Appendix C: Terrestrial Focal Species Accounts

Species accounts for the 10 terrestrial focal species selected for the Umatilla/Willow subbasin are given below. These species accounts were provided to subbasin planners at the Council website http://www.nwppc.org/fw/subbasinplanning/admin/species/Default.asp#null. The authors of each species account are listed, although some selections have been edited.

PILEATED WOODPECKER
Dryocopus pileatus

Species Account Author: Charles Gobar, United States Forest Service

LIFE HISTORY, KEY ENVIRONMENTAL CORRELATES, AND HABITAT REQUIREMENTS
Migration Status: Permanent resident

Breeding Habitat: Woodland

Nest Type: Cavity

Clutch Size: 3-5

Length of Incubation: 15-18 days

Days to Fledge: 26-28

Number of Broods: 1

Diet

Feeds extensively on carpenter ants (*Camponotus* spp.) and beetle larvae obtained by chiseling into standing trees, stumps, and logs; also digs into anthills on ground and eats other insects, fruits, and seeds (Hoyt 1957). In Wisconsin, Nicholls (1994) found the cerambycid wood borer, *Trigonarthris*, to be the major prey of pileated woodpeckers feeding at dead American elms (*Ulmus americana*). The preference of the birds for feeding at larger trees seemed related to the requirement of the beetles for larger trees as their habitat. There tends to be seasonal variation in the diet and foraging strategy to take advantage of available foods. More fruit and seeds are taken in late summer and fall (Conner 1979, Hoyt 1948, Sprunt and Chamberlain 1970); more excavation for arthropods is done in winter (Conner 1979, Hoyt 1948, Pfitzenmeyer 1956, Tanner 1942). Quantitative studies of diet include stomach content and scat analysis. In a range-wide, year-round study, Beal (1911) found 80 stomachs to include 22% beetles (*Cerambycidae, Buprestidae, Elateridae, Lucanidae, Scarabaeidae, Carabidae*), 40% ants (*Camponotus sp., Crematogaster sp.*), 11% other insects, and 27% vegetable (numerous fruits, see Bull and Jackson 1995). Analyses of 330 scats in Oregon revealed 68% carpenter ants, 29% thatching ants (*Formica*), 0.4% beetles, and 2% other. The species is opportunistic, known to take advantage of insect outbreaks (e.g., western spruce budworm (*Choristoneura occidentalis*) Bull and Jackson 1995), the progression of fruiting trees in an area (Stoddard 1978), and to visit suet feeders in many areas of eastern North America (Connecticut, Hardy 1958; Mississippi, Jackson, pers. obs.; Tennessee, Spofford 1947; Georgia, Stoddard 1978; Minnesota, Tusler 1958).

Logs and stumps are important foraging substrates in many areas (e.g., Mannan 1984, Renken and Wiggers 1989, Schardien and Jackson 1978), but Aubry and Raley (1992) rarely observed foraging on logs in closed canopy forests of western Washington. Mannan (1984) found the pileated to forage on dead wood substrates 96% of the time.

Reproduction

Pairs share a territory year round (Bull and Jackson 1995). On warm days of February and early March in the southeastern U.S. and March through early April in northern areas there is an increase in vocalizations and drumming associated with pair formation and increased territoriality. Vocalizations and drumming take place with greatest frequency in early morning and late afternoon (Hoyt 1941). Courtship behavior is described in detail by Kilham (1979, 1983), with additional details and circumstances by Arthur (1934), Hoyt (1944), and Oberman (1989). Nest construction, egg-laying, hatching, and fledging are also progressively later from south to north (Bull and Jackson 1995) and likely from lower to higher altitudes (at least in California, Harris 1982).

Early egg dates in the southern U.S. are in early March; late egg dates, from northern areas, are in mid-June. Similarly, nestlings have been found from mid-May in the southeast to mid-July in the north (Bull and Jackson 1995, Peterjohn 1989). Young remain with adults at least through late summer or early fall. Clutch size is usually 3-4 throughout the range (Bent 1939, Christy 1939); a clutch of 6 was reported by Audubon and Chevalier (1842). Incubation takes 15-19 days (Bendire 1895, Hoyt 1944, Kilham 1979), by both sexes. Young are tended by both parents, leave nest at 22-26 days (Hoyt 1944, Bull and Jackson 1995).

Longevity records thus far include several birds surviving for 9 years (Bull and Jackson 1995, Bull and Meslow 1988, Hoyt and Hoyt 1951, Hoyt 1952). However, through 1981, there had only been 15 recoveries from a total of 670 banded (Clapp et al. 1983), thus it is quite possible that this species could live much longer.

Migration

Although generally considered to be a resident species, there is evidence of some migratory movement in the northern part of its range. Hall (1983) reported a small southward movement of pileated woodpeckers in fall along the Allegheny Front of West Virginia. Sutton (1930) also noted gradual southward movement in fall through New York State. In British Columbia, the paucity of winter records in the northern half of the province indicates that many breeding individuals there move considerable distances to the south (Campbell et al. 1990).

Threats

Major threats are (from greatest to least): (1) conversion of forest habitats to non-forest habitats, (2) short rotation, even-age forestry, (3) monoculture forestry, (4) forest fragmentation, (5) removal of logging residue, downed wood, and pine straw that would ultimately put nutrients back into the ecosystem and provide foraging substrate, (6) lightning striking cavity/roost trees because they are the oldest, tallest trees around as a result of cutting priorities, (7) deliberate killing by humans, and (8) toxic chemicals. The first four threats are ones that have been a major concern for some time.

As an example of habitat losses, nonfederal forested wetlands decreased by 5 million acres in the continental U.S. between 1982 and 1987 (Cubbage and Flather 1992). Forest fragmentation has been recognized as a major problem for many wildlife species (e.g., Wilcove 1990), but it results in habitat changes within as well as between fragments. In the southeast, smaller fragments tend to become drier (hence less conducive to conditions favorable to the pileated) and also change in plant species composition and tend towards younger successional stages (Rudis 1992). Removal of logging residue, downed wood, and pine straw from forested areas is becoming increasingly common. Considerable research directed at finding ways to maximize economic returns from the forest through such actions is being conducted by the U.S. Forest Service and others (e.g., Howard and Setzer 1989) and pine straw is currently sold on some southern forests. Removing these materials not only removes the nutrients they contain and foraging substrates for pileated woodpeckers and others, but also changes the water balance of the forest floor, making the forest a drier environment less suitable for the arthropod fauna the woodpecker is dependent on.

Shooting by humans was a serious problem in the past (e.g., Sclater 1912, Stoddard 1947) and continues in some areas (Jackson, pers.obs.). The birds are an impressive and easy target and in some quarters are considered to harm trees. Becker (1942) offered one of the most detailed accounts of the disappearance of the species. Toxic chemicals can affect woodpeckers in two ways: (1) by direct poisoning and (2) by killing their arthropod prey. Careless use of agricultural chemicals and widespread control programs such as have been conducted in the past against the imported fire ant can have both affects. In addition, when woodpeckers nest in chemically treated utility poles, embryos or chicks can be killed by the fumes (Rumsey 1970).

In the eastern U.S., rat snakes (*Elaphe obsoleta*) have been reported as nestling predators (Gress and Wiens 1983, Kilham 1959, Moore 1984). Both sharp-shinned (*Accipiter striatus*; Smith 1983) and Cooper's (*A. cooperi*; Michael 1921) hawks are known as potential predators on pileated woodpeckers. Erdman (pers. comm.) has found remains of adults and juveniles at goshawk (*A. gentilis*) nests in Wisconsin. The sharp-shinned hawk is certainly more of a threat to fledglings than to adults. Todd (1944) reported predation by a gray fox (*Urocyon cinereoargenteus*) on a ground-feeding pileated in Tennessee. Because they feed extensively on the ground, woodpeckers are vulnerable to being killed by vehicles as they approach or leave feeding sites (e.g., Eifrig 1944), an argument for keeping downed wood away from highway rights-of-ways.

Habitat Requirements (Nesting, Breeding, Non-breeding)

General

Dense deciduous (favored in southeast), coniferous (favored in north, northwest and west), or mixed forest, open woodland, second growth, and (locally) parks and wooded residential areas of towns. Prefers woods with a tall closed canopy and a high basal area. Most often in areas of extensive forest or minimal isolation from extensive forest. Uses a minimum of 4 cavities per year (only one for raising brood).

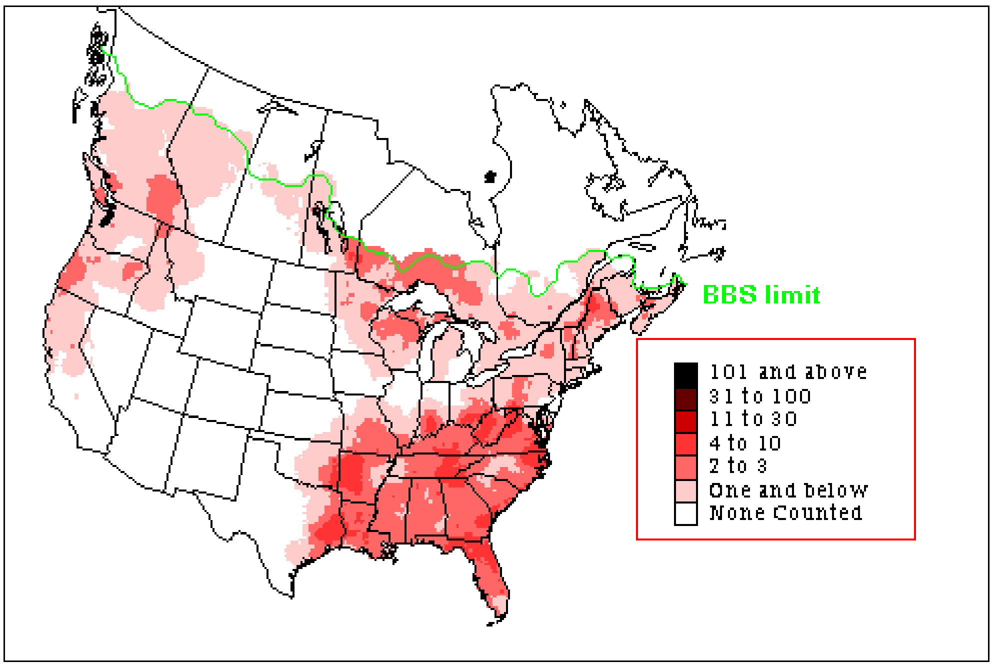
Nesting

Nests are in cavities excavated by both sexes usually in dead stubs in shaded places; cavity entrance averages about 14 m above ground (see photos and descriptions in Harrison 1975, 1979). Usually digs a new hole for each year's brood, but the same cavity may be used for several years. Nest tree species and size varies among regions and even within regions depending on site and availability. In southern British Columbia, preferred nest sites were in live aspen with heartwood decay, in trees larger than 40 cm dbh (Harestad and Keisker 1989). In northwest

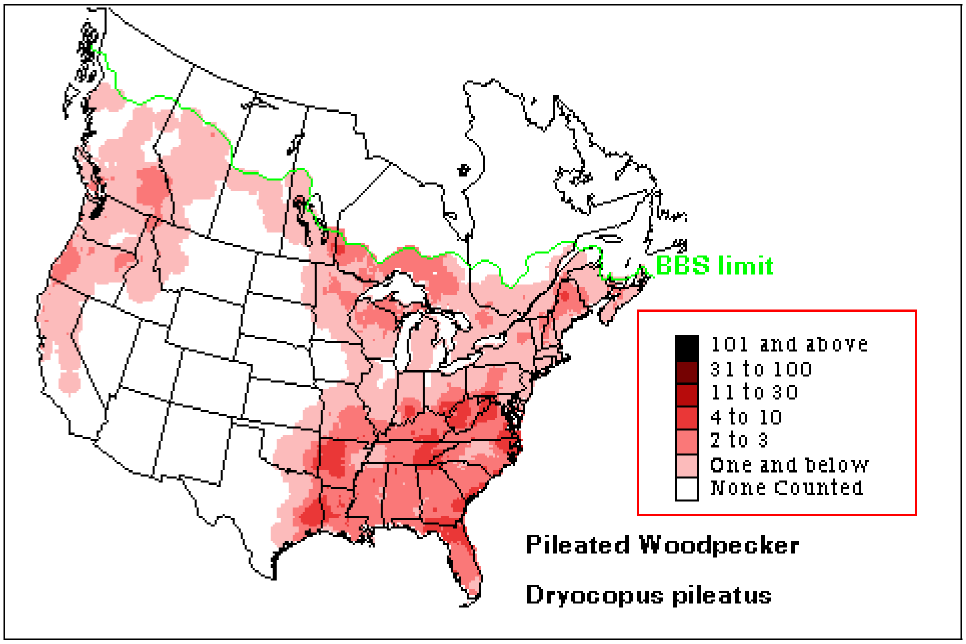
Montana, most of 54 nest trees were large western larch (*Larix occidentalis*) and nest trees averaged 74.9 cm dbh (McClelland 1979). In northeast Oregon, 75% of nest trees were ponderosa pine (*Pinus ponderosa*) and mean dbh of nest trees was 84 cm (Bull 1987). In western Oregon, 73% of nest trees were Douglas-fir (*Pseudotsuga menziesii*) and nest trees averaged 69 cm dbh (Mellen 1987). In Virginia, 28% of nest trees were hickory (*Carya spp*.), 22% red oak (*Quercus rubra*), 17% chestnut oak (*Q. prinus*) and nest trees averaged 54.6 cm dbh (Conner et al. 1975). Most studies report nests 5-17 m above ground in wood softened by fungal rot, in trees usually 100-180 years old, over 51 cm DBH, 12-21 m tall, and often near permanent water (Bushman and Therres 1988).

Population and Distribution (historic and current)

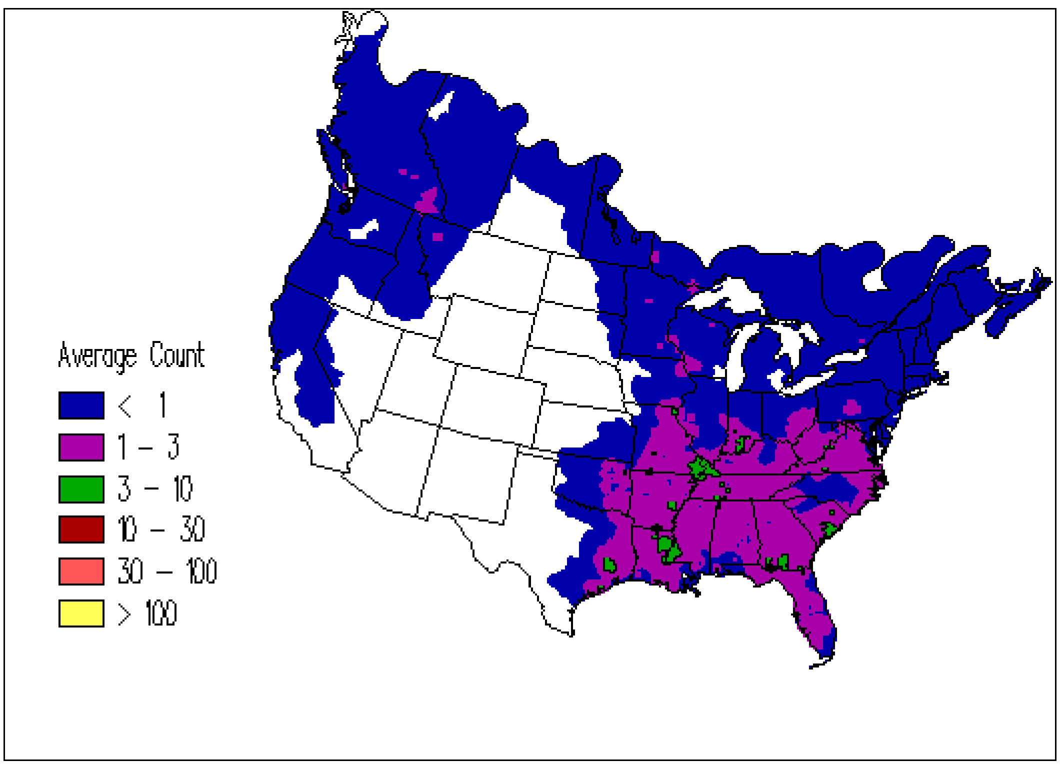
Current Summer Distribution Map and Abundance (from CBC data) (Sauer et al. 2003

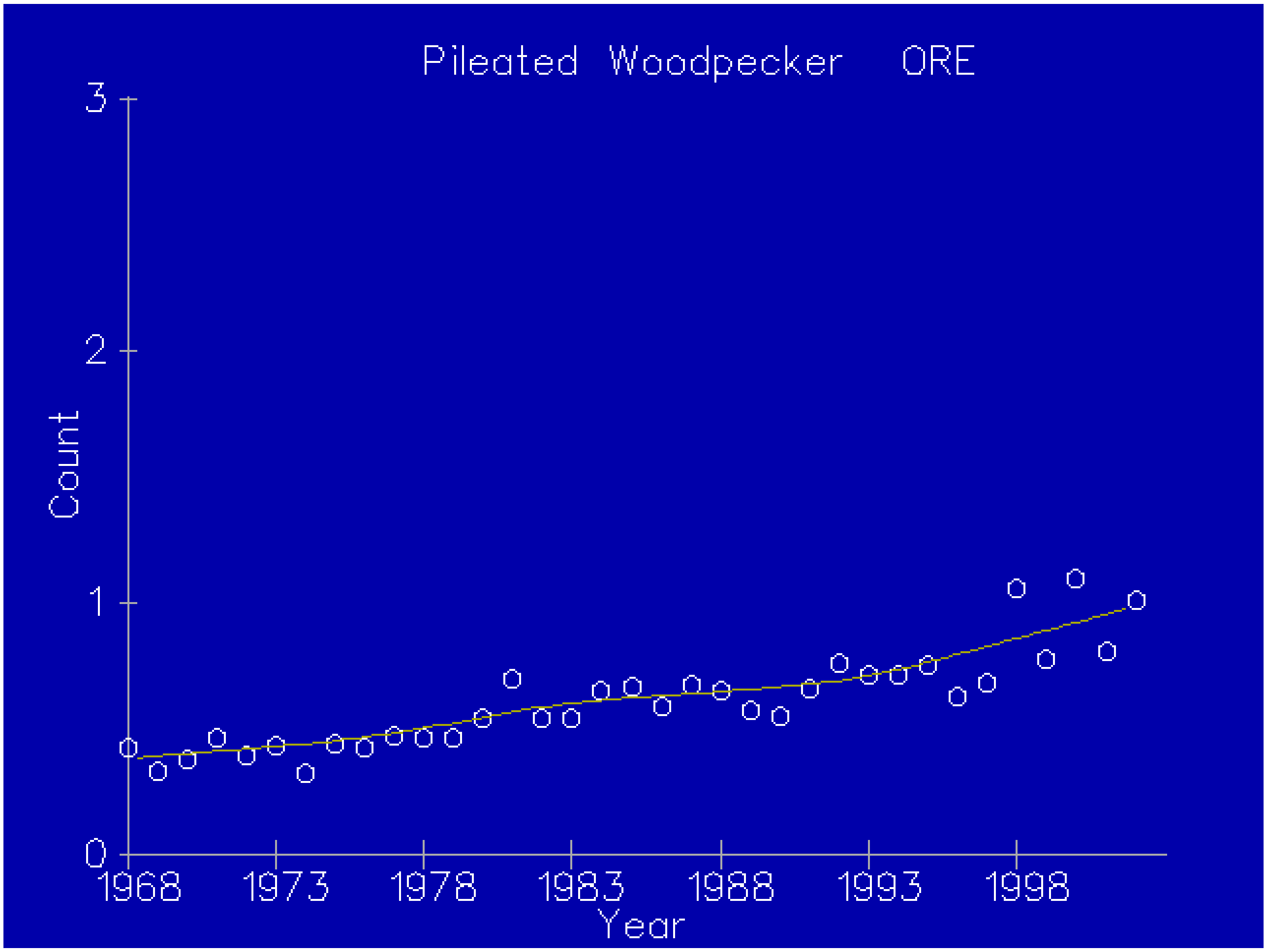


Current Breeding Distribution and Abundance (from CBC data) (Sauer et al. 2003)



Current Winter distribution from CBC





Pileated Woodpecker Population Trend Data, Oregon (From BBS)

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WHITE-HEADED WOODPECKER
Picoides albolarvatus

Original Species Account Authors: Paul Ashley and Stacy Stoval, as appeared in the Southeast Washington Ecoregional Assessment, January 2004

Introduction
The white-headed woodpecker (*Picoides albolarvatus*) is a year round resident in the Ponderosa pine (*Pinus ponderosa*) forests found at the lower elevations (generally below 950m). White-headed woodpeckers are particularly vulnerable due to their highly specialized winter diet of ponderosa pine seeds and the lack of alternate, large cone producing, pine species.

Nesting and foraging requirements are the two critical habitat attributes limiting the population growth of this species of woodpecker. Both of these limiting factors are very closely linked to the habitat attributes contained within mature open stands of Ponderosa pine. Past land use practices, including logging and fire suppression, have resulted in significant changes to the forest structure within the Ponderosa pine ecosystem.

White-headed Woodpecker Life History, Key Environmental Correlates, and Habitat Requirements
Life History
Diet
White-headed woodpeckers feed primarily on the seeds of large Ponderosa pines. This is makes the white-headed woodpecker quite different from other species of woodpeckers who feed primarily on wood boring insects (Blood 1997; Cannings 1987 and 1995). The existence of only one suitable large pine (ponderosa pine) is likely the key limiting factor to the white-headed woodpecker's distribution and abundance.

Other food sources include insects (on the ground as well as hawking), mullein seeds and suet feeders (Blood 1997; Joe *et al.* 1995). These secondary food sources are used throughout the spring and summer. By late summer, white-headed woodpeckers shift to their exclusive winter diet of ponderosa pine seeds.

Reproduction
White-headed woodpeckers are monogamous and may remain associated with their mate throughout the year. They build their nests in old trees, snags or fallen logs but always in dead wood. Every year the pair bond constructs a new nest. This may take three to four weeks. The nests are, on average 3m off the ground. The old nests are used for overnight roosting by the birds.

The woodpeckers fledge about 3-5 birds every year. During the breeding season (May to July) the male roosts in the cavity with the young until they are fledged. The incubation period usually lasts for 14 days and the young leave the nest after about 26 days. White-headed woodpeckers have one brood per breeding season and there is no replacement brood if the first brood is lost. The woodpeckers are not very territorial except during the breeding season. They are not especially social birds outside of family groups and pair bonds and generally do not have very dense populations (about 1 pair bond per 8 ha).

Nesting
Generally large ponderosa pine snags consisting of hard outer wood with soft heartwood are preferred by nesting white-headed woodpeckers. In British Columbia 80 percent of reported nests have been in ponderosa pine snags, while the remaining 20 percent have been recorded in Douglas-fir snags. Excavation activities have also been recorded in Trembling Aspen, live Ponderosa pine trees and fence posts (Cannings *et al.* 1987).

In general, nesting locations in the South Okanagan, British Columbia have ranged between 450 - 600m (Blood 1997), with large diameter snags being the preferred nesting tree. Their nesting cavities range from 2.4 to 9 m above ground, with the average being about 5m. New nests are excavated each year and only rarely are previous cavities re-used (Garrett *et al.* 1996).

Migration
The white-headed woodpecker is a non-migratory bird.

Habitat Requirements
Breeding
White-headed woodpeckers live in montane, coniferous forests from British Columbia to California and seem to prefer a forest with a relatively open canopy (50-70 percent cover) and an availability of snags (a partially collapsed, dead tree) and stumps for nesting. The birds prefer to build nests in trees with large diameters with preference increasing with diameter. The understory vegetation is usually very sparse within the preferred habitat and local populations are abundant in burned or cut forest where residual large diameter live and dead trees are present.

Highest abundances of white-headed woodpeckers occur in old-growth stands, particularly ones with a mix of two or more pine species. They are uncommon or absent in monospecific ponderosa pine forests and stands dominated by small-coned or closed-cone conifers (e.g., lodgepole pine or knobcone pine).

Where food availability is at a maximum such as in the Sierra Nevadas, breeding territories may be as low as 10 ha (Milne and Hejl 1989). Breeding territories in Oregon are 104 ha in continuous forest and 321 ha in fragmented forests (Dixon 1995b). In general, open Ponderosa pine stands with canopy closures between 30-50% are preferred. The openness however, is not as important as the presence of mature or veteran cone producing pines within a stand (Milne and Hejl 1989). In the South Okanagan, British Columbia, Ponderosa pine stands in age classes 8 -9 are considered optimal for white-headed woodpeckers (Haney 1997). Milne and Hejl (1989) found 68 percent of nest trees to be on southern aspects, this may be true in the South Okanagan as well, especially, towards the upper elevational limits of Ponderosa pine (800 - 1000m).

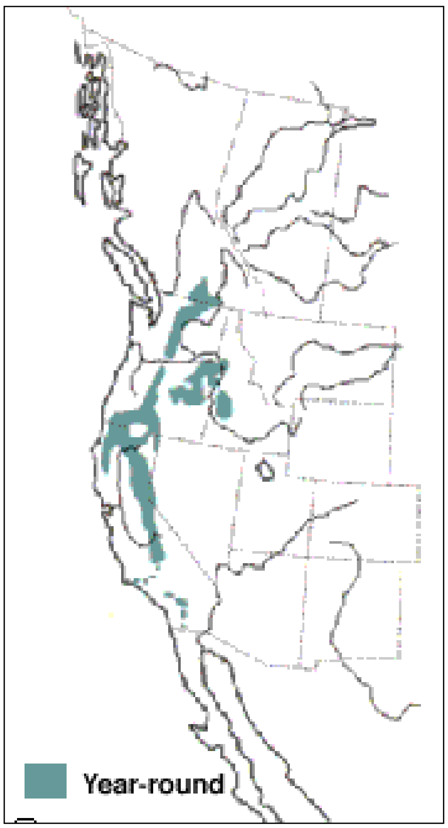
White-headed Woodpecker Population and Distribution
Population
Historic
No data are available.

Current
No data are available.

Distribution
Historic
No data are available.

Current
These woodpeckers live in montane, coniferous forests from southern British Columbia in Canada, to eastern Washington, southern California and Nevada and Northern Idaho in the United States. The exact population of the white-headed woodpecker is unknown but there are thought to be less than 100 of the birds in British Columbia.

Woodpecker abundance appears to decrease north of California. They are uncommon in Washington and Idaho and rare in British Columbia. However, they are still common in most of their original range in the Sierra Nevada and mountains of southern California. The birds are non-migratory but do wander out of their range sometimes in search of food.



White-headed Woodpecker Status and Abundance Trends
Status
Although populations appear to be stable at present, this species is of moderate conservation importance because of its relatively small and patchy year-round range and its dependence on mature, montane coniferous forests in the West. Knowledge of this woodpecker's tolerance of forest fragmentation and silvicultural practices will be important in conserving future populations.

Trends

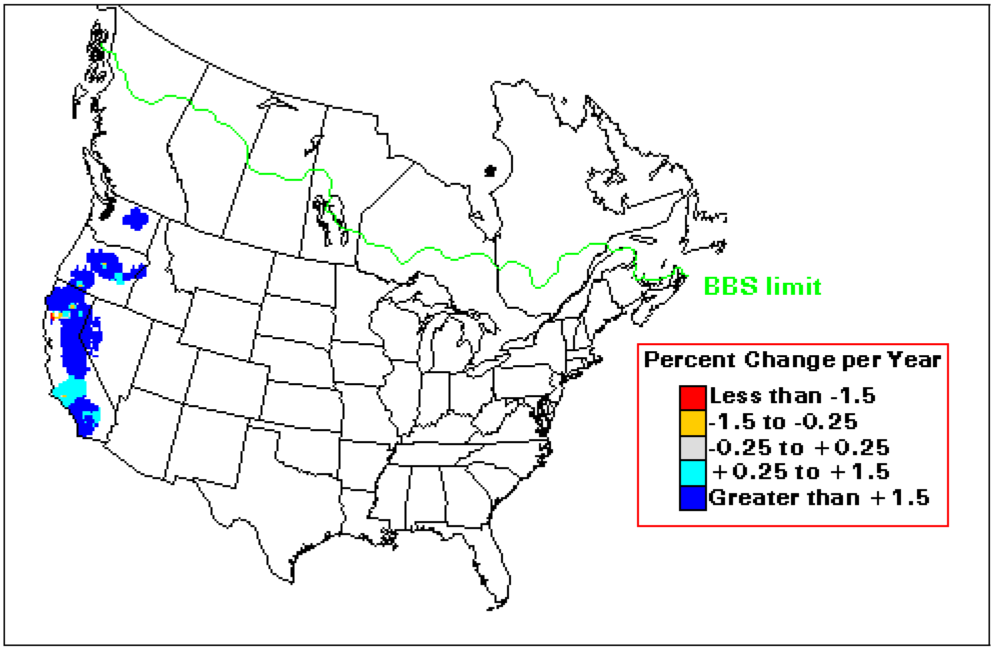


Figure 1. White-headed woodpecker Breeding Bird Survey (BBS) population trend: 1966-1996 (Sauer *et al.* 2003).

Factors Affecting White-headed Woodpecker Population Status
Key Factors Inhibiting Populations and Ecological Processes
Logging

Logging has removed much of the old cone producing pines throughout the South Okanagan. Approximately 27, 500 ha of ponderosa pine forest remain in the South Okanagan and 34.5 percent of this is classed as old growth forest (Ministry of Environment Lands and Parks 1998). This is a significant reduction from the estimated 75 percent in the mid 1800s (Cannings 2000). The 34.5 percent old growth estimate may in fact be even less since some of the forest cover information is incomplete and needs to be ground truthed to verify the age classes present. The impact from the decrease in old cone producing ponderosa pines is even more exaggerated in the South Okanagan because there are no alternate pine species for the white-headed woodpecker to utilize. This is especially true over the winter when other major food sources such as insects are not available. Suitable snags (DBH>60cm) are in short supply in the South Okanagan.

Fire Suppression

Fire suppression has altered the stand structure in many of the forests in the South Okanagan. Lack of fire has allowed dense stands of immature ponderosa pine as well as the more shade tolerant Douglas-fir to establish. This has led to increased fuel loads resulting in more severe stand replacing fires where both the mature cone producing trees and the large suitable snags are destroyed. These dense stands of immature trees has also led to increased competition for nutrients as well as a slow change from a Ponderosa pine climax forest to a Douglas-fir dominated climax forest.

Predation

There are a few threats to white-headed woodpeckers such as predation and the destruction of its habitat. Chipmunks are known to prey on the eggs and nestlings of white-headed woodpeckers.

There is also predation by the great horned owl on adult white-headed woodpeckers. However, predation does not appreciably affect the woodpecker population.

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RED-NAPED SAPSUCKER
Sphyrapicus nuchalis

Original Species Account Author: Charles Gobar, United States Forest Service

Introduction
The red-naped sapsucker (*Sphyrapicus nuchalis*) occurs in the inland West, inhabiting montane coniferous forests mixed with deciduous groves of aspen (*Populus spp*.), cottonwood (*Populus spp.*), and willow (*Salix spp*.). The sapsucker creates nest cavities and sap wells that are used by other birds, mammals and insects. Considered a double key stone species as its nest cavities are sued by secondary cavity-nesters and its sp wells provide food for a variety of other animal, from insects to other birds to squirrels (Daily et al. 1993). Locally common, populations are generally stable to increasing, but there is concern over loss of aspen and cottonwood nesting habitat and large snags for nest cavities.

Life History, Key Environmental Correlates, and Habitat Requirements
Life History
Diet
In general, the sapsucker diet includes sap, cambium and soft parts beneath the bark. Neat rows of holes are drilled in the bark or the bark may be removed in strips to collect the oozing sap and insects attracted to it (Marshall et al. Eds. 2003). Rows of small holes are drilled in conifer and broad-leaved trees and the sapsucker. The amount of sap taken and tree species used vary seasonally (Scott et al. 1977). Sap is most important in seasons when insects are not abundant. The sapsucker also feeds on insects caught in the sap. Other foods items the bird feeds on include tree cambium, ants, larvae, beetles, wasps, caterpillars, and small amounts of fruit and berries (Scott et al. 1977, Marshall et al. Eds. 2003). [NatureServe 2003]

Reproduction
Courtship and territorial displays may involve drumming and posturing and calling during the breeding season. Territories for red-naped sapsucker range from 1.6 to > 14.6 acres (Marshall et al. Eds. 2003). In the Pacific Northwest, territory size reported to be about 10 acres (Bull 1978 *in* NatureServe 2003) in size. In California, defends territories 0.6 to 6.0 hectares in size (USDA Forest Service 1994 *in* NatureServe 2003). Both sexes begin excavating a nest cavity before copulating. Three to seven eggs are laid and young are in the nest cavity from mid-May to late July (Gabrielson and Jewett 1970, and Anderson 1988e, Anderson 1989d, and Spencer 2000b *in* Marshall et al. Eds. 2003)

The red-naped sapsucker is known to hybridize with red-breasted sapsucker (*Sphyrapicus ruber*) and yellow-bellied sapsucker (*Sphyrapicus varius*) where distributions overlap. The outcome may produce viable hybrid offspring; hybrid and backcross mating (Scott et al. 1976, Johnson and Johnson 1985 *in* NatureServe 2003).

Nesting
Typically, four to five eggs are laid and incubated by both female and male sapsuckers. Eggs are incubated 12-13 days and fledging occurs in 25-26 day; both sexes attend young (Ehrlich et al. 1988 *in* NatureServe 2003). In Colorado, nests with eggs were recorded throughout June. Nestlings were noted from late June to mid-July in Montana and Wyoming (Johnsgard 1986 *in* NatureServe 2003). In central Arizona, 100 percent of 18 nests monitored successfully fledged

young (Li and Martin 1991 *in* NatureServe 2003). Re-use of same nest tree, but with a new cavity, each year suggests strong site fidelity (USDA Forest Service 1994 *in* NatureServe 2003).

Migration
The red-naped sapsucker is a local migrant and a long distance migrant. Arrives in northern Rocky Mountains mainly April-May, with peak arrival from late April to early May. Fall migration occurs from mid August o mid October (Gabrielson and Jewett 1970). The red-naped woodpecker is a transient and winter visitor in northwestern Mexico from late September to mid-April (Howell and Webb 1995 in NatureServe 2003).

Mortality
No information is available on survival rates.

Harvest
Not applicable.

Historic
Not applicable.

Current
Not applicable.

Habitat Requirements
The red-naped sapsucker responds to habitat mosaic that includes broad-leaved trees (e.g. aspen, birch, and cottonwood) for nesting and adjacent coniferous forest and/or willows for foraging (Ehrlich and Daily 1988 *in* NatureServe 2003, Tobalske 1992). Typically found in riparian habitats especially aspen, as well as cottonwoods, alders, and pine forest, and less frequently in mixed conifer forests (Marshall et al. Eds. 2003). Known to use natural edges of mature conifer and deciduous hardwood habitats. Gabrielson and Jewett (1970) and Browning (1973b *in* Marshall et al. Eds. 2003) found sapsucker nests more abundant between 6,000 and 7,000 feet in the Blue Mountains. Numerous nests were found in two area of south-central Oregon, at elevations from 5,200-6,600 feet and 6,650-7,550 feet (Dobkin et al. 1995 and Trombino 1998 *in* Marshall et al. Eds. 2003).

In a Colorado study, abundance did not vary with differences in understory (herbaceous, short shrub, tall shrub) of mature aspen stands (Finch and Reynolds 1987 *in* NatureServe 2003). In a study of Idaho cottonwoods gallery forest, there appeared to be no significant sensitivity to patch size, although birds were more often detected in large patches (more than 25-495 ac. 0.21 birds per point count visit) than in small patches (less than 2-7 acres; 0.12 birds per point count visit; Saab 1998).

Will use forest edges and logged forests, but extensive clearcuts or the removal of snags and preferred tree species would be detrimental. Also will use burns, partially cut forests and small clearcuts where snags and live hardwood trees remain and adjacent forest is available for foraging (Bock and Lynch 1970,and Tobalske 1992 *in* NatureServe 2003).

Nesting
A primary cavity nester, excavates a nest hole in a snag or a living tree with a dead or rotten interior, and shows a strong preference for aspen (Johnsgard 1986, Li and Martin 1991, and

Daily et al. 1993 *in* NatureServe 2003). The red-naped sapsucker will also use cottonwood (*Populus spp.*), alder (*Alnus spp.*), western larch (*Larix occidentalis*), ponderosa pine, lodgepole pine ((*Pinus contorta*); USDA 1991. Aspen nest trees often have heartwood decay brought about by shelf fungus (*Fomes igniarius var. populinus*), a heart rot that infects roots and dead branch stubs and spreads from the base of trees upward, but leaves the sapwood intact (Kilham 1971, Crockett and Hadow 1975, Daily et al. 1993, and Dobkin et al. 1995 *in* NatureServe 2003). Seventy-two percent of live aspen with woodpecker-excavated cavities at Hart Mountain had visible fungi. Of the 25 nests in riparian and snowpocket aspen woodlands on Hart Mountain, 92-100 percent were in aspens. Dead trees (8%) and live trees (92%) were used in proportion to availability (Dobkin et al. 1995).

In a Colorado study; sapsuckers placed the first nest cavity close to ground and then excavated progressively higher cavities in subsequent years. Nest cavities were usually freshly excavated during the season of use and most nests were in trees bearing nest cavities excavated during previous years. Nest height averaged 8.8 feet in trees with no other cavities and 19.7 feet in trees with more than one cavity (Daily et al. 1993). In a study in Colorado and Wyoming, sapsuckers used both healthy aspen and aspen infected by shelf fungus, nested in trees 6.7 to 16.5 inches dbh (mean 12.2 inches dbh) and used cavities that were 3.3 to 36 feet high (mean 16.4 feet; Crockett and Hadow 1975).

In the Hart Mountain study (Dobkin et al. 1995 *in* NatureServe 2003) mean diameter at breast height was 10.8 inches, tree height was 47.9 feet, cavity height was 13.8 feet and entrance diameter was 1.7 inches. Less than 4 percent of all aspens were greater than 33 feet in height and greater than 9 inches in diameter at breast height, yet were preferred as nest trees. No nests were located along the riparian woodland edge nor were any oriented in that direction. Nest trees on average were located 65.6 feet from edges, and the mean canopy cover was 76 percent (Dobkin et al. 1995 *in* Marshall et al. Eds. 2003).

In Oregon and Washington, the red-naped was reported to nest in snags greater than or equal to 10 inches diameter breast height and nest heights at least 15 feet in height (Thomas et al. 1979). In the Blue Mountains of northeast Oregon, of eight nests, seven (88%) were within 330 feet of open water. Nests were in western larch, lodgepole pine, Douglas-fir, grand fir, and ponderosa pine; two were in live trees. Trees retained 70-100 percent of original bark and were likely dead less than 10 years. Mean diameter at breast height was 20 inches, trees height was 66 feet, and cavity height was 30 feet (Bull 1980 *in* Marshall et al. Eds. 2003). In western larch/Douglas-fir (*Pseudotsuga menziesii*) forests of northwestern Montana, red-naped sapsuckers nested in both small and large trees, ranging from 22 to 46.8 inches diameter at breast height and averaging 22.8 inches diameter at breast height (McClelland et al. 1979 *in* NatureServe 2003).

In mixed coniferous forest in northeast Oregon, densities per 100 acres were 0-0.5 in old growth (Mannan 1982 *in* Marshall et al. Eds. 2003). In mixed coniferous and aspen forest (six sights ranging from 1-98 percent aspen) at 9,000 feet on the west slope of the Rocky Mountains, in Colorado densities ranged 0-3 birds per 100 acres (Scott and Crouch *in* Marshall et al. Eds. 2003).

Breeding

The red-naped sapsucker primarily breeds in coniferous forests that include aspen and other hardwoods vegetation types. In the Northern Rockies, most abundant in cottonwood and aspen forests, also observed in other riparian cover types and in harvested conifer forests. Of harvest types, most observations were in patch cuts, seed-tree cuts, clearcuts and older clearcuts. Birds in harvested stands and in drier conifer forests were probably associated with patches of deciduous trees (Hutto and Young 1999 *in* NatureServe 2003). In the Centennial Mountains, Idaho, the sapsucker uses xeric tall willow (*Salix spp*.) communities (Douglas et al. 1992). In Wyoming and Colorado, closely associated with aspen and mixed habitats (Finch and Reynolds 1988 *in* NatureServe 2003). In Colorado subalpine forests, significantly associated with habitats where aspen occurs near (less than 164 feet) willow, and used the willow for foraging (Ehrlich and Daily 1988, Daily et al. 1993). In the Pacific Northwest, typically breeds in aspen, riparian cottonwood, ponderosa pine, mixed conifer, and white fir (*Abies concolor*) forests (Bull 1978 *in* NatureServe 2003).

Foraging
The sapsucker drills for sap in conifer (e.g., western larch, pine) and deciduous trees (e.g. aspen, willow, cottonwood and birch (*Betula spp*.). In Oregon, aspen, willow, elm, apple, and ornamental pine trees are used often for foraging. In California, the red-naped drilled in and around pitchy bole wounds on ponderosa pine that were the result of earlier overstory removal and porcupine feeding (Oliver 1970 *in* NatureServe 2003). Sap well attract insects and are used for drinking sap.

Non-breeding
During migration and winter the sapsucker tends to use various forest and open woodland habitats, parks, orchards, and gardens (AOU 1998). In northwestern Mexico found in forests and edge feeding at mid- to upper levels; may overlap with wintering yellow-bellied sapsuckers in north-central Mexico and red-breasted sapsuckers in northern Baja California (Howell and Webb 1995 in NatureServe 2003). In western Mexico, Hutto (1992 in NatureServe 2003) found red-naped sapsucker only in pine-oak-fir forests.

Management
Sustaining populations of red-naped sapsuckers requires maintaining, enhancing, and restoring snags, riparian woodlands, and hardwood stands of aspen or cottonwood adjacent to coniferous forest. Both snags and live trees retained for the species should include a mix of hardwood and conifer species, particularly near riparian areas and mesic sites (USDA Forest Service 1994 *in* NatureServe 2003). Aspen and other trees with shelf fungus (*Fomes ignlarius populinus*) should be retained to provide optimal conditions for nest cavities. Access to conifer sap in adjacent forest is also important in the early spring, and to birches and aspens after bud-break (Tobalske 1992).

Partners in Flight have established biological objectives for this species in riparian woodland habitat for the Northern Rocky Mountains of Eastern Oregon and Washington (Altman 2000). These include providing and maintaining habitats that meet the following definition: large trees and snags, especially aspen and cottonwood, with adequate representation of younger seral stages for replacement (i.e., greater than 10 percent cover of sapling in the understory); greater than 1.5 trees (live) per acre and greater than 1.5 snags per acre, greater than 39 feet in height and 10 inches in diameter at breast height; and mean canopy cover between 30 to 70 percent, either clumped with patches and openings or relatively evenly distributed (Altman 2000). In addition, were ecologically appropriate, initiate actions in aspen habitat to provide areas with natural (e.g., fire) or mechanical disturbance to provide successional development in the stand (Altman 2000). Sustaining populations requires maintaining, enhancing, and restoring snags, riparian woodland, and hardwood stands of aspen, birch, and cottonwood adjacent to coniferous forest.

Population and Distribution
Population
Historic
Historic population data was not available for this species.

Current
The red-naped populations appear to be stable to increasing overall, with areas of local declines, perhaps related to loss of cottonwood, and aspen nesting habitats. However, North American Breeding Bird Surveys (BBS) trend estimates confounded because of changes in sapsucker taxonomy splitting red-naped from yellow-bellied sapsucker (*Sphyrapicus varius*) and BBS sampling and sample size are minimal for analysis for most states and physiographic regions. The BBS data indicates a nonsignificant population increase in North America Between 1966 and 1996 (1.3 percent average increase per year), and a steep and significant increase between 1980 and 1996 (4.5 percent average increase per year (Sauer et al. 2003).

Most likely including yellow-bellied sapsucker data (vs. only red-naped data), Thomas, et al (1979) estimated that 150 snags per 100 acres, greater than or equal to 10 diameter at breast height were necessary to support the “maximum population” in Blue Mountain forests of Oregon and southeast Washington.

Captive Breeding Programs, Transplants, Introductions
Not applicable for this species.

Historic
Not applicable for this species.

Current
Not applicable for this species.

Distribution
Historic
Historic distribution data was not available or extremely limited for this species. The species is noted in Gabrielson and Jewett (1970) as regular but not a common resident and breeding bird of eastern slope of Cascades, Blue Mountains and timbered parts of isolated ranges of eastern Oregon.

Current
The red-naped sapsucker breeds in the Rock Mountain region from southwest Canada, west and central Montana, and southwest South Dakota south, east of the Cascades and Sierra Nevada, to east-central California, southern Nevada, central Arizona, southern New Mexico, and extreme western Texas ((AOU 1983 *in* NatureServe and *in* Marshall et al. Eds. 2003). The current distribution of red-naped sapsucker is shown in **Figure 1.**

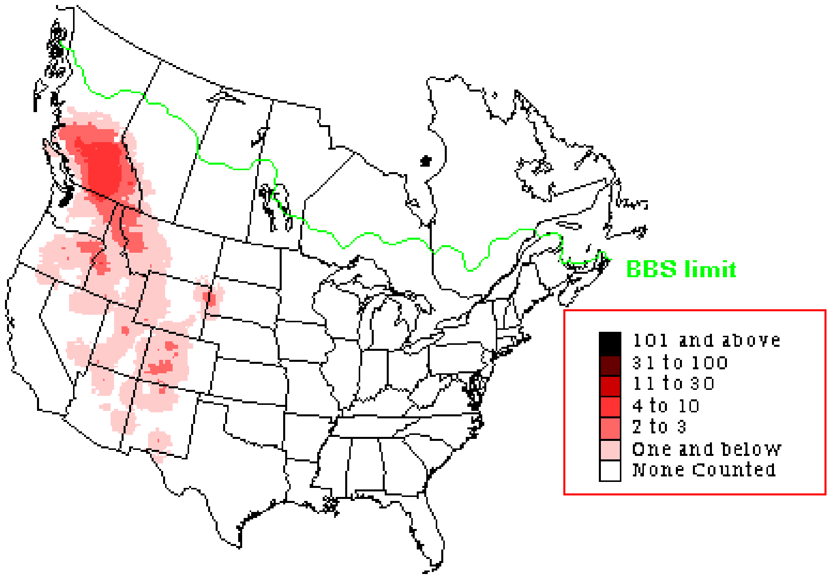

Figure 1: Red-naped sapsucker summer distribution based on Breeding Bird Surveys (Sauer et al. 2003).

Breeding

In Oregon, the sapsucker is a common summer resident throughout the eastern slope of the Cascades eastward throughout the Blue Mts., Wallowa Mtn., and lesser mountains, such as Mahogany Mtn. (Malheur Co.), Steens Mtn. (Harney Co.), and Hart Mtn. (Lake Co.) (Gilligan et al. 1994).

Non-Breeding

Winters in southern California (casually in Oregon, southern Nevada, central Arizona, and central New Mexico south to southern Baja California, and northwest and north-central Mexico, including Jalisco, Durango, Coahuila and Nuevo Leon ((AOU 1983) *in* NatureServe and *in* Marshall et al. Eds. 2003).

A common spring and fall transient through the mountains of eastern Oregon, and at lower elevations along rivers, in town, and at desert oases. Occurs rarely in winter along the east slope of the Cascades and very rare elsewhere east of the Cascades.

Red-naped Sapsucker Status and Abundance Trends

Status

Red-naped sapsuckers are demonstrably secure globally. In Oregon the species in not identified as threatened, endangered, or sensitive species (ODFW 1997). Within the state of Oregon, red-naped sapsuckers are apparently secure and are not of conservation concern (Altman 2000).

Trends

Trend estimates for other states and physiographic regions for these periods showed not statistically significant change. Mapped trends for 1966-1996 show population declines in parts of British Columbia and Alberta, central Oregon, and the central Rockies (eastern Idaho to Utah and n. Colorado), and marked increases in the Northern Rockies, southern Colorado, and northern New Mexico (Sauer et al. 2003 *in* NatureServe 2003). BBS data for Oregon showed a non-significant increase of 0.5 percent increase per year, in the population from 1966-2000 (Sauer et al. 2003).

Factors Affecting Red-naped Sapsucker Population Status

Key Factors Inhibiting Populations and Ecological Processes

- Threats are largely unknown, but sapsuckers dependency on aspen and mature riparian woodland is cause for concern because of impacts on these habitats by land management activities throughout its range (NatureServe 2003).

- Loss of aspen stands and a decline in aspen regeneration has occurred throughout the mountain west due to fire suppression, conifer invasion, cutting, and development. For example aspen has declined 100 percent (about 1,800 acres) when comparing historical and current conditions in the Umatilla sub basin (NHI 2004). In addition, many of the aspen forest in the Blue Mountains are over 100 years old and decadent or declining in vigor. Lack of tree regeneration may lead to inevitable loss of large tees, which could result in significant declines in cavity –nesting (Dobkin et al. 1995) and affect the species in the long term.

- Grazing can have detrimental effects where the health and regeneration of aspen, cottonwood, and other preferred species is compromised. Studies of grazing impacts show mixed effects in the short term. In an Idaho cottonwood gallery forest where moderate to heavy grazing reduced understory shrub cover, Saab (1998) found no significant difference between grazed and unmanaged sites, although sapsucker abundances were slightly higher in unmanaged forest. On the other hand, in western Montana cottonwood/ponderosa pine riparian habitat, were significantly more abundant on lightly grazed sites than on heavily grazed sites, where ground cover, bush cover, mid-canopy cover, and number of small trees (less than 10 centimeter dbh) were significantly reduced in the heavily grazed sites (Mosconi and Hutto 1982 *in* NatureServe 2003). In California/Nevada aspen habitat, Page et al. (1978, cited in Saab et al. 1995) also observed a negative response to grazing.

Out-of-Subbasin Effects and Assumptions

No data could be found on the migration and wintering grounds of the red-naped sapsucker. It is a long distance migrant and as a result faces a complex set of potential effects during it annual cycle. Habitat loss or conversions could be occurring along its entire migration route and winter range.

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FERRUGINOUS HAWK
Buteo regalis

Original Species Account: obtained from NatureServe Explorer website at http://natureserve.org/explorer with supplements provided by Russ Morgan, Oregon Department of Fish and Wildlife.

Distribution
The species is found in U.S. States (AZ, CA, CO, ID, KS, MN, MT, ND, NE, NM, NN, NV, OK, OR, SD, TX, UT, WA, WY) and Canadian Provinces (AB, BC, MB, SK). In some jurisdictions, the statuses for common species have not been assessed. A species is not referenced in a jurisdiction if it is not known to breed in the jurisdiction or if it occurs only accidentally or casually in the jurisdiction. Thus, the species may occur in a jurisdiction as a seasonal non-breeding resident or as a migratory transient.

Global breeding ranges include eastern Washington, southern Alberta, southern Saskatchewan, extreme southwestern Manitoba (Bechard and Schmutz 1995), south to eastern Oregon, Nevada, northern Arizona, northern New Mexico, Texas panhandle, extreme western Oklahoma, and western Kansas. Recently discovered breeding in California (Small 1994). Historic breeding range in the southwestern U.S. apparently was much greater than at present (Hall et al. 1988). Two subpopulations are recognized (Bechard and Schmutz 1995); one to the east and another to the west of the Rocky Mountains.

Non-breeding ranges occur primarily in southwestern and south-central U.S. south to Baja California and central mainland of Mexico. In the U.S., in largest numbers occur in western Texas, eastern New Mexico, and western Oklahoma (Root 1988). The species winters locally in some more northerly breeding areas (Bechard and Schmutz 1995).

Between 1991 and 1993 a total of 28 active ferruginous hawk nests were known within the Umatilla subbasin (ODFW unpubl. data). These known nest sites were distributed into two distinct population areas; higher elevation grasslands/foothill canyonlands – 15 nests (where most nests were located in rock outcroppings and cliffs), and low elevation shrubsteppe/juniper savannah areas – 13 nests (juniper tree nests). While it is unknown the status of these historical nest sites today, it is known that a number of those “active” nest trees in the lower elevation portion of the basin have been lost by fire and human removal within the past 10 years (Russ Morgan personal communication).

Habitat
According to the ICBEMP terrestrial vertebrate habitat analyses, historical source habitats for ferruginous hawk occurred throughout all three ERUs within our planning unit (Wisdom et al. in press). Within this core of historical habitat, declines in source habitats were most evident for the Columbia Plateau; over 72% of the watersheds had moderate or strongly declining trends, and source habitat has been reduced from historical levels by 53%. Relatively stable trends are apparent for source habitats in the Great Basin and Owyhee Uplands (4% and 8% declines, respectively). Within the entire Interior Columbia Basin, over 54% of the watersheds show moderate or strongly declining trends in source habitats (Wisdom et al. in press).

Low elevation shrub-steppe and grasslands with scattered juniper trees are the habitat most threatened in the Umatilla/Willow subbasin. Conversion to agriculture, habitat loss from overgrazing, conversion of juniper savannah through fire suppression, and loss of isolated mature juniper trees by fire, cutting and trampling of roots by cattle seeking shade are four primary sources of loss (Altman and Holmes, 2000). Remaining core habitat strongholds within the subbasin are the Boardman Bombing Range (US Navy), Boardman Conservation Area (The Nature Conservancy and private), and the Horn Butte and Willow Creek area (BLM and private).

Palustrine habitat is riparian. Terrestrial habitat is cliff, desert, grassland/herbaceous and savanna. Open country, primarily prairies, plains and badlands; sagebrush, saltbush-greasewood shrubland, periphery of pinyon-juniper and other woodland, desert. In the southern Great Plains, common at black-tailed prairie dog colonies in winter (Schmutz and Fyfe 1987). They nest in tall trees or willows along streams or on steep slopes, in junipers (Utah), on cliff ledges, on river-cut banks, on hillsides, on power line towers, and sometimes on sloped ground on the plains or on mounds in open desert. Generally they avoid areas of intensive agriculture or human activity.

Hawks prefer open grasslands and shrub-steppe communities, using native and tame grasslands, pastures, hayland, cropland, and shrub-steppe (Stewart 1975, Woffinden 1975, Powers and Craig 1976, Fitzner et al. 1977, Blair 1978, Wakeley 1978, Lardy 1980, Schmidt 1981, Gilmer and Stewart 1983, Green and Morrison 1983, Konrad and Gilmer 1986, MacLaren et al. 1988, Palmer 1988, Roth and Marzluff 1989, Bechard et al. 1990, Black 1992, Niemuth 1992, Bechard and Schmutz 1995, Faanes and Lingle 1995, Houston 1995, Zelenak and Rotella 1997, Leary et al. 1998). Usually occupy rolling or rugged terrain (Blair 1978, Palmer 1988, Black 1992). High elevations, forest interiors, narrow canyons, and cliff areas are avoided (Janes 1985, Palmer 1988, Black 1992), as is parkland habitat in Canada (Schmutz 1991a).

Landscapes with moderate coverage (less than 50 percent) of cropland and hayland are used for nesting and foraging (Blair 1978; Wakeley 1978; Gilmer and Stewart 1983; Konrad and Gilmer 1986; Schmutz 1989, 1991a; Bechard et al. 1990; Faanes and Lingle 1995; Leary et al. 1998). In North Dakota, hayfields and native pastures were the habitats most often used by both fledglings and adults, whereas cultivated fields rarely were used (Konrad and Gilmer 1986). Fledglings in South Dakota hunted in an area where native hay recently had been cut (Blair 1978). When prey densities were low in big sagebrush (*Artemisia tridentata*)/grassland habitat, agricultural fields served as important foraging areas (Leary et al. 1998). Foraged extensively in alfalfa (*Medicago sativa*) and irrigated potato fields in Washington and in alfalfa fields in Idaho during the breeding season presumably because of high prey densities (Wakeley 1978, Leary et al. 1998).

Breeding

Home ranges are variable, ranging from about 0.5 to about 90 square kilometers; the latter figure refers to nests where birds commuted some distance to feeding grounds. A number of studies give mean home ranges on the order of 7 square kilometers, which equates to a circle with a diameter of about 3 kilometers; three times that home range gives a separation distance of about 10 kilometers. Home ranges: Ferruginous Hawk, mean 5.9 square kilometers in Utah (Smith and Murphy 1973); range 2.4 to 21.7 square kilometers, mean 7.0 square kilometers in Idaho (Olendorff 1993); mean 7.6 square kilometers in Idaho (McAnnis 1990); mean 90 square kilometers in Washington (Leary et al. 1998); Red-tailed Hawk, most forage within 3 kilometers of nest (Kochert 1986); mean spring and summer male home ranges 148 hectares (Petersen 1979); Hawaiian Hawk, 48 to 608 hectares (n = 16; Clarkson and Laniawe 2000); Zone-tailed Hawk, little information, apparent home range 1-2 kilometers/pair in west Texas (Johnson et al.

2000); White tailed Kite, rarely hunts more than 0.8 kilometers from nest (Hawbecker 1942); Prairie Falcon, 26 square kilometers in Wyoming (Craighead and Craighead 1956), 59 to 314 square kilometers (reported by Steenhof 1998); Aplomado Falcon, 2.6 to 9.0 square kilometers (n = 5, Hector 1988), 3.3 to 21.4 square kilometers (n = 10, Montoya et al. 1997).

Nest site fidelity is high in Zone-tailed Hawk; all seven west Texas nesting territories occupied in 1975 were reused in 1976 (Matteson and Riley 1981). Ferruginous Hawk: In California, dispersal distances from natal sites to subsequent breeding sites ranged from 0 to 18 kilometers, mean 8.8 kilometers (Woodbridge et al. 1995); in contrast, none of 697 nestlings in Saskatchewan returned to the study area; three were found 190 200 and 310 kilometers away (Houston and Schmutz 1995).

In nonbreeding class, evidence of recurring presence of wintering birds (including historical); and potential recurring presence at a given location, usually minimally a reliable observation of five birds (this can be reduced to one individual for rarer species). Occurrences should be locations where the species is resident for some time during the appropriate season; it is preferable to have observations documenting presence over at least 20 days annually. Be cautious about creating EOs for observations that may represent single events.
Separation distance is somewhat arbitrary; 10 kilometers can be used to define occurrences of manageable size for conservation purposes. However, occurrences defined primarily on the basis of areas supporting concentrations of foraging birds, rather than on the basis of distinct populations.

Nests

Nest site selection depends upon available substrates and surrounding land use. Ground nests typically are located far from human activities and on elevated landforms in large grassland areas (Lokemoen and Duebbert 1976, Blair 1978, Blair and Schitoskey 1982, Gilmer and Stewart 1983, Atkinson 1992, Black 1992). Lone or peripheral trees are preferred over densely wooded areas when trees are selected as the nesting substrate (Weston 1968, Lokemoen and Duebbert 1976, Gilmer and Stewart 1983, Woffinden and Murphy 1983, Palmer 1988, Bechard et al. 1990). Tree-nesting hawks seem to be less sensitive to surrounding land use, but they still avoid areas of intensive agriculture or high human disturbance (Gilmer and Stewart 1983; Schmutz 1984, 1987, 1991a; Bechard et al. 1990).

Foothill and canyon grasslands with rock outcroppings are, by their very nature, a more stable nesting habitat and exhibit little change in nest availability from year to year. Observations of old nest structures on rock outcroppings indicate that ferruginous hawks may use and maintain a number of different nest structures over time within a territory – often rotating the actual nesting site from year to year. Virtually all of this habitat type within the subbasin is privately owned and is used for cattle ranching and, to a lesser extent, farming. Clearly the largest threats to ferruginous hawks in this habitat are human disturbance to highly visible nest sites and grassland quality as it relates to prey availability. In 1993, a number of easily visible nests were destroyed by illegal killing of nesting adult birds in the Little Butter Creek area. In addition, grazing practices which remove most or all of the native bunchgrass cover (especially during drought years) can negatively affect nest success. Even so, from 1990 to 2004, the number of active nests in this habitat type appears to be relatively stable (Russ Morgan personal communication).

In eastern Colorado, nested more frequently in grassland areas than in cultivated areas (Olendorff 1973). In North Dakota, preferred to nest in areas dominated by pasture and hayland (Gilmer and

Stewart 1983, Gaines 1985). In southwestern Montana, sagebrush (*Artemisia*) and grasslands predominated within 100 meters of nests (Atkinson 1992). Ground nests in northern Montana were located in grass-dominated, rolling (more than 10 percent slope) rangeland; in general, cropland and areas with dense (more than 30 percent cover), tall (more than 15.24 centimeters) sagebrush were avoided (Black 1992). In western Kansas, most nests were surrounded by more than 50 percent rangeland and 25-50 percent cropland, although one pair incorporated more than 75 percent cropland in its territory (Roth and Marzluff 1989). The majority of nests (86 of 99) were not in direct view of black-tailed prairie dog (*Cynomys ludovicianus*) towns, although most nest sites were within 8 kilometers of towns (Roth and Marzluff 1989). In Utah, Idaho, Oregon, and California, preferred native grassland and shrubland habitats over cropland, and preferred areas with no perches (Janes 1985). In Washington, some nests occurred in agricultural fields, but most nests were in areas with higher percentages of grassland, shrubland, and western juniper (*Juniperus occidentalis*) (Bechard et al. 1990). Nest productivity in Idaho was greater in territories with higher amounts of crested wheatgrass (*Agropyron cristatum*) fields interspersed with desert shrub than in territories with monotypic stands of crested wheatgrass or shrubland, or with greater amounts of Utah juniper (*Juniperus osteosperma*), alfalfa, and cropland (Howard 1975).

In Alberta, however, cultivated areas (11-30 percent of 4,100 hectare plots) had higher nesting densities than grassland areas with 0-11 percent cultivation (Schmutz 1989). In cultivated areas (20 percent) in northcentral Montana, nests closer to cultivated fields and roads were more successful, presumably because of higher prey densities associated with edge habitats (Zelenak and Rotella 1997). The numbers of fledglings produced in unfragmented rangeland versus a mixture of rangeland and cropland were not significantly different in Nebraska (Podany 1996).

The slope, height, and exposure of nests were mostly similar across the species' range. The mean height of ground nests (on buttes or hills) above the surrounding prairie in South Dakota was less than 10 meters, and nests were oriented toward the south and west, providing access to prevailing winds from the south and west (Blair 1978). Lokemoen and Duebbert (1976) found ground nests in South Dakota were all oriented toward the west. Nests in southwestern Montana were significantly oriented toward the south (Atkinson 1992). Nests on rock outcrops in Montana were built on slopes averaging 62.8 percent and were found on the upper 35 percent of the slope (Atkinson 1992). Ground nests in northern Montana were located either on the top of a small rise or on slopes ranging from 10 to 50 percent (Black 1992). Average height of ground nests below the highest surrounding topographic feature was 10 meters, whereas average height of ground nest sites above the valley floor was 10.4 meters, indicating that nests were placed at mid-elevation sites within the immediate topography (Black 1992). Nests in Wyoming were built on a mean slope of 14.26 degrees, and the mean height of nests was 4.55 meters (MacLaren et al. 1988).

In southeastern Washington, 86 percent of nests on outcrops and in western junipers were located less than 10 meters from the ground and had southern or western exposures (Bechard et al. 1990). In Oregon shrub-steppe, nests were in relatively short western juniper trees, were less than 10 meters from the ground, and had large support branches (Green and Morrison 1983). In Washington, Idaho, and Utah, the majority of nests also were less than 10 meters from the ground in western juniper and Utah juniper trees (Woffinden 1975, Fitzner et al. 1977, Woffinden and Murphy 1983). Howard (1975) and Howard and Wolfe (1976) also found Utah juniper trees were important nest substrates in southern Idaho and northern Utah. In Utah, nests were built 2-3 meters from the ground, were most commonly located on the sides or summits of hills, and often had southern or eastern exposures (Weston 1968). Woffinden (1975) found that the majority of nests in Utah were on slopes ranging from 15 to 80 degrees with a mean of 42.5 degrees.

Habitat Loss

Some habitat has been lost due to agricultural development. Schmutz and Schmutz (1980) reported that habitat in the breeding range in Canada has been severely depleted by agriculture, disturbance, and forest invasion (see also Jensen 1995), though recent trends suggest relative stability (Schmutz 1995). Loss of grassland is not regarded as an immediate threat (USFWS 1992), but is likely a long-term threat (Olendorff 1993). Ability of native grasslands and shrublands to support viable populations may be compromised by the invasion of exotic annuals, especially cheatgrass (*Bromus tectorum*) and Russian thistle (*Salsola iberica*). However, conversion of large areas of dense shrublands to grasslands may locally benefit Ferruginous Hawks.

Ferruginous Hawks are easily disturbed by humans during the breeding season (Olendorff 1973, Gilmer and Stewart 1983, Schmutz 1984, White and Thurow 1985, Bechard et al. 1990). Abandonment of nests occurs particularly in the early stages of nesting (Davy 1930, Weston 1968, Fitzner et al. 1977, Gilmer and Stewart 1983, White and Thurow 1985). In eastern Colorado, nests in remote locations had greater productivity compared to more accessible nests (Olendorff 1973). In South Dakota, the probability of fledging young was 11.4 percent greater in more remote nests than in nests within 2.47 kilometers of occupied buildings (Blair 1978). In North Dakota, avoided cropland and nesting within 0.7 kilometers of occupied buildings (Gaines 1985). In Alberta, rarely nested within 0.5 kilometers of farmyards (Schmutz 1984). In other instances, more tolerant of human disturbance. Nesting has occurred near active railroads and gravel roads (Rolfe 1896, Gilmer and Stewart 1983, MacLaren et al. 1988). Sensitivity to disturbance may be heightened in years of low prey abundance (White and Thurow 1985). Shooting may also be a threat, especially on the wintering grounds (Harmata 1981, Gilmer et al. 1985). Poisoning of prey species may be a threat both directly to hawks eating poisoned animals and indirectly through reduction of prey base, especially at prey concentration areas such as prairie dog colonies.

Diet

Both the immature and adult hawks are carnivorous. Mammals are the primary prey during the breeding season, although birds, amphibians, reptiles, and insects also are taken (Weston 1968, Howard 1975, Fitzner et al. 1977, Blair 1978, Smith and Murphy 1978, Gilmer and Stewart 1983, Palmer 1988, De Smet and Conrad 1991, Atkinson 1992). Primary prey in central grasslands are ground squirrels (*Spermophilus spp*.), followed by pocket gophers (*Thomomys spp*.) and white-tailed jackrabbits (*Lepus townsendii*) (Bechard and Schmutz 1995). Primary prey in western shrub-steppe are jackrabbits (*Lepus spp*.), followed by ground squirrels and pocket gophers (Smith and Murphy 1978, Bechard and Schmutz 1995). White-tailed (*Cynomys leucurus*) and black-tailed prairie dogs(*Cynomys ludovicianus*)also serve as prey items (Powers and Craig 1976, MacLaren et al. 1988). In Oregon, Janes (1985) found that the highest abundance of major prey species (white-tailed jackrabbits, Townsend's ground squirrels [*Spermophilus townsendii*], and northern pocket gophers [*Thomomys talpoides*]) occurred in native grasslands. Foraging range is variable, with three kilometers the mean diameter in several species. Hunting occurs most frequently near sunrise and sunset (Evans 1982).

Vulnerability of prey also is an important factor in habitat suitability, such that Ferruginous Hawks avoid dense vegetation that reduces their ability to see prey (Howard and Wolfe 1976, Wakeley 1978, Schmutz 1987). Prey vulnerability decreases where taller small-grain crops replace shorter grasses (Houston and Bechard 1984). Intensive agricultural practices, such as annual plowing and biennial fallowing, exclude many prey species (Wakeley 1978, Houston and Bechard 1984). In Alberta, prey abundance increases as the area of cultivation increases up to 30 percent, but abundance is reduced where agriculture is extensive, e.g., more than 30 percent (Schmutz 1989).

Global Short Term Trend

Most recent global population estimate is 5,842-11,330 compiled by Olendorff (1993). However, Schmutz et al. (1992) estimated 14,000 for the Great Plains alone. Estimated population in Canada in the early 1990s was 2000-4000 breeding pairs (Schmutz, 1994 COSEWIC report, cited by Jensen 1995). Between year movements of population centers and individuals makes estimation of actual abundance difficult.

Local declines have been noted (e.g., Woffinden and Murphy 1989), but a widespread decline was not evident as of the early-1990s (USFWS 1992, Olendorff 1993). North American Breeding Bird Survey (BBS) data for the U.S. and Canada indicate a 13.5 percent increase from 1988 to 1989 and an average annual 0.5 percent increase for 1966-1989 (Droege and Sauer 1990). Wintering data from Christmas Bird Counts also indicate an increase in numbers from 1952-1984 (USFWS 1992). Schmutz (1995) reported that the range in Canada has been reduced by half, and that habitat within the range has been severely depleted and total numbers reduced by about 95 percent. Kirk et al. (1995) indicated that populations in Canada apparently are stable in available habitat. Jensen (1995) reported a recent range re-expansion in south-central Canada. Historically, very abundant in eastern Montana but numbers were lowered by the early 1900's (Allen 1874, Cameron 1914).

Global Protection

There is one protected at Kevin Rim by BLM as an ACEC (Area of Critical Environmental Concern). Eight Key Raptor Areas are managed by BLM in Montana (Centennial Valley, Lima Foothills, Madison River, Sweetwater Breaks, Kevin Rim, Rocky Mountain East Front, Rock Creek-Thoeny Area, and Lone Tree Management Area).

Global protection needs cover extensive areas of suitable habitat throughout the breeding and wintering range, including the concentrated prey sources such as prairie dog towns.

Economic Attributes

Management Summary

Stewardship Overview

Conversion of grasslands to intensive cultivation has reduced the amount of preferred habitat that is available and has been implicated in the population decline of the species in some areas (Schmutz 1984, Faanes and Lingle 1995). Agricultural development has restricted the species to areas of greater topographic relief or other areas unsuitable for agriculture (Stewart 1975). Keys to management are providing suitable nest sites, protecting active nest areas from disturbance, and improving habitat for prey. Isolated trees and stringers should be protected from livestock in nesting habitat. Prescribed burning may increase habitat suitability in shrub-dominated areas. Practices that increase exotic plant species number or dominance should be discouraged.

Artificial nests have been used to increase number of nesting pairs in areas where suitable sites are scarce (Schmutz 1984).

Preserve Selection & Design Considerations
Land Protection
Maintain ownership of public lands that have substantial numbers of hawks (Olendorff 1993). Protect large tracts of native prairie from conversion to monotypic stands of grass or other types of agriculture (Howard and Wolfe 1976, Lardy 1980, Schmutz 1991a, Bechard and Schmutz 1995). Avoid seeding of exotic grasses and cultivating of habitat, where possible (Janes 1985). Leave scattered islands of shrubby vegetation in crested wheatgrass fields so that the islands make up a minimum of 20 percent of the total area (Howard and Wolfe 1976).

Management Requirements
Prey Consideration
Increase grassland area to increase Richardson's ground squirrel (*Spermophilus richardsonii*) abundance in Canada (Houston and Bechard 1984). Improve prey habitat by providing native shrub vegetation and increasing edge (Howard and Wolfe 1976, Bechard and Schmutz 1995). If brush is chained, windrow it to provide cover for prey (Olendorff 1993). When converting land from sagebrush steppe to herbaceous grassland (e.g., to crested wheatgrass), create a mosaic of treated (chained or disced) and untreated areas (Howard and Wolfe 1976). To attract small rodents, maintain or restore sagebrush-grass rangeland, removing pinyon pine (*Pinus edulus*)/Utah juniper stands (Howard and Wolfe 1976). If it is necessary to control lagomorph or rodent populations, try to lower the peaks of cyclic highs rather than completely exterminating them (Olendorff 1993).

Reduce Disturbance
Do not disturb nest sites from 15 March to 15 July (Howard and Wolfe 1976, Bechard and Schmutz 1995). Close public areas near nest sites to recreation during the breeding season (Lardy 1980) and close public land to firearms where dense populations of Ferruginous Hawks are particularly susceptible to shooting (Olendorff 1993). Establish buffer zones around nest sites and delay energy development until 45 days after fledging (Konrad and Gilmer 1986). White and Thurow (1985) recommended creating a buffer zone of 0.25 kilometers around nest sites. Atkinson (1992) suggested that a minimum distance of 0.45 kilometers be maintained from the nest. Olendorff (1993) suggested buffer zones of 0.25 kilometers for brief disturbances, 0.5 kilometers for intermittent activities, 0.8 kilometers for prolonged activities, and more than 1.0 kilometer for construction or similar activities. Provide information to ranchers, seismic crews, prospectors, and others to avoid disturbance to the nest (Atkinson 1992). Conduct treatments, e.g., chaining, discing, plowing, or burning, during the non-nesting season to avoid direct impacts to the hawks and their prey species during the reproductive season (Olendorff 1993). Generally, avoid treatments between 1 March and 1 August each year, especially during the incubation period when hawks are more prone to abandon nests if disturbed. Mitigate development impacts from mining, pipeline construction, and urbanization (Bechard and Schmutz 1995). Encourage rest-rotation or deferred-rotation grazing systems (Olendorff 1993). Delay grazing to allow for the completion of incubation (Atkinson 1992).

Nest Structures
Enhance, protect, and create nest substrates through fencing of nest trees, supporting heavy tree nests that are at risk of toppling, and building artificial nesting structures where nest sites are otherwise lacking (Olendorff 1973, Smith and Murphy 1978, Houston 1985, Bechard and

Schmutz 1995, Leary et al. 1998). Other successful nest structure management techniques are to remove some of the previous year's nesting material to reduce the chance of toppling, realign the nest over a vertical axis, widen the base of the nest, reinforce the base of the nest using wire netting or other materials, move the nest to a safer location, or provide protection from predators by nailing tin sheathing around the tree base (Craig and Anderson 1979). In converting tree communities to grassland, provide nest sites by leaving individual trees, a mosaic of stands of trees, or a thin scattering of trees (Olendorff 1993). Leave poles and cross-arms of unused electrical lines for hunting perches (Olendorff 1993).

Grazing provides benefits by reducing vegetative cover and making prey more visible (Wakeley 1978, Konrad and Gilmer 1986). Kantrud and Kologiski (1982) found highest densities of Ferruginous Hawks in heavily grazed areas in the northern Great Plains. These areas provided a combination of grazing and soil type (typic borolls) that resulted in abundant prey populations (Kantrud and Kologiski 1982). In South Dakota, preferentially placed ground nests in lightly grazed pasture or idle areas (Lokemoen and Duebbert 1976, Blair 1978, Blair and Schitoskey 1982). In Saskatchewan, preferred grassland habitat exists in large blocks of government pastures located along the Montana and Alberta borders (Houston and Bechard 1984). These blocks of habitat are the only remaining areas with stable populations in Saskatchewan (Houston and Bechard 1984). Livestock, however, can weaken nest trees by excessive rubbing or trampling (Houston 1982, Olendorff 1993). Bock et al. (1993) suggested negative response to grazing in shrub-steppe habitats, based on the ground cover requirements of their prey.

Biological Research Needs

Understanding of the wintering ecology, dispersal, site fidelity (breeding and winter), and possible differences between subpopulations east and west of the Rocky Mountains is needed for conservation planning. Other research needs include basic biology, color polymorphism, nomadism, and relationship between populations of hawks and prey, especially cyclic species. The effects of management actions and strategies on Ferruginous hawks is also poorly known (Bechard and Schmutz 1995).

Reproduction Comments: Occur on breeding areas from late February through early October (Weston 1968, Olendorff 1973, Maher 1974, Blair 1978, Smith and Murphy 1978, Gilmer and Stewart 1983, Schmutz and Fyfe 1987, Palmer 1988, Bechard and Schmutz 1995). See Palmer (1988) and Hall et al. (1988) for egg dates in different areas. Clutch size usually is two to four. Incubation lasts about 32-33 days, mostly by female; male provides food. Young fledge in 35-50 days (males before females), depend on parents for several weeks more. No evidence that yearlings breed. Renesting within the same year is rare (Woffinden 1975, Palmer 1988) even when clutch is lost. Territory and nest site reoccupancy is common and one of several nests within a territory may be used in alternate years (Davy 1930, Weston 1968, Olendorff 1973, Blair 1978, Smith and Murphy 1978, Palmer 1988, Roth and Marzluff 1989, Schmutz 1991b, Atkinson 1992, Houston 1995). Mate fidelity also is common. (Schmutz 1991b). Clutch size, fledging rate, and/or breeding density tend to vary with prey (especially jackrabbit [*Lepus spp*.] or ground squirrel [*Spermophilus spp*.]) availability.

Ecology Comments

Density and productivity are closely associated with cycles of prey abundance (Woffinden 1975; Powers and Craig 1976; Smith and Murphy 1978, Smith et al. 1981; Gilmer and Stewart 1983; Houston and Bechard 1984; White and Thurow 1985; Palmer 1988; Schmutz 1989, 1991a; Schmutz and Hungle 1989; Bechard and Schmutz 1995). Estimates of home range size vary from

3.14 to 8.09 square kilometers in the Columbia River Basin and Great Basin regions of the western U.S. (Janes 1985). The average home range was 90.3 square kilometers in Washington, and the variability in home range was significantly related to distance from the nest to the nearest irrigated agricultural field (Leary et al. 1998). One male that nested closest to the surrounding agricultural fields had the smallest home range, whereas another male nesting farthest from the agricultural fields had the largest home range. In Utah, mean home range recorded of 5.9 square kilometers (Smith and Murphy 1973). An area of up to 21.7 square kilometers may be required by one pair for hunting in Idaho (Wakeley 1978). Up to 8-10 nests per 100 square kilomters if local conditions are favorable (see Palmer [1988] for density data in several areas). In 11 study areas, mean nearest neighbor distance was 3.4 kilometers (range 0.8-7.2); in six study areas the mean home range size was 7.0 square kilometers (range 3.4-21.7) (Olendorff 1993). Recent studies in Idaho (McAnnis 1990) and Washington (Leary 1996) found average home ranges of 7.6 square kilometers (minimum convex polygon)/19.4 square kilometers (95 percent harmonic mean) and 70 square kilometers (95 percent minimum convex polygon)/31 square kilometers (85 percent adaptive kernel), respectively.

Mobility and Migration

Hawks arrive in northern breeding range (South Dakota) by March-early April, in Utah and Colorado mostly in late February-early March; yearlings arrive later. Adults depart northern end of breeding range by late October; young depart in August. Wintering areas of grassland and desert shrub breeders are mainly separate. (Schmutz and Fyfe 1987). Alberta populations winter mainly in Texas. In southern breeding range, may be short-distance migrant or possibly sedentary (Palmer 1988).

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GRASSHOPPER SPARROW
Ammodramus savannarum perpallidus

Original Species Account Authors: Paul Ashley and Stacy Stoval, as appeared in the Southeast Washington Ecoregional Assessment, January 2004

Introduction
Grassland ecosystems that were prominent in the Columbia Basin have suffered the greatest losses of any habitats in the Columbia Plateau (Kagan et al.1999). The Palouse Prairie has been identified as the most endangered ecosystem in the United States (Noss *et al.* 1995). Land conversion and livestock grazing coupled with the rapid spread of cheatgrass (*Bromus tectorum*) and a resulting change in the natural fire regime has effectively altered much of the grassland habitats to the effect that it is difficult to find stands which are still in relatively natural condition (Altman and Holmes 2000).

As a result, many of these steppe, grassland, species are declining in our area. BBS data (Robbins et al. 1986) have shown a decreasing long term trend for the grasshopper sparrow (1966-1998) (Sauer *et al.* 1999). Throughout the U.S., this sparrow has experienced population declines throughout most of its breeding range (Brauning 1992, Brewer *et al.* 1991, Garrett and Dunn 1981). In 1996, Vickery (1996) reported that grasshopper sparrow populations have declined by 69% across the U.S. since the late 1960s. In Washington, the grasshopper sparrow is considered a State Candidate species (http://wdfw.wa.gov/wlm/diversity/soc/candidat.htm). In Oregon it is considered as a naturally rare, vulnerable species, and a state Heritage program status as imperiled.

Focal Species Life History, Key Environmental Correlates, and Habitat Requirements
Life History
Diet
Grasshopper sparrows are active ground or low shrub searchers. Vickery (1996) states that exposed bare ground is the critical microhabitat type for effective foraging. Bent (1968) observed that grasshopper sparrows search for prey on the ground, in low foliage within relatively dense grasslands, and sometimes scratch in the litter.

They eat mostly insects, primarily grasshoppers, but also other invertebrates and seeds. In one study, grasshoppers formed 23% of the grasshopper sparrows' diet during 8 months of the year; 60% of their diet in Jan., and 37% from May to Aug. From Feb. to Oct., 63% of food taken was animals, 37% vegetable. Insects comprised 57% total food; spiders, myriapods, snails and earthworms made up 6%. Of the insects, "harmful" beetles (click beetles (*Clateridae*), weevils (Sitones *et. al*), and smaller leaf beetles (*Systens spp*.) made up 8%, caterpillars (cutworms) made up 14%. Vegetable matter eaten included waste grain, grass, weed and sedge seeds (Smith 1968, Terres 1980).

Their diet varies by season. Spring diet 60% invertebrates, 40% seeds (n=28); summer diet 61% invertebrates, 39% seeds (n=100); fall diet 29% invertebrates, 71% seeds (n=17), and no data for winter (Martin *et al.* 1951 in Vickery 1996).

Reproduction
Grasshopper sparrows are monogamous throughout the breeding season (Ehrlich 1988).

Grasshopper sparrows nest in semi-colonial groups of 3-12 pairs (Ehrlich 1988). Smith (1963) recorded breeding densities that ranged from 0.12 to 0.74 males per hectare in Pennsylvania and Collier (1994) observed breeding densities of 0.55 males per hectare in California.
Clutch size ranges from 2 to 6, with 4 most frequently (Smith 1963). The female alone has a brood patch and incubates eggs (Smith 1963, Ehrlich 1988, Harrison 1975). During incubation, the male defends the pair’s territory (Smith 1963).

Incubation period is from 11 to 13 days (Smith 1963, Ehrlich 1988, Harrison 1975), with a nestling period of 6 to 9 days after hatching (Harrison 1975, Hill 1976, Kaspari and O’Leary 1988). Hatchlings are blind and covered with grayish-brown down (Smith 1968).

Throughout most of their range, grasshopper sparrows can produce two broods, one in late May and a second in early July (George 1952, Smith 1968, Vickery 1996). However, in the northern part of its range, one brood is probably most common (Vickery et al. 1992, Wiens 1969). grasshopper sparrows frequently renest after nest failure, and if unsuccessful in previous attempts, may renest 3-4 times during the breeding season (Vickery 1996).

After the young hatch, both parents share the responsibilities of tending the hatchlings and seem more concerned over human intrusion into their territory than before (Smith 1963). Kaspari and O’Leary (1988) observed cooperative breeding by non-parental attendants ("defined as birds bringing food to the nest"). Unrelated juveniles and adults from adjacent territories made 9-50% of the provisioning visits to four of twenty-three nests. Parents facilitated visits from non-parental attendants by moving off the nest yet unrelated birds that did not bring food to the nest were vigorously chased away. Kaspari and O’Leary (1988) suggested that non-parental attendants, rare among the population observed, are likely cases of "misdirected parental care".

Nesting
Grasshopper sparrows arrive on the breeding grounds in mid-April and depart for the wintering grounds in mid-September (George 1952, Bent 1968, Smith 1968, Harrison 1975, Stewart 1975, Laubach 1984, Vickery 1996). In Saskatchewan and Manitoba, they arrive later (mid-May) and leave earlier (August) (Knapton 1979). Grasshopper sparrows may be site faithful (Skipper 1998).

With few exceptions, nests are built on the ground, near a clump of grass or base of a shrub, "domed" with overhanging vegetation (Vickery 1996). Female grasshopper sparrows build a cup nest in two or three days time. Domed with overhanging grasses and accessed from one side, the rim of the nest is flush with the ground; the slight depression inside fashioned such that the female’s back is nearly flush with the ground while brooding (Dixon 1916, Pemberton 1917, Harrison 1975, Ehrlich 1988, and Vickery 1996).

Male grasshopper sparrows establish territories promptly upon arrival to the breeding grounds and rigidly maintain them until the young hatch. Territorial defense then declines and considerable movement across territory boundaries may occur. It appears that fledglings frequently flutter into adjoining territories and the parent birds follow in answer to the feeding call. A sharp increase in territorial behavior is exhibited during the two or three days prior to re-nesting (Smith 1963). Collier (1994 in Vickery 1996) observed grasshopper sparrow territory sizes of 0.37   0.16 (SD) ha (n=41) in southern California. In other states, territories have been observed to range in size from 1.4 ha (n=6) in Michigan (Kendeigh 1941) to 0.19   0.13 (SD) ha (n=20: Piehler 1987) in western Pennsylvania.

Although average territory size for grasshopper sparrows is small (<2 ha) (George 1952, Wiens 1969, 1970, Ducey and Miller 1980, Laubach 1984, Delisle 1995), grasshopper sparrows are area sensitive, preferring large grassland areas over small areas (Herkert 1994a,b, Vickery et al. 1994, Helzer 1996). In Illinois, the minimum area on which grasshopper sparrows were found was 10-30 ha (Herkert 1991), and the minimum area needed to support a breeding population may be >30 ha (Herkert 1994b). In Nebraska, the minimum area in which grasshopper sparrows were found was 8-12 ha, with a perimeter-area ratio of 0.018 (Helzer 1996, Helzer and Jelinski 1999). Occurrence of grasshopper sparrows was positively correlated with patch area and inversely correlated with perimeter-area ratio (Helzer and Jelinski 1999).

Migration
In spring, the grasshopper sparrow is a notably late migrant, arriving in southern B.C. in early to late May (Vickery 1996). Grasshopper sparrows arrive in Colorado in mid May and remain through September. They initiate nesting in early June, and most young fledge by the end of July. They winter across the southern tier of states, south into Central America.

This species generally migrates at night, sometimes continuing into morning. Mechanisms surrounding migration are not known but probably involve similar mechanisms as in savannah Sparrow, which include magnetic, stellar, and solar compasses (Moore 1980, Able and Able 1990a, b). While in migration the grasshopper sparrow does not form large conspecific flocks; individuals are found in mixed-species flocks with other sparrows and appear to migrate in small numbers, traveling more as individuals (Vickery 1996).

Data regarding the movements of grasshopper sparrows outside of the breeding season is scarce due to their normally secretive nature (Zeiner et al.1990). Although diurnally active, grasshopper sparrows are easily overlooked as "they seldom fly, preferring to run along the ground between and beneath tufts of grass" (Pemberton 1917). Because of their secretive nature the northern limits of their winter range is poorly known. Migratory individuals have been recorded casually south to w. Panama (Ridgely and Gwynne 1989) and (in winter) north to Maine (PDV), New Brunswick, Minnesota (Eckert 1990), and w. Oregon (Vickery 1996).

Mortality
Nest predators cited include: Raccoons (*Procyon lotor*), Red Fox (*Vulpes vulpes*), Northern Black Racers (*Coluber constrictor constrictor*), Blue Jays (*Cyanocitta cristata*), and Common Crows (*Corvus brachyrhynchos*) (Johnson and Temple 1990, Wray et. al 1982). Loggerhead Shrikes (*Lanius ludovicianus*) commonly take grasshopper sparrows as prey in Oklahoma and Florida (Stewart 1990, Vickery 1996). Many other species, especially those not dependent upon sight to find nests, are likely to be predators. Seasonal flooding in some areas may be a source of mortality during the nesting season (Vickery 1996).

Mowing and haying operations be the source of mortality for grasshopper sparrows directly and indirectly. Haying may reduce height and cover of herbaceous vegetation, destroy active nests, kill nestlings and fledglings, cause nest abandonment, and increase nest exposure and predation levels (Bollinger *et al.* 1990).

Habitat Requirements
Grasshopper sparrows prefer grasslands of intermediate height and are often associated with clumped vegetation interspersed with patches of bare ground (Bent 1968, Blankespoor 1980,

Vickery 1996). Other habitat requirements include moderately deep litter and sparse coverage of woody vegetation (Smith 1963; Bent 1968; Wiens 1969, 1970; Kahl et al. 1985; Arnold and Higgins 1986). In east central Oregon grasshopper sparrows occupied relatively undisturbed native bunchgrass communities dominated by *Agropyron spicatum* and/or *Festuca idahoensis*, particularly north-facing slopes on the Boardman Bombing Range, Columbia Basin (Holmes and Geupel 1998). Vander Haegen *et al.* (2000) found no significant relationship with vegetation type (i.e., shrubs, perennial grasses, or annual grasses), but did find one with the percent cover perennial grass.

In portions of Colorado, Kansas, Montana, Nebraska, Oklahoma, South Dakota, Texas, Wisconsin, and Wyoming, abundance of grasshopper sparrows was positively correlated with percent grass cover, percent litter cover, total number of vertical vegetation hits, effective vegetation height, and litter depth; abundance was negatively correlated with percent bare ground, amount of variation in litter depth, amount of variation in forb or shrub height, and the amount of variation in forb and shrub heights (Rotenberry and Wiens 1980).

Grasshopper sparrows have also been found breeding in Conservation Reserve Program (CRP) fields, pasture, hayland, airports, and reclaimed surface mines (Wiens 1970, 1973; Harrison 1974; Ducey and Miller 1980; Whitmore 1980; Kantrud 1981; Renken 1983; Laubach 1984; Renken and Dinsmore 1987; Bollinger 1988; Frawley and Best 1991; Johnson and Schwartz 1993; Klute 1994; Berthelsen and Smith 1995; Hull et al. 1996; Patterson and Best 1996; Delisle and Savidge 1997; Prescott 1997; Koford 1999; Jensen 1999; Horn and Koford 2000). In Alberta, Manitoba, and Saskatchewan, grasshopper sparrows are more common in grasslands enrolled in the Permanent Cover Program (PCP) than in cropland (McMaster and Davis 1998). PCP was a Canadian program that paid farmers to seed highly erodible land to perennial cover; it differed from CRP in that haying and grazing were allowed annually in PCP.

Grasshopper sparrows occasionally inhabit cropland, such as corn and oats, but at a fraction of the densities found in grassland habitats (Smith 1963, Smith 1968, Ducey and Miller 1980, Basore et al. 1986, Faanes and Lingle 1995, Best *et al.* 1997).

Grasshopper sparrows are also included as members of shrub-steppe communities, occupying the steppe habitats having the habitat features shown in Table 1 (Altman and Holmes 2000).

Table 1. Key habitat relationships required for breeding grasshopper sparrows (Altman and Holmes 2000).

Conservation Focus	Key Habitat Relationships			
	Vegetative Composition	Vegetation Structure	Landscape/ Patch Size	Special Considerations
native bunchgrass cover	native bunchgrasses	bunchgrass cover >15% and >60% total grass cover; bunchgrass >25 cm tall; shrub cover <10%	>40 ha (100 ac)	larger tracts better; exotic grass detrimental; vulnerable in agricultural habitats from mowing, spraying, etc.

Focal Species Population and Distribution
Population
Historic
According to the ICBEMP terrestrial vertebrate habitat analyses, historical source habitats for grasshopper sparrow within our planning unit occurred primarily along the eastern portions of the Columbia Plateau Ecological Reporting Unit (ERU) and the northern portion of the Owyhee Uplands ERU with a small amount in the northern portion of the Great Basin (Wisdom et al. 2000). Within this core of historical habitat, the current amount of source habitat has been reduced dramatically from historical levels by 91% in the Columbia Plateau and 85% in the Owyhee Uplands. Within the entire Interior Columbia Basin, overall decline in source habitats for this species (71%) was third greatest among 91 species of vertebrates analyzed (Wisdom *et al.* 2000).

Wing (1941) described the grasshopper sparrow as occupies the edge between the *Agropyron-Poa* type and the *Festuca-Agropyron* type. Jewett *et al.* (1953) gave its distribution in summer as north to Sprague, east to Pullman, south to Anatone and Prescott, and west to Toppenish.

Current
No data are available

Distribution
Grasshopper sparrows are found from North to South America, Ecuador, and in the West Indies

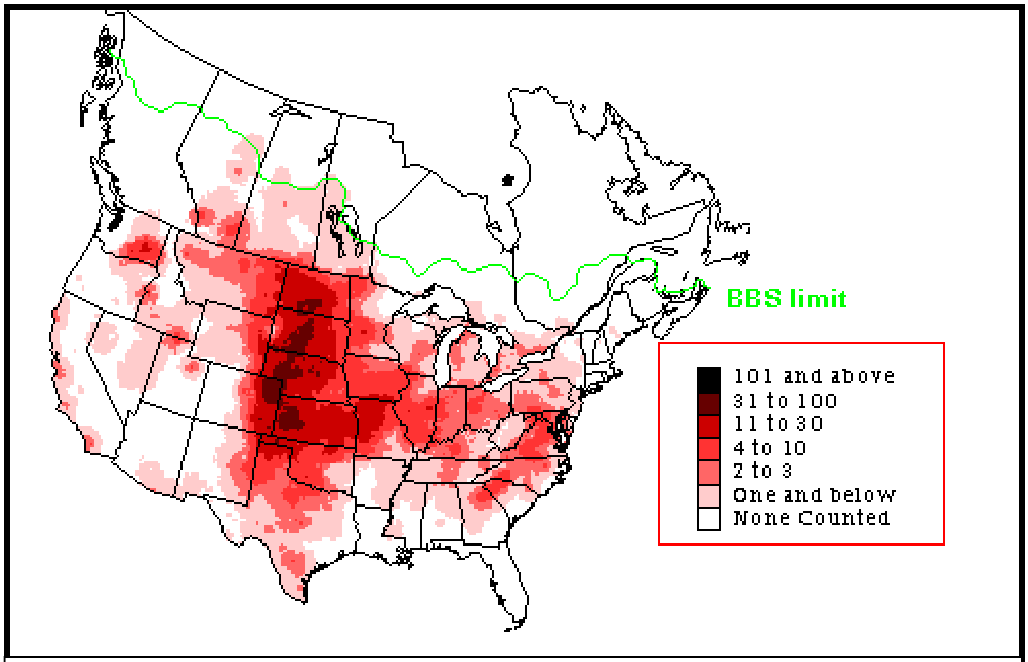

Figure 1. Breeding Range and Abudance of grasshopper sparrow in the U.S. based on Breeding Bird Survey data 1985-2001. Scale represents average number of individuals detected per route per year (Sauer 2003).

(Vickery 1996, AOU 1957). They are common breeders throughout much of the continental United States, ranging from southern Canada south to Florida, Texas, and California. Additional populations are locally distributed from Mexico to Colombia and in the West Indies (Delany et al. 1985, Delany 1996a, Vickery 1996).

The subspecies breeding in eastern Washington is *Ammodramus savannarum perpallidus* (Coues) which breeds from northwest California, where it is uncommon, into eastern Washington, northeast and southwest Oregon, where it is rare and local, into southeast B.C., where it is considered endangered, east into Nevada, Utah, Colorado, Oklahoma, Texas, and possibly to Illinois and Indiana (Vickery 1996).

Historic

Larrison (1981) called it a local irregular summer resident and/or migrant mostly through the arid interior of the Northwest and rare west of the Cascades in southwestern B.C. and Oregon. In Idaho, it was considered an uncommon irregular summer resident and migrant in the northern portion (Larrison 1981).

Jewett *et al.* (1953) classified the grasshopper sparrow as a rare summer resident between May and probably August or September locally in the bunch-grass associations of the lower Transition Zone of eastern Washington, occurring locally in the Upper Sonoran also.

Current

Grasshopper sparrows have a spotty distribution at best across eastern Washington. Over the

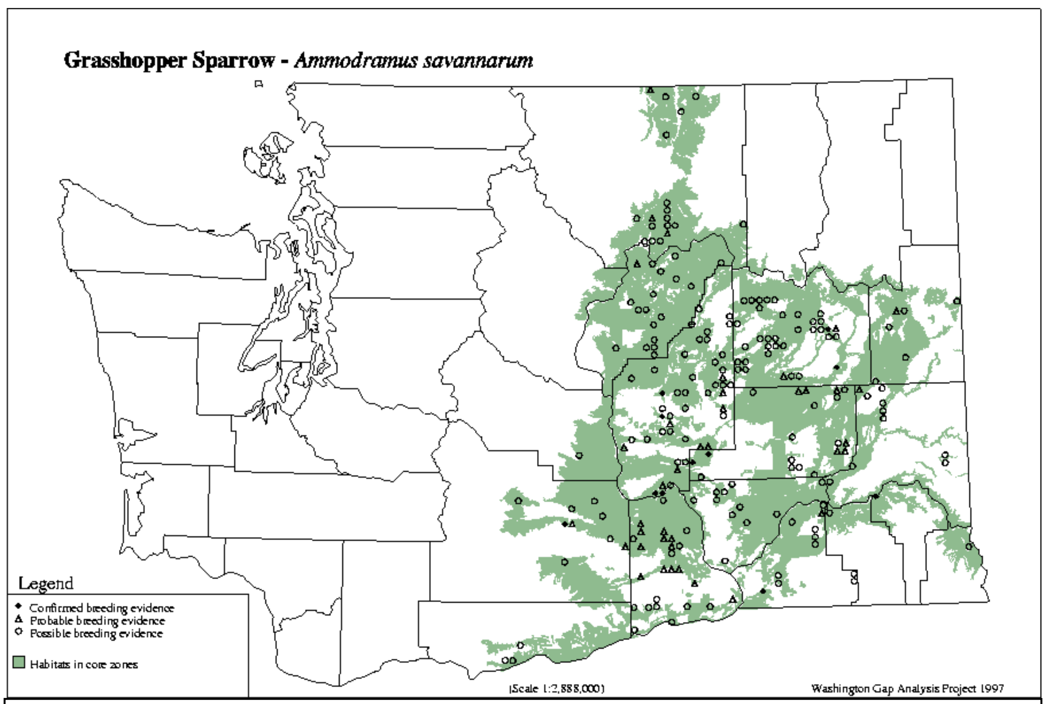

Figure 1. Current distribution of grasshopper sparrow in Washington from GAP analysis (Smith et al. 1997).

years they have been found in various locales including CRP. They appear to utilize CRP on a consistent basis in southeast Washington (Mike Denny pers. Comm).

Focal Species Status and Abundance Trends
Status
No data are available.

Trends
Throughout the U.S., this sparrow has experienced population declines throughout most of its breeding range (Brauning 1992, Brewer *et al.* 1991, Garrett and Dunn 1981). In 1996, Vickery (1996) reported that grasshopper sparrow populations have declined by 69% across the U.S. since the late 1960s.

Approximately 6 million hectares of shrub-steppe have been converted to wheat fields, row crops, and orchards in the interior Columbia Basin (Quigley and Arbelbide 1997). In Washington over 50% of historic shrub-steppe has been converted to agriculture (Dobler *et al*. 1996).

Table 2. Trends for grasshopper sparrow from BBS data 1980-2002 (Sauer *et al.* 2003).

State	1996- 2002 Trend	1980-2002 Trend
Washington	-4.9	-3.0
Idaho	-7.4	-10.7
Oregon	-4.4	-1.6
Intermountain Grassland	-13.0	-12.4

Accordingly, Breeding Bird Survey data show long term declines from 1980 through 2002 of –3.0, -1.6 and –10.7 for Washington, Oregon and Idaho, respectively (see Table 2) (see http://www.mbr-pwrc.usgs.gov/cgi-bin/atlasa02.pl?05460 for this data online). The entire Intermountain Grassland area shows large decrease of –12.4 over this same time period.

Washington, Oregon and the entire Intermountain Grassland area show an increasing negative trend when looking at the more recent time period 1996-2002 time period indicating the populations have increase even more over this time period (Sauer et al. 2003).

Factors Affecting Focal Species Population Status
Key Factors Inhibiting Populations and Ecological Processes

Habitat Loss and Fragmentation
The principal post-settlement conservation issues affecting bird populations include: habitat loss and fragmentation resulting from conversion to agriculture; and habitat degradation and alteration from livestock grazing, invasion of exotic vegetation, and alteration of historic fire regimes. Conversion of shrub-steppe lands to agriculture adversely affects landbirds in two ways: 1) native habitat is in most instances permanently lost, and 2) remaining shrub-steppe is isolated and embedded in a highly fragmented landscape of multiple land uses, particularly agriculture. Fragmentation resulting from agricultural development or large fires fueled by cheatgrass can have several negative effects on landbirds. These include: insufficient patch size for area-dependent species, and increases in edges and adjacent hostile landscapes, which can result in reduced productivity through increased nest predation, nest parasitism, and reduced pairing success of males. Additionally, fragmentation of shrub-steppe has likely altered the dynamics of dispersal and immigration necessary for maintenance of some populations at a regional scale. In a recent analysis of neotropical migratory birds within the Interior Columbia

Basin, most species identified as being of "high management concern" were shrub-steppe species (Saab and Rich 1997) which includes the grasshopper sparrow.

Approximately 6 million hectares of shrub-steppe have been converted to wheat fields, row crops, and orchards in the interior Columbia Basin (Quigley and Arbelbide 1997). In Washington over 50% of historic shrub-steppe has been converted to agriculture (Dobler et al. 1996).

Large scale reduction and fragmentation of sagebrush habitats have occurred due to a number of activities, including land conversion to tilled agriculture, urban and suburban development, and road and power-line rights of way. Range improvement programs remove sagebrush by burning, herbicide application, and mechanical treatment, replacing sagebrush with annual grassland to promote forage for livestock.

Making this loss of habitat even more severe is that the grasshopper sparrow like other grassland species shows a sensitivity to the grassland patch size (e.g. Herkert 1994, Samson 1980, Vickery 1994a b, Bock et al. 1999). Herkert (1991) in Illinois, found that grasshopper sparrows were not present in grassland patches smaller than 30 hectares (74 acres) despite the fact that their published average territory size is only about 0.3 ha (0.75 acres). Vickery et al. (1994) found the minimum requirement to be 100 hectares and Samson (1980) found the minimum to be 20 ha. in Missouri. Differences in minimum area requirements may be explained by the effect of relative population level on the selectivity of individuals, as has been shown for many species of birds (Vickery et al. 1994). Minimum requirement size in the Northwest is unknown.

Grazing

Grazing can trigger a cascade of ecological changes, the most dramatic of which is the invasion of non-native grasses escalating the fire cycle and converting sagebrush shrublands to annual grasslands. Historical heavy livestock grazing altered much of the sagebrush range, changing plant composition and densities. West (1988, 1996) estimates less than 1 percent of sagebrush steppe habitats remain untouched by livestock; 20 percent is lightly grazed, 30 percent moderately grazed with native understory remaining, and 30 percent heavily grazed with understory replaced by invasive annuals. The effects of grazing in sagebrush habitats is complex, depending on intensity, season, duration and extent of alteration to native vegetation.

Extensive and intensive grazing in w. North America has had negative impacts on this species (Bock and Webb 1984).

The legacy of livestock grazing in the Columbia Plateau has had widespread and severe impacts on vegetation structure and composition. One of the most severe impacts in shrub-steppe has been the increased spread of exotic plants (Altman and Holmes 2000, Weddell 2001)

For instance, the grasshopper sparrow has been found to respond positively to light or moderate grazing in tallgrass prairie (Risser et al 1981). However, it responds negatively to grazing in shortgrass, semidesert, and mixed grass areas (Bock et al 1984).

Invasive Grasses

Cheatgrass readily invades disturbed sites, and has come to dominate the grass-forb community of more than half the sagebrush region in the West, replacing native bunchgrasses (Rich 1996). Crested wheatgrass and other non-native annuals have also fundamentally altered the grass-forb community in many areas of sagebrush shrub-steppe, altering shrubland habitats.

The degree of degradation of terrestrial ecosystems is often diagnosed by the presence and extent of alien plant species (e.g., Andreas and Lichvar 1995); frequently their presence is related to soil disturbance and overgrazing. Increasingly, however, aggressive aliens are becoming established even in ostensibly undisturbed bunchgrass vegetation, wherever their seed can reach. The most notorious alien species in the Palouse region are upland species that can dominate and exclude perennial grasses over a wide range of elevations and substrate types (Weddell 2001).

Fire

Cheatgrass has altered the natural fire regime in the western range, increasing the frequency, intensity, and size of range fires. Fire kills sagebrush and where non-native grasses dominate, the landscape can be converted to annual grassland as the fire cycle escalates, removing preferred habitat (Paige and Ritter 1998).

The historical role of fire in the steppe and meadow steppe vegetation of the Palouse region is less clear (Weddell 2001). Daubenmire (1970) dismissed it as relatively unimportant, whereas others conclude that fires were probably more prevalent in the recent past than at present (Morgan et al. 1996). The lack of information about the presettlement fire frequency of steppe and meadow steppe ecosystems makes it difficult to emulate the natural fire regime in restored communities.

Studies on the effects of burns on grassland birds in North American grasslands have shown similar results as grazing studies: namely, bird response is highly variable. Confounding factors include timing of burn, intensity of burn, previous land history, type of pre-burn vegetation, presence of fire-tolerant exotic vegetation (that may take advantage of the post-burn circumstances and spread even more quickly) and grassland bird species present in the area. It should be emphasized that much of the variation in response to grassland fires lies at the level of species, but that even at this level results are often difficult to generalize. For instance, Mourning Doves have been found to experience positive (Bock and Bock 1992, Johnson 1997) and negative (Zimmerman 1997) effects by fire in different studies. Similarly, grasshopper sparrow have been found to experience positive (Johnson 1997), negative (Bock and Bock 1992, Zimmerman 1997, Vickery et al 1999), and no significant (Rohrbaugh 1999) effects of fire. Species associated with short and/or open grass areas will most likely experience short-term benefits from fires. Species that prefer taller and denser grasslands most likely will demonstrate a negative response to fire. (CPIF 2000).

Avoid burning during breeding season. Encroachment of woody vegetation in grassland areas will be detrimental to most grassland species. For instance, grasshopper sparrows have been found to be absent from areas with greater than 30% shrub cover. In areas of good grassland bird diversity and productivity, efforts should be made to keep woody vegetation from reducing open grassland habitat. (CPIF 2000).

Mowing/Haying

Mowing and haying affects grassland birds directly and indirectly. It may reduce height and cover of herbaceous vegetation, destroy active nests, kill nestlings and fledglings, cause nest abandonment, and increase nest exposure and predation levels (Bollinger et al. 1990). Studies on grasshopper sparrow have indicated higher densities and nest success in areas not mowed until after July 15 (Shugaart and James 1973, Warner 1992). Grasshopper sparrows are vulnerable to early mowing of fields, while light grazing, infrequent and post-season burning or mowing can be beneficial (Vickery 1996).

Brood Parasitism

Grasshopper sparrows may be multiply-parasitized (Elliott 1976, 1978; Davis and Sealy 2000). In Kansas, cowbird parasitism cost grasshopper sparrows about 2 young/parasitized nest, and there was a low likelihood of nest abandonment occurring due to cowbird parasitism (Elliott 1976, 1978). In Manitoba, mean number of host young fledged from successful, unparasitized nests was significantly higher than from successful, parasitized nests; cowbird parasitism cost Grasshopper Sparrows about 1.3 young/successful nest (Davis and Sealy 2000).

Predators

Predators of the grasshopper sparrow are hawks, Loggerhead Shrikes, mammals and snakes (Vickery 1996).

Out-of-Subbasin Effects and Assumptions

No data are available.

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SAGE SPARROW
Amphispiza belli

Original Species Account Authors: Paul Ashley and Stacy Stoval, as appeared in the Southeast Washington Ecoregional Assessment, January 2004

Introduction

Sage sparrow (*Amphispiza belli*) is a species of concern in the West due to population decline in some regions and the degradation and loss of breeding and wintering habitats. Vulnerable to loss and fragmentation of sagebrush habitat, sage sparrows may require large patches for breeding. Sage sparrow can likely persist with moderate grazing and other land management activities that maintain sagebrush cover and the integrity of native vegetation.
Sagebrush habitats may be very difficult to restore where non-native grasses and other invasive species are pervasive, leading to an escalation of fire cycles that permanently convert sagebrush habitats to annual grassland.

Sage sparrows are still common throughout much of sagebrush country and have a high probability of being sustained wherever large areas (e.g., 130 hectares observed in Washington, Vader Haegen, pers. comm.) of sagebrush and other preferred native shrubs exist for breeding. Sage sparrows are likely to return to areas where sagebrush and other native vegetation have been restored. However, sagebrush habitats can be very difficult to reclaim once invaded by cheatgrass and other noxious non-native vegetation, leading to an escalation of fire frequency and fire intensity that permanently converts shrub-steppe to annual grassland.

Life History, Key Environmental Correlates, and Habitat Requirements
Life History
Diet

Sage sparrows eat insects, spiders, seeds, small fruits, and succulent vegetation. They forage on the ground, usually under or near shrubs. They may occasionally be observed gleaning prey items from main stems and leaves. Consumed vegetation and insect prey provide most water requirements (Martin and Carlson 1998).

Reproduction

Sage sparrow clutch size usually is three to four, sometimes five. Incubation lasts about 13 days. Nestlings are altricial. Individual females produce one to three broods annually. Reproductive success is greater in wetter years (Rotenberry and Wiens 1991).
In eastern Washington, 70 percent (n = 53) of clutches examined had 3 eggs (Rotenberry and Wiens 1989). Annual reproductive success in Idaho was 1.3 fledglings/nest and probability of nest success was 40 percent (Reynolds 1981). Estimate of nest success in eastern Washington is 32 percent (M. Vander Haegen, unpub. data in Altman and Holmes 2000).

Nesting

Sage sparrows form monogamous pair bonds in early spring; nesting behavior occurs from March to July. Nests are constructed by females in or under sagebrush shrubs and pairs raise 1-2 broods a season (Martin and Carlson 1998).

Brown-headed cowbirds will parasitize sage sparrow nests; parasitized nests are often abandoned (Rich 1978).

Chicks are altricial and fledge when 9-10 days of age. Both parents feed young for more than two weeks after fledging. Fledglings often sit low in shrubs or on the ground under shrubs (Martin and Carlson 1998).

Migration
Sage sparrow populations in Washington are migratory. Sage sparrows are present only during the breeding season, arriving in late February-early March. Birds winter in shrub-steppe habitats of the southwestern United States and northwestern Mexico.

Mortality
Little information is available on estimates of annual survival rates (Martin and Carlson 1998). Typical nest predators include, common raven (*Corvus corax*), Townsend's ground squirrel (*Spermophilus townsendi*), and gopher snakes (*Pituophis catenifer*) (Martin and Carlson 1998, Rotenberry and Wiens 1989). Predators of juvenile and adult birds include loggerhead shrike (*Lanius ludovicianus*) and raptors (Martin and Carlson 1998).

Habitat Requirements
Similar to other shrub-steppe obligate species, sage sparrows are associated with habitats dominated by big sagebrush (*Artemisia tridentata*) and perennial bunchgrasses (Paige and Ritter 1999). In shrub-steppe habitat in southwestern Idaho, habitat occupancy by sage sparrows increased with increasing spatial similarity of sites, shrub patch size, and sagebrush cover; landscape features were more important in predicting presence of sage sparrows than cover values of shrub species and presence of sagebrush was more important than shadscale (Knick and Rotenberry 1995).

Nesting
Habitat in the vicinity of sage sparrow nests in southwestern Idaho was characterized by lower sagebrush cover (23 percent), greater shrub dispersion (clumped vs. uniform), and taller shrub height (18 in.) than surrounding areas. Sage sparrows preferred nesting in large, live sagebrush plants; birds frequently nested in shrubs 16-39 in. tall, shrubs < 6 in. or > 39 in. were rarely used (Petersen and Best 1985). In eastern Washington, height of sagebrush nest shrubs averaged 90 cm (35 in.) (Vander Haegen 2003). In Idaho, nests were constructed an average distance of 34 cm (13 in.) above ground, 11 in. from the top, and 8 in. from the shrub perimeter (Petersen and Best 1985). Although sage sparrows generally place nests in sagebrush shrubs they frequently nest on the ground (Vander Haegen 2003).

Breeding
Washington breeders represent the northern subspecies *A. b. nevadensis.* In the northern Great Basin, sage sparrow is associated with low and tall sagebrush/bunchgrass, juniper/sagebrush, mountain mahogany/shrub, and aspen/sagebrush/bunchgrass communities for breeding and foraging (Maser et al. 1984). In Idaho, sage sparrows are found in sagebrush of 11 to 14 percent cover (Rich 1980). Martin and Carlson (1998) report a preference for evenly spaced shrubs; other authors (Rotenberry and Wiens 1980; Peterson and Best 1985) report association where sagebrush is clumped or patchy. Sage sparrows prefer semi-open habitats, shrubs 1-2 meters tall (Martin and Carlson 1998). Habitat structure (vertical structure, shrub density, and habitat patchiness) is important to habitat selection (Martin and Carlson 1998). Sage sparrow is positively correlated with big sagebrush (*Artemisia tridentata*), shrub cover, bare ground, above-

average shrub height, and horizontal patchiness; it is negatively correlated with grass cover (Rotenberry and Wiens 1980; Wiens and Rotenberry 1981; Larson and Bock 1984).

The subspecies *nevadensis* breeds in brushland dominated by big sagebrush or sagebrush-saltbush (Johnson and Marten 1992). Sage sparrows nest on the ground or in a shrub, up to about one meter above ground (Terres 1980). In the Great Basin, nests are located in living sagebrush where cover is sparse but shrubs are clumped (Petersen and Best 1985). Nest placement may be related to the density of vegetative cover over the nest, and will nest higher in a taller shrub (Rich 1980).

Breeding territory size in eastern Washington averages 1.5-3.9 ac but may vary among sites and years (Wiens et al. 1985). Territories are located in relatively large tracts of continuous sagebrush-dominated habitats. Territory size can vary with plant community composition and structure, increasing with horizontal patchiness (see Wiens et al. 1985). Sage sparrows are absent on sagebrush patches < 325 ac (Vander Haegen et al. 2000; M. Vander Haegen unpub. data in Altman and Holmes 2000).

Non-breeding

In migration and winter, sage sparrows are found in arid plains with sparse bushes, grasslands and open areas with scattered brush, mesquite, and riparian scrub, preferring to feed near woody cover (Martin and Carlson 1998; Meents et al. 1982; Repasky and Schluter 1994). Flocks of sage sparrows in the Mojave Desert appear to follow water courses (Eichinger and Moriarty 1985). Wintering birds in honey mesquite of lower Colorado River select areas of higher inkweed (*Suaeda torreyana*) density (Meents et al. 1982).

Population and Distribution

Population

Historic

No data are available.

Current

Sage sparrow populations are most abundant in areas of deep loamy soil and continuous sagebrush cover 3.3-6.6 feet high (Vander Haegen et al. 2000). In south-central Washington sage sparrows are one of the most common shrub-steppe birds (Vander Haegen et al. 2001). Sage sparrow breeding density was estimated at 121-207 individuals/km2 over a two-year study at the Arid Lands Ecology Reservation in southern Washington (Wiens et al. 1987). Density estimates ranged from 33-90 birds/km2 in sagebrush habitat on the Yakima Training Center (Shapiro and Associates 1996), whereas Schuler et al. (1993) on Hanford Reservation, reported density from 0.23-21.03 birds/km2.

The sedentary subspecies *belli* is found in the foothills of the Coast Ranges (northern California to northwestern Baja California) and the western slope of the central Sierra Nevada in California (Johnson and Marten 1992).

The subspecies *canescens* breeds in the San Joaquin Valley and northern Mohave Desert in California and extreme western Nevada, winters in the southwestern U.S. (Johnson and Marten 1992).

The subspecies *nevadensis* breeds from central interior Washington eastward to southwestern Wyoming and northwestern Colorado, south to east-central California, central Nevada, northeastern Arizona, and northwestern New Mexico. *Nevadensis* winters in the southwestern U.S. and northern Mexico (Johnson and Marten 1992).

Distribution

Historic

Jewett et al. (1953) described the distribution of the sage sparrow as a common summer resident probably at least from March to September in portions of the sagebrush of the Upper Sonoran Zone and of the neighboring bunchgrass areas of the Transition zone in eastern Washington. They describe its summer range as north to Wilbur and Waterville, Grand Coulee; east to Connell and Wilbur; south to Kiona, Kennewick, and Lower Flat, Walla Walla County; and west to Waterville, Moxee City, Sunnyside, Yakima, and Soap Lake. Jewett et al. (1953) also note that the sage sparrow was found practically throughout the sagebrush of eastern Washington, and in a few places, notably in the vicinity of Wilbur, Waterville, Prescott, and Horse Heaven, it ranges into the bunch grass as well. Jewett et al. (1953) report that Snodgrass found it the predominant sparrow in the sagebrush west of Connell. Hudson and Yocom (1954) described the sage sparrow as a summer resident and migrant in sagebrush areas of Adams, Franklin, and Grant counties. They report that Snodgrass reported it as common in western Walla Walla County.

Current

Data are not available.

Breeding

During the breeding season, sage sparrows are found in central Washington, eastern Oregon, southern Idaho, southwestern Wyoming, and northwestern Colorado south to southern California, central Baja California, southern Nevada, southwestern Utah, northeastern Arizona, and northwestern New Mexico (AOU 1983; Martin and Carlson 1998).

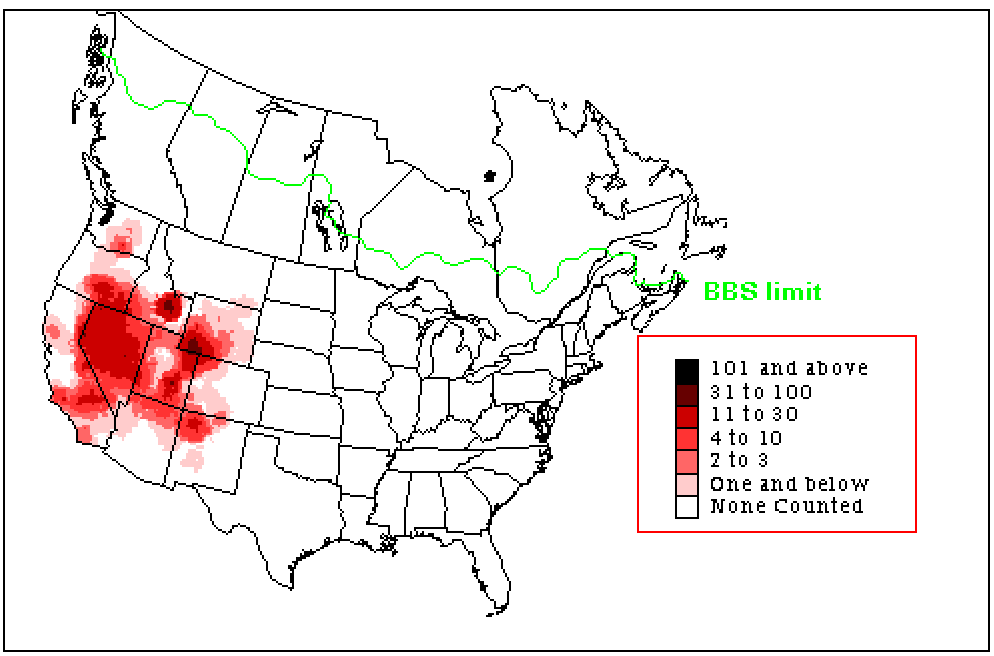

Figure 2. Sage sparrow breeding season abundance (from BBS data) (Sauer et al. 2003).

Non-breeding

Sage sparrows are found in central California, central Nevada, southwestern Utah, northern Arizona, and central New Mexico south to central Baja California, northwestern mainland of Mexico, and western Texas (AOU 1983; Martin and Carlson 1998).

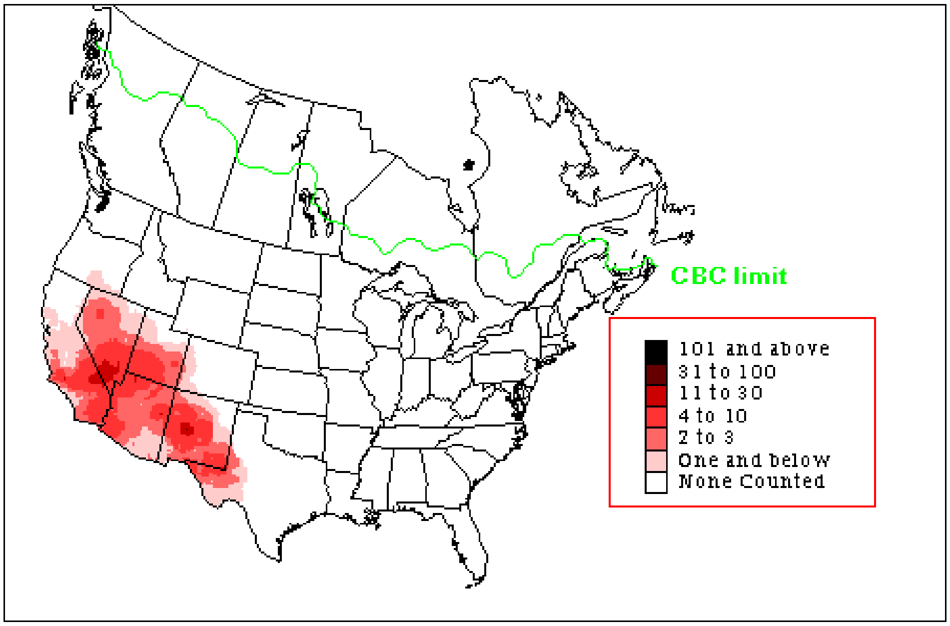

Figure 3. Sage sparrow winter season abundance (from CBC data) (Sauer et al. 2003).

Sage Sparrow Status and Abundance Trends

Status

North American Breeding Bird Survey (BBS) data indicate that sage sparrows have declined 1.0-2.3 percent in recent decades (1966-1991); greatest declines have occurred in Arizona, Idaho, and Washington (Martin and Carlson 1998). Sage sparrows are listed as a 'candidate' species (potentially threatened or endangered) by the Washington Department of Fish and Wildlife and are listed by the Oregon-Washington chapter of Partners in Flight as a priority species, and on the National Audubon Society Watch List. Based on genetic and morphometric differences, the subspecies *A. b. nevadensis* (currently found in east-central Washington) may be reclassified as a distinct species. Such an action would likely prompt increased conservation interest at the federal level.

Trends

The BBS data (1966-1996) for Washington State show a non-significant 0.3 percent average annual increase in sage sparrow survey-wide (n = 187 survey routes). There has been a significant decline of -4.8 percent average per year for 1966-1979 (n = 73), and a recent significant increase of 2.0 percent average per year, 1980-1996 (n = 154; Sauer et al. 1997). BBS data indicate recent non-significant declines in California and Wyoming, 1980-1995. Generally, low sample sizes make trend estimates unreliable for most states and physiographic regions. Highest sage sparrow summer densities occur in the Great Basin, particularly Nevada, southeastern Oregon, southern Idaho, and Wyoming (Sauer et al. 1997).

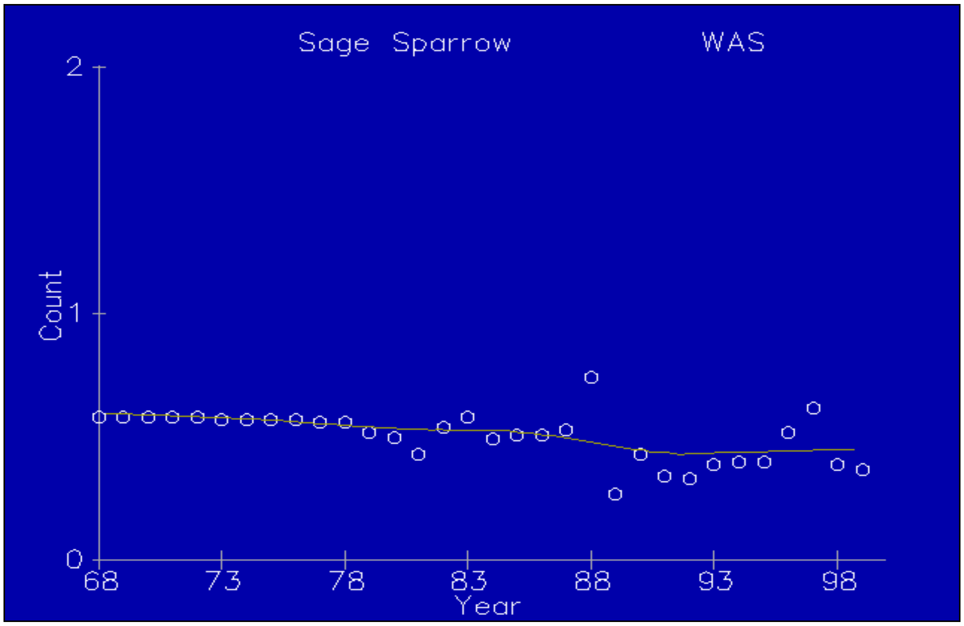

Figure 4. Sage sparrow population trend data (from BBS), Washington (Sauer et al. 2003).

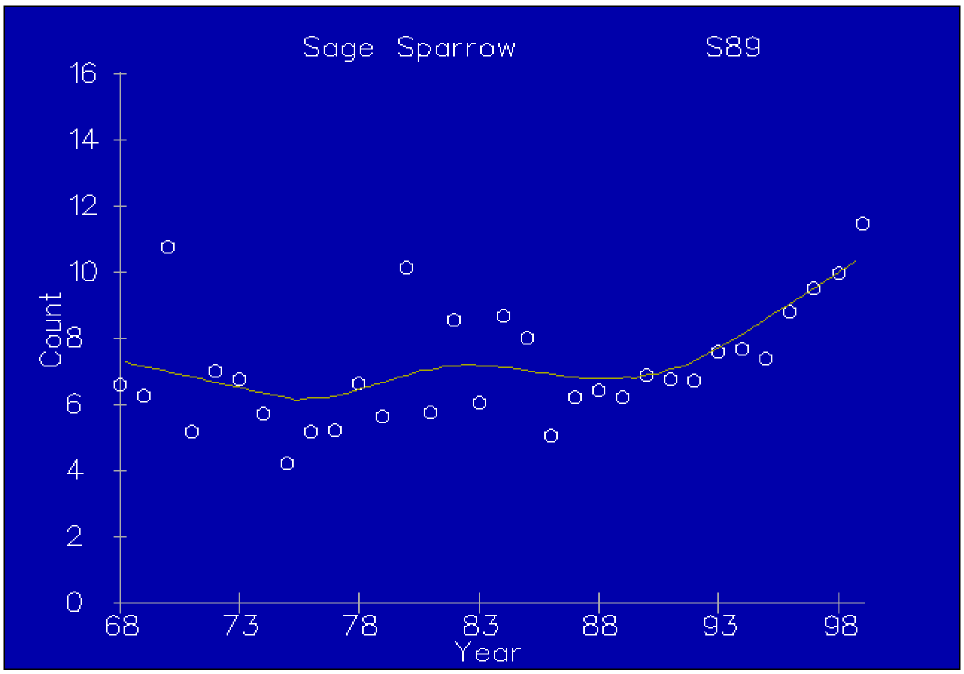

Figure 5. Sage sparrow trend results (from BBS data), Columbia Plateau (Sauer et al. 2003).

Christmas Bird Count (CBC) data show a significant decline in sage sparrows (-2.1 percent average per year; n = 160 survey circles) survey-wide for the period from 1959-1988. Sage sparrow trend estimates show declines in Arizona, New Mexico, and a significant decline in Texas (-2.2 percent average per year; n = 16). The highest sage sparrow winter counts occur in southern Nevada, southern California, Arizona, New Mexico, and west Texas (Sauer et al. 1996).

According to the ICBEMP terrestrial vertebrate habitat analysis, historical source habitats for sage sparrow occurred throughout most of the three ERUs within our planning unit (Wisdom et al. in press). Declines in source habitats were moderately high in the Columbia Plateau (40 percent), but relatively low in the Owyhee Uplands (13 percent) and Northern Great Basin (7 percent). However, declines in big sagebrush (e.g., 50 percent in Columbia Plateau ERU), which is likely higher quality habitat, are masked by an increase in juniper sagebrush (>50 percent in Columbia Plateau ERU), which is likely reduced quality habitat. Within the entire Interior Columbia Basin, over 48 percent of watersheds show moderately or strongly declining trends in source habitats for this species (Wisdom et al. in press) (from Altman and Holmes 2000).

Factors Affecting Sage Sparrow Population Status
Key Factors Inhibiting Populations and Ecological Processes
Habitat Loss
Because sage sparrows are shrub-steppe obligates. Sagebrush shrublands are vulnerable to a number of activities that reduce or fragment sagebrush habitat, including land conversion to tilled agriculture, urban and suburban development, and road and powerline rights of way. Range improvement programs remove sagebrush by burning, herbicide application, and mechanical treatment, replacing sagebrush with annual grassland to promote forage for livestock.

Agricultural set-aside programs (such as the Conservation Reserve Program [CRP]) may eventually increase the quantity of potential breeding habitat for sage sparrows but it is not clear how long this will take. Habitat objectives recommended for sage sparrows include; dominant sagebrush canopy with 10 - 25 percent sagebrush cover, mean sagebrush height >50 cm, high foliage density, mean native grass cover > 10 percent, mean exotic annual grass cover < 10 percent, mean open ground cover > 10 percent, and where appropriate provide suitable habitat conditions in patches >1000 ha (400ac) (Altman and Holmes 2000).

Fragmentation
The presence of relatively large tracts of sagebrush-dominated habitats is important as research in Washington indicates a negative relationship between sage sparrow occurrence and habitat fragmentation (Vander Haegen et al. 2000). Additionally, fragmentation of shrub-steppe habitat may increase vulnerability of sage sparrows to nest predation by generalist predators such as the common raven (*Corvus corax*) and black-billed magpie (*Pica hudsonia*) (Vander Haegen et al. 2002).

Livestock Management
Response to variation in grazing intensity is mixed. Sage sparrows respond negatively to heavy grazing of greasewood/Great Basin wild rye and shadscale/Indian ricegrass communities. They respond positively to heavy grazing of Nevada bluegrass/sedge communities, moderate grazing of big sage/bluebunch wheatgrass community, and to unspecified grazing intensity of big sage communities (see review by Saab et al. 1995). Because sage sparrows nest on the ground in early spring, and forage on the ground, maintenance of >50 percent of annual vegetative herbaceous growth of perennial bunchgrasses through the following season is recommended (Altman and Holmes 2000).

Pesticides/Herbicides
Large scale (16 km2) aerial spraying of sagebrush habitat with the herbicide 2,4-D resulted in a significant decline in sage sparrow abundance 2 years post treatment. Because sage sparrows display high site fidelity to breeding areas birds may occupy areas that have been rendered unsuitable (Wiens and Rotenberry 1985).

Fire

Cheatgrass has altered the natural fire regime in the western range, increasing the frequency, intensity, and size of range fires. Fire kills sagebrush and where non-native grasses dominate, the landscape can be converted to annual grassland as the fire cycle escalates, removing habitat for sage sparrow (Paige and Ritter 1998).

Invasive Grasses

Cheatgrass readily invades disturbed sites, and has come to dominate the grass-forb community of more than half the sagebrush region in the West, replacing native bunchgrasses (Rich 1996). Crested wheatgrass and other non-native annuals have also fundamentally altered the grass-forb community in many areas of sagebrush shrub-steppe.

Brood Parasitism

Sage sparrow is an occasional host for brown-headed cowbird (*Molothrus ater*), and may abandon the nest (e.g., see Reynolds 1981). Prior to European-American settlement, sage sparrow was probably largely isolated from cowbird brood parasitism, but is now vulnerable where the presence of livestock, land conversion to agriculture, and fragmentation of shrublands creates a contact zone between the species (Rich 1978).

Predation

In Oregon, predation by Townsend ground squirrel (*Spermophilus townsendi*) affected sage sparrow reproductive success when squirrel densities were high. Sage sparrow populations in southeastern Washington and northern Nevada incurred high rates of nest predation, probably mainly by gopher snakes (*Pituophis melanoleucus*) (Rotenberry and Wiens 1989). Loggerhead shrikes (*Lanius ludovicianus*) prey on both adults and altricial young in nest, and can significantly reduce nest production (Reynolds 1979). Feral cats near human habitations may increase predation (Martin and Carlson 1998).

Out-of-Subbasin Effects and Assumptions

No data could be found on the migration and wintering grounds of the sage sparrow. It is a short distance migrant, wintering in the southwestern U.S. and northern Mexico, and as a result faces a complex set of potential effects during it annual cycle. Habitat loss or conversions is likely happening along its entire migration route (H. Ferguson, WDFW, pers. comm., 2003). Management requires the protection shrub, shrub-steppe, desert scrub habitats, and the elimination or control of noxious weeds. Migration routes, corridors, and wintering grounds need to be identified and protected just as its breeding area.

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COLUMBIA SPOTTED FROG
Rana luteiventris

Original Species Account Author: Keith Paul, United States Fish and Wildlife Service

Introduction
The Columbia spotted frog (CSF) is olive green to brown in color, with irregular black spots. They may have white, yellow, or salmon coloration on the underside of the belly and legs (Engle 2004). The hind legs are relatively short relative to body length and there is extensive webbing between the toes on the hind feet. The eyes are upturned (Amphibia Web 2004). Tadpoles are black when small, changing to a dark then light brown as they increase in size. CSFs are about one inch in body length at metamorphosis (Engle 2004). Females may grow to approximately 100 mm (4 inches) snout-to-vent length, while males may reach approximately 75 mm (3 inches) snout-vent length (Nussbaum et al. 1983; Stebbins 1985; Leonard et al. 1993).

Life History, Key Environmental Correlates, and Habitat Requirements
Life History
Diet
The CSF eats a variety of food including arthropods (e.g., spiders, insects), earthworms and other invertebrate prey (Whitaker et al. 1982). Adult CSFs are opportunistic feeders and feed primarily on invertebrates (Nussbaum et al. 1983). Larval frogs feed on aquatic algae and vascular plants, and scavenged plant and animal materials (Morris and Tanner 1969).

Reproduction
The timing of breeding varies widely across the species range owing to differences in weather and climate, but the first visible activity begins in late winter or spring shortly after areas of ice-free water appear at breeding sites (Licht 1975; Turner 1958; Leonard et al 1996). Breeding typically occurs in late March or April, but at higher elevations, breeding may not occur until late May or early June (Amphibia Web 2004). Great Basin population CSFs emerge from wintering sites soon after breeding sites thaw (Engle 2001).

Adults exhibit a strong fidelity to breeding sites, with oviposition typically occurring in the same areas in successive years. Males arrive first, congregating around breeding sites, periodically vocalizing “advertisement calls” in a rapid series of 3-12 “tapping” notes that have little carrying power (Davidson 1995; Leonard et al. 1996). As a female enters the breeding area, she is approached by and subsequently pairs with a male in a nuptial embrace referred to as amplexus. From several hours to possibly days later, the female releases her complement of eggs into the water while the male, still clinging to the female, releases sperm upon the ova (Amphibia Web 2004). Breeding is explosive (as opposed to season-long), occurring only in the first few weeks following emergence (USFWS 2002a). After breeding is completed, adults often disperse into adjacent wetland, riverine and lacustrine habitats (Amphibia Web 2004).

CSF’s have a strong tendency to lay their eggs communally and it is not uncommon to find 25 or more egg masses piled atop one another in the shallows (Amphibia Web 2004). Softball-sized egg masses are usually found in groups, typically along northeast edges of slack water amongst emergent vegetation (USFWS 2002a). After a few weeks thousands of small tadpoles emerge and cling to the remains of the gelatinous egg masses. Newly-hatched larvae remain clustered for several days before moving throughout their natal site (USFWS 2002a). In the Columbia

Basin tadpoles may grow to 100 mm (4 in) total length prior to metamorphosing into froglets in their first summer or fall. At high-elevation montane sites, however, tadpoles barely reach 45 mm (1.77 in) in total length prior to the onset of metamorphosis in late fall (Amphibia Web 2004). As young-of-the-year transform, many leave their natal sites and can be found in nearby riparian corridors (USFWS 2002a).

Females may lay only one egg mass per year; yearly fluctuations in the sizes of egg masses are extreme (Utah Division of Wildlife Resources 1998). Successful egg production and the viability and metamorphosis of CSF's are susceptible to habitat variables such as temperature, depth, and pH of water, cover, and the presence/absence of predators (e.g., fishes and bullfrogs) (Morris and Tanner 1969; Munger et al. 1996; Reaser 1996).

Migration

[David Pilliod observed movements of approximately 2,000 m (6,562 ft) linear distance within a basin in montane habitats (Reaser and Pilliod, in press). Pilliod et al. 1996 (in Koch et al. 1997) reported that individual high mountain lake populations of *R. luteiventris* in Idaho are actually interdependent and are part of a larger contiguous metapopulation that includes all the lakes in the basin. In Nevada, Reaser (1996; in Koch et al. 1997) determined that one individual of *R. luteiventris* traveled over 5 km (3.11 mi) in a year (NatureServe 2003)].

[In a three-year study of *R. luteiventris* movement within the Owyhee Mountain subpopulation of the Great Basin population in southwestern Idaho, Engle (2000) PIT-tagged over 1800 individuals but documented only five (of 468) recaptures over 1,000 m (3,281 ft) from their original capture point. All recaptures were along riparian corridors and the longest distance between capture points was 1,765 m (5,791). Although gender differences were observed, 88 percent of all movement documented was less than 300 m (984 ft) from the original capture point (NatureServe 2003)].

[Though movements exceeding 1 km (0.62 mi) and up 5 km (3.11 mi) have been recorded, these frogs generally stay in wetlands and along streams within 0.6 km (0.37 mi) of their breeding pond (Turner 1960, Hollenbeck 1974, Bull and Hayes 2001). Frogs in isolated ponds may not leave those sites (Bull and Hayes 2001) (NatureServe 2003)].

[In the Toiyabe Range in Nevada, Reaser (2000) captured 887 individuals over three years, with average mid-season density ranging from 2 to 24 frogs per 150 m (492 ft) of habitat (NatureServe 2003)].

Mortality

Based on recapture rates in the Owyhee Mountains, some individuals live for at least five years. Skeletochronological analysis in 1998 revealed a 9-year old female (Engle and Munger 2000).

Mortality of eggs, tadpoles, and newly metamorphosed frogs is high, with approximately 5% surviving the first winter (David Pilliod, personal communication, cited in Amphibia Web 2004).

Habitat Requirements

General

This species is relatively aquatic and is rarely found far from water. It occupies a variety of still water habitats and can also be found in streams and creeks (Hallock and McAllister 2002). CSF's are found closely associated with clear, slow-moving or ponded surface waters, with little shade (Reaser 1997). CSF's are found in aquatic sites with a variety of vegetation types, from grasslands to forests (Csuti 1997). A deep silt or muck substrate may be required for hibernation and torpor (Morris and Tanner 1969). In colder portions of their range, CSF's will use areas where water does not freeze, such as spring heads and undercut streambanks with overhanging vegetation (IDFG et al. 1995). CSF's may disperse into forest, grassland, and brushland during wet weather (NatureServe 2003). They will use stream-side small mammal burrows as shelter. Overwintering sites in the Great Basin include undercut banks and spring heads (Blomquist and Tull 2002).

Breeding

Reproducing populations have been found in habitats characterized by springs, floating vegetation, and larger bodies of pooled water (e.g., oxbows, lakes, stock ponds, beaver-created ponds, seeps in wet meadows, backwaters) (IDFG et al. 1995; Reaser 1997). Breeding habitat is the temporarily flooded margins of wetlands, ponds, and lakes (Hallock and McAllister 2002). Breeding habitats include a variety of relatively exposed, shallow-water (<60 cm), emergent wetlands such as sedge fens, riverine over-bank pools, beaver ponds, and the wetland fringes of ponds and small lakes. Vegetation in the breeding pools generally is dominated by herbaceous species such as grasses, sedges (*Cares spp*.) and rushes (*Juncus spp*.) (Amphibia Web 2004).

Population and Distribution

Distribution

Populations of the CSF are found from Alaska and British Columbia to Washington east of the Cascades, eastern Oregon, Idaho, the Bighorn Mountains of Wyoming, the Mary's, Reese, and Owyhee River systems of Nevada, the Wasatch Mountains, and the western desert of Utah (Green et al. 1997). Genetic evidence (Green et al. 1996) indicates that Columbia spotted frogs may be a single species with three subspecies, or may be several weakly-differentiated species.

The FWS recognizes four distinct population segments (DPS) based on disjunct distribution: the Wasatch Front DPS (Utah), West Desert DPS (White Pine County, NV and Toole County Utah), Great Basin DPS (southeast Oregon, southwest Idaho, and northcentral/northeast Nevada), and the Northern DPS (includes northeastern Oregon, eastern Washington, central and northern parts of Idaho, western Montana, northwestern Wyoming, British Columbia and Alaska) (C. Mellison, J. Engle, pers. comm., 2004).

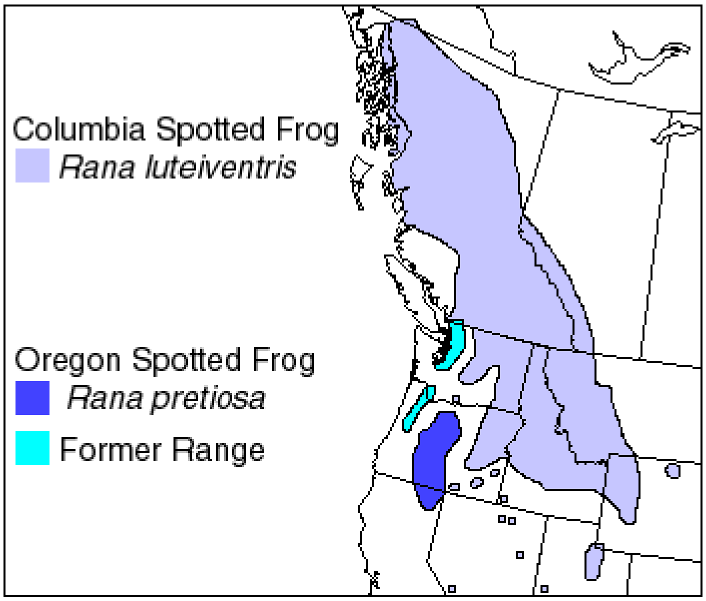
There is still some uncertainty about whether the northeast Oregon frogs and the southeastern Washington frogs are part of the Great Basin or Northern population. This group of frogs (Blue and Wallowa Mountains) is isolated from the Great Basin population based on geography, and the habitat in the Anthony Lakes area is more like that of the Northern population (montane) than the Great Basin (high desert). It has been considered to make the Snake River a boundary between the Northern and Great Basin populations, but further genetics work will need to be done to clarify the issue (J. Engle, pers. comm., 2004).

Two populations of CSFs are found within the Columbia River Basin: Northern DPS and Great Basin DPS. The Great Basin DPS is further divided into five subpopulations: southeastern Oregon, Owyhee, Jarbidge-Independence, Ruby Mountains, and Toiyabe (J. Engle, C. Mellison, pers. comm., 2004). Of the five subpopulations, only the eastern Oregon, Owyhee, and the Jarbidge-Independence occur in the Columbia River subbasin.

Historic

Historic range of the Northern population is most likely similar to that of the current range. Moving south into the southern populations (Great Basin, Wasatch Front, and West Desert) the range was most likely larger in size. Due to habitat loss and alteration, fragmentation, water diversion, dams, and loss of beaver the current distribution and abundance of CSF and suitable habitat has dramatically decreased.

Current



USGS, Northern Prairie Wildlife Research Center; range acquired from Green et al. 1997.

<u>Wasatch Front DPS</u>

[Spotted frog populations in Utah represent the southern extent of the species range (Stebbins 1985). The Wasatch Front population occurs in isolated springs or riparian wetlands in Juab, Sanpete, Summit, Utah, and Wasatch counties in Utah. These counties are located within the Bonneville Basin of Utah. The Bonneville Basin encompasses the area that was covered by ancient Lake Bonneville and which, today, lies within the Great Basin province. The largest known concentration is currently in the Heber Valley; the remaining six locations are Jordanelle/Francis, Springville Hatchery, Holladay Springs, Mona Springs Complex/Burraston Ponds, Fairview, and Vernon (USFWS 2002b)].

<u>West Desert DPS</u>

[The West Desert spotted frog population occurs mainly in four large spring complexes. One new population, Vernon, was recently discovered in the eastern-most portion of the West Desert geographic management unit (GMU). CSFs in the West Desert DPS can be found along the eastern border of White Pine County, NV and Toole County, Utah. Populations have been extirpated from the northern portions of the West Desert range (USFWS 2002b)].

<u>Northern DPS</u>

The Northern DPS includes northeastern Oregon, eastern Washington, central and northern parts of Idaho, western Montana, northwestern Wyoming, British Columbia and Alaska (J. Engle, C.

Mellison, pers. comm., 2004). Populations within the Blue and Wallowa Mountains are found within this DPS.

<u>Great Basin DPS</u>
<u>Nevada</u>
The Great Basin population of Columbia spotted frogs in Nevada is geographically separated into three distinct subpopulations; the Jarbidge-Independence Range, Ruby Mountains, and Toiyabe Mountains subpopulations (USFWS 2002c).

[The largest of Nevada's three subpopulation areas is the Jarbidge-Independence Range in Elko and Eureka counties. This subpopulation area is formed by the headwaters of streams in two major hydrographic basins. The South Fork Owyhee, Owyhee, Bruneau, and Salmon Falls drainages flow north into the Snake River basin. Mary's River, North Fork of the Humboldt, and Maggie Creek drain into the interior Humboldt River basin. The Jarbidge-Independence Range subpopulation is considered to be genetically and geographically most closely associated with Columbia spotted frogs in southern Idaho (Reaser 1997)(USFWS 2002c)].

[Columbia spotted frogs occur in the Ruby Mountains in the areas of Green Mountain, Smith, and Rattlesnake creeks on lands in Elko County managed by the U.S. Forest Service (Forest Service). Although geographically, Ruby Mountains spotted frogs are close to the Jarbidge-Independence Range subpopulation, preliminary allozyme evidence suggests they are genotypically different (J. Reaser, pers. comm., 1998). The Ruby Mountains subpopulation is considered discrete because of this difference (J. Reaser, pers. comm., 1998) and because it is geographically isolated from the Jarbidge-Independence Range subpopulation area to the north by an undetermined barrier (e.g., lack of suitable habitat, connectivity, and/or predators), and from the Toiyabe Mountains subpopulation area to the southwest by a large gap in suitable Humboldt River drainage habitat (USFWS 2002c)].

[In the Toiyabe Range, spotted frogs are found in seven drainages in Nye County, Nevada; the Reese River (Upper and Lower), Cow and Ledbetter Canyons, and Cloverdale, Stewart, Illinois, and Indian Valley Creeks. Although historically they also occurred in Lander County, preliminary surveys have found them absent from this area (J. Tull, Forest Service, pers. comm., 1998). Toiyabe Range spotted frogs are geographically isolated from the Ruby Mountains and Jarbidge-Independence Range subpopulations by a large gap in suitable habitat and they represent *R. luteiventris* in the southern-most extremity of its range. Genetic analyses of Great Basin Columbia spotted frogs from the Toiyabe Range suggest that these frogs are distinctive in comparison to frogs from the Ruby Mountains and Jarbidge-Independence Range subpopulation areas (Green et al. 1996, 1997; J. Reaser, pers. comm., 1998). Genetic (mtDNA) differences between the Toiyabe Range frogs and the Ruby Mountains frogs are less than those between the Toiyabe Range frogs and the Jarbidge-Independence Range frogs, but this may be because of similar temporal and spatial isolation (J. Reaser, pers. comm., 1998) (USFWS 2002c)].

<u>Idaho and Oregon</u>
[Surveys conducted in the Raft River and Goose Creek drainages in Idaho failed to relocate spotted frogs (Reaser 1997; Shipman and Anderson 1997; Turner 1962). In 1994 and 1995, the Bureau of Land Management (BLM) conducted surveys in the Jarbidge and Snake River Resource Areas in Twin Falls County, Idaho. These efforts were also unsuccessful in locating spotted frogs (McDonald 1996). Only six historical sites were known in the Owyhee Mountain

range in Idaho, and only 11 sites were known in southeastern Oregon in Malheur County prior to 1995 (Munger et al. 1996) (USFWS 2002c)].

Currently, Columbia spotted frogs appear to be widely distributed throughout southwestern Idaho (mainly in Owyhee County) and eastern Oregon, but local populations within this general area appear to be isolated from each other by either natural or human induced habitat disruptions. The largest local population of spotted frogs in Idaho occurs in Owyhee County in the Rock Creek drainage. The largest local population of spotted frogs in Oregon occurs in Malheur County in the Dry Creek Drainage (USFWS 2002c).

Population, Status, and Abundance Trends

Nevada

[Declines of Columbia spotted frog populations in Nevada have been recorded since 1962 when it was observed that in many Elko County localities where spotted frogs were once numerous, the species was nearly extirpated (Turner 1962). Extensive loss of habitat was found to have occurred from conversion of wetland habitats to irrigated pasture and spring and stream dewatering by mining and irrigation practices. In addition, there was evidence of extensive impacts on riparian habitats due to intensive livestock grazing. Recent work by researchers in Nevada have documented the loss of historically known sites, reduced numbers of individuals within local populations, and declines in the reproduction of those individuals (Hovingh 1990; Reaser 1996a, 1996b, 1997). Surveys in Nevada between 1994 and 1996 indicated that 54 percent of surveyed sites known to have frogs before 1993 no longer supported individuals (Reaser 1997) (USFWS 2002c)].

[Little historical or recent data are available for the largest subpopulation area in Nevada, the Jarbidge-Independence Range. Presence/absence surveys have been conducted by Stanford University researchers and the Forest Service, but dependable information on numbers of breeding adults and trends is unavailable. Between 1993 and 1998, 976 sites were surveyed for the presence of spotted frogs in northeastern Nevada, including the Ruby Mountains subpopulation area (Shipman and Anderson 1997; Reaser 2000). Of these, 746 sites (76 percent) that were believed to have characteristics suitable for frogs were unoccupied. For these particular sites there is no information on historical presence of spotted frogs. Of 212 sites that were known to support frogs before 1992, 107 (50 percent) sites no longer had frogs, while 105 sites did support frogs. At the occupied sites, surveyors observed more than 10 adults at only 13 sites (12 percent). Frogs in this area appear widely distributed (Reaser 1997). No monitoring or surveying has taken place in northeastern Nevada since 1998. The Forest Service is planning on surveying the area during the summer of 2002 (USFWS 2002c)].

[Between 1993 and 1998, 339 sites were surveyed for the presence of Columbia spotted frogs in the Toiyabe Range. Surveyors visited 118 sites (35 percent) with suitable habitat characteristics where no frogs were present. Ten historical frog sites no longer had frogs when surveyed by Reaser between 1993 and 1996 (Reaser 1997). However, at 211 other historical sites, frogs were still present during this survey period. Of these 211 sites, surveyors reported greater than 10 adult frogs at 133 sites (63 percent) (Reaser 1997). In 2000, frog mark-recapture surveys of the Toiyabe Range subpopulation was conducted by the University of Nevada, Reno. Preliminary estimates of frog numbers in the Indian Valley Creek drainage were around 5,000 breeding individuals, which is greater than previously believed (K. Hatch, pers. comm., 2001). However, during the 2000-2001 winter, Hatch (2002) noted a large population decrease, ranging between 66 and 86.5 percent at several sites. Research is currently being conducted to help understand this apparent winterkill. Lack of standardized or extensive monitoring and routine surveying has prevented dependable determinations of frog population numbers or trends in Nevada (USFWS 2002c)].

Idaho and Oregon

[Extensive surveys since 1996 throughout southern Idaho and eastern Oregon, have led to increases in the number of known spotted frog sites. Although efforts to survey for spotted frogs have increased the available information regarding known species locations, most of these data suggest the sites support small numbers of frogs. Of the 49 known local populations in southern Idaho, 61 percent had 10 or fewer adult frogs and 37 percent had 100 or fewer adult frogs (Engle 2000; Idaho Conservation Data Center (IDCDC) 2000). The largest known local population of spotted frogs occurs in the Rock Creek drainage of Owyhee County and supports under 250 adult frogs (Engle 2000). Extensive monitoring at 10 of the 46 occupied sites since 1997 indicates a general decline in the number of adult spotted frogs encountered (Engle 2000; Engle and Munger 2000; Engle 2002). All known local populations in southern Idaho appear to be functionally isolated (Engle 2000; Engle and Munger 2000) (USFWS 2002c)].

[Of the16 sites that are known to support Columbia spotted frogs in eastern Oregon, 81 percent of these sites appear to support fewer than 10 adult spotted frogs. In southeastern Oregon, surveys conducted in 1997 found a single population of spotted frogs in the Dry Creek drainage of Malheur County. Population estimates for this site are under 300 adult frogs (Munger et al. 1996). Monitoring (since 1998) of spotted frogs in northeastern Oregon in Wallowa County indicates relatively stable, small local populations (less than five adults encountered) (Pearl 2000). All of the known local populations of spotted frogs in eastern Oregon appear to be functionally isolated (USFWS 2002c)].

Legal Status

In 1989, the U.S. Fish and Wildlife Service (USFWS) was petitioned to list the spotted frog (referred to as *Rana pretiosa*) under ESA (Federal Register 54[1989]:42529). The USFWS ruled on April 23, 1993, that the listing of the spotted frog was warranted and designated it a candidate for listing with a priority 3 for the Great Basin population, but was precluded from listing due to higher priority species (Federal Register 58[87]:27260). The major impetus behind the petition was the reduction in distribution apparently associated with impacts from water developments and the introduction of nonnative species.

On September 19, 1997 (Federal Register 62[182]:49401), the USFWS downgraded the priority status for the Great Basin population of Columbia spotted frogs to a priority 9, thus relieving the pressure to list the population while efforts to develop and implement specific conservation measures were ongoing. As of January 8, 2001 (Federal Register 66[5]:1295- 1300), however, the priority ranking has been raised back to a priority 3 due to increased threats to the species. This includes the Great Basin DPS Columbia spotted frog populations

Factors Affecting Columbia Spotted Frog Population Status

Key Factors Inhibiting Populations and Ecological Processes

The present or threatened destruction, modification, or curtailment of its habitat or range

[Spotted frog habitat degradation and fragmentation is probably a combined result of past and current influences of heavy livestock grazing, spring development, agricultural development, urbanization, and mining activities. These activities eliminate vegetation necessary to protect frogs from predators and UV-B radiation; reduce soil moisture; create undesirable changes in water temperature, chemistry and water availability; and can cause restructuring of habitat zones through trampling, rechanneling, or degradation which in turn can negatively affect the available invertebrate food source (IDFG et al. 1995; Munger et al. 1997; Reaser 1997; Engle and Munger 2000; Engle 2002). Spotted frog habitat occurs in the same areas where these activities are likely to take place or where these activities occurred in the past and resulting habitat degradation has not improved over time. Natural fluctuations in environmental conditions tend to magnify the detrimental effects of these activities, just as the activities may also magnify the detrimental effects of natural environmental events (USFWS 2002c)].

[Springs provide a stable, permanent source of water for frog breeding, feeding, and winter refugia (IDFG et al. 1995). Springs provide deep, protected areas which serve as hibernacula for spotted frogs in cold climates. Springs also provide protection from predation through underground openings (IDFG et al. 1995; Patla and Peterson 1996). Most spring developments result in the installation of a pipe or box to fully capture the water source and direct water to another location such as a livestock watering trough. Loss of this permanent source of water in desert ecosystems can also lead to the loss of associated riparian habitats and wetlands used by spotted frogs. Developed spring pools could be functioning as attractive nuisances for frogs, concentrating them into isolated groups, increasing the risk of disease and predation (Engle 2001). Many of the springs in southern Idaho, eastern Oregon, and Nevada have been developed (USFWS 2002c)].

[The reduction of beaver populations has been noted as an important feature in the reduction of suitable habitat for spotted frogs. Beaver are important in the creation of small pools with slow-moving water that function as habitat for frog reproduction and create wet meadows that provide foraging habitat and protective vegetation cover, especially in the dry interior western United States (St. John 1994). Beaver trapping is still common in Idaho and harvest is unregulated in most areas (IDFG et al. 1995). In some areas, beavers are removed because of a perceived threat to water for agriculture or horticultural plantings. As indicated above, permanent ponded waters are important in maintaining spotted frog habitats during severe drought or winter periods. Removal of a beaver dam in Stoneman Creek in Idaho is believed to be directly related to the decline of a spotted frog subpopulation there. Intensive surveying of the historical site where frogs were known to have occurred has documented only one adult spotted frog (Engle 2000) (USFWS 2002c)].

[Fragmentation of habitat may be one of the most significant barriers to spotted frog recovery and population persistence. Recent studies in Idaho indicate that spotted frogs exhibit breeding site fidelity (Patla and Peterson 1996; Engle 2000; Munger and Engle 2000; J. Engle, IDFG, pers. comm., 2001). Movement of frogs from hibernation ponds to breeding ponds may be impeded by zones of unsuitable habitat. As movement corridors become more fragmented due to loss of flows within riparian or meadow habitats, local populations will become more isolated (Engle 2000; Engle 2001). Vegetation and surface water along movement corridors provide relief from high temperatures and arid environmental conditions, as well as protection from predators. Loss of vegetation and/or lowering of the water table as a result of the above mentioned activities can pose a significant threat to frogs moving from one area to another. Likewise, fragmentation and loss of habitat can prevent frogs from colonizing suitable sites elsewhere (USFWS 2002c)].

[Though direct correlation between spotted frog declines and livestock grazing has not been studied, the effects of heavy grazing on riparian areas are well documented (Kauffman et al.

1982; Kauffman and Kreuger 1984; Skovlin 1984; Kauffman et al. 1985; Schulz and Leininger 1990). Heavy grazing in riparian areas on state and private lands is a chronic problem throughout the Great Basin. Efforts to protect spotted frog habitat on state lands in Idaho have been largely unsuccessful because of lack of cooperation from the State. In northeast Nevada, the Forest Service has completed three riparian area protection projects in areas where spotted frogs occur. These projects include altering stocking rates or changing the grazing season in two allotments known to have frogs and constructing riparian fencing on one allotment. However, these three sites have not been monitored to determine whether efforts to protect riparian habitat and spotted frogs have been successful. In the Toiyabe Range, a proposal to fence 3.2 kilometers (km) (2 miles (mi)) of damaged riparian area along Cloverdale Creek to protect it from grazing is scheduled to occur in the summer of 2002. In addition to the riparian exclosure, BLM biologists located a diversion dam in 1998 on Cloverdale Creek which was completely de- watering approximately 1.6 km (1 mi) of stream. During the summer of 2000, this area was reclaimed and water was put back into the stream. This area of the stream is not currently occupied by spotted frogs but it is historical habitat (USFWS 2002c)].

[The effects of mining on Great Basin Columbia spotted frogs, specifically, have not been studied, but the adverse effects of mining activities on water quality and quantity, other wildlife species, and amphibians in particular have been addressed in professional scientific forums (Chang et al. 1974; Birge et al. 1975; Greenhouse 1976; Khangarot et al. 1985) (USFWS 2002c)].

<u>Disease or predation</u>

[Predation by fishes is likely an important threat to spotted frogs. The introduction of nonnative salmonid and bass species for recreational fishing may have negatively affected frog species throughout the United States. The negative effects of predation of this kind are difficult to document, particularly in stream systems. However, significant negative effects of predation on frog populations in lacustrine systems have been documented (Hayes and Jennings 1986; Pilliod et al. 1996, Knapp and Matthews 2000). One historic site in southern Idaho no longer supports spotted frog although suitable habitat is available. This may be related to the presence of introduced bass in the Owyhee River (IDCDC 2000). The stocking of nonnative fishes is common throughout waters of the Great Basin. The Nevada Division of Wildlife (NDOW) has committed to conducting stomach sampling of stocked nonnative and native species to determine the effects of predation on spotted frogs. However, this commitment will not be fulfilled until the spotted frog conservation agreements are signed. To date, NDOW has not altered fish stocking rates or locations in order to benefit spotted frogs (USFWS 2002c)].

[The bull frog (*Rana catesbeiana*), a nonnative ranid species, occurs within the range of the spotted frog in the Great Basin. Bullfrogs are known to prey on other frogs (Hayes and Jennings 1986). They are rarely found to co-occur with spotted frogs, but whether this is an artifact of competitive exclusion is unknown at this time (USFWS 2002c)].

[Although a diversity of microbial species is naturally associated with amphibians, it is generally accepted that they are rarely pathogenic to amphibians except under stressful environmental conditions. *Chytridiomycosis* (chytrid) is an emerging panzootic fungal disease in the United States (Fellers et al. 2001). Clinical signs of amphibian chytrid include abnormal posture, lethargy, and loss of righting reflex. Gross lesions, which are usually not apparent, consist of abnormal epidermal sloughing and ulceration; hemorrhages in the skin, muscle, or eye; hyperemia of digital and ventrum skin, and congestion of viscera. Diagnosis is by identification of characteristic intracellular flask-shaped sporangia and septate thalli within the epidermis. Chytrid can be identified in some species of frogs by examining the oral discs of tadpoles which may be abnormally formed or lacking pigment (Fellers et al. 2001) (USFWS 2002c)].

[Chytrid was confirmed in the Circle Pond site, Idaho, where long term monitoring since 1998 has indicated a general decline in the population (Engle 2002). It is unclear whether the presence of this disease will eventually result in the loss of this subpopulation. Two additional sites may have chytrid, but this has yet to be determined (J. Engle, pers. comm., 2001). Protocols to prevent further spread of the disease by researchers were instituted in 2001. Chytrid has also been found in the Wasatch Columbia spotted frog distinct population segment (K. Wilson, pers comm., 2002). Chytrid has not been found in Nevada populations of spotted frogs (USFWS 2002c)].

<u>The inadequacy of existing regulatory mechanisms</u>
[Spotted frog occurrence sites and potential habitats occur on both public and private lands. This species is included on the Forest Service sensitive species list; as such, its management must be considered during forest planning processes. However, little habitat restoration, monitoring or surveying has occurred on Forest Service lands (USFWS 2002c)].

[In the fall of 2000, 250 head of cattle were allowed to graze for 45 days on one pasture in the Indian Valley Creek drainage of the Humboldt-Toiyabe National Forest in central Nevada for the first time in 6 years (M. Croxen, pers. comm., 2002). Grazing was not allowed in this allotment in 2001. Recent mark-recapture data indicated that this drainage supports more frogs than previously presumed, potentially around 5,000 individuals (K. Hatch, pers. comm., 2000). Perceived improvements in the status of frog populations in the Indian Valley Creek area may be a result of past removal of livestock grazing. The reintroduction of grazing disturbance into this relatively dense area of frogs has yet to be determined (USFWS 2002c)].

[BLM policies direct management to consider candidate species on public lands under their jurisdiction. To date, BLM efforts to conserve spotted frogs and their habitat in Idaho, Oregon, and Nevada have not been adequate to address threats (USFWS 2002c)].

[The southernmost known population of spotted frogs can be found on the BLM San Antone Allotment south of Indian Valley Creek in the Toiyabe Range. Grazing is allowed in this area from November until June (L. Brown, pers. comm., 2002). The season of use is a very sensitive portion of the spotted frog annual life cycle which includes migration from winter hibernacula to breeding ponds, breeding, egg laying and hatching, and metamorphosing of young. Additionally, the riparian Standards and Guidelines were not met in 1996, the last time the allotment was evaluated (USFWS 2002c)].

[The status of local populations of spotted frogs on Yomba-Shoshone or Duck Valley Tribal lands is unknown. Tribal governments do not have regulatory or protective mechanisms in place to protect spotted frogs (USFWS 2002c)].

[The Nevada Division of Wildlife classifies the spotted frog as a protected species, but they are not afforded official protection and populations are not monitored. Though the spotted frog is on the sensitive species list for the State of Idaho, this species is not given any special protection by the State. Columbia spotted frogs are not on the sensitive species list for the State of Oregon. Protection of wetland habitat from loss of water to irrigation or spring development is difficult because most water in the Great Basin has been allocated to water rights applicants based on historical use and spring development has already occurred within much of the known habitat of spotted frogs. Federal lands may have water rights that are approved for wildlife use, but these rights are often superceded by historic rights upstream or downstream that do not provide for minimum flows. Also, most public lands are managed for multiple use and are subject to livestock grazing, silvicultural activities, and recreation uses that may be incompatible with spotted frog conservation without adequate mitigation measures (USFWS 2002c)].

<u>Other natural or manmade factors affecting its continued existence</u>
[Multiple consecutive years of less than average precipitation may result in a reduction in the number of suitable sites available to spotted frogs. Local extirpations eliminate source populations from habitats that in normal years are available as frog habitat (Lande and Barrowclough 1987; Schaffer 1987; Gotelli 1995). These climate events are likely to exacerbate the effects of other threats, thus increasing the possibility of stochastic extinction of subpopulations by reducing their size and connectedness to other subpopulations (see Factor A for additional information). As movement corridors become more fragmented, due to loss of flows within riparian or meadow habitats, local populations will become more isolated (Engle 2000). Increased fragmentation of the habitat can lead to greater loss of populations due to demographic and/or environmental stochasticity (USFWS 2002c)].

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YELLOW WARBLER
Dendroica petechia

Original Species Account Authors: Paul Ashley and Stacy Stoval, as appeared in the Southeast Washington Ecoregional Assessment, January 2004

Introduction
The yellow warbler (*Dendroica petechia*) is a common species strongly associated with riparian and wet deciduous habitats throughout its North American range. In Washington it is found in many areas, generally at lower elevations. It occurs along most riverine systems, including the Columbia River, where appropriate riparian habitats have been protected. The yellow warbler is a good indicator of functional subcanopy/shrub habitats in riparian areas.

Life History, Key Environmental Correlates, and Habitat Requirements
Life History
Diet
Yellow warblers capture and consume a variety of insect and arthropod species. The species taken vary geographically. Yellow warblers consume insects and occasionally wild berries (Lowther *et al.* 1999). Food is obtained by gleaning from subcanopy vegetation; the species also sallies and hovers to a much lesser extent (Lowther *et al.* 1999) capturing a variety of flying insects.

Reproduction
Although little is known about yellow warbler breeding behavior in Washington, substantial information is available from other parts of its range. Pair formation and nest construction may begin within a few days of arrival at the breeding site (Lowther *et al.* 1999). The reproductive process begins with a fairly elaborate courtship performed by the male who may sing up to 3,240 songs in a day to attract a mate. The responsibility of incubation, construction of the nest and most feeding of the young lies with the female, while the male contributes more as the young develop. In most cases only one clutch of eggs is laid; renesting may occur, however, following nest failure or nest parasitism by brown-headed cowbirds (Lowther *et al.* 1999). The typical clutch size ranges between 4 and 5 eggs in most research studies of the species (Lowther *et al.* 1999). Egg dates have been reported from British Columbia, and range between 10 May and 16 August; the peak period of activity there was between 7 and 23 June (Campbell *et al.* in press). The incubation period lasts about 11 days and young birds fledge 8-10 days after hatching (Lowther *et al.* 1999). Young of the year may associate with the parents for up to 3 weeks following fledging (Lowther *et al.* 1999).

Nesting
Results of research on breeding activities indicate variable rates of hatching and fledging. Two studies cited by Lowther *et al.* (1999) had hatching rates of 56 percent and 67 percent. Of the eggs that hatched, 62 percent and 81 percent fledged; this represented 35 percent and 54 percent, respectively, of all eggs laid. Two other studies found that 42 percent and 72 percent of nests fledged at least one young (Lowther *et al.* 1999); the latter study was from British Columbia (Campbell *et al.* in press).

Migration
The yellow warbler is a long-distance neotropical migrant. Spring migrants begin to arrive in the region in April. Early dates of 2 April and 10 April have been reported from Oregon and British Columbia, respectively (Gilligan *et al.* 1994, Campbell *et al.* in press). Average arrival dates are somewhat later, the average for south-central British Columbia being 11 May (Campbell *et al.* in press). The peak of spring migration in the region is in late May (Gilligan *et al.* 1994). Southward migration begins in late July, and peaks in late August to early September; very few migrants remain in the region in October (Lowther *et al.* 1999).

Mortality
Little has been published on annual survival rates. Roberts (1971) estimated annual survival rates of adults at 0.526 ±0.077 SE, although Lowther *et al.* (1999) felt this value underestimated survival because it did not account for dispersal. The oldest yellow warbler on record lived to be nearly 9 years old (Klimkiewicz *et al.* 1983).

Yellow warblers have developed effective responses to nest parasitism by the brown-headed cowbird (*Molothrus ater*). The brown-headed cowbird is an obligate nest brood parasite that does not build a nest and instead lays eggs in the nests of other species. When cowbird eggs are recognized in the nest the yellow warbler female will often build a new nest directly on top of the original. In some cases, particularly early in the incubation phase, the female yellow warbler will bury the cowbird egg within the nest. Some nests are completely abandoned after a cowbird egg is laid (Lowther *et al.* 1999). Up to 40 percent of yellow warbler nests in some studies have been parasitized (Lowther *et al.* 1999).

Habitat Requirements
The yellow warbler is a riparian obligate species most strongly associated with wetland habitats and deciduous tree cover. Yellow warbler abundance is positively associated with deciduous tree basal area, and bare ground; abundance is negatively associated with mean canopy cover, and cover of Douglas-fir (*Pseudotsuga menziesii*), Oregon grape (*Berberis nervosa*), mosses, swordfern (*Polystuchum munitum*), blackberry (*Rubus discolor*), hazel (*Corylus cornuta*), and oceanspray (*Holodiscus discolor*) (Rolph 1998).

Partners in Flight have established biological objectives for this species in the lowlands of western Oregon and western Washington. These include providing habitats that meet the following definition: >70 percent cover in shrub layer (<3 m) and subcanopy layer (>3 m and below the canopy foliage) with subcanopy layer contributing >40 percent of the total; shrub layer cover 30-60 percent (includes shrubs and small saplings); and a shrub layer height >2 m. At the landscape level, the biological objectives for habitat included high degree of deciduous riparian heterogeneity within or among wetland, shrub, and woodland patches; and a low percentage of agricultural land use (Altman 2001).

Nesting
Radke (1984) found that nesting yellow warblers occurred more in isolated patches or small areas of willows adjacent to open habitats or large, dense thickets (i.e., scattered cover) rather

than in the dense thickets themselves. At Malheur National Wildlife Refuge, in the northern Great Basin, nest success 44 percent (n = 27), however, cowbird eggs and young removed; cowbird parasitism 33 percent (n = 9) (Radke 1984).

Breeding
Breeding yellow warblers are closely associated with riparian hardwood trees, specifically willows, alders, or cottonwood. They are most abundant in riparian areas in the lowlands of eastern Washington, but also occur in west-side riparian zones, in the lowlands of the western Olympic Peninsula, where high rainfall limits hardwood riparian habitat. Yellow warblers are less common (Sharpe 1993). There are no BBA records at the probable or confirmed level from subalpine habitats in the Cascades, but Sharpe (1993) reports them nesting at 4000 feet in the Olympics. Numbers decline in the center of the Columbia Basin, but this species can be found commonly along most rivers and creeks at the margins of the Basin. A local breeding population exists in the Potholes area.

Non-breeding
Fall migration is somewhat inconspicuous for the yellow warbler. It most probably begins to migrate the first of August and is generally finished by the end of September. The yellow warbler winters south to the Bahamas, northern Mexico, south to Peru, Bolivia and the Brazilian Amazon.

Yellow Warbler Population and Distribution
Population
Historic
No historic data could be found for this species.

Current
No current data could be found for this species.

Distribution
Historic
Jewett *et al.* (1953) described the distribution of the yellow warbler as a common migrant and summer resident from April 30 to September 20 in the deciduous growth of Upper Sonoran and Transition Zones in eastern Washington and in the prairies and along streams in southwestern Washington. They describe its summer range as north to Neah Bay, Blaine, San Juan Islands, Monument 83; east to Conconully, Swan Lake, Sprague, Dalkena, and Pullman; south to Cathlamet, Vancouver and Bly, Blue Mts., Prescott, Richland, and Rogersburg; and west to Neah Bay, Grays Harbor, and Long Beach. Jewett *et al. (*1953) also note that the yellow warbler was common in the willows and alders along the streams of southeastern Washington and occurs also in brushy thickets. They state that its breeding range follows the deciduous timber into the mountains, where it probably nests in suitable habitat to 3,500 or perhaps even to 4,000 feet – being common at Hart Lake in the Chelan region around 4,000 feet. They noted it was a common nester along the Grande Ronde River, around the vicinity of Spokane, around Sylvan Lake, and along the shade trees along the streets of Walla Walla.

Current

The yellow warbler breeds across much of the North American continent, from Alaska to Newfoundland, south to western South Carolina and northern Georgia, and west through parts of the southwest to the Pacific coast (AOU 1998). Browning (1994) recognized 43 subspecies; two of these occur in Washington, and one of them, *D.p. brewsteri*, is found in western Washington. This species is a long-distance migrant and has a winter range extending from western Mexico south to the Amazon lowlands in Brazil (AOU 1998). Neither the breeding nor winter ranges appear to have changed (Lowther *et al.* 1999).

The yellow warbler is a common breeder in riparian habitats with hardwood trees throughout the state at lower elevations. It is a locally common breeder along rivers and creeks in the Columbia Basin, where it is declining in some areas. Core zones of distribution in Washington are the forested zones below the subalpine fir and mountain hemlock zones, plus steppe zones other than the central arid steppe and canyon grassland zones, which are peripheral.

The yellow warbler breeds across much of the North American continent, from Alaska to Newfoundland, south to western South Carolina and northern Georgia, and west through parts of the southwest to the Pacific coast (AOU 1998).

Non-Breeding

This data is not readily available; however, the yellow warbler is a long-range neotropical migrant. Its winter range is from Northern Mexico south to Northern Peru.

Status and Abundance Trends

Status

Yellow warblers are demonstrably secure globally. Within the state of Washington, yellow warblers are apparently secure and are not of conservation concern (Altman 1999).

Trends

Yellow warbler is one of the more common warblers in North America (Lowther *et al.* 1999). Information from Breeding Bird Surveys indicates that the population is stable in most areas. Some subspecies, particularly in southwestern North America, have been impacted by degradation or destruction of riparian habitats (Lowther *et al.* 1999). Because the Breeding Bird Survey dates back only about 30 years, population declines in Washington resulting from habitat loss dating prior to the survey would not be accounted for by that effort.

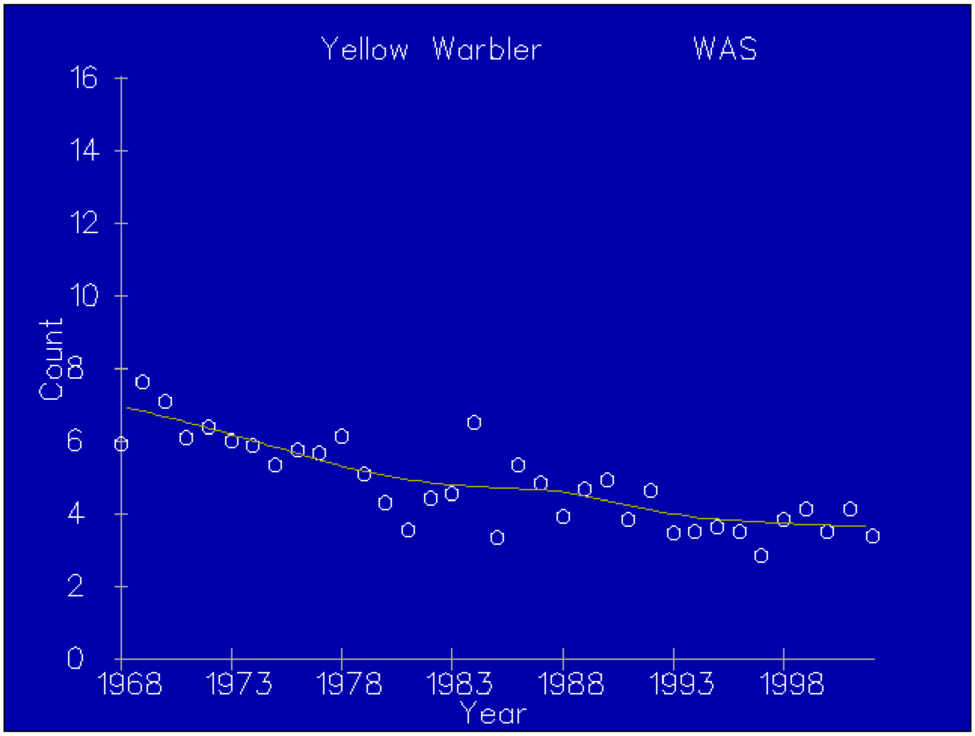

Figure 6. Breeding Bird Survey data for Washington State show a significant population decline of 2.9 percent per year ($p < .1$) from 1966 to 1991 (Peterjohn 1991).

Factors Affecting Yellow Warbler Population Status

Key Factors Inhibiting Populations and Ecological Processes

Habitat loss due to hydrological diversions and control of natural flooding regimes (e.g., dams) resulting in reduction of overall area of riparian habitat, conversion of riparian habitats, inundation from impoundments, cutting and spraying for ease of access to water courses, gravel mining, etc.

Habitat degradation from: loss of vertical stratification in riparian vegetation, lack of recruitment of young cottonwoods, ash, willows, and other subcanopy species; stream bank stabilization (e.g., riprap) which narrows stream channel, reduces the flood zone, and reduces extent of riparian vegetation; invasion of exotic species such as reed canary grass and blackberry; overgrazing which can reduce understory cover; reductions in riparian corridor widths which may decrease suitability of the habitat and may increase encroachment of nest predators and nest parasites to the interior of the stand.

Hostile landscapes, particularly those in proximity to agricultural and residential areas, may have high density of nest parasites (brown-headed cowbird) and domestic predators (cats), and be subject to high levels of human disturbance.

Recreational disturbances, particularly during nesting season, and particularly in high-use recreation areas.

Increased use of pesticide and herbicides associated with agricultural practices may reduce insect food base.

Out-of-Subbasin Effects and Assumptions

No data could be found on the migration and wintering grounds of the yellow warbler. It is a long-distance migrant and as a result faces a complex set of potential effects during it annual cycle. Habitat loss or conversions is likely happening along its entire migration route (H. Ferguson, WDFW, pers. comm. 2003). Riparian management requires the protection of riparian shrubs and understory and the elimination of noxious weeds. Migration routes, corridors and wintering grounds need to be identified and protected just as its breeding areas. In addition to loss of habitat, the yellow warbler, like many wetland or riparian associated birds, faces increased pesticide use in the metropolitan areas, especially with the outbreak of mosquito born viruses like West Nile Virus.

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AMERICAN BEAVER
Castor canadensis

Original Species Account Authors: Paul Ashley and Stacy Stoval, as appeared in the Southeast Washington Ecoregional Assessment, January 2004

Introduction
The American beaver (*Castor canadensis*) is a large, highly specialized aquatic rodent found in the immediate vicinity of aquatic habitats (Hoffman and Pattie 1968). The species occurs in streams, ponds, and the margins of large lakes throughout North America, except for peninsular Florida, the Arctic tundra, and the southwestern deserts (Jenkins and Busher 1979). In Oregon, beavers can be found in suitable habitats throughout the state (Verts and Carraway 1998). Beavers construct elaborate lodges and burrows and store food for winter use. The species is active throughout the year and is usually nocturnal in its activities. Adult beavers are nonmigratory.

Life History, Key Environmental Correlates, and Habitat Requirements
Life History
Diet
Beavers are exclusively vegetarian in diet. A favorite food item is the cambial, or growing, layer of tissue just under the bark of shrubs and trees. Many of the trees that are cut are stripped of bark, or carried to the pond for storage under water as a winter food cache. Buds and roots are also consumed, and when they are needed, a variety of plant species are accepted. The animals may travel some distance from water to secure food. When a rich food source is exploited, canals may be dug from the pond to the pasture to facilitate the transportation of the items to the lodge.

Much of the food ingested by a beaver consists of cellulose, which is normally indigestible by mammals. However, these animals have colonies of microorganisms living in the cecum, a pouch between the large and small intestine, and these symbionts digest up to 30 percent of the cellulose that the beaver takes in. An additional recycling of plant food occurs when certain fecal pellets are eaten and run through the digestive process a second time (Findley 1987). Woody and herbaceous vegetation comprise the diet of the beaver. Herbaceous vegetation is a highly preferred food source throughout the year, if it is available.

Woody vegetation may be consumed during any season, although its highest utilization occurs from late fall through early spring. It is assumed that woody vegetation (trees and/or shrubs) is more limiting than herbaceous vegetation in providing an adequate food source. In summer, a variety of green herbaceous vegetation, especially aquatic species, is eaten (Jenkins and Busher 1979; Svendsen 1980, cited in Verts and Carraway 1998). In autumn and winter as green herbaceous vegetation disappears, beavers shift their diet to stems, leaves, twigs, and bark of many of the woody species that grow near the water (Verts and Carraway 1998). Bulbous roots of aquatic species also may be eaten in winter (Beer 1942, cited in Verts and Carraway 1998). Beavers cut mostly deciduous trees such as cottonwood, will, alder, maple, and birch, but in some regions, coniferous species may be used (Jenkins 1979, cited in Verts and Carraway 1998).

Denney (1952) summarized the food preferences of beavers throughout North America and reported that, in order of preference, beavers selected aspen (*Populus tremuloides*), willow (*Salix spp.*), cottonwood (*P. balsamifera*), and alder (*Alnus spp*.). Although several tree species have often been reported to be highly preferred foods, beavers can inhabit, and often thrive in, areas where these tree species are uncommon or absent (Jenkins 1975). Aspen and willow are considered preferred beaver foods; however, these are generally riparian tree species that may be more available for beaver foraging but are not necessarily preferred over all other deciduous tree species (Jenkins 1981). In southeastern Oregon, riparian-zone trees have been reduced or eliminated in many areas by browsing herbivores. However, comparison of growth of red willow (*Salix lasiandra*) in an area inaccessible to cattle but occupied by beavers with that in an area inaccessible to both cattle and beavers, indicated that beavers were not responsible for the deterioration. Although beavers harvested 82% of available stems annually, they cut them at a season after growth was completed and reserves were translocated to roots. Subsequent growth of cut willows increase exponentially in relation to the proportion of the stems cut by beavers (Kindschy 1985, cited in Verts and Carraway 1998).

Beavers have been reported to subsist in some areas by feeding on coniferous trees, generally considered a poor quality source of food (Brenner 1962; Williams 1965). Major winter foods in North Dakota consisted principally of red-osier dogwood (*Cornus stolonifera*), green ash (*Fraxinus pennsylvanica*), and willow (Hammond 1943). Rhizomes and roots of aquatic vegetation also may be an important source of winter food (Longley and Moyle 1963; Jenkins pers. comm.). The types of food species present may be less important in determining habitat quality for beavers than physiographic and hydrologic factors affecting the site (Jenkins 1981).

Aquatic vegetation, such as duck potato (*Sagittaria spp.),* duckweed (*Lemna spp.),* pondweed (*Potamogeton spp.),* and water weed (Elodea spp.), are preferred foods when available (Collins 1976a). Water lilies (*Nymphaea* spp.), with thick, fleshy rhizomes, may be used as a food source throughout the year (Jenkins 1981). If present in adequate amounts, water lily rhizomes may provide an adequate winter food source, resulting in little or no tree cutting or food caching of woody materials. Jenkins (1981) compared the rate of tree cutting by beavers adjacent to two Massachusetts ponds that contained stands of water lilies. A pond dominated by yellow water lily (*Y. variegatum*) and white water lily (*N. odorata*), which have thick rhizomes, had low and constant tree cutting activity throughout the fall. Conversely, the second pond, dominated by watershield (*Brasenia schreberi*), which lacks thick rhizomes, had increased fall tree cutting activity by beavers.

Reproduction

The basic composition of a beaver colony is the extended family, comprised of a monogamous pair of adults, subadults (young of the previous year), and young of the year (Svendsen 1980). Female beavers are sexually mature at 2.5 years old. Females normally produce litters of three to four young with most kits being born during May and June. Gestation is approximately 107 days (Linzey 1998). Kits are born with all of their fur, their eyes open, and their incisor teeth erupted.

Dispersal of subadults occurs during the late winter or early spring of their second year and coincides with the increased runoff from snowmelt or spring rains. Subadult beavers have been reported to disperse as far as 236 stream km (147 mi) (Hibbard 1958), although average emigration distances range from 8 to 16 stream km (5 to 10 mi) (Hodgdon and Hunt 1953; Townsend 1953; Hibbard 1958; Leege 1968). The daily movement patterns of the beaver centers around the lodge or burrow and pond (Rutherford 1964). The density of colonies in favorable habitat ranges from 0.4 to 0.8/km2 (1 to 2/mi2) (Lawrence 1954; Aleksiuk 1968; Voigt *et al.* 1976; Bergerud and Miller 1977 cited by Jenkins and Busher 1979).

Home Range

The mean distance between beaver colonies in an Alaskan riverine habitat was 1.59 km (1 mi) (Boyce 1981). The closest neighbor was 0.48 km (0.3 mi) away. The size of the colony's feeding range is a function of the interaction between the availability of food and water and the colony size (Brenner 1967). The average feeding range size in Pennsylvania, excluding water, was reported to be 0.56 ha (1.4 acre). The home range of beaver in the Northwest Territory was estimated as a 0.8 km (0.5 mi) radius of the lodge (Aleksiuk 1968). The maximum foraging distance from a food cache in an Alaskan riverine habitat was approximately 800 m (874 yds) upstream, 300 m (323 yds) downstream, and 600 m (656 yds) on oxbows and sloughs (Boyce 1981).

Mortality

Beavers live up to 11 years in the wild, 15 to 21 years in captivity (Merritt 1987, Rue 1967). Beavers have few natural predators. However, in certain areas, beavers may face predation pressure from wolves (*Canis lupus*), coyotes (*Canis latrans*), lynx (*Felis lynx*), fishers (*Martes pennanti*), wolverines (*Gulo gulo*), and occasionally bears (*Ursus spp*.). Alligators, minks (*Mustela vison*), otters (*Lutra canadensis*), hawks, and owls periodically prey on kits (Lowery 1974, Merritt 1987, Rue 1967).

Beavers often carry external parasites, one of which, *Platypsylla castoris*, is a beetle found only on beavers.

Harvest

Historic

Because of the high commercial value of their pelts, beavers figured importantly in the early exploration and settlement of western North America. Thousands of their pelts were harvested annually, and it was not many years before beavers were either exterminated entirely or reduced to very low populations over a considerable part of their former range. By 1910 their populations were so low everywhere in the United States that strict regulation of the harvest or complete protection became imperative. In the 1930s live trapping and restocking of depleted areas became a widespread practice which, when coupled with adequate protection, has made it possible for the animals to make a spectacular comeback in many sections.

Current

Harvest of beavers in Oregon between 1969 and 1992 per 1,000 hectares in Union and Wallowa Counties were <1 and 1-10 respectively (ODFW, annual reports, cited in Verts and Carraway 1998). Trapping was terminated by initiative in Washington. No commercial or recreational trapping of beaver occurs in southeast Washington. Between 1991 and 1999, the beaver harvest in the four counties of southeast Washington ranged from 56 to 162/year, and averaged 107/year. Since the initiative to ban trapping, the beaver harvest has declined 95%, and has averaged about

5/year for southeast Washington. As a result of the declining harvest, populations appear to be increasing along with complaints from landowners. Beavers have become a problem in some tributaries, damming farm irrigation and causing problems for fish passage.

Harvest trends will not indicate population trend, because the price of beaver pelts often determines the level of harvest. The higher the pelt price, the higher the harvest because trappers put more effort into trapping beaver. If pelt prices are low, little effort is expended to trap beaver, regardless of population size.

Habitat Requirements

The beaver almost always is associated with riparian or lacustrine habitats bordered by a zone of trees, especially cottonwood and aspen (Populus), willow (Salix), alder (Alnus), and maple (Acer) (Verts and Carraway 1998). Small streams with a constant flow of water that meander through relatively flat terrain in fertile valleys and rare subject to being dammed seem especially productive of beavers (Hill 1982, cited in Verts and Carraway 1998). Streams with rocky bottoms through steep terrain and more subject to wide fluctuations in water levels are less suitable to beavers. In large lakes with broad expanses subject to extensive wave action, beavers usually are restricted to protected inlets (Verts and Carraway 1998).

All wetland cover types (e.g., herbaceous wetland and deciduous forested wetland) must have a permanent source of surface water with little or no fluctuation in order to provide suitable beaver habitat (Slough and Sadleir 1977). Water provides cover for the feeding and reproductive activities of the beaver. Lakes and reservoirs that have extreme annual or seasonal fluctuations in the water level will be unsuitable habitat for beaver. Similarly, intermittent streams, or streams that have major fluctuations in discharge (e.g., high spring runoff) or a stream channel gradient of 15 percent or more, will have little year-round value as beaver habitat. Assuming that there is an adequate food source available, small lakes [< 8 ha (20 acres) in surface area] are assumed to provide suitable habitat. Large lakes and reservoirs [> 8 ha (20 acres) in surface area] must have irregular shorelines (e.g., bays, coves, and inlets) in order to provide optimum habitat for beaver.

Beavers can usually control water depth and stability on small streams, ponds, and lakes; however, larger rivers and lakes where water depth and/or fluctuation cannot be controlled are often partially or wholly unsuitable for the species (Murray 1961; Slough and Sadleir 1977). Rivers or streams that are dry during some parts of the year are assumed to be unsuitable beaver habitat. Beavers are absent from sizable portions of rivers in Wyoming, due to swift water and an bsence of suitable dwelling sites during periods of high and low water levels (Collins 1976b).

In riverine habitats, stream gradient is the major determinant of stream morphology and the most significant factor in determining the suitability of habitat for beavers (Slough and Sadleir 1977). Stream channel gradients of 6 percent or less have optimum value as beaver habitat. Retzer *et al.* (1956) reported that 68 percent of the beaver colonies recorded in Colorado were in valleys with a stream gradient of less than 6 percent, 28 percent were associated with stream gradients from 7 to 12 percent, and only 4 percent were located along streams with gradients of 13 to 14 percent. No beaver colonies were recorded in streams with a gradient of 15 percent or more. Valleys that were only as wide as the stream channel were unsuitable beaver habitat, while valleys wider than the stream channel were frequently occupied by beavers. Valley widths of 46 m (150 ft) or more were considered the most suitable. Marshes, ponds, and lakes were nearly always occupied by beavers when an adequate supply of food was available.

Foraging

Beavers are generalized herbivores; however, they show strong preferences for particular plant species and size classes (Jenkins 1975; Collins 1975a; Jenkins 1979). The leaves, twigs, and bark f woody plants are eaten, as well as many species of aquatic and terrestrial herbaceous vegetation. Food preferences may vary seasonally, or from year to year, as a result of variation in the nutritional value of food sources (Jenkins 1979).

An adequate and accessible supply of food must be present for the establishment of a beaver colony (Slough and Sadleir 1977). The actual biomass of herbaceous vegetation will probably not limit the potential of an area to support a beaver colony (Boyce 1981). However, total biomass of winter food cache plants (woody plants) may be limiting. Low marshy areas and streams flowing in and out of lakes allow the channelization and damming of water, allowing access to, and transportation of, food materials. Steep topography prevents the establishment of a food transportation system (Williams 1965; Slough and Sadleir 1977). Trees and shrubs closest to the pond or stream periphery are generally utilized first (Brenner 1962; Rue 1964). Jenkins (1980) reported that most of the trees utilized by beaver in his Massachusetts study area were within 30 m (98.4 ft) of the water's edge. However, some foraging did extend up to 100 m (328 ft). Foraging distances of up to 200 m (656 ft) have been reported (Bradt 1938). In a California study, 90 percent of all cutting of woody material was within 30 m (98.4 ft) of the water's edge (Hall 1970).

Woody stems cut by beavers are usually less than 7.6 to 10.1 cm (3 to 4 inches) DBH (Bradt 1947; Hodgdon and Hunt 1953; Longley and Moyle 1963; Nixon and Ely 1969). Jenkins (1980) reported a decrease in mean stem size cut and greater selectivity for size and species with increasing distance from the water's edge. Trees of all size classes were felled close to the water's edge, while only smaller diameter trees were felled farther from the shore.

Beavers rely largely on herbaceous vegetation, or on the leaves and twigs of woody vegetation, during the summer (Bradt 1938, 1947; Brenner 1962; Longley and Moyle 1963; Brenner 1967; Aleksiuk 1970; Jenkins 1981). Forbs and grasses comprised 30 percent of the summer diet in Wyoming (Collins 1976a). Beavers appear to prefer herbaceous vegetation over woody vegetation during all seasons of the year, if it is available (Jenkins 1981).

Cover

Lodges or burrows, or both, may be used by beavers for cover (Rue 1964). Lodges may be surrounded by water or constructed against a bank or over the entrance to a bank burrow. Water protects the lodges from predators and provides concealment for the beaver when traveling to and from food gathering areas and caches.

The lodge is the major source of escape, resting, thermal, and reproductive cover (Jenkins and Busher 1979). Mud and debarked tree stems and limbs are the major materials used in lodge construction although lesser amounts of other woody, as well as herbaceous vegetation, may be used (Rue 1964). If an unexploited food source is available, beavers will reoccupy abandoned lodges rather than build new ones (Slough and Sadleir 1977). On lakes and ponds, lodges are frequently situated in areas that provide shelter from wind, wave, and ice action. A convoluted shoreline, which prevents the buildup of large waves or provides refuge from waves, is a habitat requirement for beaver colony sites on large lakes.

Population and Distribution
Population
Historic
Historically, beaver populations were more expansive until populations were reduced by unregulated trapping, as they were throughout much of the western United States (P. Fowler, WDFW, personal communications, 2003).

Current
Beaver populations exist in all major watersheds in the Blue Mountains. In the Walla Walla subbasin, beaver can be found in the Walla Walla and Touchet River drainages; Mill Creek, Coppei Creek, North Touchet, South Touchet. Beaver can be found in the Tucannon subbasin in the Tucannon River and its tributaries. Beaver can be found in the Asotin watershed, Asotin Creek and its tributaries. Beaver also occur in the Snake River.

Captive Breeding Programs, Transplants, Introductions
Historic
No data are available.

Current
No data are available.

Distribution
Historic
No data are available.

Current
The beaver is found throughout most of North America except in the Arctic tundra, peninsular Florida, and the Southwestern deserts (Allen 1983; VanGelden 1982; Zeveloff 1988).

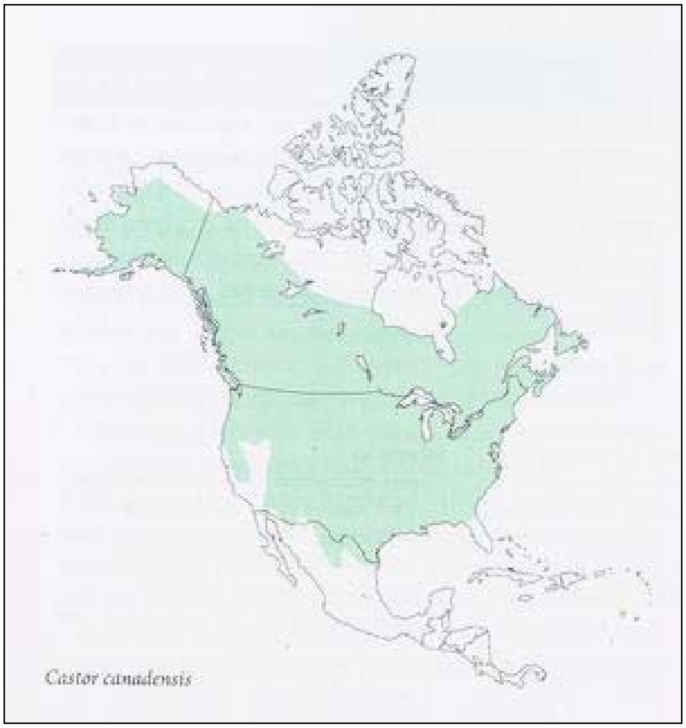

Figure 7. Geographic distribution of American beaver (*Castor canadensis*) (From Linzey and Brecht 2002).

Status and Abundance Trends
Status
Status is generally unknown, but beaver populations appear to be stable or increasing slightly in southeast Washington (P. Fowler, WDFW, personal communication, 2003).

Trends
Trend information is not available. No population data is available for northeast Oregon.

Factors Affecting American Beaver Population Status
Agriculture. Riparian habitat along many water ways has been removed in order to plant agricultural crops, thus removing important habitat and food sources for beaver in northeast Oregon.

Agricultural Conflict. Beaver may be removed when complaints are received from farmers about blocked irrigation canals or pumps.

Conflict with Fisheries. Beaver sometimes create dams that restrict fish passage, and are removed in order to restore fish passage. Beaver cutting tree planted to improve riparian habitat have also been removed.

Key Factors Inhibiting Populations and Ecological Processes
No data are available.

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Great Blue Heron
Ardea herodias

Original Species Account Authors: Paul Ashley and Stacy Stoval, as appeared in the Southeast Washington Ecoregional Assessment, January 2004

Introduction
The great blue heron (*Ardea herodias*) is the largest, most widely distributed, and best known of the American herons (Henny 1972). Great blue herons occur in a variety of habitats from freshwater lakes and rivers to brackish marshes, lagoons, mangrove areas, and coastal wetlands (Spendelow and Patton in prep.).

Life History, Key Environmental Correlates, and Habitat Requirements
Life History
Diet
Fish are preferred food items of the great blue heron in both inland and coastal waters (Kirkpatrick 1940; Palmer 1962; Kelsall and Simpson 1980), although a large variety of dietary items has been recorded. Frogs and toads, tadpoles and newts, snakes, lizards, crocodilians, rodents and other mammals, birds, aquatic and land insects, crabs, crayfish, snails, freshwater and marine fish, and carrion have all been reported as dietary items for the great blue heron (Bent 1926; Roberts 1936; Martin *et al.* 1951; Krebs 1974; Kushlan1978). Fish up to about 20 cm in length dominated the diet of herons foraging in southwestern Lake Erie (Hoffman 1978). Ninety-five percent of the fish eaten in a Wisconsin study were 25 cm in length (Kirkpatrick 1940).

Great blue herons feed alone or occasionally in flocks. Solitary feeders may actively defend a much larger feeding territory than do feeders in a flock (Meyerriecks 1962; Kushlan 1978). Flock feeding may increase the likelihood of successful foraging (Krebs 1974; Kushlan 1978) and usually occurs in areas of high prey density where food resources cannot effectively be defended.

In southeast Washington, blue herons are often seen hunting along rivers and streams. In the winter months they are often seen hunting rodents in alfalfa fields (P. Fowler, WDFW, pers. comm.. 2003).

Reproduction
The great blue heron typically breeds during the months of March - May in its northern range and November through April in the southern hemisphere. The nest usually consists of an egg clutch between 3-7 eggs, with clutch size increasing from south to north. Chicks fledge at about two months.

Nesting
Great blue herons normally nest near the tree tops. Usually, nests are about 1 m in diameter and have a central cavity 10 cm deep with a radius of 15 cm. This internal cavity is sometimes lined with twigs, moss, lichens, or conifer needles. Great blue herons are inclined to renest in the same area year after year. Old nests may be enlarged and reused (Eckert 1981).

The male gathers nest-building materials around the nest site, from live or dead trees, from neighboring nests, or along the ground, and the female works them into the nest. Ordinarily, a pair takes less than a week to build a nest solid enough for eggs to be laid and incubated. Construction continues during almost the entire nesting period. Twigs are added mostly when the eggs are being laid or when they hatch. Incubation, which is shared by both partners, starts with the laying of the first egg and lasts about 28 days. Males incubate during the days and females at night.

Herons are particularly sensitive to disturbance while nesting. Scientists suggest as a general rule that there should be no development within 300 m of the edge of a heron colony and no disturbance in or near colonies from March to August.

Mortality
The great blue heron lives as long as 17 years. The adult birds have few natural enemies. Birds of prey occasionally attack them, but these predators are not an important limiting factor on the heron population. Draining of marshes and destruction of wetland habitat is the most serious threat. The number of herons breeding in a local area is directly related to the amount of feeding habitat.

Mortality of the young is high: both the eggs and young are preyed upon by crows, ravens, gulls, birds of prey, and raccoons. Heavy rains and cold weather at the time of hatching also take a heavy toll. Pesticides are suspected of causing reproductive failures and deaths, although data obtained up to this time suggest that toxic chemicals have not caused any decline in overall population levels.

Habitat Requirements
Minimum Habitat Area
Minimum habitat area is defined as the minimum amount of contiguous habitat that is required before a species will live and reproduce in an area. Minimum habitat area for the great blue heron includes wooded areas suitable for colonial nesting and wetlands within a specified distance of the heronry where foraging can occur. A heronry frequently consists of a relatively small area of suitable habitat. For example, heronries in the Chippewa National Forest, Minnesota, ranged from 0.4 t o 4.8 ha in size and averaged 1.2 ha (Mathisen and Richards 1978). Twelve heronries in western Oregon ranged from 0.12 t o 1.2 ha in size and averaged 0.4 ha (Werschkul *et al.* 1977).

Foraging
Short and Cooper (1985) provide criteria for suitable great blue heron foraging habitat. Suitable great blue heron foraging habitats are within 1.0 km of heronries or potential heronries. The suitability of herbaceous wetland, scrub-shrub wetland, forested wetland, riverine, lacustrine or estuarine habitats as foraging areas for the great blue heron is ideal if these potential foraging habitats have shallow, clear water with a firm substrate and a huntable population of small fish. A potential foraging area needs to be free from human disturbances several hours a day while the herons are feeding. Suitable great blue heron foraging areas are those in which there is no human disturbance near the foraging zone during the four hours following sunrise or preceding sunset or the foraging zone is generally about 100m from human activities and habitation or about 50m from roads with occasional, slow-moving traffic.

A smaller energy expenditure by adult herons is required to support fledglings if an abundant source of food is close to the nest site than if the source of food is distant. Nest sites frequently are located near suitable foraging habitats. Social feeding is strongly correlated with colonial nesting (Krebs 1978), and a potential feeding site is valuable only if it is within "commuting" distance of an active heronry. For example, 24 of 31 heronries along the Willamette River in Oregon were located within 100m of known feeding areas (English 1978). Most heronries along the North Carolina coast were located near inlets, which have large concentrations of fish (Parnell and Soots 1978). The average distance from heronries to inlets was 7.0 to 8.0 km. The average distance of heronries to possible feeding areas (lakes 140 ha in area) varied from 0 to 4.2 km and averaged 1.8 km on the Chippewa National Forest in Minnesota (Mathisen and Richards 1978). Collazo (1981) reported the distance from the nearest feeding grounds to a heronry site as 0.4 and 0.7 km. The maximum observed flight distance from an active heronry to a foraging area was 29 km in Ohio (Parris and Grau 1979).

Great blue herons feed anywhere they can locate prey (Burleigh 1958). This includes the terrestrial surface but primarily involves catching fish in shallow water, usually 150m deep (Bent 1926; Meyerriecks 1960; Bayer 1978).

Thompson (1979b) reported that great blue herons along the Mississippi River commonly foraged in water containing emergent or submergent vegetation, in scattered marshy ponds, sloughs, and forested wetlands away from the main channel. He noted that river banks, jetties, levees, rip-rapped banks, mudflats, sandbars, and open ponds were used to a lesser extent. Herons near southwestern Lake Erie fed intensively in densely vegetated areas (Hoffman 1978).

Other studies, however, have emphasized foraging activities in open water (Longley 1960; Edison Electric Institute 1980). Exposed mud flats and sandbars are particularly desirable foraging sites at low tides in coastal areas in Oregon (Bayer 1978), North Carolina (Custer and Osborn 1978), and elsewhere (Kushlan 1978). Cooling ponds (Edison Electric Institute 1980) and dredge spoil settling ponds (Cooper *et al.* in prep.) also are used extensively by foraging great blue herons.

Water

The great blue heron routinely feeds on soft animal tissues from an aquatic environment, which provides ample opportunity for the bird to satisfy its physiological requirements for water.

Cover

Cover for concealment does not seem to be a limiting factor for the great blue heron. Heron nests often are conspicuous, although heronries frequently are isolated. Herons often feed in marshes and areas of open water, where there is no concealing cover.

Reproduction

Short and Cooper (1985) describe suitable great blue heron nesting habitat as a grove of trees at least 0.4 ha in area located over water or within 250m of water. These potential nest sites may be on an island with a river or lake, within a woodland dominated swamp, or in vegetation near a river or lake. Trees used as nest sites are at least 5m high and have many branches at least 2.5 cm in diameter that are capable of supporting nests. Trees may be alive or dead but must have an "open canopy" that allows an easy access to the nest. The suitability of potential heronries diminishes as their distance from current or former heronry sites increases because herons develop new heronries in suitable vegetation close to old heronries.

A wide variety of nesting habitats is used by the great blue heron throughout its range in North America. Trees are preferred heronry sites, with nests commonly placed from 5 to 15 m above ground (Burleigh 1958; Cottrille and Cottrille 1958; Vermeer 1969; McAloney 1973). Smaller trees, shrubs, reeds (*Phragmites communis*), the ground surface, rock ledges along coastal cliffs, and artificial structures may be utilized in the absence of large trees, particularly on islands (Lahrman 1957; Behle 1958; Vermeer 1969; Soots and Landin 1978; Wiese 1978). Most great blue heron colonies along the Atlantic coast are located in riparian swamps (Ogden 1978). Most colonies along the northern Gulf coast are in cypress - tupelo (*Taxodium Nyssa*) swamps (Portnoy 1977). Spendelow and Patton (in prep.) state that many birds in coastal Maine nest on spruce (*Picea spp*.) trees on islands. Spruce trees also are used on the Pacific coast (Bayer 1978), and black cottonwood (*Populus trichocarpa*) trees frequently are used as nest sites along the Willamette River in Oregon (English 1978). Miller (1943) stated that the type of tree was not as important as its height and distance from human activity. Dead trees are commonly used as nest sites (McAloney 1973). Nests usually consist of a platform of sticks, sometimes lined with smaller twigs (Bent 1926; McAloney 1973), reed stems (Roberts 1936), and grasses (Cottrille and Cottrille 1958).

Heron nest colony sites vary, but are usually near water. These areas often are flooded (Sprunt 1954; Burleigh 1958; English 1978). Islands are common nest colony sites in most of the great blue heron's range (Vermeer 1969; English 1978; Markham and Brechtel 1979). Many colony sites are isolated from human habitation and disturbance (Mosely 1936; Burleigh 1958). Mathisen and Richards (1978) recorded all existing heronries in Minnesota as at least 3.3 km from human dwellings, with an average distance of 1.3 km to the nearest surfaced road. Nesting great blue herons may become habituated to noise (Grubb 1979), traffic (Anderson 1978), and other human activity (Kelsall and Simpson 1980). Colony sites usually remain active until the site is disrupted by land use changes.

A few colony sites have been abandoned because the birds depleted the available nest building material and possibly because their excrement altered the chemical composition of the soil and the water. Heron exretia can have an adverse effect on nest trees (Kerns and Howe 19667; Wiese 1978).

Population and Distribution
Population
Historic
In the past, herons and egrets were shot for their feathers, which were used as cooking utensils and to adorn hats and garments, and they also provided large, accessible targets. The slaughter of these birds went relatively unchecked until 1900 when the federal government passed the Lacey Act, which prohibits the foreign and interstate commercial trade of feathers. Greater protection was afforded in 1918 with the Migratory Bird Treaty Act, which empowered the federal government to set seasons and bag limits on the hunting of waterfowl and waterbirds. With this protection, herons and other birds have made dramatic comebacks.

In southeast Washington, few historical colonies have been reported. The Foundation Island colony is the oldest, but has been taken over by cormorants. It appears blue herons numbers in the colony have declined significantly.

One colony was observed from a helicopter in 1995 on the Touchet River just upriver from Harsha, but that colony appears to have been destroyed by a wind storm (trees blown down), and no current nesting has been observed in the area (Fowler per. com.)

Current

The great blue heron breeds throughout the U.S. and winters as far north as New England and southern Alaska (Bull and Farrand 1977). The nationwide population is estimated at 83,000 individuals (NACWCP 2001).

In southeast Washington, three new colonies have been discovered over the last few years. One colony on the Walla Walla River contains approximately 24 nests. This colony has been active for approximately 12 years. Two new colonies were discovered in 2003, one on a railroad bridge over the Snake River at Lyons Ferry, and one near Chief Timothy Park on the Snake River. The Lyons Ferry colony contained approximately 11 nests, and the Chief Timothy colony 5 nests (P. Fowler, WDFW, personal communication, 2003).

Distribution

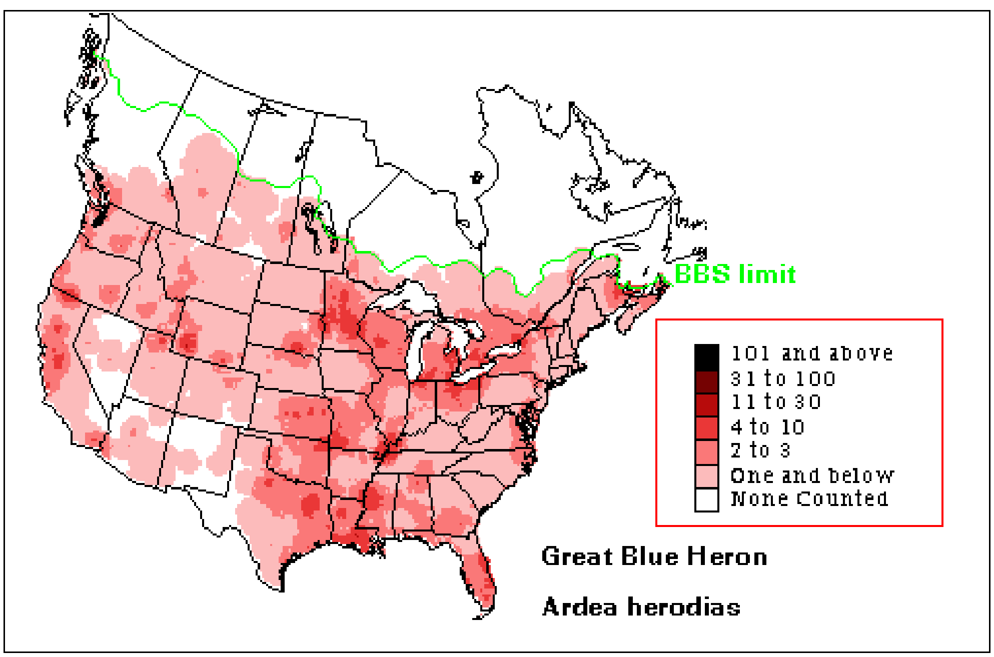
Two known heron rookeries occur within the Walla Walla subbasin, one on the Walla Walla and one on the Touchet River (NPPC 2001). The Walla Walla River rookery contains approximately 13 active nests. The Touchet River rookery contains approximately 8-10 active nests. Blue herons are observed throughout the lowlands of southeast Washington near rivers or streams (P. Fowler, WDFW, personal communication, 2003).

Historic

No data are available.

Current

Figure 8Great blue heron breeding distribution from Breeding Bird Survey (BBS) data (Sauer *et al.* 2003).



Status and Abundance Trends

Status

Surveys of blue heron populations are not conducted. However, populations appear to be stable and possibly expanding in some areas. Two new nesting colonies have been found in on the Lower Snake River (P. Fowler, WDFW, personal communication, 2003).

Trends

Populations in southeast Washington appear to be stable, and may actually be increasing.

Factors Affecting Great Blue Heron Population Status

Key Factors Inhibiting Populations and Ecological Processes

Habitat destruction and the resulting loss of nesting and foraging sites, and human disturbance probably have been the most important factors contributing to declines in some great blue heron populations in recent years (Thompson 1979a; Kelsall and Simpson 1980; McCrimmon 1981).

Habitat Loss

Natural generation of new nesting islands, created when old islands and headlands erode, has decreased due to artificial hardening of shorelines with bulkheads. Loss of nesting habitat in certain coastal sites may be partially mitigated by the creation of dredge spoil islands (Soots and Landin 1978). Several species of wading birds, including the great blue heron, use coastal spoil islands (Buckley and McCaffrey 1978; Parnell and Soots 1978; Soots and Landin 1978). The amount o f usage may depend on the stage of plant succession (Soots and Parnell 1975; Parnell and Soots 1978), although great blue herons have been observed nesting in shrubs (Wiese 1978), herbaceous vegetation (Soots and Landin 1978), and on the ground on spoil islands.

Water Quality

Poor water quality reduces the amount of large fish and invertebrate species available in wetland areas. Toxic chemicals from runoff and industrial discharges pose yet another threat. Although great blue herons currently appear to tolerate low levels of pollutants, these chemicals can move through the food chain, accumulate in the tissues of prey and may eventually cause reproductive failure in the herons.

Several authors have observed eggshell thinning in great blue heron eggs, presumably as a result of the ingestion of prey containing high levels of organochlorines (Graber *et al.* 1978; Ohlendorf *et al.* 1980). Konermann *et al.* (1978) blamed high levels of dieldrin and DDE use for reproductive failure, followed by colony abandonment in Iowa. Vermeer and Reynolds (1970) recorded high levels of DDE in great blue herons in the prairie provinces of Canada, but felt that reproductive success was not diminished as a result. Thompson (1979a) believed that it was too early to tell if organochlorine residues were contributing to heron population declines in the Great Lakes region.

Human Disturbance

Heronries often are abandoned as a result of human disturbance (Markham and Brechtel 1979). Werschkul *et al.* (1976) reported more active nests in undisturbed areas than in areas that were being logged. Tree cutting and draining resulted in the abandonment of a mixed-species heronry in Illionois (Bjorkland 1975). Housing and industrial development (Simpson and Kelsall 1979) and water recreation and highway construction (Ryder *et al.* 1980) also have resulted in the abandonment of heronries. Grubb (1979) felt that airport noise levels could potentially disturb a heronry during the breeding season.

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