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FIRE AND BIRDS IN MARITIME PACIFIC NORTHWEST

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Abstract. Resource managers face the challenge of understanding how numerous factors, including fire and fire suppression, influence habitat composition and animal communities. We summarize information on fire effects on major vegetation types and bird/fire relations within the maritime Pacific Northwest, and pose management-related questions and research considerations. Information on how fire affects birds is limited for the maritime Pacific Northwest, even though fire is an essential process within natural vegetation communities throughout the region. We describe fire regimes, vegetation succession patterns, bird communities, and fire effects on birds for 12 major vegetation types in the region. Fire regimes and fire effects vary considerably within this region due to its diverse topography and climate. Seven of the types have a low- to moderate-severity fire regime and five have a high-severity fire regime with fire-return intervals that span several centuries. Bird communities and effects of fire are best known from the western hemlock type, which has a high-severity fire regime. The postfire stand-initiation stage in this type supports a reasonably distinct avifauna compared to other successional stages, a phenomenon that has been documented for high-severity fire regimes in other regions. In general, there is a high turnover of species after high-severity fires, with a shift primarily from canopy-dwelling to ground-, shrub-, and snag-dwelling species that mostly are not associated with other successional stages. No studies exist that directly address how bird communities are affected by habitat changes from fire suppression in this region. The most likely bird communities vulnerable to these changes are in low-severity, high-frequency fire regimes that include the Douglas-fir type, drier portions of the white fir type, Oregon-oak woodlands and savannas, native grasslands and sclerophyllous shrublands. In general, prescribed fire is not being used for bird conservation in this region. Where prescribed fire is being used to restore fire as an ecological process or more often for reducing potentially hazardous fuels, bird conservation objectives can be achieved as a secondary benefit. New land management polices that will greatly accelerate fuel reduction activities throughout the Pacific Northwest, including use of prescribed fire, are currently being undertaken with limited scientific information on the ecological consequences for bird communities.

Key Words: birds, fire, fire-suppression, forest management, Pacific Northwest, succession.

EL FUEGO Y LAS AVES EN LA PORCIÓN MARÍTIMA DEL NOROESTE

Resumen. Los manejadores de recursos naturales, enfrentan el reto de entender como numerosos efectos, incluyendo incendios y supresión de estos, influyen en la composición del habitat y sus comunidades de animales. Resumimos información de los efectos del fuego en la mayoría de los tipos de vegetación y las relaciones ave-incendios alrededor del las zonas marítimas de la costa del Noroeste pacífico, y postulamos preguntas y consideraciones de investigación relacionadas al manejo. La información de cómo el fuego afecta a las aves es limitada en la región marítima del Noroeste Pacífico, a pesar de que el fuego es un proceso esencial para las comunidades vegetales naturales de dicha región. Describimos regimenes de incendios, patrones de sucesión vegetal, comunidades de aves y los efectos del fuego en las aves, en 12 tipos principales de vegetación en dicha región. Los regimenes y los efectos del fuego varían considerablemente en esta región, debido a la diversidad topográfica y climática. Siete de los tipos tienen un régimen de severidad de incendios de bajo a moderado, y cinco tienen un régimen de severidad alto, con intervalos de repetición de incendios separados por varios siglos. Las comunidades de aves y sus efectos al fuego son mejor conocidos para el tipo de bosque occidental de abeto, el cual tiene un alto régimen de severidad de incendios. El estado de iniciación post incendio de este tipo, soporta a una avifauna razonablemente distinta comparado con otros estados sucesionales, fenómeno el cual ha sido documentado por regimenes severos de incendios altos en otras regiones. En general, existe una alta recuperación de las especies después de incendios con regimenes de alta severidad, con un cambio primordialmente de especies que viven en las copas de los árboles contra las del suelo, arbustos, y tocones, las cuales principalmente no están asociadas con otro estado sucesional. No existen estudios en esta región los cuales muestren directamente cómo las comunidades de aves son afectadas por cambios en el habitat, ocasionados por supresión del fuego. Las comunidades de aves más vulnerables a estos cambios se encuentran en el tipo de baja severidad, con regimenes de incendio de alta frecuencia, los cuales incluye el tipo *Pseudotsuga menziessi*, porciones más secas del tipo de Abies blanco, bosques de encino y sabana de Oregon, en pastizales nativos y en arbustos sclerophylus. En general, las quemas prescritas no son utilizadas para la conservación de aves en esta región. Las quemas prescritas son utilizadas para restaurar, como un proceso ecológico o

más a menudo, para reducir potencialmente combustibles riesgosos. Al restaurar con fuego, objetivos para la conservación de aves pueden ser alcanzados, como beneficio secundario. Nuevas políticas de manejo de la tierra en la región del Noroeste Pacífico acelerarán enormemente actividades de reducción de combustible, incluyendo el uso de quemas preescritas, las cuales son actualmente aplicadas con información científica limitada en cuanto a las consecuencias ecológicas para las comunidades de aves.

Disturbances can modify physical and biological environments and have profound effects on ecological processes, patterns, and interactions (e.g., White 1979). Primary disturbance agents include fires, wind, insect and disease outbreaks, floods, landslides, and human-related activities (Pickett and White 1985). In the Maritime Pacific Northwest, fires have been the most wide-ranging and continuous disturbance agent (Agee 1993), except for forest harvest, over the last few decades. Fire regimes and fire effects vary considerably within this region, primarily due to a diverse climate and topography that ranges from arid lands to rainforest and sea level to mountain peaks >4,300 m. Fire severities range from slight to cataclysmic, and natural fire-return intervals from almost annually to >1,000 yr (Agee 1993). In general, the north and coastal environments of western Washington and northwest Oregon have infrequent, stand-replacing fires that are part of a high-severity fire regime. In southwest Oregon and northwest California, and on the east side of the Cascade Mountains, fire occurs more frequently, a low- to moderate-severity fire regime, often with effects that are less severe, but more variable.

With effective fire suppression that began 80–100 yr ago, the natural patterns of fire have changed, especially in areas where fires burn most frequently (Agee 1993). In areas of low- to moderate-fire severity regimes, effective fire prevention may change habitat composition, shift the composition of biological communities, and lead to unnatural fuel accumulations associated with severe fires that cannot be withstood by historical ecosystems. When fire suppression increases fuel loads and the risk of severe fires, it becomes difficult to address the social and ecological concerns to protect property and lives, sensitive species and their habitat, and air and water quality, thus reinforcing widespread suppression of fires. However, consequences of continuing to suppress fires (i.e., passive management) without corrective measures also are high (Agee 2002). Shifts in fire prevention strategies are underway that propose to ameliorate potentially hazardous fuel conditions in areas of low- to moderate-fire severity regimes by steadily increasing prescribed fires (a fire ignited under known fuel conditions, weather, and topography to achieve specified objectives); thinning tree canopies to create shaded fuel breaks (Agee et al. 2000); and introducing other fuel reduction activities as part of a revised National Fire Plan (USDI et al. 2001). Such actions, however, often are planned and implemented with minimal understanding or considerations for effects on biota. Ecological objectives, if stated in planning documents, are almost always secondary to those for reducing hazardous fuels.

The purpose of this paper is to summarize information on bird-fire relations within the Maritime Pacific Northwest (hereafter Pacific Northwest), an area that encompasses the east slope of the Cascade Ranges, and the western portions of Washington, Oregon, and northern California (Fig. 1). This area is roughly equivalent to the range of the Northern Spotted Owl (*Strix occidentalis caurina*), the Northwest Forest Plan (USDA and USDI 1994b), and similar to the Southern Pacific Rainforest and Cascade Mountains physiographic stratifications used for the Breeding Bird Survey (Droege and Sauer 1989; map at http://www.mbr-pwrc.usgs.gov/bbs/physio.html [23 July 2004]). Our paper is organized into four parts: (1) a description of environmental conditions, vegetation communities, the role of fire in the major habitats and bird communities of the Pacific Northwest, and a summary of information on the effects of fire on birds; (2) a discussion of the major alterations to low- and moderate-severity fire regimes and their known or hypothesized effects on birds in this region; (3) a discussion of the role of prescribed fire in low- and moderate-severity fire regime and the implications for bird conservation; and (4) the critical management issues and research questions for this region.

ENVIRONMENTAL SETTING AND ROLE OF FIRE

Fire Regimes

Fire regimes have been described for the Pacific Northwest forests by Agee (1981, 1990, 1993, and 1998). Fires in high-severity fire regimes usually occurred >100 yr apart with >70% of the vegetative basal area removed; in moderate-severity fire regimes, fires were 25–100 yr apart with 20–70% basal area removed; and in low-severity fire regimes, they averaged 1–25 yr apart with <20% of the basal area removed. Grasslands and shrublands typically are part of a high severity regime, in which fires are

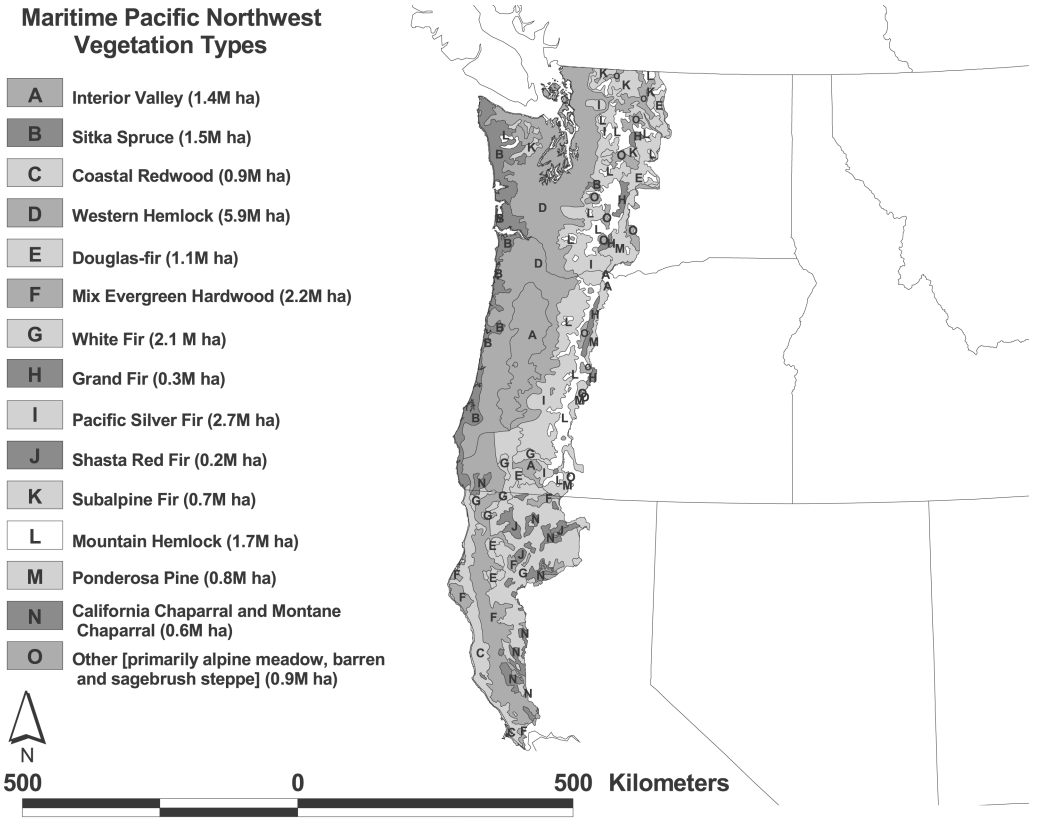

FIGURE 1. Generalized distributions of major vegetation types in the maritime Pacific Northwest (adapted from Kuchler 1964). Two types displayed on the map, ponderosa pine and chaparral (California and montane types combined), are not covered in this chapter because they have large geographic distributions primarily outside the maritime Pacific Northwest.

considerably more frequent than in most forested areas in this regime.

Vegetation patterns after fire depend on fire severity (Agee 1998). Agee found that for the low-, moderate-, and high-severity fire regimes, respectively, burn patch size created by fire tends to be small (~1 ha), medium (up to 300+ ha), and large (up to 10,000+ ha). He also found that the amount of patch edge, or contrasting conditions created by fire, tends to be low, high, and moderate, according to fire regime.

In high-severity fire regime areas of the Pacific Northwest, post fire stand-initiation can be prolonged for many decades as trees reestablish slowly after a fire (e.g., Hemstrom and Franklin 1982, Huff 1995). After burning or logging, this stage can be hastened through replanting trees and suppressing competing vegetation. Once trees establish in high-severity fire regime areas, the stem-exclusion and re-initiation of understory stages may continue for a century or two, leading to an old-growth stage that can persist for centuries depending on the natural fire rotation. Moderate-severity fire regimes sustain a highly variable forest structure by creating stands of uneven size and age trees and patchiness at landscape scales. Of the three fire regimes, low severity is most likely to create a balanced tree age-class distribution, where each fire only affects pattern and process on a small portion of a landscape (Agee 1998).

Ecological Units

Geographic distribution of vegetation and fire regimes in the Pacific Northwest are closely linked to complex topographic, moisture, and temperature gradients that can change rapidly with elevation, latitude, geological formations, substrate, and proximity to the Pacific Ocean (Agee 1993). In classifying the environments of Washington, Oregon, and California, Franklin and Dyrness (1973) and Barbour and Major

(1977) divided the states into broad ecological units using two approaches: (1) physiographic provinces based on geography, geology, and soils, and (2) vegetation zones (hereafter types) based on associations of natural plant communities.

In this paper, we use the physiographic area delineated in the Northwest Forest Plan (Forest Ecosystem Management Team 1993, USDA and USDI 1994a, 1994b, 2000) (Fig. 1) to describe distributions of broad vegetation types, fire regimes, and avifauna (Table 1). We aggregated vegetation types into five major ecosystems: lowlands and foothills, coastal forests, lower montane, upper montane, and subalpine. We describe the distribution of 12 vegetation types (Fig. 1), fire regimes, and establishment patterns after fire, using Franklin and Dyrness (1973, 1988) (vegetation of Washington and Oregon), Barbour and Major (1977) (vegetation of California), and Agee (1993) (fire regimes and effects) as our initial source. About 59% of the Pacific Northwest is covered by a high-severity fire regime and about 41% by a low- or moderate-severity fire regime based on vegetation types and fire regimes in Table 1 and Fig. 1. Below we describe the major vegetation types within the five ecosystems.

Lowlands and Foothills Ecosystem

This ecosystem occurs in a relatively dry environment within and near interior valley bottomlands. The diverse vegetation within the lowlands and foothills is aggregated into one vegetation type, interior valley. It is a mosaic of grasslands (e.g., Johannessen et al. 1971, Franklin and Dyrness 1973) and oak savannas, oak woodlands, mixed oak-conifer forests (e.g., Habeck 1961, Thilenius 1968, Smith 1985, Riegel et al. 1992); and sclerophyllous shrublands (i.e., chaparral) dispersed sporadically across southwestern Oregon and northern California (Whittaker 1954, Barbour and Major 1977). Generally, fire history has not been well documented in the lowlands and foothills because fires carried by grass and herbs are short duration, low intensity, and high-severity in which most vegetation is consumed and because much of the vegetation has been converted to other uses, such as urban and suburban, cropland, pasture, and forestry.

Oak woodlands and savannas are dominated by Oregon white oak (*Quercus garryana*) in Oregon; California black oak (*Quercus kellogii*) is an important species in the southern portion of the lowlands and foothills. At present, the ground flora has been so altered in these communities, especially through livestock grazing, that benefits of using fire for restoration are uncertain. Sprouting appears to be the primary process for recruitment. The role fire plays in perpetuating these communities by stimulating sprouts or influencing acorn germination and seedling survival is not well understood (Harrington and Kallas 2002). Sclerophyllous shrublands,

TABLE 1. FOREST VEGETATION TYPES OF THE MARITIME PACIFIC NORTHWEST BY FIVE ECOSYSTEM TYPES AND BY MOISTURE AND FIRE REGIMES.

	Forest vegetation types	
Ecosystem type	Wet to mesic environments/ high-severity fire regime	Dry to mesic environments/ low- to moderate-severity fire regime
Lowlands and foothills		Interior valley—a mosaic of oak woodlands and savannas; oak-conifer; grasslands[a], and sclerophyllous shrublands[a].
Coastal forests	Sitka spruce	Coast redwoods.
Lower montane	Western hemlock	Douglas-fir.
		Mix evergreen hardwood.
		White fir.
		Grand fir.
		(Ponderosa pine[b]).
		(California and montane chaparral[a,b]).
Upper montane	Pacific silver fir	Shasta red fir.
Subalpine	Subalpine fir	
	Mountain hemlock	

[a] Occurs in dry to mesic environments and has a high-severity fire regime.

[b] Ponderosa pine and California and montane chaparral types were not covered in this chapter because they have large geographic distributions primarily outside the maritime Pacific Northwest.

established near grasslands and oak communities in valley bottoms and serpentine soils, are maintained by fire, and dominated mostly by shrubs, such as *Ceanothus* spp., *Arcostaphyllus* spp, and *Baccharis* spp. High-severity fires are common, and burned areas are readily re-colonized by these shrub species when they resprout and their long-lived seeds quickly respond to scarification from burning. These shrublands appear to be declining in the Pacific Northwest because human development and fire suppression favor tree establishment (Chappell and Kagan 2000).

Coastal Forests Ecosystem

Two major vegetation types occur within the coastal forests, Sitka spruce (*Picea sitchensis*) and coast redwood (*Sequoia sempervirens*), hug the coastline and extend up major river valleys inland where summer fog lingers. Important tree species found in the Sitka spruce type are Sitka spruce western hemlock (*Tsuga heterophylla*) and western red cedar (*Thuja plicata*); grand fir (*Abies grandis*) and Douglas-fir (*Pseudotsuga menziesii*) are minor components (Fonda 1974, Henderson et al. 1989). Fire history has not been documented for this type (Agee 1993). The climatic conditions and the extent of older age classes indicate that it burned infrequently (likely even less often than the wettest areas of the western hemlock type), and belongs to a high-severity fire regime. Fire is more likely to spread into the coastal Sitka spruce type from nearby areas within the lower montane western hemlock type by the occasional dry east winds during rare climatic conditions, than from fires that originate from within the type (Agee 1993). Because fire occurs so infrequently, wind is probably a more important disturbance agent in this type.

Coast redwood, the tallest tree the world, is the principal species of the coast redwood type (Waring and Major 1964, Zinke 1977, Noss 1999). Other important trees within the type's broad moisture gradient include western hemlock, western red cedar, Sitka spruce, tan oak (*Lithocarpus densiflorus*), and Douglas-fir. The type's fire regime is moderate severity, with fires mostly occurring as low-to-moderate severity with infrequent high-severity events (Veirs 1982; Jacobs et al. 1985; Stuart 1987; Finney and Martin 1989, 1992). Fire-return intervals may exceed 500 yr in moist areas, and be as low as 20–50 yr at drier inland locations. Structurally diverse, multi-aged forests maintained by fires are characteristic of the type.

Lower Montane Ecosystem

This is the most diverse ecosystem type and covers the largest geographic area. We recognize five major vegetation types within the lower montane ecosystem: western hemlock; Douglas-fir; mixed evergreen hardwood; white fir; and grand fir. In the western hemlock type, which encompasses the largest area, western hemlock and Douglas-fir commonly co-occur, with the hemlock more prevalent at the wet end and the fir at the dry end of the moisture gradient (Franklin and Dyrness 1973, Zobel et al. 1976, Topik et al. 1987, Topik 1989, Henderson et al. 1989, Hemstrom and Logan 1986, Ruggiero et al. 1991, Atzet et al. 1996). Other important species are Sitka spruce in the coastal areas and Pacific silver fir (*Abies amabilis*) within a wide band of vegetation that transitions into the upper montane. Fire frequency is variable across the type, reflecting diverse moisture and temperatures conditions found within the type. In the areas of high moisture fires burn very infrequently, with 250 to >500-yr return intervals (Hemstrom and Franklin 1982, Agee and Flewelling 1983, Agee and Huff 1987, Henderson et al. 1989, Agee 1991a, Huff 1995, Impara 1997, Agee and Krusemark 2001). A high-severity fire regime predominates throughout much of the type. Fires often create large patches of killed trees with a few very widely scattered Douglas-firs that survived, as well as, an occasional small island of trees that was spared. Trees may reestablish slowly in these burned patches, and may remain a forest opening for many decades. In the drier southern portion of the type, average fire-return interval drops quickly from about 200 yr down to 100–30 yr as the fire regime grades into a moderate severity (Means 1982, Teensma 1987, Morrison and Swanson 1990, Cissel et al.1998, Weisberg 1998, Van Norman 1998, Olson 2000, Weisberg and Swanson 2003). In these more fire-prone areas, overlapping fires of varying severities create a complex age-class structure. Western hemlock (in wetter environments), Douglas-fir, and western red cedar are early seral species that attain great sizes, dominate for centuries, and, in late-successional conditions, show substantial structural diversity. In mesic and drier environments, Douglas-fir dominates after fire, sometimes in pure stands.

The Douglas-fir type usually occupies the warmest and driest environments in the lower montane (Franklin and Dyrness 1973; Atzet et al. 1992, 1996). It is often found upslope and adjacent to types dominated by ponderosa pine (*Pinus ponderosa*) or oaks. Douglas-fir is commonly an early and late-successional species throughout the Douglas-fir type,

with ponderosa pine distributed widely in dry and warm environments. Other locally important species are lodgepole pine (*Pinus contorta*), western larch (*Larix occidentalis*), Jeffrey pine (*Pinus jeffreyi*), and incense-cedar (*Libocedrus decurrens*). A low-severity fire regime is common throughout the type; characteristics of a moderate-severity regime emerge as moisture increases (Agee 1990, 1991b; Chang 1996; Skinner and Chang 1996; Taylor and Skinner 1997; Everett et al. 2000). Historically, fires probably burned often over relatively small areas and were confined mostly to the understory. Infrequent, larger fires that burned with varied severity eclipsed effects of these smaller fires. Fire history studies from across the region indicate that frequencies in dry locations averaged <10 yr but ranged up to 50 yr with increasing moisture. In this type, fire severity varied considerably by topographic position. Here, the most severe fires occur on upper slopes, ridge tops, and south- and west-facing slopes. These fires resulted in simple forest structures, whereas lower slope positions and riparian areas with less severe fires created complex forest structures, including the largest trees (e.g., Taylor and Skinner 1998).

The mixed evergreen hardwood type has the most restricted distribution in the lower montane ecosystem, but the most diverse tree composition (Whittaker 1960, 1961; Sawyer et al. 1977; Atzet et al. 1992, 1996). The type is most prominent in the coastal mountains and at mid slope and elevation, and moderate aspects. Important tree species are Douglas-fir, usually an overstory dominant, and tan oak. Tan oak has increased its density and cover throughout the type, presumably a result of fire suppression (Atzet et al. 1992). Other major species are Pacific madrone (*Arbutus menziesii*) and ponderosa pine.

The few fire-related studies in the mixed evergreen hardwood type show a wide range of fire severities, frequencies, and sizes (Thornburgh 1982, Wills and Stuart 1994, Agee 1991b), indicating that the type largely belongs in a moderate-severity fire regime. The mean fire-return intervals in pre-settlement times cluster around 15–35 yr, but range from 3 to >70, and the size of some fires have been quite large. Complex successional patterns have developed that parallel the variable nature of fire in this type. In general, after low- to moderate-severity fires, tanoak regenerates beneath a canopy of Douglas-fir. After more severe fires, Douglas-fir regenerates beneath a canopy of tanoak, a species that can sprout profusely after fire and have rapid early growth. Older forests usually have one to three age classes resulting from past fires, with Douglas-fir as an emergent canopy above various hardwoods, mostly evergreen.

The white fir type occurs in a range of environments wider than all other types in the Lower Montane except western hemlock (Rundel et al. 1977; Sawyer et al. 1977; McNeil and Zobel 1980; Atzet et al. 1992, 1996). White fir (*Abies concolor*), the most dominant tree species, can form pure stands in the coolest environments. The importance of Douglas-fir, relative to white fir, increases with increasing dryness and temperature. Ponderosa pine becomes a major species in drier environments. The white fir type typically exhibits a diverse and lush understory; yet, the fire regime likely fits a moderate or moderate-to-low severity. Documented mean fire-return intervals of ~10–65 yr and up to 160 yr (McNeil and Zobel 1980, Bork 1985, Agee 1991b, Stuart and Salazar 2000), broadly overlap other lower montane vegetation types in southwest Oregon and northwestern California, yet the type has not been studied extensively considering its breadth of environments. Fires are less frequent at higher elevations where white fir forms pure or nearly-pure stands. White fir has increased in cover and density with decades of fire suppression. In the white fir type, fires typically overlap to create a patchy mosaic of multiple structures and age classes, providing conditions for white fir to persist.

The grand fir type is found at mid-slopes in eastern Washington and Oregon on moist to dry sites (Hopkins 1979, Topik et al. 1988, Topik 1989, Lillybridge et al. 1995). In addition to grand fir, major tree species include ponderosa pine in warm and dry locations, lodgepole pine in cool and dry, Douglas-fir in environments broadly across the type, and western larch in the northern reaches of the type. Information on fire frequency and effects is scarce from this type (Agee 1993). Behavior of natural fires can be quite unpredictable, varying quickly from intense crown fires to surface fires. Higher severity fires may be an important part of forest development in the grand fir type, but to what degree is unclear. Preliminary indications are that a moderate-severity fire regime might be more likely than a low- or high-severity fire regime. Forest succession after fire should reflect fire-severity patterns and species present at the time of the fire: frequent low-to-moderate severities would favor, if present, Douglas-fir, ponderosa pine, and western larch survival and regeneration, while high severities would favor establishment of lodgepole pine in areas that are cool and dry. Grand fir can establish in open to partially shaded environments with moderate moisture, probably over a wide range of fire severities with low to moderate frequencies (Hall 1983).

Upper Montane Ecosystem

The vegetation types of the upper montane are found upslope from the lower montane, as conditions become cooler and wetter at higher elevations. There are two major upper montane vegetation types in the Pacific Northwest, Shasta red fir (*Abies magnifica*) and Pacific silver fir. The Shasta red fir type occupies a narrow ~500 m elevation band, typically above the white fir type and below the mountain hemlock type where Shasta red fir is common in the overstory and understory in a broad range of environments (Rundel et al. 1977; Sawyer et al. 1977; Atzet et al. 1992, 1996). Other important tree species include lodgepole pine and western white pine (*Pinus monticola*). Studies from Shasta red fir type and California red fir type (adjacent to the Pacific Northwest region), indicate that mean fire-return intervals are about 20–50 yr, with ranges of ~5–65 yr and fire-free periods spanning ~150 yr (McNeil and Zobel 1980, Pitcher 1987, Taylor and Halpern 1991, Agee 1993, Taylor 1993, Chappell and Agee 1996, Bekker and Taylor 2001, Taylor and Solem 2001). The type has a moderate-severity fire regime that creates a very patchy environment of diverse patch sizes and stand structures in close proximity that range from closed- and open-canopied late-successional forests to young, regenerating stands and open meadows.

In the Pacific silver fir type, Pacific silver fir is the dominant species; western and mountain hemlocks (*Tsuga mertensiana*) co-dominate at the lower and upper limits of the type, respectively (Fonda and Bliss 1969, Franklin and Dyrness 1973, Hemstrom et al. 1982, Packee et al. 1983, Brockway et al. 1983, Franklin et al. 1988, Henderson et al. 1989). Other important tree species are Douglas-fir and noble fir (*Abies procera*) in relatively dry and warm climates, Alaska-cedar (*Chamaecyparis nootkatensis*) in cooler and wetter sites, and western white pine in moderate environments. Subalpine fir (*Abies lasiocarpa*) is an important species in the eastern Cascades where the Pacific silver fir type transitions into a subalpine ecosystem. The type has a high-severity fire regime, in which fires burn infrequently (Agee 1993). Although fire history data are scant for this type, known fire-return intervals were about 100–200 yr along the drier eastern edge of the type that is influenced by a continental climate (Agee et al. 1990), and about 300–550 yr in the moister westside of the Cascades (Hemstrom and Franklin 1982). Pre-settlement fires in this type were usually associated with large fires that swept through adjacent types, creating large patches where most of the vegetation in the Pacific silver fir type was killed. This type, however, can act as a barrier to fire spread, unless extreme conditions associated with prolonged drought and dry east winds are met. Pacific silver fir, a fire-sensitive species, rarely survives where fires have burned and postfire conditions are often too harsh for it to establish, except in very cool and wet locations. Stand establishment after fire can last for decades, and be very prolonged if seed sources are absent for species that function as pioneers, such as Douglas-fir, western white pine, noble fir, subalpine fir, and lodgepole pine. Pacific silver fir is more prominent as forests mature.

Subalpine Ecosystem

Two major vegetation types occur in subalpine ecosystems: subalpine fir and mountain hemlock. Subalpine fir dominates throughout the subalpine fir type and forms nearly pure stands (Franklin and Mitchell 1967; Fonda and Bliss 1969; Henderson 1973, 1982; Agee and Kertis 1987; Franklin et al. 1988; Henderson et al. 1989; Lillybridge et al. 1995). Other species that typically co-occur with subalpine fir are Pacific silver fir in transition upslope into the type; whitebark pine (*Pinus albicaulis*) at higher elevations; Engelmann spruce (*Picea engelmanii*) as the type transitions to the east and lodgepole pine in harsh environments and locations with higher fire frequencies. A high-severity fire regime characterizes the subalpine fir type (Fahnestock 1977, Agee and Smith 1984, Agee et al. 1990, Taylor and Fonda 1990, Huff and Agee 1991, Agee 1993). Fires in this type typically burn infrequently, though more often than in the mountain hemlock type because precipitation and snow pack are lower and summer months are warmer. The fire-return interval ranges about 100–250 yr from relatively dry to wet environments. Tree reestablishment can be slow and inconsistent (except where lodgepole pine is prevalent) due to severe climatic and site conditions for establishment and insufficient distribution of seed, leaving the postfire environment in a park-like setting for decades.

In the mountain hemlock type, mountain hemlock tends to occur in mixed stands with Pacific silver fir as a co-dominant, and with subalpine fir, lodgepole pine, and Alaska-cedar (Fonda and Bliss 1969; Henderson 1973, 1982; Sawyer et al. 1977; Agee and Kertis 1987; Franklin et al. 1988; Henderson et al. 1989; Atzet et al. 1992, 1996). Fire history is poorly understood in the mountain hemlock type; however, it is likely that fires burn very infrequently, suggesting a high-severity fire regime (Agee 1993, Dickman

and Cook 1989). Fire-return interval for similar mountain hemlock forests in nearby Canada were found to be >1,000 yr (Lertzman and Krebs 1991). All major tree species in this type are easily killed by fire, although older mountain hemlocks can survive with fire scars where fires burned at lower intensities. Stand establishment after fire is slow, inhibited by harsh climatic conditions and seeds that are poorly dispersed (Agee and Smith 1984).

FIRE AND BIRD COMMUNITIES IN THE PACIFIC NORTHWEST

Computer simulations of Pacific Northwest forests have demonstrated that over the last 3,000 yr, historical fire regimes maintained highly variable forest age-class distributions. Wimberly et al. (2000) estimated that the proportion of older forest age classes varied from 25–75% at various times during this period. Such fire-related fluctuations in habitat availability have important implications for understanding the dynamics of bird populations across the Pacific Northwest. We took a two-step approach to describe interactions between fire and birds. First, we identified bird species that were common and indicative of each vegetation type. When literature was not available for a specific vegetation type, we used general sources that synthesize bird distribution and habitat relations in the Pacific Northwest (e.g., Johnson and O'Neil 2001). Second, we characterized the responses of birds to short- and long-term effects of fire using existing literature from the Pacific Northwest; however bird-related information was not available for many of the vegetation types.

Lowlands and Foothills Ecosystem

Bird communities of the interior valley vegetation type are difficult to characterize because vegetation prior to European settlement has been dramatically altered over the last 150 yr. Contemporary species' habitat relations in natural dry grasslands or prairies and oak savannahs are based on reports from agriculturally dominated landscapes, where only small, scattered remnants of original habitat remain. Using natural history information and historical accounts, Altman et al. (2000) developed a list of bird species historically associated with dry grassland and oak savannah in this region. They list six species as highly associated (abundance is likely to be significantly higher in this habitat during the breeding season than elsewhere in the Pacific Northwest) with dry grasslands and oak savannas, including the: Burrowing Owl (*Athene cunicularia*) (probably extirpated from the region), Western Meadowlark (Sturnella neglecta), Vesper (*Pooecetes gramineus*) and Grasshopper Sparrows (*Ammodramus savannarum*), Horned Lark (*Eremophila alpestris*), and Western Bluebird (*Sialia mexicana*). Most of these species are not widely distributed elsewhere within the Pacific Northwest. No information was available regarding fire effects on birds inhabiting native grassland and savannah habitats.

To document possible long-term changes in oak woodland bird communities where fire was excluded, Hagar and Stern (2001) re-sampled three of Anderson's (1970) oak woodland bird study sites about three decades later. Results of both studies were expressed in terms of density (birds 100 acres^{-1}). Hagar and Stern (2001) found 14 species that were rare or absent in 1967–1968 but more abundant in 1994–1996, and three species detected in 1967–1968 that were absent in 1994–1996. These authors hypothesized that fire suppression had allowed more coniferous trees and a thicker understory to develop, favoring birds that the authors characterized as forest-dwelling species (e.g., Swainson's Thrush [*Catharus ustulatus*], Purple Finch [*Carpodacus purpureus*], House Wren [*Troglodytes aedon*], and Pacific-slope Flycatcher [*Empidonax difficilis*]) over birds favoring partially open habitat (e.g., Chipping Sparrow [*Spizella passerina*], Bushtit [*Psaltriparus minimus*], and Yellow Warbler [*Dendroica petechia*]).

Bird populations closely linked to oaks could be in jeopardy if fire exclusion continues. Oak-dominated woodlands of the Pacific Northwest are maintained by fire (Agee 1993), which has been controlled tenaciously because oaks are in close proximity to rural communities. Because fire exclusion in this vegetation has caused major structural and compositional changes (e.g., a shift to dominance by conifers) (Tveten and Fonda 1999), thus potentially jeopardizing bird populations closely linked to oaks. Bird species most likely to be affected by declining oak habitat resulting from fire suppression are White-breasted Nuthatch (*Sitta carolinensis*), Western Scrub-Jay (*Aphelocoma californica*), and Acorn Woodpecker (*Melanerpes formicivorus*), and in southern Oregon and northern California the Ash-throated Flycatcher (*Myiarchus cinerascens*) and Oak Titmouse (*Baeolophus inornatus*; Altman et al. 2000).

Western Scrub-Jay, California Towhee (*Pipilo crissalis*), and Lesser Goldfinch (*Carduelis psaltria*) appear to be the most common species in fire-dependent sclerophyllous shrublands of the interior valley type, based on informal field observations (Altman et al. 2000). These shrublands support a unique assem-

blage of bird species that seldom breed elsewhere in the Pacific Northwest, including Wrentit (*Chamaea fasciata*), Green-tailed Towhee (*Pipilo chlorurus*), and Fox Sparrow (*Passerella iliaca*). The effects of fire on this vegetation type probably are similar to other high-severity fire regimes where most of the aboveground vegetation is killed, although the amount and distribution of mortality can vary considerably within and among fires. Studies from burned and unburned shrublands suggest that after fire, bird occupancy shifts to species that appear to favor open environments, and that shrub-dwelling species persist at moderately to substantially lower populations or leave and re-colonize later when the shrub cover is suitable (e.g., Moriarty et al. 1985).

Coastal Forests Ecosystem

The bird community of coastal forest types are similar, and include: Olive-sided Flycatcher in early seral with residual trees; Hutton's Vireo in shrub with small tree regeneration; Winter Wren (*Troglodytes troglodytes*), Wilson's Warbler (*Wilsonia pusilla*), Pacific-slope Flycatcher, and Hermit Warbler (*Dendroica occidentalis*) in young and mature forest; Varied Thrush (*Ixoreus naevius*)and Pileated Woodpecker (*Dryocopus pileatus*) in mature forest; Red Crossbill (*Loxia curvirostra*) in mature and old-growth forest; and Vaux's Swift (*Chaetura vauxi*) in old-growth forests with snags (Oregon-Washington Partners in Flight (PIF) http://community.gorge.net/natres/pif.html). Also, the federally listed Northern Spotted Owl breed in these types, as well as federal species of concern including Olive-sided Flycatcher (USDI Fish and Wildlife Service 2002).

Historically, fire played an important role in maintaining a mosaic of seral stages or habitat structures (e.g., snags) that have facilitated the persistence of these species. The Sitka spruce and wetter portions of the western hemlock types have high-severity fires and birds probably respond to fire similarly in these types (see section on *Lower Montane Ecosystem*). Fire suppression probably has had little effect on bird communities in the Sitka spruce type because the fire-return interval is so long. The direct effects of fire on coast redwood forest bird communities are not well documented. Where fires are infrequent in the redwoods, fire is probably less important than other disturbances, such as wind-throw. In areas with frequent low to moderate severity fires, such as drier coastal forest sites, fire played a role in maintaining forest structure by limiting the amount of downed woody debris, or reducing understory vegetation.

Lower Montane Ecosystem

Bird communities have been studied extensively in the western hemlock type (e.g., Morrison and Meslow 1983a and 1983b, Manuwal and Huff 1987, Carey et al. 1991, Gilbert and Allwine 1991, Huff and Raley 1991, Manuwal 1991, McGarigal and McComb 1995, Bettinger 1996, Chambers et al. 1999). The most widespread and abundant bird species of older forests (>200 yr after fire) are Chestnut-backed Chickadee (*Poecile rufescens*), Pacific-slope Flycatcher, Winter Wren, Hermit Warbler, and Golden-crowned Kinglet (*Regulus satrapa*). Other important species are Varied Thrush (wet environments), Wilson's Warbler (with increasing importance of deciduous trees and shrubs), Red-breasted Nuthatch (*Sitta canadensis*; drier environments), and Northern Spotted Owl—for which a large proportion of its habitat lies within the western hemlock type (Bart et al. 1992).

In the wet Olympic Mountains where fires are rare, breeding bird communities were examined at two sites covering a long stand initiation stage, 1–3 and 19 yr after fire, and compared to surveys from the other successional stages (Huff 1984, Huff et al. 1985). Winter Wren and Dark-eyed Junco (*Junco hyemalis*) were the most abundant species during the first three years after fire. At year 19 after fire, Winter Wren abundance was 3–4 times below the three most abundant species, Dark-eyed Junco, Rufous Hummingbird (*Selasphorus rufus*), and American Robin (*Turdus migratorius*). Year 19 had the highest species richness and diversity, including the highest amount of woodpecker species. In general, the stand initiation stage was most favorable for ground- and brush-foraging species, while unfavorable for canopy-feeding species. A high proportion of species bred only in the stand initiation stage after fire (30%), as observed in other regions (e.g., Taylor and Barmore 1980). Pacific-slope Flycatcher, abundant in older forests, was negatively affected by fire during the stand initiation stage; more so than any other species (Huff et al. 1985).

In a high severity fire regime, wildfires and timber harvest followed by site preparation with fire have a few similar effects on bird habitat: loss of live tree overstory followed by re-colonization of herbs, shrubs, and trees. After logging followed by broadcast burning in the western hemlock type in Oregon, the most common birds in open managed stands (<15 yr-old and planted with conifer regeneration) were mostly ground- and brush-foraging species that included White-crowned Sparrow (*Zonotrichia leucophrys*) and Song Sparrow (*Melospiza melodia*), Swainson's

Thrush, Rufous Hummingbird, Spotted Towhee (*Pipilo maculates)*, American Goldfinch (*Carduelis tristis*), Dark-eyed Junco, and MacGillivray's (*Oporornis tolmiei*) and Orange-crowned Warbler (*Vermivora celata*; Morrison and Meslow 1983b, Bettinger 1996). In contrast to the stand initiation stage after a fire (e.g., Wyoming, Taylor and Barmore 1980; Washington, Huff 1984), primary and secondary cavity nesting birds were nearly absent in logged areas because past practices in this region typically removed most or all standing dead trees. Hermit Warbler, Chestnut-backed Chickadee, Golden-crowned Kinglet, Swainson's Thrush and Dark-eyed Junco that are common in young-tree thickets of rapidly maturing managed forests ~15–35 yr old (Bettinger 1996), are probably associated with similar conditions after wildfires but take longer to develop. About 80 yr after wildfires, bird species composition in the western hemlock type tends to stabilize, (i.e., Huff and Raley 1991), that is, generally not changing until the next wildfire, which could be centuries. During this extended period between fires, relative abundance among species appears to be regionally distinct and varies as forests grow older (Huff and Raley 1991). Bark foragers, such as Brown Creeper (*Certhia americana*), Hairy Woodpecker (*Picoides villosus*), Pileated Woodpecker, and Red-breasted Nuthatch tend to increase overtime as forests develop old-growth characteristics.

Relative to other lower montane forest types, the bird community of mixed evergreen hardwood type had more insectivorous species, foliage gleaning species, and snag nesting species (Alexander 1999). The mixed evergreen hardwood type also provides habitat for the federally listed Northern Spotted Owl and federal species of concern such as Olive-sided Flycatcher (USDI Fish and Wildlife Service 2002).

No studies have directly measured the response of mixed evergreen hardwood type bird communities to fire. Like the coastal forests, bird communities in this type vary among forest age classes (Raphael et al. 1988, Ralph et al. 1991). However, unlike the coastal forests, fire was probably far more widespread than any other disturbance. As a result, most authors agree that the historical fire regime has created and maintained high spatial and biological heterogeneity of vegetation in this type (Wills and Stuart 1994, Agee 1991b, 1993). Therefore, fire likely played an important role in maintaining the richness and diversity of these bird communities. Specifically, fires may influence communities by maintaining a heterogeneous seral composition, creating snags for foraging and nesting, and increasing availability of limiting food resources. The relative importance of these mechanisms is likely to be highly variable among species.

Upper Montane and Subalpine Ecosystems

The bird community of upper montane Shasta red Fir type is less species rich than those in Lower Montane Ecosystem types (Alexander 1999). The federally listed Northern Spotted Owl use this type, as do species of concern such as Olive-sided Flycatcher (USDI Fish and Wildlife Service 2002).

In the Shasta red fir type, the relatively mixed effects of fire may be important for understanding the composition of the bird community. Red-breasted Nuthatch and Golden-crowned Kinglet, both strongly associated with this vegetation type, decrease in response to fire (Kreisel and Stein 1999) and management practices that reduce canopy cover (Chambers et al. 1999). However, Alexander (1999) also documented a high proportion (relative to lower montane conifer forests) of canopy seed-eating, ground-foraging, and ground-nesting species in this type. The persistence of both canopy dependent and shrub-dependent species may be facilitated by landscape-scale patchiness created by fire.

Bird communities of the Pacific silver fir and mountain hemlock types of the Pacific Northwest are likely comparable to those in the same vegetation types in nearby areas of southern British Columbia, Canada, studied by Waterhouse et al. (2002). The most common species found at >900 m were Red Crossbill and Pine Siskin (*Carduelis pinus*); other relatively common species were Dark-eyed Junco, Winter Wren, Golden-crowned Kinglet, Chestnut-backed Chickadee, Townsend's Warbler(*Dendroica townsendi*), and Varied Thrush, which also were important species at lower elevations. Birds characteristic of the subalpine fir type in Washington include Pine Siskin, Mountain Chickadee (*Poecile gambeli*), Yellow-rumped Warbler (*Dendroica coronata*), and Clark's Nutcracker (*Nucifraga columbiana*; Manuwal et al. 1987).

Bird response to fire in the high-severity regime of the mountain hemlock, Pacific silver, and subalpine fir types probably parallels high-severity regimes of the western hemlock type. After fire, forest reestablishment is slower in these types than the western hemlock type due to short growing seasons, favoring ground- and brush-dwelling species over canopy feeder for up to a century or more. In subalpine environments, species that use edges and forest openings, such as Olive-sided Flycatcher, may benefit from fires (Altman and Sallabanks 2000, but see Meehan and George 2003).

ALTERATIONS TO FIRE REGIMES

Southwest Oregon and northwest California have a long history of anthropogenic influence on fire regimes (Frost and Sweeney 2000). This influence began with burning by American Indians (Boyd 1999), continued with fires set by Euro-American settlers, and then shifted to policies of fire suppression during the 20th century. However, establishing how and to what extent these activities have changed the natural fire regime is difficult. Such alterations in fire regimes can influence the local and landscape patterns of vegetation structure and community composition that determine habitat availability and quality for many forest birds.

The use of fire by American Indians in low- to moderate-severity fire regimes of the Pacific Northwest is accepted (Frost and Sweeney 2000), but the extent of these fires is not well known. Most burning by Indians probably occurred in oak woodland and pine forests (LaLande and Pullen 1999), possibly leading to a pattern where ignition of fires by American Indians was greatest at low elevations and decreased at higher elevations; there is little evidence that coniferous forests at higher elevations were significantly affected by aboriginal burning (Frost and Sweeney 2000).

Although burning by American Indians declined after the initiation of European settlement in the early 1800s (Frost and Sweeney 2000), a number of anecdotal accounts suggest that Euro-American settlement lead to large and severe fires throughout many areas of the West (Biswell 1989). Although more frequent fires may have been the case in some areas, quantitative evidence has not shown this effect to be ubiquitous. In Douglas-fir dominated forests of the California Klamath Mountains, Taylor and Skinner (1998) found no difference between pre-settlement (1626–1849) and post-settlement (1850–1904) fire-return intervals. Similarly, at Kinney Creek in the eastern Oregon Klamath Mountains, Agee (1991b) documented a pre-settlement (1760–1860) fire frequency of 16 yr that was not substantially different from a nearby post-settlement (1850–1920) estimate of 12 yr.

As in most of the West, fire suppression became the policy towards forest fires in the Pacific Northwest in the early 1900s (Atzet and Wheeler 1982, Biswell 1989, Agee 1993). However, most evidence suggests that this policy did not become highly effective until after World War II, when fire fighting became more mechanized (Pyne 1982). Additionally, increases in cattle and sheep grazing in the California and Oregon Klamath Moutains (Atzet and Wheeler 1982) may have facilitated effective fire suppression if it reduced fuels that carried fires (Biswell 1989).

Given the inherent variability within and among the different forest types, drawing generalizations about the effects of fire suppression on low- to moderate-severity fire regimes is difficult. At Oregon Caves National Monument in the Oregon Klamath Mountains, Agee (1991b) found that the fire-free period between 1920 and 1989 was the longest in over 300 yr. Perhaps more convincingly, in mixed conifer forests of the California Klamath Mountains, fire-return intervals increased from 12.5 yr during 1850–1904 (during the pre-suppression period) to 21.8 yr from 1905–1992 (during suppression) (Taylor and Skinner 1998), and fire rotation (the time required for the entire study area to burn) increased 10 fold from 20–238 yr (Taylor and Skinner 2003). Although increases in fire severity are often ascribed to policies of suppression (Biswell 1989), the evidence to support this hypothesis is limited and requires more information. Because fire suppression has reduced fire frequency, fuel levels are likely to be greater than they were historically. In areas where fire-return intervals were short, such build-ups may result in fires that are more severe and larger than they would have been historically (Agee 1993). However, in an analysis of fires in the Klamath Mountains between 1909 and 1997, Frost and Sweeney (2000) determined that although high severity fires were more common than they were prior to 1950, there was no conclusive evidence that this trend was outside of the historical range. More data are needed before we understand how fire suppression has affected fire severity in low- and moderate-severity fire regimes.

Effects of 20th century fire suppression may be responsible for changes in forest structure and landscape composition at several spatial scales in the Pacific Northwest (Kauffmann 1990). Generally, such effects can be more pronounced in forest types where historical fire-return intervals were shorter, low- and moderate-severity fire regimes, than where they were longer (Agee 1993). Fire suppression can alter the habitat potential for species associated with the composition and structure maintained by recurring fires by, for example, creating dense canopy layers that can substantially reduce herbaceous ground cover (e.g., Thilenius 1968, Hall 1977) and altering important roost characteristics used by birds of prey (e.g., Dellasala et al. 1998). At larger spatial scales, the effect of fire suppression on landscape composition may be more pronounced (Agee 1993). It has been hypothesized that in the mixed-conifer forests types with low- and (to a lesser extent) moderate-severity

fire regimes in the Pacific Northwest, fire suppression has decreased stand heterogeneity and promoted a forest landscape that is more even-aged than was historically present (Agee 1993). In the Klamath Mountains and California Cascade Range, this hypothesis is supported by a comparison of forest openings measured in 1945 and again in 1985 (Skinner 1995). He found that during this time period, the median distance from random points to the nearest opening doubled, suggesting that the spatial diversity of forest structure has declined since 1945. Such changes are likely to reduce variation in fire severity, an important source of structural diversity among stands in forested landscapes (Taylor and Skinner 2003), and may decrease biological richness.

In forest types that have experienced fire-return intervals >40 yr, structural and biological changes may be relatively minor because the period of fire suppression has been shorter than the typical fire-free intervals (Chappell and Agee 1996), though this is difficult to validate. Because large areas of Sitka spruce, western hemlock, subalpine fir, and mountain hemlock forest types have fire-return intervals that greatly exceed fire-free intervals fire suppression effects to date are probably minimal. In the Shasta red fir type, Chappell and Agee (1996) found little evidence that fuel loads or vegetation structure of stands were outside of the range of natural variation at the stand scale. They suggested, however, that at larger spatial scales fire suppression has allowed more stands to develop late-successional characteristics, and thus creating more structural homogeneity across the landscape.

Effects of Fire Suppression on Birds

No studies have directly addressed effects of fire suppression on bird abundance in the Pacific Northwest. However, we do know that many bird species vary in abundance across successional vegetation gradients (Marcot 1984, Raphael et al. 1988, Huff and Raley 1991, Ralph et al. 1991). Thus, if fire suppression has changed forest composition, regional patterns of bird abundance may have also changed (e.g., Raphael et al. 1988). If subtle demographic mechanisms are important in the birds and operate even in the absence of successional changes, then the effects of fire suppression may be more complex and difficult to quantify.

Fire suppression may be one factor contributing to changes in bird community composition in the interior valley type. In the absence of fire, coniferous trees and non-native shrubs encroach upon these habitats (Tveten and Fonda 1999). Such changes have been implicated in changes in bird community composition between the late 1960s and early 1990s in oak woodlands (Hagar and Stern 2001). Fire has also been suppressed in sclerophyllous shrublands enabling trees to establish (Altman et al. 2000). Although the addition of trees to sclerophyllous shrublands could increase bird species diversity, converting the shrublands to forest is likely detrimental to shrubland bird populations.

Identifying general mechanisms by which fire suppression may influence changes in bird abundance may be difficult in coniferous forests. Information from nearby regions suggests that changes in forest structure caused by fire have important consequences for bird abundance. Studies in the Sierra Nevada and northeastern Washington reported canopy-dwelling and foraging species, including Golden-crowned Kinglet, Red-breasted Nuthatch, and Brown Creeper are consistently less abundant in burned areas (Bock and Lynch 1970, Kreisel and Stein 1999). These results are supported by similar responses of these species to commercial thinning (Chambers et al. 1999) or stand age (Marcot 1984, Raphael et al. 1988) that were studied in our region. To the extent that fire suppression has allowed denser forest canopies to develop over broad areas within the low-severity fire regime, species that use dense canopy characteristics should increase, while species strongly associated with fire-maintained vegetation composition and structure (e.g., canopy openings or diverse herb and shrub layers) should decrease. Olive-sided Flycatchers, for example, which typically increase in response to forest openings and open understory conditions created by fire (Hutto 1995; Altman, unpublished data) or commercial thinning (Chambers et al. 1999), declined broadly throughout the western North America from 1966 to 1996 (Altman, unpubl. data). These declines may be associated with habitat loss from fire suppression, but such an effect is difficult to verify over broad areas. Furthermore, changes in abundance may not be the best measure of habitat quality. In a study of Olive-sided Flycatchers in northern California, Meehan and George (2003) showed that although burned sites were more likely to be occupied than unburned sites, nest success was greater on the unburned sites.

ROLE OF PRESCRIBED FIRE AND FIRE MANAGEMENT

Prescribed Burning for Habitat Restoration and Maintenance

Prescribed fire is increasingly recognized as a tool for restoring and maintaining forest health and

reducing fuels (Biswell 1989), yet the risks of escape and smoke production have limited its application. Estimates of how much area to should be treated with fuel reduction activities were made during revisions to the Northwest Forest Plan (USDA and USDI 2000). The estimated area burned historically in this region (~190,000 ha yr^{-1}) was used as a starting point, and this was reduced by the average area currently burned by wildland fire and what is feasible to treat considering budget constraints to arrive at a goal of treating 76,000 ha annually.

In areas that burn frequently, fire may play an important role in maintaining species composition of some unique vegetation communities. Fire has been reintroduced to grasslands within the Pacific Northwest, for example, to maintain endangered plant species (Pendergrass et. al. 1999) and to restore prairie-like conditions (Clark and Wilson 2001). Such applications of fire are considered experimental because so little is known about the natural fire regime and the desired results have not necessarily been achieved. Using prescribed fire to mimic natural fire regimes in these habitats has the potential to restore and maintain vegetation communities at watershed scales (e.g., Taylor and Skinner 2003) and associated bird communities.

Bird Conservation Strategies Using Prescribed Fire

Bird conservation plans developed by Oregon-Washington Partners in Flight (2000) recommend prescribed burning in oak woodlands and savannas of the interior valley type in western Oregon and Washington. Burning in oak woodlands is being used as a conservation strategy for birds by reducing encroachment of Douglas-fir, stimulating oak seedling recruitment, and creating multi-aged stands. Prescribed fire is also being used to restore oak woodlands in other regions of the United States (e.g., Abrams 1992). Although prescribed fire may restore forest plant communities, the effects on bird communities are not well documented. Potential declines may occur as a result of changes in vegetation structure or in response to fires that take place during the breeding season (e.g., Artman et al. 2001). Although the timing of prescribed fire relative to the breeding season should be considered, these relatively short-term disturbances should not outweigh the long-term conservation benefits of restoring oak woodlands (Oregon-Washington Partners in Flight 2000).

Although the effects of prescribed burning on bird abundance in the Pacific Northwest have not been documented, observations from the Sierra Nevada (e.g., Kilgore 1971) have been incorporated into the Oregon-Washington PIF conservation strategy for coniferous forests on the east slope of the Cascade Range (Oregon-Washington Partners in Flight 2000). This strategy recommends use of prescribed fire to reduce fuel loads and accelerate development of late-seral conditions. Such conditions are hypothesized to enhance habitat for Olive-sided Flycatchers as well as Cassin's Finch (*Carpodacus cassinii*), Western Wood-Pewee (*Contopus sordidulus*), Mountain Bluebird (*Sialia currucoides*), Northern Flicker (*Colaptes auratus*), American Kestrel (*Falco sparverius*), and American Robin (Oregon-Washington Partners in Flight 2000). Similarly, the California Partners in Flight conservation strategy for coniferous forests has identified the restoration of fire cycles as potential conservation measures for Black-backed Woodpeckers (*Picoides arcticus*), Fox Sparrows, and Olive-sided Flycatcher (California Partners in Flight 2002a).

Awareness is growing that prescribed fire and other forest management activities need to be considered within the context of natural wildfire on the landscape. The ability of birds to recruit to a site after fire depends on management activities that occur before and after fires. In particular, snag-nesting species, which typically increase following fire, are affected by postfire snag availability (e.g., Haggard and Gaines 2001). Snag-nesting bird communities were compared among three salvage-logged treatments (high, moderate, and low amounts of snags), 4–5 yr after a high-severity fire in the Douglas-fir type of the east-central Washington Cascades (Haggard and Gaines 2001). Distribution patterns of snags (e.g, clustering) and snag size affected cavity-nesting species response. Intermediate densities (15–35 snags ha^{-1}) of snags >25 cm diameter at breast height (dbh) were associated with the highest abundance, species richness, and nesting densities of cavity nesters.

Another example of the interaction between fire and forest management is the potential effect of fire and fire management on habitat for endangered species. This is illustrated by the habitat requirements of the Northern Spotted Owl in areas of moderate-severity fire regimes throughout the Pacific Northwest. Historically, much of the habitat in this fire regime was spatially and temporally dynamic because of fires. More recently, decades of fire suppression and selective harvest of large, fire-resistant trees have created multi-canopied forests with thick understories of fire-intolerant species, which

possibly has accelerated a shift towards larger and more severe fires (Agee and Edmonds 1992). As a result, the amount of suitable Northern Spotted Owl habitat (USDI 1992, Agee 1993) may have increased in some forest types, particularly the grand fir type. Because these areas provide high quality owl habitat, they have been protected as late successional reserves designed to sustain owl populations. Consequently, high-severity fires are expected to burn a greater portion of the landscape than they did historically, increasing the probability that Northern Spotted Owl habitat will be altered for longer periods and at larger spatial scales. Some level of low-severity fire that maintains high canopy cover may have modest, or even beneficial effects for Northern Spotted Owls (Bond et al. 2002). The owl is more likely to be present and persist in areas of mixed successional stages and forest structures in northern California where fire is likely a major contributing factor to the mosaic of conditions that are favorable (Thome et al. 1999, Zabel et al. 2003). Severely-burned areas, however, are avoided and may reduce owl productivity (Bevis et al. 1997, Gaines et al. 1997, cf. Bond et al. 2002). If the extent of severely burn area increases substantially, the amount of Northern Spotted Owl habitat in reserves and managed areas may not be sufficient to sustain owl populations.

MANAGEMENT QUESTIONS AND FUTURE RESEARCH CONSIDERATIONS

Managing fire and fuels to meet the ecological, economic, and public safety concerns of society is challenging. From an ecological perspective, the ability to apply effective management is mostly limited by lack of relevant information to make informed decisions. In the Pacific Northwest, where fire has undoubtedly had a profound effect on bird communities, few studies have addressed how fire affects birds (Table 2), and basic information about bird species composition and relative abundance patterns was lacking for most vegetation types in this region. To effectively manage fire and fuels in this region while considering short- and long-term bird conservation will require substantial investments to accurately portray potentially affected bird communities across vegetation types, monitor response to varied management actions, and develop models to predict related bird response. Monitoring within the context of adaptive management (Holling 1978, Walters and Holling 1990) will be critical to assure that ecological goals of fire management are met (Tiedemann et al. 2000). In some cases, monitoring programs may be designed in such a way to contribute information to important research questions, and vice-versa. With this potential in mind, we have outlined ten questions that, if answered, will, at least, provide critical information for the application of fire management toward effective bird conservation.

What Were Characteristics of Natural Fire Regimes in the Pacific Northwest?

Clearly one of the most crucial steps to understanding the relationships between fire and bird abundance is to understand their interactions at a large spatial scale. This requires understanding, not just of the central tendency of fire regimes, but also their variability (Gill and McCarthy 1998). Given the complex nature of fire regimes in the Pacific Northwest, more data on the frequency, severity, and spatial distribution of fires is needed, especially in regions with low- to moderate-severity fire regimes.

How Do Bird Populations Change in Response to Fire?

Even relatively basic information on the response of birds to fire is still lacking. Few studies have distinguished between changes in behavior (e.g., habitat use or nest-site selection) versus larger-scale changes in population density. Studies that measure behavioral responses to fire (e.g., foraging activity [Kreisel and Stein 1999]), and nesting density and demographics (e.g., Saab and Dudley 1998, Saab et al. 2002) could provide these data. Using geographic information system (GIS) data to link these patterns with landscape level patterns may provide useful insights (Dettmers and Bart 1999, Saab et al. 2002). If fire affects important demographic patterns (e.g., Saab and Vierling 2001), then we should consider how source/sink dynamics might be influenced at large spatial scales.

When Bird Populations Change in Response to Fire, What Are the Driving Factors?

With a better understanding of how bird abundance changes in response to fire, it will be important to determine causal explanations. A number of mechanisms have been hypothesized to influence postfire changes in bird communities, including food availability (Apfelbaum and Haney 1981), nest-site availability (Hutto 1995), predator abundance (Altman and Sallabanks 2000, Saab and Vierling 2001), and vegetation structure (Bock and Lynch 1970). An evaluation of the relative importance of these factors

TABLE 2. SUMMARY OF BIRD RESPONSES TO FIRE BY SPECIES.

Species	State	Year after fire	Size (ha) and No. of fires[a]	No. of replicate sites	Response[b]	Reference[c]	Comments[c]
Blue Grouse (*Dendragapus obscurus*)	WA	4	4000 (2)	13 burned, 9 unburned	0	1	1
Spotted Owl (*Strix occidentalis*)	CA, AZ, NM	1	>540 (1)	11 territories	0	2	2, Postfire estimates not outside range of unburned estimates.
Rufous Hummingbird (*Selasphorus rufus*)	WA	1–515	no data (7)	7 plots	m	3	3, Most abundant 19 yr postfire.
Downy Woodpecker (*Picoides pubescens*)	WA	4	4000 (2)	13 burned, 9 unburned	0	1	1
Hairy Woodpecker (*Picoides villosus*)	WA	4	4000 (2)	13 burned, 9 unburned	+	1	1
American Three-toed Woodpecker (*Picoides dorsalis*)	WA	4	4000 (2)	13 burned, 9 unburned	+	1	1
Black-backed Woodpecker (*Picoides arcticus*)	WA	4	4000 (2)	13 burned, 9 unburned	+	1	1
Pileated Woodpecker (*Dryocopus pileatus*)	WA	4	4000 (2)	13 burned, 9 unburned stations	0	1	1
Pacific-slope Flycatcher (*Empidonax difficilis*)	WA	1–515	np (7)	7 plots	–	3	3
Gray Jay (*Perisoreus canadensis*)	WA	4	4000 (2)	13 burned, 9 unburned	0	1	1
Steller's Jay (*Cyanocitta stelleri*)	WA	4	4000 (2)	13 burned, 9 unburned	0	1	1
Clark's Nutcracker (*Nucifraga columbiana*)	WA	4	4000 (2)	13 burned, 9 unburned	0	1	1
Black-billed Magpie (*Pica hudsonia*)	WA	4	4000 (2)	13 burned, 9 unburned	0	1	1
Common Raven (*Corvus corax*)	WA	4	4000 (2)	13 burned, 9 unburned	0	1	1
Black-capped Chickadee (*Poecile atricapilla*)	WA	4	4000 (2)	13 burned, 9 unburned	0	1	1
Mountain Chickadee (*Poecile gambeli*)	WA	4	4000 (2)	13 burned, 9 unburned	0	1	1
Chestnut-backed Chickadee (*Poecile rufescens*)	WA	1–515	np (7)	7 plots	–	3	3
Red-breasted Nuthatch (*Sitta canadensis*)	WA	4	4000 (2)	13 burned, 9 unburned	–	1	1
White-breasted Nuthatch (*Sitta carolinensis*)	WA	4	4000 (2)	13 burned, 9 unburned	0	1	1
Brown Creeper (*Certhia Americana*)	WA	4	4000 (2)	13 burned, 9 unburned	0	1	1
Brown Creeper	WA	1–515	np (7)	7 plots	m	3	3
Winter Wren (*Troglodytes troglodytes*)	WA	4	4000 (2)	13 burned, 9 unburned	0	1	1
Winter Wren	WA	1–515	np (7)	7 plots	+	3	3
Golden-crowned Kinglet (*Regulus satrapa*)	WA	4	4000 (2)	13 burned, 9 unburned	–	1	1
Golden-crowned Kinglet	WA	1–515	np (7)	7 plots	–	3	3
American Robin (*Turdus migratorius*)	WA	1–515	np (7)	7 plots	m	3	Most abundant 3–19 yr postfire.
Varied Thrush (*Ixoreus naevius*)	WA	4	4000 (2)	13 burned, 9 unburned	0	1	1
Townsend's Warbler (*Dendroica townsendi*)	WA	1–515	np (7)	7 plots	m	3	3
Dark-eyed Junco (*Junco hyemalis*)	WA	1–515	np (7)	7 plots	+	3	3
Red Crossbill (*Loxia curvirostra*)	WA	4	4000 (2)	13 burned, 9 unburned	0	1	1

[a] Only wildland fires are reported in this table.

[b] + = increase; – = decrease; 0 = no effect or study inconclusive; m = mixed response.

[c] References: 1 = Kreisel and Stein 1999, compared 13 stations in two 80-ha stand replacement burns with nine unburned stations; 2 = Bond et al. 2002, compared survival, site fidelity, and reproductive success for owls 1 yr postfire to unburned estimates; 3 = Huff et al. 1985, chronosequence comparing seven sites ranging from 1–515 yr postfire, positive fire response if birds more abundant in years immediately after fire.

for different nesting and foraging guilds is needed. An effective evaluation of these hypotheses could include experimental manipulations or large sample sizes in many treatments. Understanding the mechanisms responsible for these changes will provide a better understanding of the unique ecological effects of fire-mediated habitat change.

How Do Fire Regimes Influence the Bird Community Structure at Local and Landscape Levels?

With data on the response of bird populations to fire, researchers and managers will be in a better position to understand how fire regimes structure bird communities. To date, most studies of bird communities and fires have been performed at the scale of single forest stands (Bock and Lynch 1970, Kreisel and Stein 2000, but see Saab et al. 2002). In order to understand the relationship of fires and birds, it is important not only to measure the effect of fire at burned and unburned points, but to quantify the effect of fire, and different spatial patterns of fire, at larger spatial scales, such as watersheds or regions. Fire likely influences abundance and distribution of birds throughout entire regions, not simply within the immediate area of disturbance.

How Do Changes to Fire-return Intervals Affect Bird Populations?

It is clear that fire suppression has lengthened fire-return intervals in the low- and to some degree moderate-severity fire regimes, and may have altered patterns of fire severity. The ecological effects of these changes are not well understood (Chappell and Agee 1996, Frost and Sweeney 2000), but changes in composition and spatial distribution of large-scale habitat characteristics are probably important (Skinner 1995). Effects of reducing fire frequencies on bird abundance and demographics should be evaluated if a policy of suppression continues, and at least outside of wilderness areas it probably will. Such evaluations should be conducted at multiple spatial scales by expanding analyses beyond comparisons of burned and unburned points to consider how spatial distribution of landscape characteristics (e.g., edge/patch ratios or habitat composition around points). Inferences drawn from microhabitat and landscape-scale characteristics may be substantially different and vary depending on life-history characteristics (e.g., migration strategy) of the birds in question (Mitchell et al. 2001).

Have the Effects of Wildfires on Bird Populations Changed?

Fire suppression may have created fires that are less frequent, larger, and more severe than those occurring previously. The potential shift to large and more severe fires may affect bird abundance and community composition. Little is known about potential changes in bird populations as a result of changing fire regimes. Comparing long-term data on bird abundance (e.g., Breeding Bird Survey data) with demographic models that can evaluate the effect of landscape composition on bird populations may be one way to evaluate the effect of changes created by fire suppression. Such an approach may be especially useful in evaluating the degree to which long-term declines of species that respond positively to fire may be a result of fire suppression policies.

How Does Prescribed Fire Change Conditions for Birds?

A return to more natural fire regimes is often advocated, but given the constraints imposed by concerns of public safety, economics, and air quality, it is more likely that this will be achieved through the application of prescribed fire in combination with manual and mechanical treatments (e.g., thinning of trees and removal of invasive shrubs). The application of these management tools may occur during seasons when fires did not normally burn. In the Pacific Northwest, fires burned naturally from June through September, with most occurring in late summer or early fall (Taylor and Skinner 1998, Brown et al. 1999). To maximize fire control, however, late winter or early spring is usually a better time to apply prescribed burns (Biswell 1989) to prevent escape. Effects of such burning schedules on bird abundance may be immediate, especially for ground- and shrub-nesting species, when they conflict with the breeding season. Such effects may be relatively short-lived and must be considered within the context of long-term benefits of prescribed fire (California Partners in Flight 2002). These changes are poorly understood in the Pacific Northwest, but are presumed to include vegetation changes that lead to longer-lasting changes in bird abundance (Kilgore 1971) or nest success (Jones et al. 2002). Prescribed fire can be applied in many different ways and have many different effects on vegetation and fuel and are most likely to be applied to the low- and moderate-severity fire regime vegetation types. Use and effects of prescribed fire in these types, which include Douglas-fir, grand fir, mixed evergreen hardwood, white fir, Shasta red fir, and

coast redwood, is complex and generalizations are not easily drawn from one type that can be applied easily to birds in another type.

How Do Fire Surrogates (Fuels Treatments) Affect Bird Populations?

An alternative to prescribed fire is the use of fire surrogates (usually mechanical methods) to reduce fuels and mimic other structural and possibly ecological effects of fire. Such management activities vary widely in intensity, from collection and removal of fuels by hand to large-scale mechanical processes that have high amounts of incidental disturbance. Evidence suggests that these activities do not create the same structural or ecological conditions as a natural or prescribed fire (Imbeau et al. 1999, and see Hannon and Drapeau, this volume). For example, when reproductive success was examined for Olive-sided Flycatchers associated with recent burns and logged areas in the western Oregon Cascade Mountains (Altman and Sallabanks 2000), nest success at burned sites was nearly twice that in logged area, suggesting that openings created by fire are better habitat for the flycatcher than logged areas. In contrast, a similar study by Meehan and George (2003) showed that the probability of nest-loss was greater in burned than unburned areas. Clearly even the effect of fire on bird reproduction, let alone the ability of mechanical fuels reduction to mimic these effects, is poorly understood. Mechanical methods reduce fuels and are certainly likely to reduce fire risk. After fire, however, standing dead trees remain. Trees removed by mechanical methods create habitat that is structurally and ecologically different from postfire conditions. These activities should be monitored with respect to their effect on bird populations in order to evaluate their ecological effects.

What Are Effects of Postfire Salvage?

Postfire salvage logging is a controversial practice that removes some amount of dead and at least apparently damaged live trees from burned areas. After fire, snags provide important nest sites and foraging opportunities for many cavity-nesting and bark-foraging birds (Hutto 1995, Kreisel and Stein 1999). Salvage logging has negative consequences for some bird species (Hutto 1995, Wales 2001). Studies from the Washington Cascades (Haggard and Gaines 2001) and ponderosa pine forests of southwestern Idaho (Saab and Dudley 1998) have both suggested that snag density and distribution are important factors influencing the abundance of cavity-nesting birds. As in many other situations, the effects of such treatments may interact with burn severity. Removing structure from severely burned areas may have different effects than the same level of extraction from less severely burned areas (Kotliar et al. 2002). Investigations of the effects of salvage should consider these types of interactions.

If Climate Change Alters Fire Regimes, What Is the Effect on Bird Communities?

The possibility that climate change may alter local fire regimes should be considered (Johnson and Larsen 1991). Such changes may affect bird abundance, community composition, distribution, and diversity. Predicting the effects of climate change on fire regimes and bird communities may provide us with the opportunity to test hypotheses about the relative importance of deterministic and stochastic processes in community assembly.

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