions, and cool temperatures (South Dakota Bat Working Group 2003).

Day roosts include rock crevices in steep cliffs, stone piles, man-made structures, and trees (Cohen 2001). Roost sites also include riprap around culverts, mud cliffs, and talus slopes (Arizona Game and Fish Department 2002). Summer day roosts of the pallid bat are primarily snags of large-diameter trees in early to middle stages of decay. These snags are typically more abundant in older forests (USFS and BLM 2002). Roost snags were larger in diameter and more likely to have exfoliating bark than other snags. Roost snags were surrounded by forest with larger basal areas, higher tree densities and greater tree species diversity than other snags. Forests immediately surrounding roost snags have higher densities of snags and logs than other areas (New Mexico Game and Fish 2002). Cross et al. (1996) observed that the pallid bat preferred very large ponderosa pine snags for day roosts, but also used bridges. On snags, roost sites for bats can include the space behind loose bark, vacated woodpecker and natural cavities, and cracks (Bull et al. 1997). Woodpeckers typically use large-diameter snags (e.g., > 50 cm [20 in.] dbh) (Bull et al. 1997). Trees in California that can develop basal hollows include redwoods, incense cedars, Douglas-firs, and giant sequoias. Gellman and Zielinski (1996) concluded that forests lacking trees large enough to contain large basal hollows would provide fewer roosting opportunities that could affect the abundance and diversity of bats. Preferred summer roosting sites included horizontal rock crevices with stable 24-hr temperatures around 30°C (86°F). In cooler seasons, vertical rock crevices that were warmest towards the evening were selected (Cohen 2001).

Night roosts are located in close proximity to foraging areas (Cohen 2001), and include ponderosa pines, rock shelters, mines and caves, buildings, and the underside of bridges (BLM 1999; Cohen 2001; Cross et al. 1996; Lewis 1994). Night roosts provide protection from rain and wind, and are spacious enough to allow free flight into and out of the roost. A bridge is suitable for night roosts if it has vertical concrete roosting surfaces located between beam spaces and receives full sun for a majority of the day; bats prefer to night roost in a bridge's warmest locations (Keeley 1998). Bats appear to be faithful to particular night roosts both within and between years. However, those bats that roost together at night do not necessarily roost together during the day (Lewis 1994).

Pallid bats change roosts about every 1.4 days throughout the summer (Lewis 1996). They generally use a group of trees in relatively close proximity to one another (New Mexico

Game and Fish 2002). Bats may switch roost sites in response to disturbance, to interrupt parasite life cycles, or to take advantage of differing microclimate and structural conditions. The distance between roosts was usually less than 200 m (656 ft) and frequently less than 50 m (164 ft), while the distance between roosts and foraging areas was about 1 to 4 km (0.6 mi to 2.5 mi), so roost switching would not be effective at decreasing energetic costs of commuting to foraging areas. Also, the short distance involved in most roost switching would not be effective in deterring predators (Lewis 1996). Having several roost sites within a small area may minimize costs associated with roost switching (e.g., time and energy and disruption of the social structure of the colony; Vonhof and Barklay 1996).

Data are not available on the use of large standing old-growth trees by the pallid bat for foraging or hibernation. However, it is known that large snags and trees of old-growth forests are used for maternity sites and day roosts (USFS and BLM 2002). Maternity colonies have been documented from bole hollows of black and blue oaks (Stillwater Sciences 2003). Both pregnant and lactating pallid bats frequently changed their diurnal roost location, but lactating females tended to travel shorter distances between consecutive roosts. They were also more likely to continue to be associated with particular roost mates and less likely to roost alone (Lewis 1996).

Female pallid bats reach sexual maturity during their first year, whereas males reach maturity in their second year. Mating occurs from October to December. Ovulation and fertilization occurs in spring. Pallid bats generally give birth to two young in late May to mid-July. The young are capable of flight at four to five weeks of age, and attain adult size at eight weeks (Cohen 2001).

The pallid bat roosts in groups of 20 to over 200. The largest groups consist of summer maternity colonies composed mostly of females and young (Hermanson and O'Shea 1983). The pallid bat is known to roost with other bats, particularly *Myotis* species and the Brazilian free-tailed bat (*Tadarida brasiliensis*) (Harris 2003). However, in summer, bachelor and nursery colonies are separate (Miller 2000). Groups of up to 69 males have been observed together in diurnal roosts and over 100 in night roosts during summer months (Hermanson and O'Shea 1983). Colony size varies seasonally and maximum numbers coincide with peaks in mid-summer insect densities (Hermanson and O'Shea 1983). The pallid bat does not migrate. It is known to hibernate close to summer roosts. Winter groups generally number only one to four individuals, although as many as 100 have been observed (Hermanson and O'Shea 1983).

Pallid bats leave their day roosts to feed about 45 minutes after sunset. In addition to capturing prey from the ground, they will also take food from foliage of trees and shrubs. It is mostly insectivorous, and feeds on large (20 to 70 mm [0.8 to 2.8 in.] body length) arthropods such as beetles, crickets, and moths (Cohen 2001). The pallid bat forages along river channels within a very broad riparian zone (Stillwater Sciences 2003). The pallid bat tends to forage over open ground, usually 0.5 to 2.5 m (1.6 to 8.0 ft) above ground level (Harris 2003). They also hunt in riparian oak woodlands and, occasionally, over agricultural fields (Arizona Game and Fish Department 2002). After catching prey, pallid bats return to their night roost to eat their catch. Foraging peaks at the beginning and end of the nocturnal activity cycle (Arizona Game and Fish Department 2002).

3.10.2 Status, Population Trends, and Factors Affecting Populations of the Pallid Bat

The pallid bat exists in relatively low numbers in California and is therefore given special or protected status by a number of agencies and conservation groups. The CDFG (2003b) lists the following conservation status rankings for the pallid bat:

- *California Special Concern Species*—listed due to declining population levels, limited ranges, and/or continuing threats.
- *U.S. Forest Service*—listed as sensitive due to concerns for population viability.
- *BLM*—listed as sensitive due to small and widely dispersed populations or due to inhabiting ecological refugia or other specialized or unique habitats.
- *Western Bat Working Group*—listed as high priority due to being imperiled or at high risk of imperilment.
- *California Natural Diversity Database*—listed as S3 (restricted range and rare within California).

The pallid bat is not listed by either the State of California or FWS.

Studies to determine the densities or activities of pallid bats in the Northern California Province have not been conducted (USFS and BLM 2002). The nocturnal and secretive behaviors of bats combined with the inability to effectively study high-canopy forest habitats creates a challenge for researchers to develop reliable information about the numbers of bats in an area (Keeley 1998).

Bat populations can be affected by many factors. These include natural environmental factors such as temperature, predation, disease and parasitism; or human-influenced factors such as loss and degradation of habitat, loss of individual bats or colonies, inadequate enforcement of regulations or policies associated with protecting bat species and roost sites, insufficient interagency cooperation and funding sources for bat management, inadequate standardized monitoring methods for monitoring or surveying bats, insufficient data and knowledge on factors affecting bat populations and on the natural history of the bat species of concern (South Dakota Bat Working Group 2003).

Temperature in general, and roost microclimate (especially for maternity colonies) specifically, may be among the most important ecological factors for forest-roosting bats in the Northwest Forest Plan area (USFS and BLM 2002). Lower spring temperatures noted in 1991 in central Oregon were correlated with a higher percentage of nonreproductive female pallid bats, delayed parturition dates, reduced synchrony of parturition, and lower body mass of lactating females. Also, both females and juveniles were probably in poorer condition when entering hibernation (Lewis 1993). In Oregon, temperature appears to play a role in roost selection by pallid bats, and it is likely that social thermoregulation is a primary factor causing pallid bats to live in groups. Nevertheless, pregnant pallid bats were found roosting alone in over 25% of all observations (Lewis 1996). Bats may seek optimal temperatures by moving around under bark sheets during the day. Bats may choose large snags because they have thicker exfoliating bark than smaller snags, and the thicker bark provides more insulation (Rabe et al. 1998).

High ectoparasite loads were correlated with lower body weights of lactating females (Lewis 1996). Roost switching in pallid bats was positively correlated with ectoparasite loads. Roost switching may be a strategy to decrease ectoparasite loads by interrupting the reproductive cycles of those parasites that spend at least a portion of their life cycle on the walls of the roost (Lewis 1996).

The pallid bat's habit of capturing prey on the ground makes them susceptible to injury and predation. Major predators include snakes, owls, and domestic cats (*Felis silvestris*; Cohen 2001). Young bats tend to have higher mortality rates than adults. Fatalities occur from crashing into objects during first flights, being knocked out of the air by large gusts of wind, and being eaten by owls and other night predators. Bats also suffer high mortality rates during the first year of hibernation, possibly due to inadequate foraging success and low body weights when they enter their first winter cycle (South Dakota Bat Working Group 2003).

Northwestern California has been subjected to locally heavy timber harvest for about 100 yr. Little old-growth forest is left in coastal areas that are primarily in private ownership. Forests of varying ages, including late-successional reserves, remain on USFS land, particularly at mid to higher elevations (Pierson and Rainey 1998). Many of the areas most suitable for pallid bats (e.g., xeric forest types located in low-elevation areas) were logged historically and are susceptible to large wildfires that occur in xeric forests (USFS and BLM 2002). Exclusion of wildfire preserves some snags, but also removes an important mechanism of snag creation. Natural fire regimes remove smaller trees, promote growth of larger trees, and create snags by killing trees (Rabe et al. 1998).

Relatively large clearcuts can remove large portions of available bat roosting habitat. The remaining forested areas are often deficient in suitable roosting habitat because older forest stands are the ones most often targeted in forest-harvesting operations. If forest stands are intensely managed or are on a relatively short rotation cycle, the availability of large, older trees that are suitable for roosting will decrease (Vonhof and Barklay 1996). This could lead to a reduction in population size as a result of competition for available sites (Cross et al. 1996).

Common timber-harvest practices, including the emphasis on large-diameter cuts and selective removal of defective trees (including snags), favor low-diversity stands with truncated age and size distributions, and almost certainly reduce available roosting habitats for bats (Pierson and Rainey 1998; Thomas 1998). Snags can be lost through windthrow, wildfire, prescribed burn, felling for safety concerns, or fuel harvest. Also, once bark is lost or becomes too loose, the conditions under it may no longer be dry and warm (Rabe et al. 1998). Loss of mature oak woodlands and ponderosa pine forests, and elimination of large snags in southern Oregon and northern California, has reduced populations of pallid bats by eliminating roosting and foraging habitat (USFS and BLM 1994).

Other threats to the pallid bat throughout its range include destruction of structures, removal of bats from public buildings, urban expansion, loss or destruction of roosts, destruction of foraging habitat, closure of mines, renewed mining in historic districts, recreational use (e.g., rock climbing), and agriculture (including pesticide use) (BLM 1999; Cohen 2001). Since pallid bats often occur in buildings, human harassment and killing rank among the greatest

threats (Texas Parks & Wildlife 2003). Rock features may be especially important for pallid bats in areas where large snags have been eliminated from the landscape through timber harvest or fire. However, these areas are increasingly popular for recreational climbing and continue to be blasted for sources of road-surface material (USFS and BLM 2002). Any disturbance, even hiking, can cause the pallid bat to abandon a roosting area completely (Miller 2000). Pesticide accumulation in insect populations may affect all bat populations (USFS and BLM 1994).

3.10.3 Suitability of the Pallid Bat as a Management Indicator Species

The pallid bat satisfies most of the criteria we established for a suitable management indicator species for the Forests of the Northern California Province (see Section 1). It occurs throughout the year in the Province and does not migrate. The range of the species includes at least some portions of all four Forests of the Province. However, there is currently little information regarding the specific status, geographic distribution, reproductive ecology, and roosting and foraging habits for the pallid bat, particularly within the Northern California Province (USFS and BLM 1994). Systematic and statistically valid surveys have not been conducted for any bat species in the Northwest Forest Plan area (USFS and BLM 2002).

The pallid bat tends to have a spotty and unpredictable distribution within its range relative to other bat species, it has a low reproductive potential, and the number of known sites is low to moderate compared to other bats in the northern California. Furthermore, the pallid bat is considered “over-dispersed” due to its habit of roosting in small groups across the landscape and moving often among them. Methods are not available to provide numerical estimates of the numbers of individuals of such “over-dispersed” bats (USFS and BLM 2002). Specific studies to determine densities or activity levels of pallid bats in forest stands of differing ages have not been conducted (USFS and BLM 2002).

3.10.4 Monitoring Protocols for the Pallid Bat

Several documents describe protocols for conducting surveys of pallid bats. These include Kunz (1988) and the Bat Conservation Trust (2001). The Western Bat Working Group is in the process of developing a comprehensive survey manual that can serve as a standard bat-survey protocol (Western Bat Working Group 2003). Credible bat-survey techniques include ultrasonic-call identification, genetic testing of guano, and mist-net sampling (USFS and BLM 2002). The use of basal hollows could be indexed using the amount of guano deposited while bats roost (Gellman and Zielinski 1996).

Traditionally, surveying or monitoring of bats has been done by capturing with mist nets near water or localized foraging areas or by hand-netting, trapping, mist netting, or observing at roost sites. These techniques can yield data concerning the presence of bats, but inconsistent application often limits their use in long-term monitoring of population trends (Cross and Kerwin 1995). A successful long-term monitoring program requires reliable methods to estimate trends in relative abundance that balance disturbance to the species, survey effort and cost, and geographic coverage.

Obtaining precise counts of the pallid bat will be difficult because of its nocturnal habit, low numbers, and over-dispersion. As a consequence, any population or trend estimates for this species will be uncertain. Fortunately, if counts are consistently wrong for any reason, temporal changes can still be measured accurately using repeatable methods to achieve high precision (Bat Conservation Trust 2001). More certainty in population estimates can be obtained by marking captured bats. Punch-marking and banding can be used for short-term and long-term studies, respectively (Cross and Waldien 1994). The potential for encounters with rabid bats must be considered when conducting bat surveys. Analyses done in Colorado revealed one out of 21 pallid bats sampled had rabies. For all bats species tested, 15% of 4,470 bats tested were rabid (Pape et al. 1999).

Pallid bat colonies are easy to detect in man-made roost sites, but difficult in most natural roosts (e.g., trees and rock crevices) (Western Bat Working Group 2003). This makes it difficult to assess population estimates and trends (BLM 1999). Also, the remoteness and inaccessibility of areas utilized by pallid bats may allow only a general determination of location to be made (Cross and Waldien 1994).

Although all bat sampling methods have some advantages and disadvantages, mist netting remains the most effective technique overall for sampling bat diversity. However, while mist netting may provide information on selection of foraging areas, it does not provide information on the selection of roost sites. Mist netting does allow positive species identification and assessment of reproductive condition and age. Standardization for site and collection comparisons is based on number of bats captured per unit net area per hour (Pierson and Rainey 1998). However, as mist nets capture only a small percentage of the bats present in an area, and only those bats close to the ground or water over which nets are placed, this method is not the best indicator of overall activity level. Bat activity can be more readily determined by the use of acoustic surveys (e.g., use of Anabat or Pettersen D980 detectors) run in parallel with netting efforts (Pierson and Rainey 1998).

Bat detectors can be used to determine whether all species groups are being successfully sampled using mist nets, and for detecting the presence of species that are difficult to capture (Vonhof and Hobson 2001). Bat detectors can provide an index to relative activity of bats in different habitats or locations (Humes et al. 1999). It is preferable to sample all survey locations simultaneously to enable valid comparisons among sites that control for potentially confounding factors (e.g., weather conditions). The pallid bat has a readily identifiable echolocation call, however it does not call as frequently as some bats, so absence of pallid bat calls does not necessarily mean the species is absent from a site (USFS and BLM 2002). Even though the pallid bat has a fairly distinctive call, acoustic identification based on echolocation calls alone can be problematic. Bats can alter call slope, shape, and duration depending on habitat. Distinguishing the pallid bat from the big brown bat (*Eptesicus fuscus*) can be difficult without visual identification (Pierson and Rainey 1998).

No technique is available to measure the absolute abundance of bats, except in extremely localized areas such as single roosts (Thomas and LaVal 1988). The assessment of populations in hibernation sites has been the most consistently and widely employed technique for population monitoring (in Europe), and has been successful in highlighting declines and local extinctions

(Bat Conservation Trust 2001). Roost sites can be located by watching trees at dusk for emerging bats. Trees with good roost site potential can be observed for a set period of time prior to and following sunset.

Another method of locating roost trees is by use of radio transmitters attached to bats (Vonhof and Barklay 1996). Radio-tagged bats can be tracked to their roosts daily until the transmitter falls off or stops working. Roost locations can be recorded using a hand-held Global Positioning System (GPS) unit. Data collected with radio-telemetry can effectively delineate the landscape used by a population of bats (Adam et al. 1994). Once a roost site is identified, site and tree characteristics can be measured for both the roost trees and other available (nonroost) trees. These would include measurements of the diameter and decay classes of the trees. At some of the roost sites, it may be possible to census or monitor the population and to protect the site (Cross and Kerwin 1995).

Natural roost sites of bats in forests (e.g., basal hollows, other cavities and crevices, and foliage) should not be overlooked in favor of artificial sites (e.g., mines, bridges, and buildings) that may be easier to locate and study (Gellman and Zielinski 1996). However, studies at bridges (or other man-made sites) would enable the development of effective long-term monitoring and research programs. Bridges can be inspected during the day for roosting bats, guano accumulations, and urine staining. Those that have guano or staining, but no bats, could be assumed to be night roosts. Night-roosting bats can then be sampled with hand nets on telescoping poles (Pierson and Rainey 1998). Night-roost surveys of bridges was found to be the most effective, time efficient method for detecting certain bat species including the pallid bat (Pierson and Rainey 1998). While sampling bridges does not meet the goals of assessing forest-management practices on the pallid bat, it could be used as an initial means to determine areas where the pallid bat occurs. Subsequent surveys could then be made for natural roosting sites that occur in the area.

Comparing bat-capture success can be difficult or impossible due to variation in environmental conditions (e.g., temperature, humidity, wind, moon, etc.) and bat condition (e.g., reproductive) on different sampling occasions, which may influence capture success (Cross et al. 1996). Due to the fact that night-to-night variation in bat activity is sufficiently high, valid comparative data can be obtained only by surveying on a number of nights at the same sites or by establishing multiple survey points that are sampled concurrently (Stillwater Sciences 2003). Ambient conditions can have a significant effect on capture success. For example, higher capture success generally occurs during warmer, drier, and calmer conditions (Cross and Waldien 1995). Furthermore, the failure to find evidence for the presence of a species should be viewed with caution as it may reflect the rarity of a species or a sampling artifact, rather than the true absence of a species.

Bat population numbers can fluctuate naturally because of the effects of weather on reproduction and survival and potentially because of density-dependent feedback. Therefore, the ability to identify and distinguish between natural and anthropogenic population changes is essential. Thus, examining trends over a period of less than five years is inadvisable since short-term population fluctuations may be confounding (Bat Conservation Trust 2001).

The goal of a standard protocol for surveying bats in the Northern California Province would be to collect the same types of information in standard formats among the four Forests. A standard protocol would promote centralized tracking of bat studies to identify information gaps, facilitate monitoring of bat distributions and population trends using a centralized database, promote information-sharing, and enable spatial and temporal comparisons (Vonhof and Hobson 2001).

4 CONCLUSIONS

Our evaluation of ten candidate management indicator species included a review of their distribution, habitat requirements, life history, status, population characteristics, limiting factors, and the availability of monitoring programs for those species. Six criteria were considered in determining if each species would make a suitable management indicator species:

1. *Distribution*: the species occurs in all four Forests of the Northern California Province;
2. *Population size*: the species occurs in sufficient numbers to support a statistically robust monitoring program;
3. *Ability to be field identified and detected*: the species is easy to identify and detect;
4. *Habitat requirements*: populations of the species are known to track particular habitat types or characteristics;
5. *Affected by forest-management practices*: the species is sensitive to the effects of forest management practices; and
6. *Monitoring methodology*: methodologies exist or could be developed to effectively monitor populations of the species.

Table 5 summarizes the results of this evaluation for the 10 candidate species. Douglas-fir, ponderosa pine, black oak, northern spotted owl, acorn woodpecker, and oak titmouse satisfy all of our criteria for suitable management indicator species. The white-headed woodpecker, pileated woodpecker, and pallid bat meet most of the criteria, but have populations that may be too small to support a statistically robust monitoring program. Small populations make it difficult to develop a program that can effectively and economically detect the effects of forest-management practices. Smaller populations result in an increase in the variability among sample counts and can require a substantial increase in sampling intensity to ensure that enough data are gathered to overcome the effects of this variability. The mountain chickadee and pallid bat utilize a wide variety of habitat types in the Province, and thus their populations may be less sensitive to the effects of forest-management practices than other candidate species.

Protocols are available that can be used to monitor candidate management indicator species. The ongoing FIA program provides data that could be used to monitor population trends of Douglas-fir, ponderosa pine, and black oak. The FIA program has a number of advantages including (1) an existing database that can be used to describe baseline conditions, (2) wide coverage across the Province, (3) institutional acceptance, and (4) statistical validity. Existing protocols for the northern spotted owl have gained wide acceptance and could be used to monitor populations of this species. A variety of survey protocols are available for monitoring populations of the other candidate species. Each protocol has advantages and disadvantages, but point count and line transect techniques have a number of significant advantages. Collocating sampling locations for vertebrate species with FIA plots would facilitate testing and development of specific quantitative relationships between populations of management indicator species and habitat parameters using simple- or multiple-regression techniques. This will be an important next step in development of any monitoring program.

Table 5 Summary Evaluation of Candidate Management Indicator Species for Forests of the Northern California Province

	Evaluation Criteria					
Species	**Occurs in All Forests of the Province**	**Exists in Sufficient Numbers**	**Easy to Identify or Detect**	**Tracks Habitat Types or Characteristics**	**Affected by Forest-Management Practices**	**Monitoring Protocols Exist or Could be Developed**
Douglas-fir	Yes.	Abundant across Province.	Yes.	Occurs on moist, well-drained soils in extensive, stands along coast, and in mixed conifer forests farther inland. Fire adapted.	Management practices target production of species.	USFS FIA program is currently used to monitor species.
Ponderosa pine	Yes.	Abundant in eastern and southern portions of Province.	Yes.	Occurs in mesic mixed conifer forests or pure stands, particularly in inland areas. Fire adapted.	Management practices target production of species.	Same as above.
Black oak	Yes.	Abundant only in eastern portion of Shasta-Trinity and portions of Six Rivers and Mendocino Forests.	Yes.	Occurs on dry soils in mixed conifer forests, mixed hardwood forests, or as scattered pure stands. Fire adapted.	Traditional management practices favored replacement of black oak by conifers, but current practices target stand re-establishment.	Same as above.
Northern spotted owl	Yes.	Uncommon, but regularly detected throughout Province.	Yes, but nocturnal habits can make detection difficult.	Populations track availability of late-successional forests in association with younger forest patches.	Changes in the availability and distribution of late-successional forest could affect population size.	Nocturnal calling surveys; roost tree surveys; nest tree surveys using baiting, call playback in suspected nest areas, or observation of adults.

Table 5 (Cont.)

Species	Evaluation Criteria					
	Occurs in All Forests of the Province	**Exists in Sufficient Numbers**	**Easy to Identify or Detect**	**Tracks Habitat Types or Characteristics**	**Affected by Forest-Management Practices**	**Monitoring Protocols Exist or Could be Developed**
White-headed woodpecker	Yes.	Numbers may be too low to allow statistical determination of effects.	Yes.	No quantitative relationships between habitat and population established, but species uses variety of conifer forest types with snags.	Changes in the availability of conifer habitat and snags could affect population size.	Point counts and line transects for absolute density of all individuals. Spot-mapping for absolute density of breeding birds. Call-playback surveys for relative abundance.
Pileated woodpecker	Yes.	Numbers may be too low to allow statistical determination of effects.	Yes.	No quantitative relationships between habitat and population established, but species prefers late-successional conifer forests with abundant snags.	Populations are expected to respond to changes in the representation of late successional conifer habitat.	Same as above.
Acorn woodpecker	Yes.	Abundant across Province in habitats with oaks.	Yes.	Populations track the availability of oak woodlands and acorn production.	Changes in the availability and distribution of oak woodland could affect population size.	Same as above. Surveys of granary trees for relative abundance.
Mountain chickadee	Yes.	Common across Province.	Yes.	No quantitative relationships between habitat and population established, but species uses variety of conifer types.	Changes in the availability of conifer habitats could affect population size.	Point counts and line transects for absolute density of all birds. Spot-mapping for absolute density of breeding birds.

Table 5 (Cont.)

	Evaluation Criteria					
Species	**Occurs in All Forests of the Province**	**Exists in Sufficient Numbers**	**Easy to Identify or Detect**	**Tracks Habitat Types or Characteristics**	**Affected by Forest-Management Practices**	**Monitoring Protocols Exist or Could be Developed**
Oak titmouse	Yes.	Common across Province.	Yes.	No quantitative relationships between habitat and population established, but species is associated with oak woodland habitat.	Populations are expected to respond to changes in oak woodland availability.	Same as above.
Pallid bat	Yes.	Numbers may be too low to allow statistical determination of effects.	Yes, but requires capture or special equipment.	No quantitative relationships between populations and habitats have been established. Species uses a variety of habitats, and requires suitable roosting habitat including snags.	Given the wide variety of habitat types used, it may be difficult to attribute population changes to the effects of forest management.	Mist-net surveys to conclusively identify and collect information on individuals; acoustic surveys to survey more areas, habitats, and bats.

Given the information currently available, it is difficult to estimate the level of effort that would be needed to establish and conduct a monitoring program for management indicator species in the Province. There are inherent differences in the cost of different survey methodologies that are related to the time required to conduct the survey (e.g., point counts and line transects are less time consuming to conduct than spot mapping). There are also costs associated with the distribution of survey locations across the Province and the logistics of travel to survey locations. Cost will also depend on the number of samples (e.g., the number of point count plots or transects) collected. A well-designed monitoring program will have enough samples to detect a change in population size through time. To determine the sample size needed, an estimate of the variability among samples is needed as well as a determination of the desired "power" of the monitoring program (i.e., the percentage change in the population that the program is intended to detect). Variability among samples can be determined by pre-sampling, i.e., performing a pilot study using the methodology of choice with a relatively small number of plots or transects. Guidance on determining adequate sample size is provided in a number of sources including Sokal and Rolf (1981), Hayek and Buzas (1997), Thomas and Krebs (1997), MELP (1998b), and Huff et al. (2000).

Verner (1984) considered the sample size needed to detect a 10% difference in abundance in an indicator species between years in pine–oak woodlands of California. Previous studies had indicated that 5-min point counts were the most efficient survey procedure in that habitat type. On the basis of sample variability, he determined that 12,300 counts and 615 person-days per year would be needed for this program. When designing a monitoring program for the Province, similar determinations should be made for each management indicator species. Trade-offs will exist between feasibility, cost, and desired power. The 10% detection difference targeted by Verner (1984) in his calculations is perhaps a bit stringent and higher values could substantially reduce the number of samples needed. In Verner's example, targeting a 25% difference would require 1,968 counts while a 50% difference would require only 492 counts (Verner 1984). These lower count numbers would have proportionately lower costs.

Savings could be realized by developing an integrated sampling program that surveys all species using the same sampling locations and consistent survey techniques. As suggested above, there would be advantages to collocating vertebrate sample locations with FIA plots used to survey trees. Similarly, standardization of a single inventory approach (e.g., point counts) for use with all birds (except for the spotted owl) would save time and money because an individual sample location could be surveyed for all management indicator species at the same time. Using this approach, it would be important to estimate conservatively the sample size needed and base this on the species with the greatest variability.

Existing survey campaigns could be used to monitor the population status of management indicator species. Breeding bird surveys are conducted throughout the U.S. (Sauer et al. 2003) including the Province. This program provides a standardized, statistically valid sampling scheme for much of North America. To ensure inclusion of important portions of the Forests and to increase sample size, routes could be added and staffed by USFS personnel and volunteers. The advantage to such an approach would be the ability to coordinate data collection and analysis with the overall breeding bird survey program across North America. Any trends detected within Northern California Province Forests could be compared to trends seen

continent-wide. Similarly, the Audubon Christmas Bird Count approach could be applied to examine winter populations of management indicator species, although this program uses less standardized techniques. In addition to the national programs, CPIF coordinates survey activities among volunteers, and a number of survey areas exist in the northern California region.

A long-term monitoring program was recently developed by the Northern California North Coast Region Wildlife Programs Branch to assess forest habitat conditions and conduct wildlife inventories at sample points in the Southern Cascades ecological province (CDFG 2003h). The program examines a wide variety of plant and animal species using a 4.8 km (3 mi) grid across the Province. Grid points are randomly selected and surveyed on an annual basis (wildlife surveys) or every five years (plant surveys). Bird surveys conducted at each sample point employ electronic recording equipment to record singing males. Bats are surveyed at each location using mist nets. This program is in the pilot stage and may be subject to modification as preliminary results are evaluated.

The effectiveness of a monitoring program will increase as data are collected each year. Populations may respond relatively slowly to changes caused by forest-management practices, and a lag of a year or more between a change in conditions and population size can be expected. Populations naturally vary among years for a number of reasons and this natural variability can make it difficult to detect population trends or management effects (Thompson et al. 1998). Expected levels of natural variability should be considered when determining the desired power of the monitoring program. A well-designed monitoring program can detect the effects of forest-management activities, but confounding factors should be carefully evaluated before conclusions are drawn.

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