

FINAL SPECIES REPORT Fisher (*Pekania
pennanti*), West Coast Population

U.S. FISH AND WILDLIFE SERVICE
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INTRODUCTION

The purpose of this species report is to synthesize the best available scientific and commercial information regarding the fisher, throughout the range of its West Coast Distinct Population Segment (DPS) in the United States. This biological report has been prepared to support the review of the species under the Endangered Species Act (Act or ESA) so that we can evaluate whether or not the fisher West Coast DPS continues to warrant listing under the Act.

On October 7, 2014, the U.S. Fish and Wildlife Service (Service) published a proposed rule in the Federal Register to list the West Coast DPS of fisher as threatened (79 FR 60419). Prior to

the proposed rule, the Service published a 12-month finding in the Federal Register on April 8, 2004, stating that listing the West Coast DPS of the fisher under the Act was warranted, but precluded by other higher priority listing actions (69 FR 18770). We have annually reviewed this finding and monitored the status of the fisher, as required under 16 U.S.C. 1533(b)(3)(C)(i) and (iii), as reflected in the annual Candidate Notices of Review (CNORs). See the November 21, 2012, Federal Register (77 FR 69994) for the most recent CNOR.

In our proposed rule (79 FR 60419, p. 60426) we described the West Coast DPS of the fisher as:

the Cascade Mountains and all areas west to the coast in Oregon and Washington; the North Coast from Mendocino County, California, north to Oregon; east across the Klamath, Siskiyou, Trinity, and Marble Mountains, and across the southern Cascade Mountains; and south through the Sierra Nevada. Not included are the mountainous areas east of the Okanogan River in Washington and the Blue Mountains west to the Ochoco National Forest, in eastern Oregon, because of the naturally occurring geological conditions that isolate them from the western portions of Washington and Oregon. Figure 1 depicts our analysis area for this species report.

Figure 1. Analysis area for west coast population of fishers (*Pekania pennanti*).

ABBREVIATIONS USED

oC degrees Celsius oF degrees Fahrenheit ac acres ACEC Area of Critical Environmental Concern Act Endangered Species Act of 1973, as amended AR anticoagulant rodenticide cm centimeters BGEPA Bald and Golden Eagle Protection Act of 1940, as amended BIA Bureau of Indian Affairs BLM Bureau of Land Management CAL FIRE California Department of Forestry and Fire Protection CDFW California Department of Fish and Wildlife (formerly CDFG) CCAA Candidate Conservation Agreement with Assurances CDFG California Department of Fish and Game (now CDFW) CEQA California Environmental Quality Act CESA California Endangered Species Act CI confidence interval CNOR Candidate Notice of Review dbh diameter at breast height DNA genetic material DPS Distinct Population Segment ECOS Environmental Conservation Online System EPA U.S. Environmental Protection Agency ESA Endangered Species Act of 1973, as amended FEMAT Forest Ecosystem Management Assessment Team FGAR first-generation anticoagulant rodenticide FIFRA Federal Insecticide, Fungicide, and Rodenticide Act of 1947, as amended FLPMA Federal Land Policy and Management Act of 1976, as amended FPA Z'Berg Nejedly Forest Practice Act of 1973 FPR Forest Practice Rules FR Federal Register ft feet g grams GHG greenhouse gas GIS geographic information system ha hectares HCP Habitat Conservation Plan in. inches INFISH Inland Native Fish Strategy IPCC Intergovernmental Panel on Climate Change JBLM Joint-Base Lewis-McChord kg kilograms km kilometers km² square kilometers

lbs. pounds LD50 median lethal dose LRMP Land and Resource Management Plan m meters mi miles mi² square miles MBTA Migratory Bird Treaty Act of 1918, as amended MMMA marbled murrelet management area MOA memorandum of agreement NEPA National Environmental Policy Act of 1969, as amended NFMA National Forest Management Act of 1976, as amended NWFP Northwest Forest Plan OAR Oregon Administrative Rules ODFW Oregon Department of Fish and Wildlife ODF Oregon Department of Forestry oz. ounces PACFISH Interim management of anadromous fish-producing watersheds on Federal lands in eastern Oregon and Washington, Idaho and portions of California. PRC California Public Resources Code PSQ probably sales quantity RCW Revised Code of Washington RPF registered professional forester RMP Resource Management Plan Service U.S. Fish and Wildlife Service SGAR second-generation anticoagulant rodenticide SNAMP Sierra Nevada Adaptive Management Project SNFPA Sierra Nevada Forest Plan Amendment SPI Sierra Pacific Industries SSFCA Southern Sierra Fisher Conservation Area SWAP State Wildlife Action Plan SWGP State Wildlife Grants Program THP Timber Harvest Plan USDA U.S. Department of Agriculture USDI U.S. Department of Interior USDOJ U.S. Department of Justice USDOT FHWA U.S. Department of Transportation Federal Highway Administration USFS U.S. Forest Service USNRM U.S.

SPECIES DESCRIPTION

The fisher, as described by Powell (1981, p. 1), is a medium-sized light brown to dark blackish-brown mammal, with the face, neck, and shoulders sometimes being slightly gray. The chest and underside often has irregular white patches. The fisher has a long body with short legs and a long bushy tail. At 3.5 to 5.5 kilograms (kg) (7.7 to 12.1 pounds [lbs.]), male fishers weigh about twice as much as females (1.5 to 2.5 kg [3.3 to 5.5 lbs.]). Males range in length from 90 to 120 centimeters (cm) (35 to 47 inches [in.]), and females range from 75 to 95 cm (29 to 37 in.) in length. Fishers show regional variation in typical body weight. For example, fishers from western North America weigh more in the northern parts of their range than those living in the southern extent of their range (Lofroth *et al.* 2010, p. 10).

TAXONOMY

The fisher (*Pekania pennanti*) is classified in the order Carnivora, family Mustelidae; this family also includes weasels, mink, martens, and otters (Anderson 1994, p. 14). Initially described by Erxleben (p. 470) as *Mustela pennanti* in 1777, taxonomists during the twentieth century placed the fisher in the genus *Martes* (Goldman 1935, pp. 176–177; Powell 1981 pp. 1, 4; Powell 1993, pp. 11–12) but kept the specific epithet *pennanti* (Hagmeier 1959, p. 185). Recent genetic research has led to a reclassification of the fisher into the genus *Pekania* (Koepfli *et al.* 2008, p. 5; Sato *et al.* 2012, p. 755) and shows that fishers are more closely related to the tayra (*Eira barbara*) and the wolverine (*Gulo gulo*) than to other species in the genus *Martes* (Hosoda *et al.* 2000, p.264; Stone and Cook 2002, p. 170; Koepfli *et al.* 2008, p. 5; Sato *et al.* 2009, p. 916; Wolsan and Sato 2010, p. 179; Nyakatura and Bininda-Emonds 2012, p 13; Sato *et al.* 2012, p. 754). The Service adopts this recent name change, which places the fisher in a monotypic genus.

Characteristic of the genus *Pekania* include its large body size compared with *Martes* species and the presence of an external median rootlet on the upper carnassial (fourth) premolar (Anderson 1994, p. 21).

In 1935, Goldman (1935, p. 177) described three subspecies of fisher based on differences in skull dimensions, although he stated they were difficult to distinguish: (1) *Martes pennanti pennanti* in the east and central regions; (2) *M. p. columbiana* in the central and northwestern regions; and (3) *M. p. pacifica* in the Pacific States. A subsequent analysis questioned whether there was a sufficient basis to support recognition of different subspecies based on numerous factors, including the small number of samples available for examination (Hagmeier 1959, p. 193). Regional variation in characteristics used by Goldman to discriminate subspecies appears to be clinal (varying along a geographic gradient), and the use of clinal variations is “exceedingly difficult to categorize subspecies” (Hagmeier 1959, pp. 192–193). Although subspecies taxonomy is often used to reference fisher populations in different regions, and studies of genetic variation show patterns of population subdivision similar to the subspecies (Kyle *et al.* 2001, p. 2345; Drew *et al.* 2003, p. 59), it is not clear whether the subspecies are valid. Additional support for the uncertainty regarding the taxonomic validity of fisher subspecies is provided by Knaus *et al.* (2011, p. 5) who examined the entire mitogenomes of fishers from all three purported subspecies and found no evidence of monophyly. In other words, they did not find evidence to support a genetic tree that places each subspecies on a single branch, with a common ancestor and common descendants for each subspecies, and where all members of each subspecies are genetically distinct from the members of the other subspecies.

New information since Service (2014)

The reclassification of the fisher to the genus *Pekania* has been accepted by Bradley *et al.* (2014, pp. 4, 6, 13) and added to the Revised Checklist of North American Mammals North of Mexico, 2014.

LIFE HISTORY

Reproduction

Fishers live to be about 10 years of age in the wild and captivity (Arthur *et al.* 1992, p. 404; Powell *et al.* 2003, p. 644) with both sexes reaching maturity their first year but often not becoming effective breeders until 2 years of age (Powell and Zielinski 1994, p. 46; Powell *et al.* 2003, p. 638). Fishers are solitary except females with kits and during the breeding season, which is generally from late February to the middle of May (Wright and Coulter 1967, p. 77; Frost *et al.* 1997, p. 607). The breeding period in California and Oregon begins in late February and lasts through April based on observations of significant changes of fisher movement patterns (reviewed by Lofroth *et al.* 2010, p. 56). Uterine implantation of embryos occurs 10 months after copulation; active gestation is estimated to be 36 days and birth occurs nearly 1 year after copulation (Wright and Coulter 1967, pp. 74, 76; Frost *et al.* 1997, p. 609; Powell *et al.* 2003, p. 639).

The proportion of adult female fishers that den each year in western North America is 0.64 (range = 0.39–1.00) (Lofroth *et al.* 2010, pp. 55–57; Matthews *et al.* 2013, pp. 103–104). Individual fishers may not give birth every year and reproductive rates may change as females age (Weir and Corbould 2008, p. 28). Among fishers who do give birth, the mean litter size for fishers is between one and three kits (litter size range from one to six kits) (Powell 1993, p. 53; Powell *et al.* 2003, pp. 639–640; Sweitzer *et al.* 2015b, p. 10). The average litter size for 19 females during 4 den seasons on the Hoopa study area in Northern California was 1.9 kits (Matthews *et al.* 2013, p. 103). Within the analysis area, females give birth between mid-March and mid-April (Truex *et al.* 1998, p. 36; Aubry and Raley 2006, p. 12; Higley and Matthews 2006, p. 8; Self and Callas 2006, p. 9; Weir and Corbould 2008, p. 78; Spencer *et al.* 2015, p. 12; Sweitzer *et al.* 2015b, p.9). Newborn kits are entirely dependent on the mother and are weaned at about 10 weeks of age (Powell 1993, p. 67). At about 4 months of age, kits are mobile enough to travel with their mothers (Aubry and Raley 2006, p. 13).

Throughout their range, fishers use tree or snag cavities (Paragi *et al.* 1996a, entire; Truex *et al.* 1998, p. ii; Weir 2003, p. 12; Aubry and Raley 2006, p. 16; Higley and Matthews 2006, p. 10;

Self and Callas 2006, p. 6; Weir and Corbould 2008, pp. 105–106; Davis 2009, p. 23) to give birth and raise their young (Coulter 1966, p. 81). Kits may be moved to numerous den locations (Arthur and Krohn 1991, p. 382; Paragi *et al.* 1996a, p. 80; Higley and Matthews 2006, p. 7) before they are weaned (Powell 1993, p. 67). Once weaned, the kits stay with the female, utilizing multiple structures (for example, tree cavities, hollow logs, log piles) (Truex *et al.* 1998, p. 35; Aubry and Raley 2006, pp. 7, 16–17; Higley and Matthews 2006, pp. 6–7) within the female’s home range until juveniles disperse in the fall or winter following their birth (Aubry and Raley 2006, p. 12; Matthews *et al.* 2009, p. 9). Kits become independent of their mother and develop their own home ranges by 1 year of age (Powell *et al.* 2003, p. 640).

Natural Causes of Mortality

Natural sources of mortality besides predation and disease include interspecific and intraspecific conflict (Lofroth *et al.* 2010, p. 63; Sweitzer *et al.* 2015a, p. 6), drowning (Lewis 2014, p. 67), and starvation. One death attributed to starvation was determined to be caused by old age, since the animal’s teeth were worn to the gum line (Aubry and Raley 2006, p. 11) while another starved after suffering an infection in its throat from a porcupine quill (Weir and Corbould 2008, p. 24). Among 128 fishers necropsied in California, seven (five percent) died of nutritional deficiencies, although the specific reasons for the nutritional deficiencies were not identified (Gabriel 2013, p. 99; Gabriel 2013b, pers. comm.). These seven fishers included four adults, a juvenile, and two kits recovered from abandoned den sites. For a discussion on other causes of natural mortality, see the *Disease* and *Predation* sections below. For a discussion on anthropomorphic causes of mortality, see the *Trapping and Incidental Capture*, *Research Activities*, *Collision with Vehicles*, and *Exposure to Toxicants* sections below.

New information since Service (2014)

Gabriel *et al.* (2015, entire) investigated the causes of mortality for 167 fishers in California. Their investigations used a combination of gross necropsy, histology, toxicology, and molecular methods. Of the 167 fishers collected from 2007-2014 they had sufficient material to perform

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necropsies on 123 fishers. They reported the results of their investigations by grouping the causes of mortality into six categories. These categories included predation, natural disease, poisoning, vehicular strike, human-caused (other than vehicular strike), and unknown. This new information did not identify natural causes of mortality other than predation or disease. Further discussion of this new information is included in the *Disease* and *Predation* sections below.

Sweitzer *et al.* (2015a, p. 6) similarly documented four fisher deaths in the southern Sierra

Nevada related to starvation or illness from a debilitating injury that prevented foraging. One additional fisher mortality documented during this study resulted from a rattlesnake bite (Sweitzer *et al.* 2015a, p. 6).

Survivorship

Adult female survival has been shown to be the most important single demographic parameter determining fisher population stability (Truex *et al.* 1998, p. 52; Lamberson *et al.* 2000, pp. 6, 9, Spencer *et al.* 2011, p. 794, 798). From 2005 to 2009, Higley and Matthews (2009, pp. 15, 62) documented that adult female survival varied from 58.9 percent to 94.4 percent for all female fishers marked on the Hoopa Valley Reservation in California. On the eastern Klamath study area, Swiers (2013, p. 19) estimated that the annual survival rate of 64 percent did not vary from 2007 to 2011 and did not vary by sex. Truex *et al.* (1998, p. 32) documented an annual survival rate, pooled across years from 1994 to 1996, of 61.2 percent of adult female fishers in the southern Sierra Nevada, 72.9 percent for females in their eastern Klamath study area, and 83.8 percent for females in their North Coast study area. Addressing the population in the southern Sierra Nevada, Truex *et al.* (1998, p. 52) concluded that, “High annual mortality rates raise concerns about the long-term viability of this population.” From spring 2007 to winter 2011, Sweitzer *et al.* (2011) reported adult female survival for two study areas in the southern Sierra Nevada as 72 percent (95 percent confidence interval of 56 percent to 88 percent) in the north and 74 percent (95 percent confidence interval of 60 percent to 87 percent) in the south.

New information since Service (2014)

In the Hoopa study area in the NCSO population Higley *et al.* 2013 analyzed capture-mark-recapture (CMR) data collected from 2004-2005 to 2012-2013 to estimate population size, apparent survival, and lambda using “Closed Captures”, “Recaptures Only”, and “Pradel Models,” respectively in program MARK (White and Burnham 1999). Estimates of female annual survival indicate a stable to slight increase in annual survival while male-only survival was declining during the same time period (Higley *et al.* 2013, p. 100, Figure 27).

The most recent annual monitoring data for fishers reintroduced to the northern Sierra Nevada report monthly survival estimates for females and males during reproduction as 0.97 (95% CI = 0.95–0.99). Outside reproductive time periods, estimated monthly survival rates for both sexes is 0.99 (95% CI = 0.97–1.0). The estimate for annual survival rate for adult fishers, including breeding and non-breeding periods is 0.80 (95% CI = 0.55–0.84) (Powell *et al.* 2014, p. 14) Sweitzer *et al.* (2015b, pp. 784–785) found that “change in fisher survival was more important than fecundity for deterministic population growth.”

A recent study by Sweitzer *et al.* (2015b, p. 779) in the Sierra National Forest reported adult survival rates from fall 2008 to early summer 2013 as 72 percent (95 percent confidence interval of 62 percent to 82 percent). Additionally, Sweitzer *et al.* (2015c, p. 9) reported variation in projected fisher survival rates from this area dependent upon season, sex, and age. In particular, the authors noted lower survival in males and differences between male and female survival based on season Sweitzer *et al.* (2015c, p. 7).

Recruitment

The estimated recruitment rate we used for this analysis is defined as the number of juveniles alive per adult female at the time of juvenile dispersal during the fall of the year. Very little is known about fisher recruitment and often data are derived by piecing together various sources of information (for example, denning rates of adult females, telemetry and capture data, aging data, etc.). In central interior British Columbia, Weir and Corbould (2008, p. 21) estimated that the average fall recruitment rate of juveniles per adult female was 0.58, suggesting very little recruitment of new individuals into that population.

New information since Service (2014)

Matthews *et al.* (2013, p. 104) reported seasonal recruitment rates for fishers. Recruitment rates were: 1.0 juveniles per adult female at weaning (0.51 for female kits and 0.49 for male kits), 0.32 juveniles per adult female after the fall–winter live trapping period (0.28 for females and 0.05 for males), and 0.19 kits per adult female at home range establishment (0.16 for females and 0.02 for males) (Matthews *et al.* 2013, p. 104).

SPACING PATTERNS AND MOVEMENT

Home Range and Territoriality

An animal's home range is the area traversed by the individual in its normal activities of food gathering, mating, and caring for young (Burt 1943, p. 351). Fisher home range size most likely increases with increasing latitude (Lofroth *et al.* 2010, p. 69; Weir *et al.* 2013, p. 121) and with body size (Lindstedt *et al.* 1986, p. 416). The abundance or availability of prey and their vulnerability to predation may play a role in home range size and selection (Powell 1993, p. 173; Powell and Zielinski 1994, p. 57). Only general comparisons of fishers' home range sizes can be made, because studies across the range have been conducted by different methods. Generally, fishers have large home ranges, with male home ranges typically larger than female home ranges. Fisher home ranges vary in size across North America and range from 16 to 122 square

kilometers (km²) (4.7 to 36 square miles (mi²)) for males, and from 4 to 53 km² (1.2 to 15.5 mi²) for females (reviewed by Powell and Zielinski 1994, p. 58; Lewis and Stinson 1998, pp. 7– 8; Zielinski *et al.* 2004b, p. 652; Sweitzer *et al.* 2015d, p. 90; Weir *et al.* 2013, p. 117). West of the Rocky Mountains in the U.S. and Canada, male home ranges tended to be three times larger than females, averaging 18.8 square kilometers (km²) (7.3 mi²) for females and 53.4 km² (20.6 mi²) for males (Lofroth *et al.* 2010, pp. 67–68).

Fishers exhibit territoriality, with little overlap between members of the same sex; in contrast, overlap between opposite sexes is extensive, and the extent of overlap is possibly related to the density of prey (Powell and Zielinski 1994, p. 59). It is not known how fishers maintain territories; it is possible that scent marking plays an important role (Leonard 1986, p. 36; Powell 1993, p. 170). Direct aggression between individuals in the wild has not been observed, although combative behavior has been observed between older littermates and between adult females in captivity (Powell and Zielinski 1994, p. 59).

Fishers are polygynous (Powell 1993, p. 54) with males typically seeking out females in estrus. During the breeding season, male fishers may expand their home ranges as much as 2.4-fold or temporarily abandon their territories by taking long excursions and moving up to 22 km (13.7 mi) within 48 hours to increase their opportunities to mate (Buck 1982, p. 28; Aubry and Raley 2006, p. 13; Arthur *et al.* 1989, p. 677; Jones 1991, pp. 77–78). However, males who maintained their home ranges during the breeding season were more likely to successfully mate than were nonresident males encroaching on an established range (Aubry *et al.* 2004, p. 215). Adult females do not make pronounced breeding season movements, particularly in those years that they are raising kits, and appear to maintain relatively consistent home ranges year-round (Arthur *et al.* 1993, p. 872).

New information since Service (2014)

Home ranges of four male fishers in northeastern British Columbia averaged 210 km² (81 mi²) (Weir *et al.* 2013, p. 117). In Weir's British Columbia study area, he concludes that home range size may also be a function of the availability and distribution of particular resources needed for reproduction (Weir *et al.* 2013, pp. 121–122).

Lewis (2014, p. 29) reported mean home ranges of 128.3 km² (49.5 mi²) and 63.5 km² (24.5 mi²) for male and female fishers, respectively, in a reintroduced population in the Olympic Peninsula. The author notes that future research is needed to determine if mean home range sizes become

smaller as the population becomes established (Lewis 2014, p. 39).

One study in the southern Sierra Nevada observed extensive overlap of annual home ranges of female fishers, with reduced overlap in core use areas (Sweitzer *et al.* 2015d, pp. 88–89).

Dispersal

Dispersal, the movement of juveniles from their natal home range to establish a breeding territory, is the primary mechanism for the geographic expansion of a population. Long distance dispersal has been documented for fishers with males moving greater distances than females. Arthur *et al.* (1993, p. 872) reported an average maximum dispersal distance of 14.9 km (9.3 mi) and 17.3 km (10.7 mi) for females and males, respectively [range = 7.5 km (4.7 mi) to 22.6 km (14.0 mi) for females and 10.9 km (6.8 mi) to 23.0 km (14.3 mi) for males] in a low density population in Maine with relatively high trapping mortality. In areas such as this, with high trapping mortality, young fishers may not have to disperse as far in order to find unoccupied home ranges (Arthur *et al.* 1993, p. 872). York (1996) reported dispersal distances for juvenile male and female fishers averaging 33 km (20 mi) [range = 10 km (6 mi) to 107 km (66 mi)] for a

high-density population in Massachusetts. On the Hoopa Valley Indian Reservation study area, the mean dispersal distance between natal dens and the centroids of newly established subadult home ranges was 4.0 km (2.5 mi) [range = 0.8 km (0.5 mi) to 18.0 km (11.2 mi)] for 7 females and 1.3 km (0.81 mi) for 1 male (Matthews *et al.* 2013, p. 104). However, the mean maximum travel distance was greater for males, 8.1 km (5.0 mi) [range = 5.9 km (3.7 mi) to 10.3 km (6.40 mi)], than for females, 6.7 km (4.1 mi) [range = 2.1 km (1.3 mi) to 20.1 km (12.5 mi)] (Matthews *et al.* 2013, p. 104). Notably, only two females dispersed far enough from their natal home ranges to avoid overlapping with their mothers' home ranges (Matthews *et al.* 2013, p. 104).

Juveniles dispersing from natal areas are capable of moving long distances and navigating various landscape features such as highways, rivers, and rural communities to establish their own home range (York 1996, p. 47; Weir and Corbould 2008, p. 44). Dispersal characteristics may be influenced by factors such as sex, availability of unoccupied areas, turnover rates of adults, and habitat suitability (Arthur *et al.* 1993, p. 872; York 1996, pp. 48–49; Aubry *et al.* 2004, pp. 205–207; Weir and Corbould 2008, pp. 47–48). Long distance dispersal by juveniles is made at a high cost and is usually not successful. Fifty-five percent of fishers in a British Columbia study died before establishing home ranges, and only 17 percent successfully established a home range (Weir and Corbould 2008, p. 44). Those individuals that traveled longer distances were subject to greater mortality risk (Weir and Corbould 2008, p. 44).

Based on field observation and microsatellite genotype analyses of the fisher population in the southern Cascades, Aubry *et al.* (2004, p. 217) found empirical evidence of male-biased juvenile dispersal and female philopatry (the drive or tendency of an individual to return to, or stay in, its home area) in fishers, which may have a direct bearing on the rate at which fishers can colonize formerly occupied areas within their historical range. Tucker's (2013, p. 65) use of bi-parentally inherited genetic markers to investigate sex-biased dispersal of southern Sierra Nevada fishers yielded mixed results, but suggested that males disperse more often than do females. Research at the Hoopa study area also supports the theory that fishers have male-biased dispersal and female philopatry (Matthews *et al.* 2013 p. 105).

New information since Service (2014)

Dispersal by juvenile fisher begins during or after their first fall or winter when they are about seven to 10 months old (Aubry and Raley 2006, p. 14; Naney *et al.* 2012, p. 72). Juveniles in the southern Oregon Cascade Range began dispersing at about 10 months old in early February (Aubry and Raley 2006, p. 14). In the southern Sierra Nevada, juvenile dispersal likely begins in March (Sweitzer *et al.* 2015b, p. 5; Sweitzer *et al.* 2015d, pp. 36).

Mean juvenile dispersal distance in the southern Sierra Nevada was 4.89 km (3.04 mi) for females and 8.48 km (5.27 mi) for males (Sweitzer *et al.* 2015d, p. 82). The maximum juvenile dispersal distances for this area were 22.26 km (13.83 mi) for a female and 36.17 km (22.48 mi) for a male (Sweitzer *et al.* 2015d, p. 82). However, Sweitzer *et al.* (2015c, p. 9) did not find that dispersal reduced survival among dispersal-aged fishers in the southern Sierra Nevada.

Food Habits

Fishers are opportunistic predators, primarily of squirrels (*Tamiasciurus*, *Sciurus*, *Glaucomys*, and *Tamias* spp.), mice (*Microtus*, *Clethrionomys*, and *Peromyscus* spp.), snowshoe hares (*Lepus americanus*), and birds (numerous spp.) (reviewed in Powell 1993, pp. 18, 102; reviewed in Lofroth *et al.* 2010, pp. 74–76, 161–163). Fishers may indirectly shape forest plant communities through their influence on the population dynamics of prey species that are important seed predators in western coniferous forests (for example, tree squirrels and other rodents that cache or hoard seeds) (for example, Roemer *et al.* 2009, p. 170). Carrion and plant

material (for example, berries) also are consumed (Powell 1993, p. 18). The fisher is one of the few predators that successfully kills and eats porcupines (*Erethizon dorsatum*), (Powell 1993, p. 135).

While snowshoe hares and porcupines are important prey items across much of North American range of fishers, within the analysis area the ranges of these prey species do not extensively overlap the range of the fisher (Powell 1981, p. 3; Bittner and Rongstad 1982, pp. 146–163; Dodge 1982, p. 355; Ellsworth and Reynolds 2006, p. 10). Fishers in the analysis area have a diverse diet with the dominant component in Oregon and California being small and mid-sized mammals (Zielinski *et al.* 1999, entire; Aubry and Raley 2006, pp. 25–27; Golightly *et al.* 2006, entire). Diet studies in California have indicated that fishers prey predominantly on mammals, but their diet also includes birds, insects, and reptiles (Zielinski *et al.* 1999, entire; Golightly *et al.* 2006, entire).

Golightly *et al.* (2006, entire) examined diet and energetic return based on body size, to infer daily energy demands for fishers in the Klamath/North Coast Bioregion. He concluded that an average-weight Douglas squirrel (*Tamiasciurus douglasii*) would supply a female fisher with a 1.6-day supply of energy and a woodrat (*Neotoma* spp.) could supply 2 days of energy. A fisher would need to find and consume 10 to 26 smaller prey items (for example, mice (*Peromyscus maniculatus*) or western fence lizard, *Sceloporus occidentalis*) per day to meet their energetic needs (Golightly *et al.* 2006, pp. 40–41).

New information since Service (2014)

Fishers in coastal Washington also prey upon mountain beaver (*Aplodontia rufa*) (Lewis 2014, p. 109).

HABITAT ASSOCIATIONS

The occurrence of fishers at regional scales is consistently associated with low- to mid-elevation environments of coniferous and mixed conifer and hardwood forests with abundant physical structure (reviewed by Hagmeier 1956, entire; Arthur *et al.* 1989, pp. 683–684; Banci 1989, p. v; Aubry and Houston 1992, p. 75; Jones and Garton 1994, pp. 377–378; Powell 1994, p. 354; Powell *et al.* 2003, p. 641; Weir and Harestad 2003, p. 74, Raley *et al.* 2012, pp. 238–245). Within the analysis area, current fisher populations inhabit forested areas from sea level to approximately 2,600 meters (m) (8,530 feet [ft]) (Lofroth *et al.* 2010, p. 88; Lewis 2014, p. 98; Sweitzer *et al.* 2015d, pp.59–60). Historically, fishers in the analysis area were distributed in 15

similar elevation ranges as current populations even though they are now considered likely

extirpated in many areas of Oregon and Washington (Bailey 1936, pp. 298–299; Aubry and Houston 1992, pp. 69–70, 74–75; Lewis and Stinson 1998, pp. 4–5; Aubry and Lewis 2003, p. 79; 85–86; Lofroth *et al.* 2010, pp. 41–43, 47, and references therein).

Snow conditions and ambient temperatures may affect fisher activity and habitat use. Fishers in eastern parts of the taxon's range may be less active during winter and may avoid areas where deep, soft snow inhibits movement (Leonard 1980, pp. 108–109; Raine 1983, p. 25). Historical and current fisher distributions in California and Washington are consistent with forested areas that receive low or lower relative snowfall (Krohn *et al.* 1997, p. 226; Aubry and Houston 1992, p. 75). Fishers in Ontario, Canada, moved from low-snow areas to high-snow areas during population increases, indicating a possible density-dependent migration to less suitable habitats factored by snow conditions (Carr *et al.* 2007, p. 633). These distribution and activity patterns suggest that the presence of fishers and their populations may be limited by deep snowfall. However, the reaction to snow conditions appears to be variable across the range, with fishers in some locations appearing unaffected by snow conditions or increasing their activity with fresh snowfall (Jones 1991, p. 94; Roy 1991, p. 53; Weir and Corbould 2007, p. 1512). Thus, fishers' reaction to snow may be dependent on a myriad of factors, including, but not limited to: local freeze-thaw cycles, the rapidity of crust formation, snow interception by the forest canopy, lower rates of primary forest productivity, less complex forest structure, and prey availability (Krohn *et al.* 1997, p. 226; Mote *et al.* 2005, p. 44; Weir and Corbould 2007, p. 1512; Raley *et al.* 2012, p. 248–249).

Fishers in the analysis area occur in a wide variety of forest plant communities (Buck *et al.* 1994, pp. 368–370; Klug 1997, p. 32; Self and Kerns 2001, p. 3; Zielinski *et al.* 2004b, pp. 650–651; Aubry and Raley 2006, pp. 3–4). Some of the most productive habitats for fishers are within floristically diverse landscapes that likely provide for a wide variety of prey species (Buskirk and Powell 1994, pp. 285–287). Raley *et al.* (2012, p. 249) hypothesize that it may benefit fishers to include a diversity of available forest conditions within their home ranges to increase their access to a greater diversity and abundance of prey species as long as important habitat features supporting reproduction and thermoregulation are available. In California, fishers occur in a wider array of plant communities (for example, mixed conifer-hardwood forests) than are or would have been available to historical populations to the north in Oregon and Washington where many of these plant communities do not occur. Historically and currently, fishers do not occupy high elevation sub-alpine and alpine environments (Roy 1991, p. 42; Aubry and Lewis 2003, p. 82).

The key aspects and structural components of fisher habitat are best represented in areas that are comprised of forests with diverse successional stages containing a high proportion of mid- and late-successional characteristics (Buskirk and Powell 1994, pp. 286–287; Zielinski *et al.* 2004b, pp. 652–653, 655). Natural forest development is a dynamic continuum that begins with a disturbance event, such as wildfire or windthrow (areas of downed trees due to high winds), that

alters major components of the forest, initiating an array of successional stages across the landscape. Over time, the disturbance-affected forest grows and experiences a series of successional stages in vegetation species occurrence and stand structure. Timber harvest can also be considered a disturbance event that, if the harvesting techniques mimic or maintain some

of the attributes of natural forest development processes, may also be able to develop late successional characteristics. In the absence of major disturbance (changes in successional stage) over many decades depending on the forest type, the structure and species composition of mature or late-successional forest forests may result. Late successional forests are generally characterized by more diversity of structure and function than younger forest developmental stages and the specific characteristics of structural diversity vary by region, forest type, and local conditions.

To support fishers' successful reproduction and protection from predation, forest structure must provide both natal and maternal den and rest sites (Powell and Zielinski 1994, p. 53). The extent to which late successional forests and forest structure is required to support fishers may depend on scale (Powell *et al.* 2003, p. 641), because fishers select habitat at multiple spatial scales for different activities or behaviors (Powell and Zielinski 1994, p. 54; Weir and Harestad 1997, p. 260; Garner 2013, p. 41; Niblett 2015, p. 10). Female fishers are more selective than males in the use of various forest conditions and structures in order to successfully give birth and rear their kits (Lofroth *et al.* 2010, pp. 91, 101, 106, 115). Landscapes that support the establishment of fisher home ranges provide habitat attributes necessary for resting and denning based at the individual tree and site scales; these landscapes also provide foraging opportunities at forest stand and larger scales that contain an abundance and diversity of prey (Powell 1993, p. 89; Buskirk and Powell 1994, p. 284; Weir and Corbould 2008, p. 103, Raley *et al.* 2012, p. 237). Overall, fishers appear to be more selective in the habitat and structures that provide rest and den sites than the habitat types selected for foraging (Lofroth *et al.* 2010, p. 121).

Throughout their range, fishers are obligate users of tree or snag cavities for dens where they give birth (reviewed by Lofroth *et al.* 2010, p. 119; Coulter 1966, p. 81). Kits may be moved from their natal den to numerous maternal den locations before they are weaned; as a result, a denning female requires multiple den trees per year (Arthur and Krohn 1991, p. 382; Paragi *et al.* 1996a, p. 80; Higley and Matthews 2006, p. 7; Powell 1993, p. 67). Once weaned, the kits stay with the female, and consequently the family unit utilizes multiple structures (for example, tree cavities, hollow logs, and log piles) within the female's home range until juvenile dispersal in the fall or winter (Truex *et al.* 1998, p. 35; Aubry and Raley 2006, p. 7, 12, 16–17; Higley and Matthews 2006, p. 6–7; Matthews *et al.* 2009, p. 9).

Cavities in large-diameter live or dead trees are selected for natal dens and more often for maternal dens than other structures (Powell and Zielinski 1994, pp. 47, 56). Dens are in larger diameter trees because they need to be large enough to provide a cavity with an inside diameter of greater than 30 cm (12 in.) (Weir and Corbould 2008, p. 142; Weir *et al.* 2012, p. 230). Furthermore, female fishers select den trees with very specific dimensions of the cavity entrance (Weir *et al.* 2012, p. 237). All entrances to den cavities in British Columbia ranged from 4.5 to 9.5 cm (1.8 to 3.8 in.) to allow the female fisher access to the cavity, but exclude larger animals such as potential predators or male fishers (Weir *et al.* 2012, p. 237).

Similar to den site selection, fishers select resting sites with characteristics of late successional forests: large diameter trees, coarse downed wood, and singular features of large snags, tree cavities, or deformed trees (Powell and Zielinski 1994, p. 54; Lofroth *et al.* 2010, pp. 101–103, Aubry *et al.* 2013, entire). Live trees, snags, and logs used for resting were, on average, 1.4 to 17

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3.4 times larger in diameter than average available structures (Weir and Harestad 2003, pp. 77–78; Zielinski *et al.* 2004a, p. 475; Purcell *et al.* 2009, p. 2700). When fishers use younger forest types, they select large-diameter trees or snags, if present, that are remnants of a previously existing older forest stage (Jones 1991, p. 92). In addition, similar to den site use, fishers utilize multiple rest sites per day distributed throughout their home range, and rest site selection and use changes daily and seasonally (Lofroth *et al.* 2010, p. 72). The type of site and structure selected may be dictated by weather conditions, proximity to available prey, and potential predators (Lofroth *et al.* 2010, p. 119). Because of all of these factors and selectivity for mature forest type structure, resting and denning sites may be limiting to fisher distribution (Powell and Zielinski 1994, pp. 56–57).

Rest sites may be selected for their insulating or thermoregulatory qualities and for their effectiveness at providing protection from predators (Weir *et al.* 2004, pp. 193–194, Raley *et al.* 2012, pp. 244–245). Raley *et al.* (2012, p. 240) summarizes the “overwhelmingly consistent” characteristics of over 2,260 resting structures selected by fishers throughout western North America, stating:

Fishers rested primarily in deformed or deteriorating live trees (54–83% of all rest structures identified in individual studies), and secondarily in snags and logs (Weir and Harestad 2003; Zielinski *et al.* 2004b; Aubry and Raley 2006; Purcell *et al.* 2009). The species of trees and logs used for resting appeared to be less important than the presence of cavities, platforms, and other microstructures. In live trees, fishers rested primarily in rust brooms in more northern study areas (Weir and Harestad 2003; Weir and Corbould 2008; Davis 2009) and mistletoe brooms or other platforms elsewhere (e.g., Self and

Kerns 2001; Yaeger 2005; Aubry and Raley 2006). In contrast, fishers primarily used cavities when resting in snags (e.g., Self and Kerns 2001; Zielinski *et al.* 2004b; Purcell *et al.* 2009). Fishers used hollow portions of logs or subnivean spaces [formed beneath logs and packed snow] more frequently in regions with cold winters (e.g., Weir and Harestad 2003; Aubry and Raley 2006; Davis 2009) than those with milder winters (e.g., Yaeger 2005; Purcell *et al.* 2009; Thompson *et al.* 2010). These results suggest that fishers use structures associated with subnivean spaces to minimize heat loss during cold weather (Weir *et al.* 2004; Weir and Corbould 2008).

In most cases, cavities in live trees, snags, and down logs used as reproductive dens (natal and maternal) and rest sites are a result of heartwood decay (Weir 1995, p. 137; Aubry and Raley 2006, p. 16; Weir and Corbould 2008, p. 105; Reno *et al.* 2008, p. 19; Davis 2009, pp. 26–27). Fishers do not excavate their own natal or maternal dens; therefore, other factors (such as heartwood decay of trees, excavation by woodpeckers, broken branches, frost, or fire scars) are important in creating cavities and narrow entrance holes (Lofroth *et al.* 2010, p. 112). Depending on tree species and ecological conditions, cavity formation in large trees or snags (for denning and resting) may require over 100 years to develop (Raley *et al.* 2012, pp. 242–244, Weir *et al.* 2012, pp. 234–237). The tree species selected for den and rest sites may vary from region to region based on local influences. In regions where both hardwood and conifers occur, hardwoods are selected more often, even if they are only a minor component of the area (Lofroth *et al.* 2010, p. 115), due to their propensity to develop cavities from structural damage to the tree. Den and rest cavities tend to be in older and larger diameter trees than other available trees in the

vicinity, particularly when they are in conifer tree species, where the larger size of these structures is likely related to tree age and the long time periods required for cavities to develop (reviewed by Lofroth *et al.* 2010, pp. 115, 117; Zhao *et al.* 2012, p. 118).

The strongest and most consistent predictor of fisher occurrence in western North America is an association with moderate to dense forest canopy at larger spatial scales (reviewed by Lofroth *et al.* 2010, p. 119, and Raley *et al.* 2012, p. 245; Sweitzer *et al.* 2016, p. 218). This is emphasized by the fishers' avoidance of non-forested habitats with little or no cover (Powell and Zielinski 1994, p. 39; Buskirk and Powell 1994, p. 286) such as open forest, grassland (Powell and Zielinski 1994, p. 55), and wetland habitats (Weir and Corbould 2010, p. 408). An abundance of coarse woody debris, boulders, shrub cover, or subterranean lava tubes sometimes provide suitable overhead cover in non-forested or otherwise open areas for daily movements, seasonal movements by males and juvenile dispersal (Buskirk and Powell, 1994, p. 293; Powell *et al.* 2003, p. 641). In the understory, the physical complexity of coarse woody debris such as

downed trees and branches provides a diversity of foraging and resting locations (Buskirk and Powell 1994, p. 295).

Fishers also occupy and reproduce in managed forest landscapes and forest stands not classified as mature or late-successional if those managed forest landscapes provide sufficient amounts of and an adequate distribution of the key habitat and structural components important to fishers (Self and Callas 2006, entire; Reno *et al.* 2008, pp. 9-16; Clayton 2013, pp. 7–8; Garner 2013, p. 41). Younger and mid-seral forests may be suitable for fishers if complex forest structural components such as trees with cavities, large logs, and snags are maintained in numbers fulfilling life history requirements (Lewis and Stinson 1998, p. 34). Studies in British Columbia (Weir and Corbould 2010, p. 406) and California (Klug 1997, pp. 5, 33; Self and Kerns 2001, pp. 7–8, 10; Lindstrand 2006, pp. 50–51) have shown that fishers occur in heavily managed forested landscapes that may contain few stands of mature or late-successional forest. These studies report “a mosaic of seral stages” (Weir and Corbould 2010, p. 406), with “significant older residual components in harvested stands” (Klug 1997, pp. 5–7) or patches of dense-canopy and dead wood habitat elements that most likely provide the structural complexity required by fishers (Klug 1997, p. 42) Lindstrand 2006, pp. 50-51; Clayton 2013, pp. 7–8; Niblett 2015, pp. 9–10).

In addition, forest structure that provides high quality fisher habitat should supply a high diversity and density of prey vulnerable to fisher predation. According to Buskirk and Powell (1994, p. 286), the physical structure of the forest and prey associated with those forest structure types are thought to be the critical features that explain fisher habitat use, rather than specific forest types. In the analysis area, large old trees, a diversity of tree species, and snags provide habitat elements important for populations of northern flying squirrels (*Glaucomys sabrinus*), tree squirrels (*Sciuridae* spp.), and other arboreal rodents (*Arborimus* spp.) (Carey 1991, entire; Aubry *et al.* 2003, pp. 412–413, 426–429). Additionally brushy understory vegetation provides key habitat for many other important fisher prey species: snowshoe hares (*Lepus americanus*; Hodges 2000, pp. 137–140), brush rabbits (*Sylvilagus bachmani*; Verts and Carraway 1998, p. 133), dusky footed woodrats (*Neotoma fuscipes*; Carey *et al.* 1999a, pp. 67–70, Carey *et al.* 1999b pp. 74–77), and chipmunk species (*Tamias* spp.; Verts and Carraway 1998, pp. 168, 170– 171). As stated by Powell (1993, pp. 73, 89, 96–97), the structure and species composition of mature or late-successional forest are probably not as important to fishers as the vegetative and

structural aspects that lead to abundant and diverse prey populations and reduced fisher vulnerability to predation.

Abiotic factors have also been considered by some researchers and in some habitat modeling efforts to be important components of assessing habitat suitability and distribution of fishers. In many previous reviews and summaries of fisher habitat, riparian areas and buffers have often been highlighted as one of the key habitat features that improve a landscape's ability to support fishers (Service 2004, p. 18773; USFS and BLM 1994a, pp. J2-54, J2-56–J2-57, J2-79). However, more recent analyses of information across the west indicate that fishers' patterns of use of riparian areas are not consistent among studies (reviewed by Lofroth *et al.* 2010, p. 94). For example, ongoing studies that are investigating denning habits and habitat of female fishers indicate that a substantial number of den sites are located on south and east facing slopes and ridges early in the denning season (Thompson 2013, pers. comm.; Chatel *et al.* 2013, pers. comm.; Clayton 2013, pp. 11, 18–19). The researchers' current hypothesis is that thermoregulation considerations by female fishers and their kits (warmer in the late winter and early spring and cooler in the summer) influences seasonal and regional den and rest site selection, and therefore that the availability of den and rest structures in suitable habitat located in a diverse set of abiotic factors is important (Raley *et al.* 2012, pp. 244–245).

In summary, the physical structure of the forest and prey associated with forest structures are thought to be critical features that explain fisher habitat use (Buskirk and Powell 1994, p. 286), and the composition of individual fisher home ranges is usually a mosaic of different forested environments and successional stages (reviewed by Lofroth *et al.* 2010, p. 94). Further, fishers are opportunistic predators with a relatively general but carnivorous diet, and the vulnerability of prey may be more important to the use of an area for foraging than the abundance of a particular prey species (Powell and Zielinski 1994, p. 54). Fishers will use a variety of successional stages when active, reflecting those of their primary prey (Powell 1993, p. 92; Buskirk and Powell 1994, p. 287, Raley *et al.* 2012, p. 241), but fishers appear to be more often associated with stands containing complex forest structure for resting and denning (Buskirk and Powell 1994, pp. 286–287; Powell and Zielinski 1994, p. 53). Thus, a forested landscape that includes sufficient numbers, diversity, and distribution of structural elements suitable for denning, resting, and prey habitat, with moderate to dense overhead canopy for fishers, may be adequate habitat for occupancy. Currently, there are no data available reporting the fitness of fisher populations located in intensively managed landscapes or landscapes composed mostly of older, less intensively managed forests (Raley *et al.* 2012, pp. 252–253).

New information since Service (2014)

Fishers on the Sequoia National Forest have been documented at slightly higher elevations, up to 2,740 m (9,000 ft) (Spencer *et al.* 2015, p. 7). The majority of the higher elevation detections occurred on the Kern Plateau, which receives less snow than other areas at similar elevations (Spencer *et al.* 2015, p.7).

Zielinski and Schlexer (2015 p. 151) found that rest sites in live trees maintained their condition

class and were still available for use 10 years later. They further concluded, “growing trees to 20

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large size is also the best way to guarantee a supply of adequately-sized dead structures” (Zielinski and Schlexer 2015 p. 151).

The type of treatment and amount of area treated are also important factors in determining fisher use of managed forests (Clayton 2013, pp. 12–22; Garner 2013, p. 41; Zielinski *et al.* 2013a, p. 825).

Habitat Models

Numerous large scale habitat models have been developed for various regions within the west coast analysis area (Lewis and Hayes 2004, entire; Carroll *et al.* 1999, entire; Carroll 2005, entire; Davis *et al.* 2007, entire; Zielinski *et al.*, 2010, entire; Spencer *et al.* 2008, entire; Spencer *et al.* 2011, entire; Spencer *et al.* 2012, entire) but none provide a seamless habitat suitability depiction for the entire west coast analysis area. We developed a model (hereafter “fisher analysis area habitat model”) of potential habitat quality for fishers across the west coast analysis area (Figures 2, 3). We provide an overview of the model details below.

We obtained reports of fisher from more than 5,000 points across the analysis area (Figure 4) and selected points for model development that were verified detections (they had physical evidence to verify fisher identification; see the **Distribution and Abundance** section below) and that occurred after 1970. To ensure the spatial independence necessary for model development, if two or more detections were within 5 km of one another, the most reliable and recent detection was retained, or in case of a tie, by random selection. Our detection selection process resulted in 456 verified fisher detection localities for model development.

The analysis area was subdivided based on eco-regional subsection divisions into six overlapping model regions. We subdivided the analysis area to account for potential differences in habitat conditions due to differing ecological conditions and modeled habitat conditions based on 22 environmental predictors (for example, vegetation, climate, elevation, terrain). We did not consider urban and open water areas as having the potential to provide fisher habitat conditions. Three regions of the analysis area (Washington, the northern two-thirds of Oregon, and the central Sierra Nevada) had at the time insufficient numbers and distribution of fisher detections to calibrate the models.

To portray potential fisher habitat for areas with insufficient verified detection data

(Washington, the northern two-thirds of Oregon, and the central Sierra Nevada), we projected modeled habitat from areas with verified detection data onto the adjacent regions with insufficient data. Throughout much of the Cascade Range of Washington and Oregon and parts of the Olympic Peninsula, we developed an expert model to inform potential habitat spatial attributes necessary for this analysis. The modeling resulted in spatial representations of predicted probability of fisher occurrence or potential habitat suitability for each modeling region. We then created three categories of habitat, based on strength of fisher habitat selection in each area populated by fishers. Model values corresponding to habitat preferentially used by fishers were considered to be "high quality"; model values corresponding to habitat avoided by fishers were considered to be "low quality"; and habitat that was neither avoided nor selected was considered to be "intermediate" habitat. In regions where fisher location data were not available to calibrate the

Figure 2. Fisher analysis area habitat model (north half).

Figure 3. Fisher analysis area habitat model (south half).

Figure 4. Fisher locality data for the analysis area; reports are from 1896 to 2013. This figure has not been updated since 2013. We are in the process of updating our database with the new data and will use this information to update locality maps in future fisher status reviews.

habitat categories, habitat was categorized to match neighboring regions. Note that the "low quality" category may include non-habitat as well as areas with some habitat value, but that fishers use infrequently relative to their availability on the landscape. Although our final model predicts the probability of detection, we assume that areas with a higher probability of detection fulfill a greater number or quality of life-requisite needs for fishers and may therefore be used as an index of relative habitat suitability.

New information since Service (2014)

Additional information on the fisher analysis area habitat model is provided in Appendix C of this document.

DISTRIBUTION AND ABUNDANCE

Prehistorical and Historical Distribution across the Range of the Species

Fishers are found only in North America (Anderson 1994, pp. 22–23). The earliest dated occurrence of the genus *Pekania* comes from fossil beds in north-central Oregon and indicates that ancestors of present-day fishers were in North America by at least 7.05 million years ago (Samuels and Cavin 2013, pp. 451–452). Fishers appear in the Pleistocene fossil record approximately 30,000 years ago in the eastern United States throughout the Appalachian Mountains, south to Georgia, Alabama, and Arkansas, and west to Ohio and Missouri (Anderson 1994, p. 18). No fossil evidence of a fisher range expansion to the north or west exists until the middle Holocene (4,000 to 8,000 years ago) in southern Wisconsin, and only within the past 4,000 years is there evidence that present-day fishers inhabited northwestern North America (Graham and Graham 1994, pp. 46, 58). Although there is limited fossil evidence available from central Canada, fishers' expansion westward and northward likely coincided with glacier retreat and the subsequent development of the boreal spruce forests (Graham and Graham 1994, p. 58). Fossil remains of fisher in the northwest occur in paleontological and archeological sites in British Columbia, Washington, and Oregon dating from 4,270 years before present (Graham and Graham 1994, pp. 50–55).

Our present understanding of the historical (before European settlement) distribution of fishers is based on the accounts of natural historians of the early twentieth century and general assumptions of what constitutes fisher habitat. The presumed fisher range prior to European settlement of North America (circa 1600) was throughout the boreal forests across North America in Canada from approximately 60 degrees north latitude, extending south to the Great Lakes area and also along the Appalachian, Rocky, and Pacific Coast Mountains (Figure 5) in the United States (Hagmeier 1956, entire; Hall 1981, pp. 985–987; Powell 1981, pp. 1–2; Douglas and Strickland 1987, p. 513; Gibilisco 1994, p. 60; Lewis *et al.* 2012a, p. 9). The distribution of fishers has been described by numerous authors who delineate different distribution boundaries depending on the evidence used for occurrences.

The presumed presence of fishers has been drawn along the lines of forest distribution, and the species has been consistently described as an associate of boreal forest in Canada, mixed

deciduous-evergreen forests in eastern North America, and coniferous forest ecosystems in the west (Lofroth *et al.* 2010, p. 39). For this reason, range maps of historical distribution typically portray large areas of continuous occurrence, although it is likely that the suitability of habitat to support fishers within the portrayed range varied over time and spatial scales, subject to climatic variation, large-scale disturbances, and other ecological factors (Gibilisco 1994, p. 70; Graham and Graham 1994, pp. 57–58). Fishers do not occur in all forested habitats today, and evidence would indicate they did not occupy all forest types in the past (Graham and Graham 1994, p. 58). Likewise, recent genetic investigations point to the lack of a ubiquitous presence of fishers across the landscape. Tucker *et al.* (2012, entire) identified an apparent break in the distribution and a range reduction along the length of the Sierra Nevada, which they estimated occurred prior to the influence of European settlement.

Unregulated trapping, predator-control efforts, habitat loss and fragmentation, and climatic changes in eastern North America likely contributed to a reduction in range and distribution of fishers in the late 1800s and early 1900s. As a result, the extent of the range contracted in all Canadian Provinces except the Northwest Territory and Yukon Territories (Lewis *et al.* 2012a, p. 11) and only remnant populations remained in the United States in Maine, Minnesota, New Hampshire, New York, and the Pacific States (Powell and Zielinski 1994, p. 41). At its most contracted state in the early 1900s, Lewis *et al.* (2012a, p. 6) estimated that fishers occupied approximately 43 percent of their historical range before European settlement.

Current Distribution Outside of the Analysis Area

Since the 1950s, fishers have recovered in some of the central (Minnesota, Wisconsin) and eastern (New England) portions of their historical range in the United States as a result of trapping closures, habitat regrowth, and reintroductions (Brander and Brooks 1973, pp. 53–54; Powell 1993, p. 80; Gibilisco 1994, p. 61; Lewis and Stinson 1998, p. 3; Proulx *et al.* 2004, pp. 55–57; Lewis *et al.* 2012a, p. 11). Fisher distribution is expanding into Virginia from West Virginia in the Appalachian Mountains, but it is unclear whether they are establishing breeding populations (VDGIF 2012, p. 1).

Presently, fishers are found in all Canadian provinces and territories except Newfoundland, Labrador, and Prince Edward Island (Proulx *et al.* 2004, p. 55, Lewis *et al.* 2012a, p. 11) (Figure 5). The fisher range in Quebec, Ontario, and eastern Manitoba is contiguous with currently occupied areas in New England, northern Atlantic states, Minnesota, Wisconsin, and the Upper Peninsula of Michigan in the United States (Proulx *et al.* 2004, pp. 55–57; Lewis *et al.* 2012a, p. 11). In Saskatchewan and Alberta, fishers are found primarily north of 52 degrees and 54 degrees north latitude, respectively, and are not connected to breeding populations of fishers in the United States (Proulx *et al.* 2004, p. 58; Lewis *et al.* 2012a, p. 11). Fishers occupy low- to mid-elevation forested areas throughout British Columbia, but are rare or absent from the coast and from the southern region of the province for at least 200 km (125 mi) to the border with the United States (Weir *et al.* 2003, p. 25; Weir and Lara Almuedo 2010, p. 36). Eighty-eight fishers were legally harvested from the South Thompson Similkameen area of south-central British Columbia, bordering north-central Washington, between 1928 and 2007; and of these only 13 were harvested since 1985 (Lofroth *et al.* 2010, p. 48). This region is south of the established fisher population distribution in the province (Weir and Lara Almuedo 2010, p. 36); therefore,

Figure 5. The fisher's historical, most-contracted, and current ranges. (Adapted from Lewis *et al.* 2012a, Figure 8.)

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the significance of the trapping data in this region is not clear without more specific location information. These harvest data could indicate that individuals were captured at the periphery of larger, established populations, that there is a low-density population in south-central British Columbia, or that individuals represent transient or extralimital (outside an established population area) records. Contemporary fisher distribution in U.S. Northern Rocky Mountains of western Montana and Idaho covers an area similar to that depicted in the historical distribution synthesized by Gibilisco in 1994 (p. 64). The historical and contemporary distribution of fishers in the U.S. Northern Rocky Mountains is described in detail in our 12-month finding for the Northern Rocky Mountain DPS (76 FR 38504, June 30, 2011) including forested areas of western Montana and north-central to northern Idaho.

Distribution within the Analysis Area

At the beginning of the twentieth century in the Pacific States and Provinces, the fisher's range and distribution were described as "broadly distributed," but "generally rare" (Lofroth *et al.* 2010, p. 39). Hagmeier (1956, p. 152) reported fishers to be "common throughout most of the forested regions" of British Columbia, apparently supporting a regular fur harvest across 90 percent of the province (Rand 1944, p. 79). In Washington, fishers historically occurred throughout densely forested areas both east and west of the Cascade Crest, on the Olympic Peninsula, and probably in southwestern and northeastern Washington (Dalquest 1948, pp. 187–189; Aubry and Houston 1992, pp. 69–70; Lewis and Stinson 1998, pp. 4–5). In Oregon, Bailey (1936, pp. 298–299) reports fishers occurred in the boreal forest zones of the Cascade Range from Washington to California, west to the coniferous coastal forests and cool humid Coast Ranges; this report also extends their range to the northeastern portion of the state near the Washington and Idaho borders. In the forested, higher mountain masses of California, Grinnell *et al.* (1937, pp. 214–215) describe fishers as ranging from the Oregon border southward through the Coast Range to Lake and Marin Counties, east through the Klamath Mountains to Mount Shasta, and south throughout the main Sierra Nevada to Greenhorn Mountain in northern Kern County. Recent genetic research (Knaus *et al.* 2011, p. 11; Tucker *et al.* 2012, entire) contradicts the Grinnell *et al.* (1937, p. 216) assumption that there was a continuous population from Mt. Shasta through to the southern Sierra Nevada.

To describe the current distribution of fishers in the analysis area, we used various sources of

information. We compiled fisher locality data from published and unpublished literature (Zielinski *et al.* 1995, entire; 1997a, entire; 1997b, entire; 2000, entire; 2005, entire; 2010, entire; Zielinski and Stauffer 1996, entire; Slauson and Zielinski 2007, p. 19; Beyer and Golightly 1996, p. 18; Dark 1997, p. 31; Carroll *et al.* 1999, p. 1347; Zielinski *et al.* 2000, p. 28; Zielinski *et al.* 2010, pp. 41,47; Slauson and Zielinski 2001, p. 12; Hamm *et al.* 2003, p. 203; Slauson *et al.* 2003, pp. 20–21; Farber and Criss 2006, p. 11; Thompson 2008, entire; Lindstrand 2006, p. 49, 2010, p. 18; Spencer *et al.* 2008, p. 44; Spencer *et al.* 2011, entire), and telemetry research studies conducted between 1977 and 2013 (Buck *et al.* 1979, p. 171; Self and Kerns 2001, p. 24; Zielinski *et al.* 2004b, p. 652; Yaeger 2005, p. 4; 2008; Self and Callas 2006, p. 10; Thompson *et al.* 2010, entire; Clayton 2011, pers. comm.; Sweitzer and Barrett 2010, entire); submissions from the public during the information collection period; and information from individual fisher researchers, private companies, and agency databases, including entries to the U.S. Department of Agriculture Forest Service (USFS)'s *Forest Carnivore Surveys in the Pacific States* database.

The *Forest Carnivore Surveys in the Pacific States* database provided an archive and retrieval system for data from standardized forest carnivore surveys conducted in the Pacific states, regardless of their success or failure to detect target species. Figure 4 depicts locality information from reports of the species in the analysis area from 1896 through 2013

In compiling the location information to describe the fisher's current distribution, we considered the biology of this cryptic species and the differing amount and type of information associated with each locality point. Like most forest mesocarnivores, fishers are difficult to detect. They also are wide ranging animals with males making regular long distance movements, particularly during the breeding season (Leonard 1986, p. 41; Arthur *et al.* 1989, p. 678) and when dispersing (York 1996, p. 49; Aubry and Raley 2006, p. 14; Weir and Corbould 2008, p. 47; Matthews *et al.* 2013, p. 105; Sweitzer *et al.* 2015d, p. 82). Such movements can make it difficult to distinguish with certainty between occurrence records that represent established populations in suitable habitats and records that represent short-term occupancy or exploratory movements without the potential for establishment of home ranges, reproduction, or populations.

Determining that an area is unoccupied by fishers is also difficult. Fishers within the analysis area tend to live in remote locations where they are seldom encountered, documented, or studied. They naturally occur at low population densities and are rarely and unpredictably encountered where they do occur. They are territorial and require expansive areas of forested habitat for each individual, meaning large areas may be occupied by just a few individuals, thus reducing their likelihood of detection. In addition, many mobile species are difficult to detect in the wild

because of morphological features (such as camouflaged appearance) or elusive behavioral characteristics (such as nocturnal activity) (Peterson and Bayley 2004, pp. 173, 175). While positive fisher detections, using techniques such as sooted track plates and remotely triggered cameras, are conclusive, non-detections (inferred absence) are based on detection probability, which in turn is strongly influenced by survey effort. Slauson *et al.* (2009, p. 35) recommend using caution when interpreting the results of previous surveys because the use of inconsistent survey protocols has resulted in varying survey effort. Slauson *et al.* (2009, p. 35) recommend a minimum effort of at least 200 functional days for summer season surveys, and a minimum of 60 functional days of survey effort per sample unit during non-summer surveys to achieve a probability of detection greater than 95 percent. Surveys below these thresholds may be insufficient to conclude that fishers are absent.

Due to the challenge associated with survey efforts in relatively remote and inaccessible areas, as well as the lack of sufficient resources, we often lack adequate information to definitively determine whether fishers occupy an area or not. It is also difficult to precisely determine their current range or estimate past trends in range contraction or expansion. Assumptions about whether an area is occupied or unoccupied must be based on limited information, which can also be interpreted in several ways. Therefore, we used multiple lines of evidence to determine where fisher populations occurred in the past and where they presently occur.

Lines of Evidence for Past and Current Distributions of Fishers

As we stated previously, our present understanding of the historical distribution of fishers is based on the accounts of natural historians of the early twentieth century and their general

assumptions of what constitutes fisher habitat. These historical efforts did not typically have the rigorous standards imposed on today's information. With the passage of environmental legislation in the 1970s, such as the National Environmental Policy Act of 1970 and ESA, scientifically defensible information about the status of wildlife has become increasingly required to support management decisions. The development of rigorous non-invasive survey methods for carnivores such as sooted track plates and remotely triggered cameras became prevalent in the mid-1990s. In 1995, Zielinski *et al.* (1995, entire) published a manual that described protocols for detecting forest carnivores. This manual allowed for a standardization of surveys and provided a means for comparison between verified records of detections of various forest carnivores, including the fisher.

Verifiable records are records supported by physical evidence such as museum specimens, harvested pelts, DNA samples, sooted track plate impressions, and diagnostic photographs. Documented records are those based on accounts of fisher being killed or captured. Use of only verifiable and documented records avoids mistakes of misidentification often made in eyewitness accounts of visual encounters of unrestrained animals in the wild. Visual-encounter records often represent the majority of occurrence records for elusive forest carnivores, and they are subject to inherently high rates of misidentification of the species involved, including fishers (McKelvey *et al.* 2008, pp. 551–552). Visual-encounter records of a fisher itself, or its sign, by the general public or untrained observer may be found in agency databases; however, correct identification of fisher or its sign can be difficult by an untrained observer. Thus, these unverified records or anecdotal reports need to be viewed cautiously (Aubry and Lewis 2003, p. 81; Vinkey 2003, p. 59; McKelvey *et al.* 2008, p. 551). Other animals that are similar in appearance and share similar habitats, such as the American marten (*Martes americana*), mink (*Mustela vison*), or domestic cat (*Felis catus*), may be mistaken for fishers (Aubry and Lewis 2003, p. 82; Lofroth *et al.* 2010, p. 11; Kays 2011, p. 1). Animal signs, such as snow tracks, can be significantly altered by environmental conditions, and difficult to identify (Vinkey 2003, p. 59). On natural substrates, fisher tracks can be confused with those of the more common American marten.

We assigned a numerical reliability rating (following Aubry and Lewis 2003, p. 81) to each fisher occurrence record as follows:

- 1) Specimens, photographs, video footage, or sooted track-plate impressions (records of high reliability that are associated with physical evidence); 2) Reports of fishers captured and released by trappers or treed by hunters using dogs (records of high reliability that are not associated with physical evidence); 3) Visual observations from experienced observers or from individuals who provided detailed descriptions that supported their identification (records of moderate reliability); 4) Observations of tracks by experienced individuals (records of moderate reliability); 5) Visual observations of fishers by individuals of unknown qualifications or that lacked detailed descriptions (records of low reliability); and 6) Observations of any kind with inadequate or questionable description or locality data (unreliable records).

The development and use of rigorous survey methods to collect data on fisher began approximately 20 years ago, just prior to the publication of Zielinski *et al.*'s (1995, entire) survey

protocol manual; therefore, we have chosen 1993 as the beginning of the contemporary period. We evaluated all records with reliability ratings 1 through 6 for insight into past population distribution (prior to 1993). We consider reliability ratings 1 and 2 as the best available information on fisher locations. Because the use of unreliable records to support distribution and population extent has led to overestimation of current ranges (Aubry and Lewis 2003, p. 86; McKelvey *et al.* 2008, p. 551), we used only the most reliable and verified data from 1993 to 2013 in our analysis of the current distribution of fisher populations in the analysis area. A 20-year timeframe provides for the most recent evaluation of contemporary fisher distribution because of the substantial efforts made over the last 20 years to assess the status of fisher and other forest carnivores in the analysis area using opportunistic surveys and systematic grids of baited track and camera stations (Figure 6). We base the contemporary (1993 to 2015) distribution of fisher populations on verifiable or documented records of physical evidence such as animals captured for scientific study, genetic analysis of biological samples, and photographs or track plate impressions (reliability ratings 1 and 2; Figure 7). Note that Figures 6 and 7 do not reflect additional surveys, or detections with reliability ratings 1 and 2 we received as a result of public comment.

Past (1896 to 1993) and Current Distribution within the Analysis Area

All locality data prior to 1993 demonstrates a distribution that generally conforms to the presumed historical distribution (Figure 8). A map showing the dataset constrained to reliability codes 1 through 4 from 1953 to 1993 suggests fishers still occurred at various locations on the landscape throughout their historical distribution (Figure 9). However, in much of the analysis area, especially in Washington and northern Oregon, the scarcity of reports suggests that fishers were quite rare during these decades. For the period prior to 1993, the most reliable data from these areas come from reports of incidental capture of fishers. There have been few fishers captured in Washington in recent decades (1 each in 1969, 1971, 1987, 1990, and 1992) (Lewis and Stinson, 1998, pp. 23, 53). Three of these fishers were captured incidental to bobcat, marten, and coyote trapping efforts since 1985, in approximately 2.4 million trap-nights, which in part led Lewis and Stinson (1998, p. 23) to conclude, “The fisher is rare in Washington. Infrequent sighting reports and incidental captures indicate that a small number may still be present. However, despite extensive surveys, the Department has been unable to confirm the existence of a population in the state,” and “We believe that remaining fishers in Washington are unlikely to represent a viable population, and without recovery activities, the species is likely to be extirpated from the state” (Lewis and Stinson 1998, p. 36). However, the Washington Department of Fish and Wildlife (WDFW) clarified during the open comment period that they are concerned with the conclusions that could be made from the fisher records in Washington in recent decades. Specifically, they pointed out that three of those recent detections were escapees from a wildlife park and a fisher that had been reintroduced in Montana, and that they consider the 1969 fisher record to be the last verified record of a native fisher in Washington (WDFW 2015).

In the same time period in Oregon, few incidental captures were reported and all either appeared to be associated with the Southern Oregon Cascades Reintroduced Population (see below), or 31

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occurred to the south of this reintroduced population (Robart 1982, pp. 8–9). Fisher locations in northern Oregon are therefore exclusively derived from the less reliable visual sightings and unverified track locations.

Throughout the Coast Ranges of Oregon and Washington and the Cascades north of the reintroduced Southern Oregon Cascades Population, infrequent verified detections, all prior to 1993, suggest the species has been reduced to scattered individuals or remote isolated populations. Based on the available verified detection data, two native populations of fishers were identified in the southern portion of the analysis area: one in the southern Sierra Nevada (Southern Sierra Nevada Population) and the other in northern California and southwestern Oregon (Northern California-Southwestern Oregon Population) (Figure 7; Table 1). Reports resulting from systematic surveys suggested that fishers appeared to occupy less than half of the range in California than they did in the early 1900s (Zielinski *et al.* 1995, p. 108; Zielinski *et al.* 2005, p. 1394; CDFW 2015, p. 23), based on the assumption that the two populations had until recently been connected. However, Tucker *et al.* (2012, p. 3) estimated that the two populations have been separated for more than 1,000 years. The new information provided in Tucker *et al.* (2012, entire) makes drawing conclusions about the extent of the loss of historical range within California difficult.

New information since Service (2014)

In preparation of their fisher status review, California Department of Fish and Wildlife (CDFW 2015) staff reviewed historical records and reported anecdotal evidence concerning the presence of fishers in the northern and central Sierra Nevada. Records collected came from naturalists' reports, CDFW trapping records, and USFS reports (CDFW 2015, pp.17-21). The approach used by CDFW to describe the historical distribution of fishers in California is consistent with our previous approach. We consider these records that rely on both naturalists' reports and trapping records to constitute reliability ratings of 2 or 3 and therefore indicate fisher presence in the northern and central Sierra Nevada at least until the 1920s.

Tucker *et al.* (2014) reexamined genetic data for fishers in the Southern Sierra Nevada (SSN) population. This reanalysis of genetic data was inconclusive relative to whether or not genetic data could be used to indicate an expansion of the SSN population northward. However, a summary of recent survey data was provided. The summarized survey information states that surveys conducted in the 1990s resulted in few detections of fishers in the north genetic group of

the SSN population (1 to 2 fishers depending on study area). Current surveys in the north genetic group that are designed to collect genetic data on individuals are detecting 25–44 fishers depending on study area (Tucker *et al.* 2014, p. 131). Reliability ratings for the data collected for this analysis are considered to be highly reliable (reliability rating 1).

Figures 6- 9 have not been updated. We received additional locality data during public and peer review comment periods and the public hearing for our proposed listing rule. Location data we received was categorized with reliability ratings of 1 through 5. Information with reliability ratings of 1 do not include new location data beyond the current population boundaries with the exception of detections in the Southern Oregon Cascades and the southern Cascades of California (see the **Current Distribution of Reintroduced Populations** section below). We 32

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Figure 6. Opportunistic and systematic surveys (with both positive and negative results), fisher trapping efforts for research, and other verifiable records (for example, fisher telemetry data) from 1993–2013. Figure has not been updated. We are in the process of updating our database with the new data and will use this information to update locality maps in future fisher status reviews.

Figure 7. Locality records 1993 to 2013 for reliability ratings 1 and 2. Please note that the ONP population here is represented by a single dot, and this representation is based on the information we received from WA Department of Fish and Wildlife. Figure has not been updated. We are in the process of updating our database with the new data and will use this information to update locality maps in future fisher status reviews.

Figure 8. All records prior to 1993. This map displays records with reliability ratings 1 through 6. Here we have presented fisher detections locations with all reliability ratings (1-6) to illustrate the probable historical distribution of fishers.

Figure 9. Fisher records 1953 to 1993 with reliability ratings 1 to 4. These detections are presented to illustrate that fishers still occurred at various locations throughout their historical distribution. Reliability ratings of 5 and 6 are not depicted due to their low reliability.

recently obtained data from ORBIC (2015) of fishers in the central and northern Oregon Coast Range, either pre-1993 records or more recent non-verifiable records. As part of that same data set, we obtained a record of a fisher skull (reliability rating 1) found southwest of Roseburg, Oregon, in Douglas County. We are in the process of updating our database with the new data and will use this information to update locality maps in future fisher status reviews. Three reintroduction efforts have resulted in repeated detections of fishers: one in the northern Sierra Nevada (Northern Sierra Nevada Reintroduced Population), one in the southern Oregon Cascade Range (Southern Oregon Cascades Reintroduced Population), and one on the Olympic Peninsula of Washington (Olympic Peninsula Reintroduced Population). The Southern Oregon Cascade Population is separated from the next known populations to the north in British Columbia by more than 800 km (500 mi) and from the Olympic Peninsula by over 400 km (250 mi). As discussed in the **Current Distribution of Reintroduced Populations** section below, the reintroduced Southern Oregon Cascades Population is well established but the other two reintroduced populations are very new and their long-term stability is not yet certain. It is encouraging to note from ongoing monitoring efforts that fishers are persisting and reproducing after the last year of fisher releases on the Olympic Peninsula (4 years) and in the northern Sierra Nevada (3 years).

Table 1. Population occurrences and estimates of current range extent.

| Population State Native / Reintroduced | | Range Extent (km ²) | |
|--|--|---|-----------|
| Percent of Analysis Area | | Olympic Peninsula Washington Reintroduced | 11,000 3% |
| Oregon Reintroduced | | Oregon Cascades | 5,000 1% |
| Northern California- Southwestern Oregon | | California and Oregon | |
| Native | | 40,000 | 11% |
| Northern Sierra Nevada California Reintroduced | | 2,000 | 1% |
| Southern Sierra Nevada California Native | | 12,700 | 4% |
| Analysis Area | | 353,956 | 100% |

New Information Between December 2015 and February 2016, 23 fishers were released into the southern Washington Cascades. These fishers were released as the first stage of the reintroduction effort being implemented by WDFW, which ultimately plans to release a total of 160 fishers between two recovery zones in the Washington Cascades.

Current Distribution of Populations (1993 to Present)

A scarcity of verifiable sightings in Washington, northern Oregon, and central Oregon suggests that these populations appear to be likely extirpated, except on the Olympic Peninsula where they have been recently reintroduced (see the **Current Distribution of Reintroduced Populations** section below). However, we cannot be sure that a lack of detections in Washington and much of Oregon indicates the species is entirely absent. In Washington,

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cumulative years of trapping, fisher and other carnivore survey efforts, and review of fisher sighting reliability information led Lewis and Stinson (1998, p. 36) to conclude, “The fisher is rare in Washington. Infrequent sighting reports and incidental captures indicate that a small number may still be present. However, despite extensive surveys, the Department has been unable to confirm the existence of a population in the state.” In addition to the survey efforts in Washington mentioned above, there are large areas in coastal Oregon and Washington and in the central Oregon Cascades where surveys have not been conducted, and survey efforts are relatively sparse in the Cascades of Washington and northern Oregon (Figure 6) (Oregon Department of Fish and Wildlife (ODFW) 2015 pers. comm.). Although functioning populations like those we see in southern Oregon and California appear not to be present, it is possible, particularly in unsurveyed areas, that an isolated remnant population could be overlooked (Hudgens and Garcelon 2013, pp. 2-4, 9-10). WDFW’s current assessment of fisher presence in Washington is that they consider the 1969 fisher record to be the last verified record of a native fisher in Washington (WDFW 2015) until reintroductions began in 2008. The Service considers populations in Washington to be likely extirpated other than in reintroduction areas.

For example, in the U.S. Northern Rocky Mountains (USNRM), fishers were thought to be extirpated by 1930 from Montana and Idaho, as they were in other parts of the United States (Newby and McDougal 1964, p. 487; Weckwerth and Wright 1968, p. 977). Several reintroductions were initiated by Montana and Idaho Departments of Fish and Game, resulting in the release of 188 fishers originating from central British Columbia, Minnesota, and Wisconsin between 1959 and 1991 in north-central Idaho and northwestern and west-central Montana (Weckwerth and Wright 1968, p. 979; reviewed by Vinkey 2003, p. 55; Roy 1991, p. 18; Heinemeyer 1993, p. i). Subsequent to these reintroductions, genetic analyses revealed a remnant native population of fishers in the USNRM that survived the presumed extirpation thought to have occurred early in the twentieth century (Vinkey *et al.* 2006, p. 269; Schwartz 2007, p. 924). Fishers in the USNRM today reflect a unique genetic legacy and identity from this remnant native population combined with the genetic contributions from fishers introduced from British Columbia and the Midwest.

Northern California-Southwestern Oregon (NCSO)

The NCSO population includes the original native fisher population in northern California and southern Oregon, the Southern Oregon Cascades (SOC) and Northern Sierra Nevada

(NSN) Reintroduced Populations.

The fishers in the Southern Oregon Cascades are descendants of fishers that were introduced from British Columbia and Minnesota in 1961 and from 1977 to 1981 (Aubry and Lewis 2003, pp. 82–85, 87; Drew *et al.* 2003, p. 57, 59). This population occurs in portions of Douglas, Jackson, and Klamath Counties with verified detections from near Lemolo Lake in the north to Hyatt Reservoir in the south. Information on the current distribution of this population on the western boundary of Crater Lake National Park is from data collected during a 6-year telemetry effort (Aubry and Raley 2006, p. 5). On the eastern extent of the range of this population, we have trail camera photographs documenting fisher use of the western shore of Upper Klamath Lake. The Southern Oregon Cascades Population appears to be persisting without additional augmentations.

Fishers in the Southern Oregon Cascades are relatively close (within 40 km (25 mi)) to the Northern California-Southwestern Oregon Population, but are separated by a relatively narrow band of forested habitat and the heavily traveled Interstate 5. No genetic exchange has been documented (Aubry *et al.* 2004 p. 214; Drew *et al.* 2003, p. 59; Wisely *et al.* 2004, p. 646; Farber *et al.* 2010, p. 12) between these populations. However, one male fisher from the Northern California-Southwestern Oregon Population was detected east of Interstate 5, approximately 30 km (19 mi) south of the Southern Oregon Cascades Population in 2012 (Pilgrim and Schwartz 2012, pp. 4-5).

The NCSO population occurs in the Klamath Mountains of southwestern Oregon in Josephine, Jackson, and Curry Counties in Oregon and extends south into California through the Klamath Mountains and Coast Ranges of Del Norte, Siskiyou, Humboldt, Trinity, western Tehama, northeastern Mendocino, western Glenn, northern Lake, and western Colusa Counties and in the Cascade Range of southern Siskiyou and Shasta Counties. Surveys conducted in 2011 and 2012 at the eastern edge of this population in eastern Shasta County detected fishers where prior surveys conducted in 2003 did not. It is unclear if these recent detections represent an expansion front or are just wide ranging or dispersing males. At the southwestern edge of this population in southern Lake County, a photograph of a fisher over 60 km (37 mi) south of any previous reports was taken by a remote camera in March 2013. We have no other survey efforts occurring in this vicinity, so it is unknown whether this single detection represents an established population or represents a wide-ranging male during the breeding season.

In California, fishers were introduced into the northern Sierra Nevada from 2009-2011. The introduction was as a cooperative venture between the Service, the California Department of Fish and

Wildlife (CDFW), and Sierra Pacific Industries (SPI). Two of the 11 objectives of this reintroduction were to implement an experimental design and monitoring effort to assist with determining and describing mortality, movement patterns, and habitat use of released fishers on private industrial timberlands and to return fishers to their historical range in the northern Sierra Nevada (Service 2008, pp. 2-3). Forty fishers (16 males and 24 females) were relocated from northwestern California to the northern Sierra Nevada in the vicinity of Butte, Plumas, and Tehama Counties (Callas and Figura 2008, entire). Project plans call for monitoring these fishers for 7 years to determine the extent of their distribution into the northern portion of the Sierra Nevada (Callas and Figura 2008, p. 65). The success of this introduction will not be known for several years. Before this introduction, the Southern Sierra Nevada population was separated from the Northern California-Southwestern Oregon Population by approximately 400 km (250 mi) (Zielinski *et al.* 1995, pp. 107–108; 2005, p. 1394). With the reintroduction, this distance has been reduced to approximately 280 km (175 mi).

New information since Service (2014)

The NCSO population of fishers is the largest of the fisher populations and is located in Curry, Douglas, Josephine, Jackson, Klamath, and Lane counties in southern Oregon and in Butte, Del Norte, Humboldt, Lake, Mendocino, Plumas, Shasta Siskiyou, Tehama, and Trinity counties in northern California. In our proposed rule, we considered that the NCSO and SOC populations may be connected by dispersing fishers.

Recent surveys conducted in the southern Oregon Cascades, Jackson County, have detected two individuals with genetic haplotypes consistent with the NCSO population. One male fisher and one female fisher from the NCSO population were detected east of Interstate 5, approximately 30 km (19 mi) south of the SOC population in 2012 and approximately 56 km (35 mi) east of Interstate 5 in 2014, respectively (Pilgrim and Schwartz 2012, pp. 4–5; Pilgrim and Schwartz 2015, entire). This recent detection of a male and a female fisher is where individuals from the SOC population were also found, indicating that these populations may be in the initial stages of convergence (Pilgrim and Schwartz 2014, entire; 2015, entire).

Until the reintroduction of fishers as part of the Sierra Pacific Industries (SPI) Stirling Management Unit Candidate Conservation Agreement with Assurances (CCAA), the southern boundary of the NCSO fisher population was Shasta County, California. Ongoing monitoring of these reintroduced fishers indicates that they are reproducing and have expanded their occupancy northward into the surrounding forested areas beyond the original footprint of the reintroduction. The distance between the NSN and original NCSO population is now approximately 40 km (25

mi) which is within the dispersal distance potential of male fishers. Unlike the SOC reintroduced fishers, NSN fishers are the same genetic haplotypes as those in NCSO, thus we will be unable to confirm when successful reproduction has occurred between the original and reintroduced fishers. Based on the new information submitted since the proposed rule we now consider that the NCSO fisher population includes areas formerly identified as being the SOC and NSN reintroduced populations.

Southern Sierra Nevada

The current extent of occurrence of the Southern Sierra Nevada Population in California includes portions of Mariposa, Madera, Fresno, Tulare, and Kern Counties. While historically the population extended farther north, today the northern limit is the Merced River in Yosemite National Park in Mariposa County. The southern limit is the forested lands overlooking the Kern River Canyon, while the eastern limit is the high elevation, granite-dominated mountains, and the western limit is the low elevation extent of mixed conifer forest. This population currently occupies the west slope of the southern Sierra Nevada from the Merced River drainage in Yosemite National Park, south through the Greenhorn Mountains at the southern extent of the Sierra Nevada.

Current Distribution of Reintroduced Populations

Lewis *et al.* (2012b, entire) reviewed data from 38 translocations of fishers in North America. Their analysis also included population modeling and field data from actual reintroduction efforts to provide insight into what factors influence the success or failure of efforts to restore fisher populations. Their results and management recommendations for influencing success of reintroductions include efforts that are slightly female biased, adult biased, release 60 or more fishers, and utilize source populations close to release sites. Based only on the parameter of total number of fishers released, large releases such as the Olympic Peninsula reintroduction (>80 fishers) have a predicted index of success of 80% while those that release fewer than 60 fishers are predicted to have less than a 50% success rate (Lewis *et al.* 2012b, pg. 7). Overall, the

success rate for fisher reintroductions in North America is 77 percent which is twice the probability of success documented in western North America (Lewis *et al.* 2012b, pg. 10).

Olympic Peninsula

The Washington Department of Fish and Wildlife (WDFW), in cooperation with the Olympic National Park, United States Geological Survey, and others, began to reintroduce fishers onto Park Service lands on the Olympic Peninsula in Washington in January 2008 (Lewis and Happe 2008, p. 7). These reintroductions were complete at the end of 2010 with a total of 90 fishers (40 males and 50 females) relocated from British Columbia to Olympic National Park (Lewis *et al.* 2011, p. 4). These fishers will be monitored for a number of years to determine both the extent of their distribution and success in establishing a population of fishers on the Olympic Peninsula. The success of this introduced Olympic Peninsula population will not be known for several years.

Southern Oregon Cascades

New information since Service (2014)

See the *Northern California-Southwestern Oregon (NCSO)* section above, which includes the original native fisher population and the Southern Oregon Cascades (SOC) and Northern Sierra Nevada (NSN) Reintroduced Populations

Northern Sierra Nevada

New information since Service (2014)

See the *Northern California-Southwestern Oregon (NCSO)* section above, which includes the original native fisher population and the Southern Oregon Cascades (SOC) and Northern Sierra Nevada (NSN) Reintroduced Populations

Washington Cascades

The WDFW began a fisher reintroduction project in the South Cascades of Washington State on December 3, 2015. Since February 10, 2016, 23 fishers have been released from the Cispus Learning Center along the Cispus River. This project is the second phase of WDFW's efforts to recover fishers in Washington according to the Washington Fisher Recovery Plan (Hayes and Lewis 2006, p. 39). The reintroduction plan (Lewis 2013, p. v) calls for a total of 160 fishers to be released into the Cascade Mountains at a rate of 40 per year for four years (two years in the South Cascades, two years in the North Cascades). The source population for the fishers (British Columbia) is the same as for the Olympic Peninsula reintroduction. The WDFW Fisher Recovery Plan (Hayes and Lewis 2006, p. vii) has the goal of establishing multiple self-sustaining populations of fishers in Washington. We are not referring to this group of fisher individuals in the South Cascades as a population at this time because they have not yet had the

opportunity to successfully interbreed. These animals are not physically or demographically connected to any other populations of fishers.

Population Status

Estimates of fisher abundance and vital rates are difficult to obtain and often based on harvest records, trapper questionnaires, and tracking information (Douglas and Strickland 1987, p. 522), and recent information is limited. Habitat modeling and behavioral or other natural history characteristics (for example, home range sizes) also are used to estimate population sizes over a geographic area (Lofroth 2004, pp. 19–20; Lofroth *et al.* 2010, p. 50). Fisher densities over areas of suitable habitat have been reported, but there are no total or comprehensive population sizes for the fisher in the eastern United States or Canada. In the western range, fisher population size has been estimated using habitat models and home range size estimates. Habitat-based methods likely overestimate population sizes because some apparently suitable habitat may not be occupied. A combination of habitat modeling, protocol surveys, and occupancy modeling can improve habitat-based population estimates.

Based on trapping records from the 1920s, Grinnell *et al.* (1937, p. 227) provided an estimate of 1 fisher per 259 km² (100 mi²), equating to 300 fishers in California. The Grinnell *et al.* population estimate for California is incorrect by modern standards due to the lack of a significant sample size, survey bias, and inadequate knowledge of the historical baseline, although they employed accepted methodologies at the time they conducted their research.

Despite the lack of precise empirical data on fisher numbers in the analysis area, the reduction in the range of the fisher on the west coast, as indicated by the lack of detections or sightings over much of its historical range, and apparent isolation from the main body of the species range (Drew *et al.* 2003, p. 59; Wisely *et al.* 2004, p. 646; Knaus *et al.* 2011, p. 11; Lewis *et al.* 2012a, p. 11; Tucker *et al.* 2014, pp. 132-133), reveal that the extant fisher populations are reduced in size relative to our understanding of their historical distribution.

Northern California-Southwestern Oregon

As described above, the NCSO population includes the original native fisher population in northern California and southern Oregon, the Southern Oregon Cascades (SOC), and the Northern Sierra Nevada (NSN) Reintroduced Populations.

No published population or density estimates are available for the entire NCSO Population. There are density estimates for several individual study areas (Zielinski *et al.* 2004b, p. 654; Thompson 2008, entire; Matthews *et al.* 2011, entire; Swiers 2013, entire; Table 2). These studies, with population density estimates varying by two orders of magnitude from 18 to 52 animals per 100 km², show how difficult it is to extrapolate to an overall population estimate.

In studies that have measured fisher populations over time, some have observed stable densities and others have recorded substantial changes. Using genetic mark-recapture techniques, Swiers (2013, pp. 19-20) estimated a stable annual population ranging from 29 to 35 from 2007 to 2011 on the 510 square kilometers (km²) (197 square miles [mi²]) Eastern Klamath Study Area in northern Siskiyou County, California, and southern Jackson County, Oregon, with an estimated population growth rate of 1.06 (95% confidence interval [CI] 0.97–1.15). Using mark-recapture

techniques, Matthews *et al.* (2011, p. 72) reported a decline in population density estimates from 52 (95 percent CI = 43–64) fishers per 100 km² (38.6 mi²) in 1998, to 14 (95 percent CI = 13–16) fishers per 100 km² (38.6 mi²) in 2005 on the Hoopa Valley Indian Reservation in the Klamath Mountain Range (eastern Humboldt County, California). The authors speculated that this 73 percent decline may have been a result of increased predator densities, disease, decreased prey availability due to changes in prey habitat, or some combination of these (Matthews *et al.* 2011, pp. 72–73). Higley and Matthews (2009, p. 22) reported that the 2005 Hoopa study may have begun when the local population was rebounding from an unknown devastating effect, but a population growth rate of 1.03–1.12 (95% CIs span 1; Higley and Matthews 2009, p. 66) and shift in age structure since then indicate the population is showing signs of stability or increase. It remains unclear, however, if this was a localized decrease in what may have been temporarily a very dense population in 1998 on the Hoopa Reservation, or something occurring over a larger geographic area. While using different techniques, fisher surveys on adjacent land owned by industrial timber landowner, Green Diamond Resource Company (Humboldt County, California), did not detect declines over a similar time period, suggesting that the declines seen in the Hoopa study may have been localized (Thompson 2008, p. 23).

It should be noted that both the Hoopa and Eastern Klamath study area population growth rate estimates within this population have 95 percent confidence intervals spanning 1, which indicates a declining population if less than 1 and a stable to slightly increasing population if equal to 1 or greater. These growth rates were measured in study areas where fishers were abundant enough to generate adequate sample sizes for statistical analysis. Other studies in the NCSO population had insufficient data, were not designed to estimate population growth rates, or were not conducted over a long enough time period to assess population parameter. Given the

small portion of the NCSO population sampled by the two study areas (0.62% of the entire area, 1.08% of modeled intermediate and high probability fisher habitat), it is difficult to determine whether the NCSO population as a whole is increasing, decreasing, or stable.

There have been several approaches used to estimate NCSO population size. One unpublished study, by Self *et al.* (2008, pp. 3–5), used fisher density estimates derived from a variety of study areas within the NCSO population, and calculated that 4,018 fishers might be present in the population. However, this is likely a large overestimate, because the analysis assumes that habitat is occupied at the same densities as observed within the study areas, which may not be representative of fisher density throughout the area occupied by the population. A preliminary analysis based on spatially explicit habitat and population models, with parameters chosen to best match actual fisher occupancy and breeding (Matthews 2013, pers. comm.), suggests an equilibrium population size of approximately 2,790 to 3,990 individuals (Spencer 2014, pers. comm.; Rustigian-Romsos 2013, pers. comm.). However, there is no information on whether or not the current population is near its equilibrium size. Tucker *et al.* (2012, pp. 7, 9–10) used genetic data to calculate an effective population size of 129, which corresponds to an actual population size between 258 and 2,850. This number could be influenced by small population sizes over a number of past generations, likely including the time period when fisher trapping was legal (Tucker 2013, pers. comm.). Based on these various approaches, the NCSO population estimates range from a population size of 258 to 4,018.

Additional insight into the status of the NCSO population comes from occupancy modeling and from protocol surveys located both inside and outside the study areas listed above. A positive survey indicates that fishers were present at the survey location, but a negative survey can result either from the absence of fishers or from a failure to detect fishers that were present.

Occupancy modeling is a method to correct for these false-negative survey results. The California Department of Fish and Wildlife surveyed 86 sites, each consisting of 2 stations separated by 1.6 km, within forested lands of the Klamath and California Coast Ranges. They observed fishers at approximately 41 percent of these sites (Furnas 2014, pers. comm.). Using occupancy modeling, Furnas (2014, pers. comm.) estimated that fishers were present at 65 percent (90 percent CI 53–79 percent) of the survey sites.

We mapped our database of fisher surveys (Figure 6) onto a hexagonal, 1,000-ha grid depicting hypothetical fisher home ranges within the area occupied by the NCSO population (Figure 10). There were 1,274 hexagons that contained at least one survey location between 2003 and 2013; 34 percent of these hexagons contained at least one positive survey, whereas 66 percent included

only negative surveys. Within high-value modeled habitat, the percentage of hexagons with at least one positive survey was higher, 47 percent. If we assume a detection probability of 60 percent, we estimate that fishers may have been present within approximately 56 percent of all surveyed hexagons and within 78 percent of hexagons with high habitat value. Fisher detection probabilities are affected by latitude, season, type of survey, and survey effort (Furnas 2014, pers. comm.; Slauson *et al.* 2009, entire), but given reported fisher detection probabilities (reviewed by Slauson *et al.* 2009, pp. 15-19), we believe that 60 percent detection probability is a conservative estimate that does not place undue confidence in the accuracy of negative results. An assumption of higher detection probabilities would lend greater credibility to negative survey results and would therefore lead us to estimate that fishers occupied less of the available habitat.

These analyses indicate that a significant amount of high quality habitat remains unoccupied within the current boundaries of the NCSO population. There are several potential explanations for this. It is possible that relatively low survival rates, such as those observed on the Eastern Klamath Study Area (Swiers 2013, p. 19), are preventing this population from fully occupying the available habitat, much less expanding northward into Oregon. Unoccupied areas identified as high quality habitat by the habitat model may contain sources of mortality not identified by the model, such as high disease or predation rates, or the presence of anticoagulant rodenticides at nearby marijuana plantations. Alternatively, although the model identifies high quality habitat distributed through much of the area occupied by this population, some areas of good habitat are separated from others by roads, rivers, areas of low quality habitat, or other filters. These filters can impede connectivity within the population, which may depress occupancy rates, although interconnected fisher populations occur in spite of perceived filters such as roads, rivers, and landscape features (Swiers 2013, p. 13; Tucker *et al.* 2013, p. 12). Preliminary habitat-based population models suggest that the configuration of habitat affects population numbers in this region, and that some areas with high quality habitat may remain unoccupied even at equilibrium population sizes, probably due to restricted connectivity between these locations and the main body of the population (Rustigian-Romsos 2013, pers. comm.). Furthermore, since fishers' life histories are strongly influenced by adult survival, it may take longer time periods of stable conditions or environments for population growth and recovery of fisher populations into areas of higher quality habitat (Buskirk *et al.* 2012, p. 91).

Table 2. Density estimates.

Location Density (N per 100

km² [38.6 mi²])

Source

British Columbia, Canada (outside analysis area) British Columbia, high quality habitat 1.0-

1.54 Weir 2003, p. 20 Central British Columbia, industrial forest, 1996- 2000
 Weir and Corbould 2006, p. 124 **Northern California-Southwestern Oregon** Green Diamond
 Resource Company, Humboldt County, California, 2002-2003
 0.88 ± 0.11 to 1.12 ± 0.21
 Thompson 2008, p. 23 North Coast Study Area, Six Rivers and Shasta- Trinity National Forests,
 Humboldt and Trinity Counties, California
 7 males 11 females

5 Zielinski *et al.*
 2004b, p. 654
 Eastern Klamath Study Area, Siskiyou County , California and Jackson County, Oregon, 2007-
 2011
 5.7-6.9 Swiers 2013, p. 19
 Hoopa Valley Indian Reservation, Klamath Mountains, Humboldt County, California, 2005
 14 Matthews *et al.*
 2011, p. 72 Hoopa Valley Indian Reservation, Klamath Mountains, Humboldt County,
 California, 1998
 52 Matthews *et al.*
 2011, p. 72 **Southern Sierra Nevada** Sequoia National Forest, Tulare County, California 8
 females Zielinski *et al.*
 2004a, p. 654 Sierra National Forest, Fresno County, California, 2002, camera trapping study
 13.4 (95% CI: 7.6-24.2) Jordan 2007, p. 25 Sierra National Forest, Fresno County, California,
 2003, camera trapping study
 9.5 (95% CI: 5.6-17.0) Jordan 2007, p. 25 Sierra National Forest, Fresno County, California, 2004,
 camera trapping study
 10.0 (95% CI: 6.7-14.4) Jordan 2007, p. 25 New Information: Sierra National Forest, Fresno
 County, California, 2008-2012, camera and mark recapture study

7.2-9.7 Sweitzer *et al.*
 2015d, p. 78
 45
 022643

Figure 10. Hypothetical 1000-ha fisher home ranges that contain positive survey results since 2003 (green); that were surveyed since 2003 but contain only negative survey sites (red and pink); or that were not surveyed between 2003 and 2013 (gray). The purple outline buffers all positive detections of native animals (not including animals within the Northern Sierra Nevada or Southern Oregon Cascades Reintroduced Populations), by 41 km to represent a maximum likely dispersal distance.

New Information

In the southern Oregon Cascades, a fisher was detected on the Willamette National Forest in Lane County, just north of the county line, along the Middle Fork of the Willamette River on January 27, 2014. This is the first verifiable contemporary detection of fishers on the Willamette National Forest. However, Aubry and Raley (2006, p. 5) had a juvenile male disperse about 55 km (34 mi) to the northeast to the Big Marsh area on the Deschutes National Forest, which would be east of and across the Cascade crest from the Willamette National Forest sighting, indicating the Willamette sighting may be a potential disperser from the known Southern Oregon Cascades population.

Fisher observations from Crater Lake National Park (Park) include data from 1990, and 2013 through 2015 and are not limited to the Southern Oregon Cascades study area (S. Mohren 2016, pers. comm.). In 1990, the USFS located a radio-collared fisher near the panhandle in the southern portion of the Park. Observations from 2013 include an incidental sighting and observations of tracks southwest and north of the lake, respectively. Fishers were also detected at one camera location in the northeast portion of the Park in 2014 and at three camera stations in the southern portion of Park in 2015. A report of fisher observations east of the lake from 2015 lacks verification; however, the Park will place a camera station in the area in 2016 as part of its ongoing monitoring for carnivores.

The Klamath and Chiloquin Ranger Districts of the Fremont-Winema National Forest (FWNF) conducted surveys for fishers in 2012, 2013, and 2015 (Albert 2014, entire; Albert 2015, entire) within and outside of the Southern Oregon Cascades study area. The FWNF deployed baited camera stations and observed four fishers on the Klamath Ranger District in 2013; however, the FWNF was not successful in obtaining hair samples at these locations to further identify individuals or determine genetic descent. Surveys in 2015 detected fisher at two bait stations on the Klamath Ranger District; results from hair samples are pending. In addition to the camera bait station detections, two dead fishers were collected from the Klamath Ranger district in 2013 and 2015. Surveys will continue on the Chiloquin Ranger District in 2016.

In October 2015, the Klamath Falls Recreation Area (KFRA) and the Ashland Recreation Area, Medford BLM initiated a fisher telemetry project in partnership with USFS Pacific Northwest Research Station and Oregon State University. Five fishers (three female and two male) were collared using a combination of GPS and VHF radio collars. That project, in conjunction with baited camera and hair snare surveys, is currently ongoing and fieldwork is planned to be completed in 2016.

There are not enough data available from the Southern Oregon Cascades to determine population trends. Recent detections of fisher in areas where they were not previously recorded (for example, northern and eastern portions of Crater Lake National Park and portions of the Lakeview and Medford BLM study areas) may or may not represent an expansion of this population. However, based on the current survey efforts along with multiple unsolicited sightings of fisher in the past few years on KFRA where fisher were previously known to be absent, fisher appear to be expanding into the KFRA (S. Hayner 2016, pers. comm.).

In the California portion of NCSO, a recent 2015 estimate of 632–1,165 fishers was based on data collected by California Department of Fish and Wildlife (CDFW) as part of a mesocarnivore monitoring program in northern California (Furnas *et al.* 2015, pers. comm.). It is important to note that the sampling area for the CDFW study excluded southwest Oregon and the coastal redwood of California; thus, this estimate is not representative of the entire area within the NCSO population.

Population trend information for the approximately 45,000 km² (17,375 mi²) NCSO population is based on two long-term studies. The NCSO population represents approximately 12 percent of the West Coast DPS and includes the area in both the SOC and NSN reintroduced fisher populations:

- The Hoopa study area is approximately 145 mi² (370 km²) in size and represents the more mesic portion of the NCSO population area; fisher studies have been ongoing since 1996. The population trend from 2005–2012 indicates a lambda (population growth rate) of 0.992 (C.I. 0.883–1.100) with a higher lambda rate for females (1.038 [C.I. 0.881– 1.196]) than for males (0.912 [C.I. 0.777-1.047]) (Higley et al. 2014, p. 102, Higley 2015, pers. comm.). Demographic parameters are showing a decrease in annual male fisher survival. A lambda of approximately 1.0 indicates a stable overall population trend.
- The Eastern Klamath Study Area is approximately 510 km² (200 mi²) in size and represents the more xeric portion of the NCSO population area. Monitoring has been

conducted since 2006. Estimates for lambda from 2006–2013 are 1.06 (C.I. 0.97–1.15) (Powell et al. 2014, p. 23). This lambda of approximately 1.0 indicates a current stable population within the study area.

- Fishers in the NSN portion of NCSO population stem from a 2009 to 2012 translocation of 40 fishers from Humboldt, Siskiyou, and Trinity counties, California to the SPI Stirling Management Unit in Butte, Plumas, and Tehama counties, California. Ongoing monitoring of fishers that were reintroduced has confirmed that fishers born on site have established home ranges and have successfully reproduced. Trapping efforts in the fall of 2015 as part of ongoing monitoring of the reintroduced population indicates that a minimum of 49 fishers (34 females, 15 males) were alive, nine more individuals than were originally introduced.

Southern Sierra Nevada

Several approaches have been taken to understanding the population status of the Southern Sierra Nevada (SSN) population. Density estimates are available from three study sites (Zielinski *et al.* 2004b, p. 654; Jordan 2007, pp. 12–44; Sweitzer *et al.* 2015d, p. 78; see also Table 2). There has been one preliminary population viability analysis, with parameters based on expert opinion (Lamberson *et al.* 2000, entire), and another spatially explicit population model based on a combination of empirical data and expert opinion (Spencer *et al.* 2011, entire). One monitoring program has enabled researchers to measure trends in occupancy within the SSN population over a period of eight years (Zielinski *et al.* 2013b, entire). By all estimates, the isolated SSN population is small.

For the purpose of modeling population viability, Lamberson *et al.* (2000, p. 2) used expert opinion to estimate a population size between 100 and 500 individuals in the SSN population. They then used a deterministic, Leslie stage-based matrix model to gauge risk of extinction for the SSN population of fisher and found that the population has a very high likelihood of extinction given reasonable assumptions with respect to demographic parameters (2000, pp. 10, 16). For an initial population of 200, when all demographic parameters are low, extinction is predicted to occur in about 15 years, and when all demographic parameters are at medium levels, extinction is predicted to occur in about 45 years (Lamberson *et al.* 2000, pp. 18–20). When all demographic parameters are at their highest levels, the population increases regardless of whether the initial population is 50, 100, or 200 animals. It is important to note that the authors chose demographic parameters to represent a biologically realistic range of values based on literature reviews and preliminary data (Lamberson *et al.* 2000, p. 6), rather than through robust

demographic measurements of the population they were modeling. Therefore, it is not clear which, if any, of their parameter levels best represents the demography of the population. In light of more recent empirical studies, the true demographic parameters likely fall in between the medium and high parameter levels, and the population growth rate on the Sierra Nevada Adaptive Management Project study area is estimated to be 1.1 (95 percent CI 1.04–1.19), which indicates a stable or slightly increasing population (Sweitzer 2013a, pers. comm.; Sweitzer 2013b, pers. comm.). The authors note that population growth rates for a study area, where fishers are abundant enough to generate adequate sample sizes for research, may not be representative of the entire population.

Spencer *et al.* (2011, entire) created a spatially explicit population model that combined an empirically derived fisher probability-of-occurrence model with demographic parameters derived from literature review and expert opinion. Based on the modeled number of female home ranges that could be supported by the available habitat, they concluded that the carrying capacity of the currently occupied areas was approximately 125–250 adults (Spencer *et al.* 2011, p. 788), and that the population was probably less than 300 adult fishers (Spencer *et al.* 2011, p. 801). They also extrapolated the density estimates measured by Jordan (2007, p. 25; see Table 2 above) to arrive at a figure of 276–359 fishers (Spencer *et al.* 2011, p. 802), including juveniles and subadults, in this population. However, as discussed above for the NCSO population, this type of extrapolation is likely to result in an overestimate of the true population. Spencer *et al.* (2011, p. 797) further concluded that a 10–20 percent reduction in survivorship from the parameters used in their initial model would interfere with population expansion.

In 2002, USFS initiated a regional monitoring program to track occupancy trends of fishers in the SSN population. A power analysis for the program (Zielinski and Mori 2001, entire) determined a sampling design that targeted an 80 percent probability of detecting a 20 percent decline in occupancy in the population over a 10-year period. The sampling scheme was not designed to detect increases in occupancy (Zielinski *et al.* 2013b, p. 3). After 8 years of monitoring, Zielinski *et al.* (2013, entire) used occupancy modeling techniques, not available at the time of the original program design, to investigate occupancy, persistence rates, and trend in occupancy. They found no trend or statistically significant variations in occupancy during the 8-year period of the program (Zielinski *et al.* 2013b, p. 8) and concluded the SSN population was not decreasing. Subsets of their study area varied in occupancy rates and persistence, with the 49

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southwestern portion of their study area the most densely occupied, but none showed a significant trend (Zielinski *et al.* 2013b, p. 11). However, the annual target sampling size (288 units/year) was unattainable, due to logistical and financial constraints, and the average sample size was instead 139.5 units/year (Tucker 2013, p. 82). As a result of this smaller sample size

and shorter duration, the results of this study must be considered inconclusive. Recreating the sampling scheme of this monitoring program and using the implemented average annual sample size at the Sierra Nevada Carnivore Monitoring Program, Tucker (2013, pp. 80–97) investigated the link between occupancy and abundance, showing that a 43 percent decline in abundance over an 8-year period only resulted in a 23 percent decline in occupancy reported. This estimate was derived using a spatially explicit simulation approach with an assumed initial population size of $n=300$, as the relationship between occupancy and abundance varies depending on population density; the same simulation using an initial population size of $n=150$ would yield a slightly greater decline in occupancy (Tucker 2015, pers. comm.). This effort demonstrates the complexities in determining population trend and identifies important cautions in extrapolating the conclusion of no trend in occupancy to a conclusion of no trend in abundance over 8-years of monitoring of the SSN population.

New information since Service (2014)

A recent study of radio collared fishers monitored from 2008 through 2014 in the SSN population showed the survival rate (calculated using demographic parameters) of adult males, but not females, is lower than other populations in the DPS, and estimates a λ of 0.97 (C.I. 0.79–1.16) (Sweitzer *et al.* 2015a, pp. 781–783; Sweitzer *et al.* 2015b, p. 10). A more recent analysis from this study (Sweitzer *et al.* 2015d, p. 77), however, suggests a lower population growth rate of 0.90 (95 percent C.I. 0.71–1.12) from 2008 to 2014; however, the population growth rate was at 1.0 or above for the period from 2010 to 2014 (Sweitzer *et al.* 2015d, p. 77). Population growth in the SSN population area is thus estimated to trend less than 1.0; the authors suggest the population is not in persistent decline, however, but is offset by periods of stability or growth (Sweitzer *et al.* 2015a, p. 784).

Reintroduced Populations

Translocations, the intentional transport and release of animals to augment, reestablish, or introduce a population, have been used in attempts to recover extirpated or depleted populations of many species. Recovery efforts throughout much of the fisher's North American range have relied heavily on translocations, and the fisher has proven to be one of the most successfully reintroduced carnivores (Powell 1993, pp. 80–85, Breitenmoser *et al.* 2001, p. 242; Lewis *et al.* 2012a, p. 9). Translocations, however, are not always successful (Breitenmoser *et al.* 2001, p. 242) and many fisher translocations in eastern and western North America failed to reestablish populations (Powell 1993, p. 84; Aubry and Lewis 2003, pp. 82–85; Lewis 2006, pp. 28–29). Lewis and Hayes (2004, pp. 4–5) report at least 31 fisher reintroductions attempted throughout their range in the U.S. and Canada from 1947 to 2003 with 21 (68 percent) considered successful (fishers persisted more than 10 years following first release), 7 considered failures (22 percent), 2 were not evaluated (6 percent), and 1 is ongoing. Reintroductions have been more successful in eastern states and provinces (79 percent) than in western states and provinces (58

percent) (Lewis and Hayes 2004, p. 5). Within the Analysis Area, six separate translocations have been

attempted during the last 53 years (Aubry and Lewis 2003, p. 82; Lewis *et al.* 2012a, p. 8). Two of these reintroduction efforts were unsuccessful, one resulted in an established population (Southern Oregon Cascades), and the three most recent reintroductions (Olympic Peninsula and Northern Sierra Nevada) have not reported that they have met their criteria for success.

During the 1950s, the USFS and Weyerhaeuser Corporation asked the Oregon State Game Commission to reintroduce fishers to Oregon as a means of controlling porcupine populations (Aubry and Lewis 2003, p. 82). In 1961, two attempts were made to reintroduce fishers to Oregon, involving a total of 24 fishers translocated in 1961 from British Columbia. Of these 24, 11 were released near Klamath Falls in the southeastern Cascade Range, and 13 near La Grande in the Wallowa Mountains (Aubry and Lewis 2003, p. 82; Lewis and Hayes 2004, p. 7). The lack of observations or incidental captures of fishers after the 1961 releases suggested that the translocations were unsuccessful, and that additional releases would be required to reestablish fishers and reduce porcupine damage (Aubry and Lewis 2003, pp. 82–86).

Olympic Peninsula

From 2008 to 2010, 90 fishers were translocated from central British Columbia to the Olympic Peninsula. By monitoring translocated fishers with radio-telemetry, project researchers evaluated post-release survival, home range establishment, reproduction, and resource selection of founding individuals. Initial findings indicate that survival was highly variable among release years (Lewis *et al.* 2012b, pp. 5–8; Lewis 2014, p. 63). Project researchers confirmed reproduction seven times from 2009 to 2011 (Lewis *et al.* 2012b, pp. 9–10).

Wilderness constraints provide logistical difficulties for researchers, which lead to additional uncertainties about the current status of reintroduced fishers in the Olympic Peninsula. A second monitoring phase consisting of non-invasive surveys of fisher distribution and relative abundance was initiated in the summer of 2013 and will help determine whether a self-sustaining population of fishers has been established in the Olympic Peninsula. In early 2013, biologists from many agencies and Tribes began a 4-year investigation of the success of the Olympic Fisher Restoration Project (Happe 2013a, pers. comm.). By late October of 2013, the project partners had detected fishers at 12 percent of sampling units, and there were indications of survival of translocated individuals (photos of radio-collared individuals) and of reproduction (for example, one road-killed female was lactating and had four placental scars) (Happe 2013b,

pers. comm.).

New information since Service (2014)

In 2013 and 2014, the monitoring team detected fishers in 14 of 132 areas sampled, including six of the founding fishers and seven new recruits to the population (Happe *et al.* 2014, pp. 13–14; Happe *et al.* 2015, pp. 10–12). Sixteen fishers were also detected with non-project cameras, by trapping, and as carcasses (Happe *et al.* 2014, p. 16; Happe *et al.* 2015, pp. 14–15). These fishers will continue to be monitored for a number of years to determine both the extent of their distribution and success in establishing a population of fishers on the Olympic Peninsula. Preliminary results showing wide distribution and documentation of reproduction are encouraging, but the success of this reintroduced population will not be known for several years.

The Olympic Peninsula population occurs in three percent of the analysis area, and has not been observed to have spread beyond the Peninsula. This population is not physically or demographically connected to any other population of fishers. Population size estimates and trend information are not known at this time.

Southern Oregon Cascades

From 1977 to 1981, 24 fishers from British Columbia (n=11) and Minnesota (n=13) were released west of Crater Lake in the southern Oregon Cascades (Aubry and Lewis 2003, p. 84). An ecological study from 1995 to 2002 (Aubry and Raley 2006, entire) indicated fisher presence in the vicinity of these releases still occurred. Subsequent work (Drew *et al.* 2003, p. 57; Wisely *et al.* 2004, p. 646) found that these fishers exhibited genetic traits in common with British Columbia and Minnesota fishers, but did not exhibit traits consistent with native Oregon or California fishers (Aubry *et al.* 2004, pp. 211–215).

Although this population was reestablished >30 years ago, and is about 40 km (25 mi) from the native Northern California-Southwestern Oregon Population, no genetic exchange between the 2 populations has been documented (Aubry *et al.* 2004, p. 214; Drew *et al.* 2003, p. 59; Wisely *et al.* 2004, p. 646; Farber *et al.* 2010, p. 12). Fishers in the Cascade Range of Oregon may be geographically isolated from those in southwestern Oregon because of ecological (extensive areas of open grassland and oak savannahs) and anthropogenic (Interstate 5 corridor, urban, and agricultural development) barriers in the intervening area (Aubry and Lewis 2003, pp. 86–87; Aubry *et al.* 2004, p. 204). One male fisher from the NCSO population was detected in the vicinity of the southern extent of the Southern Oregon Cascades reintroduced Population

(Stephens 2012, pers. comm.; Pilgrim and Schwartz 2012, pp. 4-5). Therefore, it is possible the Southern Oregon Cascades Reintroduced and NCSO populations may have become interconnected by dispersing fishers.

There are no reliable estimates of population size. Based on verifiable occurrence records since the 1977–1981 reintroductions, it appears that this population has not expanded its range much beyond a relatively small area (Aubry and Lewis 2003, p. 85) of about 2,500 km² (~950 mi²; Aubry and Raley 2006, p. 3). A winter 2012-2013 survey effort on the Fremont-Winema National Forest, just south of the Crater Lake National Park boundary, failed to find fishers (Albert 2013, p. 1; Ackerman 2013, pers. comm.), but trail camera photographs captured in late 2013 indicate that this population of fishers persists (Broyles 2013, pers. comm.).

New information since Service (2014)

See New Information in the *Northern California-Southwestern Oregon (NCSO)* section above, which includes the original native fisher population and the Southern Oregon Cascades (SOC) and Northern Sierra Nevada (NSN) Reintroduced Populations

Northern Sierra Nevada

From late 2009 through late 2011, 40 fishers were released into the northern Sierra Nevada and

southern Cascade Mountains of California. All animals were equipped with radio telemetry and monitored for survival, reproduction, dispersal, and home range development (Powell *et al.* 2013, p. 2). The released fishers experienced high survival during both the initial post-release period (4 months) and for up to 2 years after release (Powell *et al.* 2013, p. 2). Released fishers produced kits in all three springs since translocation (Powell *et al.* 2013, p. 18). A trapping effort conducted in the fall of 2013 determined that at minimum, 28 fishers were known to be alive within the study area (total fishers captured as well as non-captured, telemetered fishers) (Swiers 2013, pers. comm.). Population estimates from the 2013 trapping effort had not yet been calculated as of this reporting, but a fall 2012 trapping effort returned a minimum population size of 37 and population estimates averaging 33 fishers (95 percent CI 22– 44) across all model types used (Powell *et al.* 2013, p. 13). Note that this value (33) is less than the known minimum population size for fall of 2012, and the confidence interval suggests that the population in the fall of 2012 was slightly larger than in the fall of 2011, when it was estimated to include between 18 and 40 fishers (Powell *et al.* 2013, p. 13).

New information since Service (2014)

See New Information in the *Northern California-Southwestern Oregon (NCSO)* section above, which includes the original native fisher population and the Southern Oregon Cascades (SOC) and Northern Sierra Nevada (NSN) Reintroduced Populations

Reintroduction

Summary

The Southern Oregon Cascades Reintroduced Population has persisted for over 30 years, despite estimates of a small population size. Various agency survey efforts over the past five years have resulted in verified sightings, including both photographs and DNA evidence, north, south, and east of the Aubry and Raley (2006, entire) study area. These recent agency surveys, while not systematic in design, do indicate evidence of potential population expansion.

For both the Olympic Peninsula Reintroduced Population, Washington Cascades Reintroduction area, and the Northern Sierra Nevada Reintroduced Population, it is too early to determine if the populations will persist. Current indications are encouraging, but it will take time to determine population trend and stability of these three new reintroductions.

REVIEW OF STRESSORS

In the following section, we will review and evaluate potential stressors that may be affecting fishers in the analysis area based on past, current and future impacts. Our approach draws upon methodologies put forth by NatureServe (Master *et al.* 2012, entire) and the fisher threat assessment conducted by Naney *et al.* (2012, entire), and we adopt various terms and descriptions that assist our analysis. When information is available, we may describe impacts according to geographic areas (for example, we use 8 of the 11 geographic areas as described by Naney *et al.* (2012, pp. 13–14) within the analysis area based on differences in biophysical environment, human modifications to those environments, current fisher distribution, and political jurisdiction (Table 3; Figure 11)). The NCSO population occurs in the Western Oregon Cascades, Eastern Oregon Cascades, Northern California-Southwestern Oregon and Sierra

Nevada subregions. The SSN population occurs in the Sierra Nevada subregion and the reintroduced ONP population is present in Coastal Washington. The recent reintroduction of fishers in the Cascades of Washington occurs in the Western Washington Cascades sub region.

Definition of Terms

Stressor

s

Stressors are the activities or processes that are causing or may cause in the future the destruction, degradation, or impairment of west coast fisher populations or their habitat. In some instances, these stressors could be resulting in residual impacts as a result of past activities. Stressors are primarily related to human activities, but can be natural events and act on fishers at various scales and intensities throughout the analysis area. Stressors may be observed, inferred, or projected to occur in the near term.

Classification of Stressors

Timing (immediacy) of the Stressor

The timing (immediacy) of each stressor was assessed independently based upon the nature of the stressor and time period that we can be reasonably certain the stressor is acting on fisher populations or their habitats. In general, we considered that the trajectories of the stressors acting on fisher populations within the analysis area could be reasonably anticipated over the next 40 years. This is the period of time over which we concluded we can reasonably rely on predictions about the future in making determinations about the future conservation status of fisher, as described below.

Stressors that directly cause mortalities were assessed in terms of their contribution to annual mortality rates. Without performing an additional population viability analysis, we could not precisely determine the effects of each stressor on total population numbers over the next 40 years. However, annual mortality rates allow us to compare the effects of the stressor with changes in mortality examined hypothetically in previous population models (Lamberson *et al.* 2000, entire; Spencer *et al.* 2011, entire). We also addressed the likely trend of each stressor over the next 40 years to evaluate whether the impacts of the stressor were likely to increase, decrease, or remain the same in the future.

Table 3. Analysis area sub-regions.

| State/ Province | Analysis Area Sub- Region | |
|--------------------------|--|--------------------------------|
| | Geographic Description | Populations Proportion |
| Federal | | |
| | | Proportion Non- Federal |
| Coastal WA Washington | | |
| | *The Olympic Peninsula Reintroduced Population (ONP) occurs in a portion of this sub-region. | 0.38 0.62 |
| Western Cascades WA | | |
| Washington | | |
| | Canadian border south to the Columbia River and west of Interstate 5 but excluding the Puget Trough. Includes the west and east sides of the Olympic Mountains. | |
| | *Cascade Fisher reintroduction site occurs in a portion of this sub-region? | 0.66 0.34 |
| Eastern WA | | |
| Cascades | Washington East side of the Cascade Range from the Canadian | |
| | border south to the Columbia River. | 0.66 0.34 |
| Coastal OR Oregon | | |
| | West side of the Cascade Range from the Canadian border south to the Columbia River and east of Interstate 5, but excluding the Puget Trough. | |
| | West of Interstate 5 from the Columbia River south to about the main stem of the Rogue River but excluding the Willamette Valley. | |
| | | 0.25 0.75 |
| Western OR | | |
| Cascades | Oregon | |
| | * The NCSO Population occurs in a portion of this sub-region. | 0.76 0.24 |
| Eastern OR | | |
| Cascades | Oregon East side of the Cascade Range in Oregon. *The NCSO Native Population occurs in | |
| | a portion of this sub-region. | 0.70 0.30 |
| | Northern California- Southwestern Oregon | |
| | West side of the Cascade Range from the Columbia River south to the Upper Rogue River drainage basin (about Crater Lake National Park) and east of Interstate 5, excluding the Willamette Valley | |
| | In Oregon, from about the Rogue River south to the Oregon / | |
| | California border and west of Interstate 5 to the coast. *The California | |
| | In California, the southern Cascade Range to Lassen County, west to the coast and south into Lake County. | |
| | NCSO Population occurs throughout this sub-region. | 0.49 0.51 |
| Sierra Nevada California | | |
| | *The NCSO Population occurs in From the southern end of the Cascade Range in | |
| | northern portion of this sub-region. California (Lassen County) to the southern extent of | |
| | *The SSN Population occurs in the the Sierra Nevada. | |
| | southern portion of this sub-region. | 0.57 0.43 55 |

Figure 11. Graphical representation of analysis area sub-regions used to evaluate potential stressors.

Stressors that affect fisher habitat may often have a more persistent effect than stressors that cause direct mortality. When habitat is lost, it may take many decades to return. Therefore, even though habitat loss has an immediate impact on fisher populations, its effects are also expected to continue in the future, possibly for many decades until trees become large and old enough to generate the structures needed for fisher denning and resting. Land management regimes are also planned on a multi-decade timescale. For example, most USFS Land and Resource Management Plans were developed between 1983 and 1993 (USFS 2012, p. 21164); under California Forest Practice Rules, one avenue for private land management relies on Sustained Yield Plans, which project timber production over a 100-year timeframe (CAL FIRE 2013a, pp. 14, 218–223). In general, however, we found that most management plans project actions out over a period of several decades. Climate change is underway, but its effects are likely to be long lasting and moreover are likely to accelerate at some point in the future. Climate change models show considerable agreement until mid-century, but diverge thereafter depending partly on assumptions about whether greenhouse gas emissions are curtailed or continue to increase.

Timing (immediacy) Categories: **Past/Historical**—only in the past and unlikely to return, or no direct effect. **Ongoing**—continuing (a stressor now). **Future**—expected to occur in the short-term future, such as over the next 40 years. **Long-term future**—in the future beyond the next 40 years. The effects of some ongoing stressors may be projected for the late 21st century, which is outside of the future time period as defined above; therefore, we report them as the long-term future effects. For climate change, we also reviewed information over the next 100 years.

Magnitude of a Stressor's Impact

In addition to identifying the timing of the impact, we considered the scope and magnitude of those impacts. The scope of an impact is based on the proportion of suitable habitat within a known population area (such as ONP, SOC, NCSO, NSN, or SSN) or the proportion of those populations that can reasonably be expected to be affected by a stressor based on the best available information. The magnitude of impact refers to the estimated risk level or degree of decline that a stressor may cause to one or more of the populations, or by the amount of suitable habitat that may be lost, degraded, or fragmented based on the best available information.

For each stressor, we summarized the best available scientific information relating to its potential direct and indirect impacts on the West Coast DPS of fisher. If significant information gaps existed, resulting in high levels of uncertainty in determining the scope and magnitude of impact for particular stressors, we used our best professional judgment. We used three impact level classes—low, medium, and high—to represent the likely impact of stressors to the fisher in the West Coast DPS. We defined the impact level classes as follows:

Low-level impact: Stressor is impacting individual fishers within the West Coast DPS currently or in the future, or stressor is resulting in a minor amount of habitat impacts currently or in the future.

Medium-level impact: Stressor is impacting fishers within the West Coast DPS at the population level (one or more of the five populations) currently or in the future, or stressor is resulting in more serious impacts to fisher habitat at the population level (as compared to a low-level impact) currently or in the future.

High-level impact: Stressor is significantly impacting the West Coast DPS of fishers at the rangewide level currently or in the future, or stressor is causing significant impacts to fisher suitable habitat at the rangewide level currently or in the future.

Stressors Related To Habitat Loss and Fragmentation

Introduction

Habitat components important to a fisher's use of stands and the landscape can be identified broadly as structural elements (for example, snags, down wood, live trees with cavities, and mistletoe brooms), overstory cover (dominant, co-dominant, and intermediate trees), understory cover (vertical and horizontal diversity), and vegetation diversity (floristic species) (Lofroth *et al.* 2010, pp. 119–121). The reduction in, or losses of, these components are outcomes of natural disturbance events (for example, wildfire, forest insects, and disease) and various vegetation management activities (for example, timber harvest, silvicultural practices, herbicide application, and fuel reduction techniques). Depending on the scale, intensity, and distribution of disturbance events (for example, if the areas of disturbance are larger or more extensive than the natural pattern and scale of disturbance), then overall ability of the landscape to support fishers and to

restore or connect fisher populations may be diminished (Agee 1991, p. 33; 69 FR 18770, April 8, 2004, entire; Powell and Zielinski 1994, p. 64; Franklin *et al.* 2002, pp. 7–10, 20–21; Weir and Corbould 2008, pp. 127, 161–162; Wisdom and Bate 2008, pp. 2091–2092; Naney *et al.* 2012, entire).

The loss of and reduction in the availability and distribution of structural elements and the processes that create them (for example, mistletoe, heart rot fungi, age-related decadence, primary cavity excavators) can negatively affect fisher reproduction and energy budgets (Lofroth *et al.* 2010, pp. 123–130, Naney *et al.* 2012, p. 22). Furthermore, in many of the ecosystems in the analysis area, these structural elements are important habitat components for fisher prey (Aubry *et al.* 1991, pp. 292–294; Carey and Johnson 1995, pp. 347–349; Bowman *et al.* 2000, p. 123). Timber harvest and silvicultural techniques such as regeneration harvest; selective harvest of insect damaged and diseased trees; and thinning to promote vigorous stands of trees often removes the largest trees or focuses on the removal of older, diseased, or decadent trees. This further results in the removal and/or limitation of future recruitment of rest and den trees. In addition, application of herbicides to reduce competition for conifers can remove the shrub and hardwood layer that provides understory cover, structural complexity, and a valuable mast crop for fisher prey, and over the long term removes hardwoods that would provide future fisher den

and rest sites. Fuels reduction and fire suppression techniques that focus on the removal or salvage of snags and fire damaged trees may similarly diminish the distribution, abundance, and recruitment of den and rest sites across the landscape (Naney *et al.* 2012, pp. 29–37).

Wimberly and Ohmann's (2004, p. 643) analysis of forest trends in the Oregon Coast Range found that land ownership historically had the greatest influence on changes in forest structure between 1936 and 1996, with State and Federal ownership retaining more large-conifer structure than private lands. Loss of forest and change in forest structure was primarily due to timber harvest, with fires accounting for a small portion of the loss (Wimberly and Ohmann 2004, pp. 643–644). Between 1972 and 1995, timber clearcut harvest rates in all stand types were nearly three times higher on private land (1.7 percent of private land per year) than public land (0.6 percent of public land per year), with the Coast Range dominated by private industrial ownership and having the greatest amount of timber harvest as compared to the adjacent Klamath Mountain and Western Cascades Provinces (Cohen *et al.* 2002, pp. 122, 124, 128).

Past loss and fragmentation of fisher habitat may contribute to the decline of fisher populations (Aubry and Lewis 2003, p.82). Fragmentation occurs when there is a change in habitat

configuration (Sauder and Rachlow 2014, p. 75). Fragmentation can be caused by several anthropogenic factors (for example, vegetation management, conversion to agriculture, residential construction, and highways) and natural sources, such as large rivers, mountain ridgelines, and valley deserts or grasslands between forested areas (Green *et al.* 2008, pp. 19, 27, 29; Naney *et al.* 2012, p. 15). Anthropogenic factors causing fragmentation may compound habitat loss by isolating patches of suitable habitat within area of unsuitable or less suitable habitat, within which fishers may not be able to establish home ranges, forage (by affecting prey species composition, abundance, and availability), find suitable rest and den sites, or simply travel through (Buskirk and Powell 1994, p. 288; Hayes and Lewis 2006, p. 34; Weir and Corbould 2008, p. 148). Fragmentation can also increase energetic costs to fishers, which may result in nutritional stress that can reduce animal condition, ultimately affecting survival, reproduction, and recruitment (Lehmkuhl and Ruggiero 1991, pp. 35–44). Predation risk may be increased due to the need to travel through low suitability habitat (for example, lack of cover or rest sites) or additional travel time needed to circumnavigate unsuitable habitat (Weir and Corbould 2008, p. 31). This may be exacerbated by an increased abundance of predators associated with fragmented and early-seral habitats (Lehmkuhl and Ruggiero 1991, pp. 38–39). Fragmentation from timber harvest or fire (depending on harvest method, fire intensity, and site potential) ranges in time from one fisher lifetime (about 10 years) after low-intensity disturbances in forested systems that regenerate quickly (for example, three to five years in coastal California; Klug 1997, p. 39), to more than 80 years in the in the drier areas of California and southern Oregon (Agee 1991, p. 32; Franklin and Spies 1991, p. 108).

Timber harvest and other vegetation management treatments are expected to continue on private, state, tribal, and Federal lands. Some forms of vegetation management may not exert a significant negative effect on forest structure and stand conditions important to fishers. For example, vegetation management that implements thinning with the goal of maintaining or enhancing late-successional characteristics or increases structural and species diversity in young stands may provide or improve fisher habitat. In other cases, some vegetation management activities may actually increase prey abundance and diversity, possibly benefitting fishers (Carey

and Wilson, 2001 pp. 1019-1029; Waldien 2005, pp. 25-35; Klenner and Sullivan 2009, pp. 1081-1083). Although there is no published work explicitly testing and evaluating the direct effects of vegetation management or fuel treatments on fishers, various studies indicate that management to reduce fire risk or restore ecological resilience may be consistent with maintaining landscapes that support fishers in both the short and long term, provided that treatments retain appropriate habitat structures, composition, and configuration (Spencer *et al.* 2008, entire; Scheller *et al.* 2011, entire; Thompson *et al.* 2011, entire; Truex and Zielinski 2013, entire; Zielinski 2013, pp. 17-20; Zielinski *et al.* 2013a, p. 825; Clayton 2013, entire; Garner

2013, pp. 29, 41–43; Niblett *et al.* 2015, pp. 9–10; Sweitzer *et al.* 2016, p. 219).

New information since Service (2014): Recent literature is increasing our understanding of how fishers might use managed landscapes and of the attributes of those areas that fisher may be selecting for or against. For example, researchers are documenting fishers in managed landscapes composed of multiple seral stages with legacy structures and varying degrees of forest openings and connectivity (Niblett *et al.* 2015, p. 11; Sauder and Rachlow 2015, p. 54). The scale of analysis (for example, landscape, home range, den site, etc.) and the degree of “edge” (two adjacent habitat types) is an important consideration when assessing the suitability of managed stands as fisher habitat (Aubry and Raley 2006, p. 15; Niblett *et al.* 2015, p. 11; Sauder and Rachlow 2014, p. 80; Sauder and Rachlow 2015, pp. 52–54; Sweitzer *et al.* 2016, p. 220). Zielinski *et al.* (2013, p. 825) also found that the rate at which treatments occur is extremely important in understanding fisher tolerance to vegetation treatments.

Below, we address stressors that affect the forest vegetation types most readily used by fishers and most likely to contain the habitat components fishers rely upon. Large-scale loss of important habitat components resulted from previous forest management practices that began in the 1800s and ended in the early 1990s. Although forest management practices have changed, effects to habitat still occur due to wildfire, climate change, current forest management, human development, and construction of linear features such as roads and power lines. All of these changes in habitat may affect the landscape’s overall ability to support fishers and may also fragment habitat, limiting fisher movement and dispersal. In both the historical and current analysis of stressors related to habitat, we address each stressor individually for the convenience of describing its potential effects to fishers and fisher populations, but these stressors act together, both additively and synergistically, to affect the species.

Historical loss of late-successional forest from past activities and disturbances

Within the analysis area, late-successional forest is associated with important fisher habitat elements. In the west, the habitat components most often associated with smaller scales of fisher habitat (for example, large diameter trees, live trees with cavities, complex cover and floristic species) are represented more frequently in late-successional forests and many studies indicate that fishers select for late-successional forests and select against early-successional forests (Rosenberg and Raphael 1986, pp. 269–271; Jones and Garton 1994, pp. 382–383; Zielinski *et al.* 2004b, pp. 654–655; Matthews *et al.* 2008, p. 49; Weir and Corbould 2008, pp. 124–125). Although fisher home ranges comprise a range of seral stages, they often include high proportions of mid- to late-seral stage forests (Raley *et al.* 2012, p. 248). Consequently, many 60

fisher researchers have suggested that the magnitude and intensity of past timber harvest is one of the primary causes for fisher declines across the United States (Douglas and Strickland 1987, p. 512; Powell 1993, pp. 77–80, 84; Powell and Zielinski 1994, p. 41), and this has been offered as one of the main reasons fishers have not recovered in Washington, Oregon, and portions of California as compared to the northeastern United States (Aubry and Houston 1992, p. 75; Powell 1993, p. 80; Powell and Zielinski 1994, pp. 39, 64; Lewis and Stinson 1998, p. 27; Truex *et al.* 1998, p. 59).

Sharp declines in late-successional forests in Washington, Oregon, and California began with the harvest of these forests in the 1800s (55 FR 26114, June 26, 1990; McKelvey and Johnston 1992, pp. 225–232; Bolsinger and Waddell 1993, p. 2; FEMAT 1993, pp. 6–8; Franklin and Fites-Kaufmann 1996, p. 648; Beardsley *et al.* 1999, p. 21). Late successional forests comprised about 50 percent of forests in Washington, Oregon, and California in the 1930s and 1940s, but by 1992 they comprised less than 20 percent (4,168,269 hectares [ha]) (10.3 million acres [ac]) of those forests (Bolsinger and Waddell 1993, p. 2). Franklin and Spies (1986, p. 80) estimated that 6 million ha (15 million ac) of late successional forest existed west of the Cascade Range in Washington and Oregon in the 1800s. Most of the forest (perhaps 80 percent) probably occurred in relatively large contiguous areas (greater than 405 ha [1,000 ac]) (Bolsinger and Waddell 1993, p. 2). In western Washington and Oregon, modern estimates suggest that 82–87 percent of the late successional forests present at the time of settlement have now been logged (Booth 1991, p. 1).

The conversion of low-elevation forests in western Washington to tree plantations and non-forest uses removed a large portion of potential fisher habitat west of the Cascades (Lewis and Hayes 2004, p. 4). During the last 50 years, the structure, composition, and landscape of much of western Washington's commercial timberlands have significantly changed because of intensive timber harvesting activities (Lewis and Hayes 2004, p. 4). Most of the remaining younger low and mid-elevation forest has reduced amounts of large live trees, snags, and coarse woody material, and is not likely to be able to sustain fisher populations (Lewis and Stinson 1998, p. 27; Lewis and Hayes 2004, p. 4).

In northwestern California, the pattern of timber harvest has historically differed from harvest patterns in Washington and Oregon (Franklin and Fites-Kaufmann 1996, p. 630). Rosenberg and Raphael (1986, p. 272) emphasize that the fragmentation of northwestern California Douglas-fir (*Pseudotsuga menziesii*) forests is relatively recent in comparison with forests of other regions (redwoods of California and Douglas-fir forests of Washington and Oregon), and that the true long-term responses of species to the break-up of their habitat cannot yet be discerned.

In the Sierra Nevada of California, Franklin and Fites-Kaufmann (1996, p. 648) found that forests with high late successional and old-growth structural rankings are now uncommon (14 percent of mapped area). Late successional forests of mixed conifer are a particularly poorly

represented forest type as a result of past timber harvesting, and key structural features such as large-diameter trees, snags, and logs, are generally at low levels (Franklin and Fites-Kaufmann 1996, p. 648). This loss of structurally complex forests has likely played a significant role in both the loss of fishers from the central and northern Sierra Nevada, as well as the fishers' failure to recolonize these areas (USFS 2000, p. 5).

Although there has been a dramatic loss of older forests through much of the 20th century, since 1990, timber harvest has sharply declined throughout the historical west coast fisher range. Total volume of timber harvested in California in 2010 was 73 and 74 percent below what it was in 1988 and 1972, respectively. Timber harvested from Sierra Nevada national forests in 2010 was 86 percent lower than that harvested in 1988. Though much of the decline in timber harvest has been the result of declines on federal lands, harvests from private lands has also dropped (Charnley and Long 2014, pp. 631–632). Federal timber harvest volume in Oregon has dropped by over 90 percent since the late 1980s, with harvest from other ownerships also declining 20 percent (Gale *et al.* 2012, pp. 4, 10, 11). Although the Oregon data include forests outside of the fisher analysis area, about 80–90 percent of the timber harvest volume in Oregon occurs in the western part of the state (Gale *et al.* 2012, p. 17), which is where the majority of Oregon's fisher analysis area occurs. Similar declines have occurred in Washington, including western Washington where the majority of the fisher analysis area is, with declines of timber volume on private lands since the late 1900s and dramatic declines on Federal lands similar to Oregon and California (WDNR 2016, entire). Kennedy *et al.* (2012, entire) measured amounts of disturbed forest area within the Northwest Forest Plan (NWFP) area between 1985 and 2008, and similarly found a substantial decline in the magnitude of disturbance on federal lands, coinciding with NWFP implementation (Kennedy *et al.* 2012, p. 128). In summary, harvest volume levels that resulted in the widespread loss and fragmentation of historical fisher habitat have declined across ownerships over the past two decades, but most dramatically on Federal lands (47 percent of the west coast fisher analysis area). Declining levels of timber harvest volume can be a reflection of declining tree size being harvested, a decline in actual numbers of acres of fisher habitat being harvested, or both.

1.0 Wildfire, Emergency Fire Suppression Actions, and Post-Fire Management Actions

1.1 Wildfire

Definition

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The analysis area encompasses regions subject to several different fire regimes; that is, each region experiences wildfires of differing sizes, frequencies, and severities. Within a region, different land cover types also burn with varying frequency and severity. These fire regimes are affected by naturally occurring climate and vegetation conditions as well as by human management decisions.

Fire severity has often been described in categories as high, mixed, or low severity. Low-severity fire burns at ground level and does not kill most overstory trees, although it may consume understory vegetation and downed woody debris (Jain *et al.* 2012, p. 47). High severity fire, also called stand-replacing fire, kills all or nearly all vegetation within a stand and may extend across a landscape (Jain *et al.* 2012, p. 47). Mixed-severity fires are characteristic of many western forests, and are a highly complex disturbance regime that produces unique patch dynamics and ecosystem responses. Characteristics of mixed-severity fire include widely varying fire intervals and combinations of surface, torching, and crown fire behavior both within and between fires, resulting in intermixed patches of live and dead understory and overstory vegetation (Halofsky *et al.* 2011, pp. 1–2). Mixed-severity fires can result in intricately mixed

patches of vegetation of varied age at a relatively fine scale, resulting from the variations in fire frequency and severity as well as species responses to this variation (Halofsky *et al.* 2011, p. 13).

Fire frequency is generally expressed in terms of the fire return interval, or average time between fires at the same location), or fire rotation interval (the time required before every part of a given area would be expected to burn at least once). Historical fire return intervals in the analysis area vary from 6–9 years in some areas of northern California to 1,000 years or more for some forest types in western Washington (Agee 1993, pp. 228–231; Stuart and Stephens 2006, pp. 159–161; Lofroth *et al.* 2010, pp. 22–23). In general, the forests of western Washington and northwestern Oregon have burned infrequently, with a fire return interval of 200 years or more, but when they have burned, the fire was most often stand-replacing (Agee 1991, p. 32; Lofroth *et al.* 2010, pp. 22–23). In much of the Eastern Cascades, Klamath bioregion, and Sierra Nevada, historical fire return intervals prior to the era of fire suppression were typically in the range of 11–35 years, and fires were most often low or mixed-severity (Lofroth *et al.* 2010, pp. 22–23; Sensenig *et al.* 2013, p. 105). In the current era of fire suppression, the average fire return interval has lengthened dramatically in regions and forest types that historically had short fire return intervals (Skinner *et al.* 2006, p. 178).

New information since Service (2014)

Specific to high-severity fire, historical high severity fire rotations in California's Sierra Nevada

are estimated at 281 years in the north and 354 years in the south (Baker 2014, p. 18). In the Sierra Nevada, Baker (2014, p. 25) suggests that high severity fires were historically relatively extensive, and may have covered from 31 percent (southern Sierra Nevada) to 39 percent (northern Sierra Nevada) of the area, but mixed-severity fires were the dominant fire class in these forests (from 48 percent in the north to 43 percent in the south).

Effects of fire on fisher habitat elements

Fires can cause reductions to or removal of important elements of fisher habitat, including vegetative diversity, over-story canopy cover, understory cover, and key structural elements (large hollow trees, large down logs, large live trees). Fire can also create or maintain some structural elements used by fishers; in other words, the consequences of fire for fisher habitat are complex and not subject to generalization. Low-severity fire may reduce some habitat elements, such as understory cover, while increasing others, such as vegetative diversity, and both remove and create dead wood elements such as snags and down wood. High-severity fire is more likely to remove forest cover from large blocks of habitat and potentially result in loss of habitat.

New information since Service (2014)

Mixed-severity fire may contribute to the regeneration of the hardwood component of mixed conifer forest used by fishers (Cocking *et al.* 2014, entire). Fishers may benefit from the increases in the abundance of mammalian prey species following mixed-severity fire (for example, Hanson 2013, p. 27; Ganey *et al.* 2014, p. 47). Such habitat may therefore serve as favorable foraging habitat for fishers if situated in proximity to sufficient areas of habitat that provide adequate denning and resting structures.

Low-severity fires decrease the density, diversity, and abundance of understory vegetation, at least over the short term. These understory reductions may diminish prey habitat quality and quantity, decrease prey abundance and availability, or remove cover for effective foraging, although abundance of some prey species may increase (Lehmkuhl *et al.* 2006, pp. 596–597; Monroe and Converse 2006, pp. 237–238; Fontaine and Kennedy 2012, p. 1553). The recovery of understory, however, especially on productive sites, can occur within one fisher lifetime (Naney *et al.* 2012, p. 6). In addition, low severity fires can be critical in the creation or maintenance of reproductive habitat for fishers, as fire scars enhance the formation of cavities that serve as denning sites (Weir *et al.* 2012, pp. 237–238). When evaluated using a fisher habitat model derived from fisher location data, sites recently treated with prescribed burning

showed similar foraging habitat value to sites that were not burned (Truex and Zielinski 2013, p. 90). Some low-severity fires may eliminate large downed wood (Innes *et al.* 2006, p. 3184), or reduce canopy cover enough to diminish the value of the stand as resting habitat (Truex and Zielinski 2013, p. 90). In forest types subject to frequent fires that remove woody structures near the ground, fishers are closely associated with riparian areas (Powell *et al.* 2003, p. 641), which do not burn as often.

Resting and denning sites are likely to be lost as a result of stand-replacing fires. Mixed- and high-severity fires can reduce or destroy key biological legacies and other structural habitat elements, like large snags or large downed wood. These elements, which are already uncommon in some areas, are used as resting and denning structures for fishers. Typically, decades are required for these elements to develop, and it may take more than a century to develop large, hollow trees that are suitable for reproductive dens (Naney *et al.* 2012, p. 7). Therefore, the loss of these elements could render habitat unsuitable as resting or denning habitat for a century or more.

New information since Service (2014)

Habitat may be degraded or lost in the short-term by fire; fires can promote the development of fisher habitat in certain portions of the analysis area, particularly those regions characterized by mixed oak-conifer forests (Cocking *et al.* 2014, entire; but also see Collins and Roller 2013, p. 1810). In Sierra mixed-conifer forests, a historical fire regime characterized by mixed-severity fires, with high severity fires occurring at moderate to long intervals, is believed to have produced the heterogeneous forests with abundant, dense, late-successional habitat characteristics favored by fishers (Baker 2014; Cocking *et al.* 2014; Hanson 2013).

Forests characterized by highly variable natural disturbances, such as mixed-severity fire regimes, are relatively resilient to recurrent severe fire, and severe, short interval fires did not result in loss of species richness, including hardwood and conifer species (Shatford *et al.* 2007, pp. 144–145; Donato *et al.* 2009, p. 142; Halofsky *et al.* 2011, p. 14; Baker 2014, p. 26; Cocking *et al.* 2014, pp. 94, 102–104). Mixed-severity fires promote vigorous regeneration of mixed conifer forest and that such regeneration is not precluded by native shrub cover that may initially recolonize following fire. In the Sierra Nevada, Baker (2014, pp. 14, 24) found that historical mixed-conifer forests were dominated by relatively younger and smaller trees. Large trees were still a key feature of these forests, but were not numerically dominant in the forest assemblage.

form of the replanting of conifers may be required to ensure the return of mixed-conifer forests following stand-replacing fire.

When overstory canopy is markedly reduced, as in mixed- or moderate-severity fires, important microclimate characteristics are altered (for example, increased temperature or reduced shelter from wind and precipitation). Additionally, conflicts with other species or conspecifics may increase due to the open stand structure and absence of rest sites. Landscapes with reduced canopy cover may provide decreased protection from predation, raise the energy costs of traveling between foraging sites, and provide unfavorable microclimate and decreased abundance or vulnerability of preferred prey species (Lofroth *et al.* 2010, p. 85). Once overstory is removed, it may take many decades to reestablish (Naney *et al.* 2012, p. 2)

When stand-replacing fire removes canopy cover altogether, and at a large enough scale, habitat is likely rendered unsuitable for fishers, as these early successional stands may lack canopy cover and the structural elements for rest and den sites required by fishers (Jones and Garton 1994, pp. 380–382; Weir and Harestad 1997, pp. 257–258; Weir and Corbould 2008, p. 2). If large-scale loss of canopy occurs due to large stand replacing fires, the number of fisher home ranges is reduced. Fragmentation due to fire may lead to increased energy expenditures and could ultimately affect survival, reproduction, and recruitment of fishers (Naney *et al.* 2012, p. 7). Predation risk may increase due to the lack of cover and the relatively high abundance of predators in fragmented landscapes (Naney *et al.* 2012, p. 7–8). Large enough areas of early seral vegetation after fire may present a temporary barrier to dispersing fishers, thereby reducing connectivity within and between populations.

Some fires may lead to vegetation type conversion from forest to shrublands, which may permanently change landscape permeability for fishers (Naney *et al.* 2012, p. 7; Collins and Roller 2013, p. 1801). In areas dominated by mixed-severity fire regimes, past fire history can play a significant role in shaping future fire behavior, and vegetation types that are either relatively vulnerable or resistant to stand-replacing fire can result in a self-reinforcing dynamic (Perry *et al.* 2011, pp. 703, 715). However, fire regimes derive from complex interactions among vegetation, climate, topography, and other biotic and abiotic factors that vary over space and time. As Perry *et al.* (2011, p. 709) notes, the mixed severity fire dynamic is too complex to be neatly pigeonholed. The research of Perry *et al.* (2011, pp. 707, 709) suggests that if the fire return interval is sufficiently short, the high-severity fire in the shrublands may erode the forested patches, eventually causing conversion of the entire landscape to shrublands. Conversion of forested areas to shrubland may present a long-term barrier to dispersing fishers, causing populations to become fragmented or preventing migration between populations.

*Fisher use of burned
landscapes*

Fishers evolved in forests that were subject to wildfire, leading Powell and Zielinski (1994, p. 64) to hypothesize that management regimes mimicking small stand-replacing fires will not harm fisher populations, as long as enough late-successional conifer forest remains available nearby. In Ontario, fishers were described as being practically absent from logged and burned areas (de Vos 1951, p. 500), but were occasionally observed in burned areas, particularly during the

breeding season (de Vos 1952, pp. 12-13). However, large stand-replacing fires in Wisconsin and Michigan are believed to have played a role in the extirpation of fishers in that region (Williams *et al.* 2007, p. 1). Fishers' ability to use burned landscapes likely depends on the size and severity of the fire, as well as pre- and post-fire vegetation conditions.

New information since Service (2014)

Relatively few studies have been conducted on the degree to which fishers use post-fire landscapes. In the southern Sierra Nevada, Hanson (2013, entire) observed fisher scat within areas that had experienced mixed severity fire 10–12 years previously, in areas where the fires had caused over 50 percent tree mortality. Fishers may use previously burned forests for foraging, in response to an increase in small mammal prey (Hanson 2013, p. 27). Potential benefits to fishers were found when such fires occur in unlogged mature/old forest with moderate to high pre-fire canopy cover and high structural complexity. Hanson (2013, p. 28) suggests that mixed-severity and even high severity fire is not at odds with fisher conservation in this area.

Sweitzer *et al.* (2016, p. 221) found no negative association between local colonization or persistence of fishers and fire, and also observed a female fisher denning within a patch of forest burned by a low severity fire four years earlier. Similar to other findings, these researchers also suggest that 5–10 years of succession in forests disturbed by fire produces conditions suitable for fisher prey species (Sweitzer *et al.* 2016, p. 222). Overall, they conclude their research does not identify a consistent negative effect of fire on fisher habitat use, but additional research is needed before concluding that fire is not damaging foraging and denning habitats used by fishers in the southern Sierra Nevada (Sweitzer *et al.* 2016, p. 222).

Surveys following the Fountain Fire, which burned 64,000 acres in Shasta County, California in 1992, suggest fisher use of burned area following high severity fire and salvage logging, followed by replanting. Observations of fishers at bait stations in February and March 2015, 15 years after replanting ended, revealed four fisher detections inside the fire perimeter, two detections adjacent to the fire perimeter, and two within riparian leave (buffer) areas approximately a mile inside the fire perimeter. The authors concluded that fishers are making use of previously burned, even-aged regenerating stands, at least for dispersal and foraging

(Engstrom 2015, pers. comm.).

Martens are close relatives of fishers and have similar habitat requirements (Purcell *et al.* 2012, pp. 47–50), so studies on martens' post-fire habitat use provide the best indication of fishers' post-fire habitat use, given the scarcity of studies on fishers. In the Northwest Territory, 21 years after a large, high-severity fire, martens used forested areas in preference to burned areas, though both were included in home ranges (Latour *et al.* 1994, entire). Compared with other northern marten populations, this population used abnormally large home ranges, suggesting that the burned areas provided suboptimal habitat (Latour *et al.* 1994, p. 353). In contrast, trappers in Alaska reported that martens reached high densities in burned areas 3–10 years post-fire, and believed that marten abundance was related to small mammal abundance within the burned area (Stephenson 1984, pp. 2–19). Recently burned areas may provide habitat that does not support reproduction but is adequate for dispersing juvenile martens; for example, in Alaska, young martens dispersed through but did not reproduce or establish home ranges in a study area

consisting mostly of burned areas 7 and 26 years post-fire (Paragi *et al.* 1996b, entire). This latter observation appears to be consistent with the scant data available for fishers.

Spotted owls (*Strix occidentalis* spp.) use many of the same habitat elements and forest conditions as fishers (for example, forest stands with older forest structure such as snags, hollow trees and down logs); therefore, research on spotted owl use of post-fire landscapes may provide clues for potential fisher response. Some studies have suggested that there is little or no change in occupancy by spotted owls after fires, especially those burned at low to moderate severity but also sometimes including high severity burns (Bond *et al.* 2002, pp. 1025–1026; Keane *et al.* 2010, pp. 11–12; Roberts *et al.* 2011, p. 616; Lee *et al.* 2012, pp. 798–800). Other studies have documented reductions in occupancy due to high severity fire (Gaines *et al.* 1997, p. 126; Jenness *et al.* 2004, p. 769; Clark 2007, pp. 40–45; Keane *et al.* 2010, pp. 11–12). Telemetry studies indicate that spotted owls use recently burned habitat for foraging and sometimes even nest in areas burned at low or moderate severity (Bond *et al.* 2009, pp. 1120–1122; Clark 2007, pp. 99–116), although they may shift their core nesting and foraging areas away from burned areas (King *et al.* 1998, p. 3, Clark 2007, pp. 40–41). Unfortunately, all of these studies are of short duration post-fire or their results are confounded by salvage logging or the effects of past timber harvest (for example, Clark *et al.* 2013, p. 686; see the Post-Fire Management Activities section below). It is possible that due to high site fidelity, spotted owls may occupy areas that are not otherwise suitable to meet all of their life requirements and that they occupy these areas despite a reduction in fitness (Clark 2007, p. 41; Clark *et al.* 2011, pp. 43–44). In contrast to spotted owls' site fidelity, fishers travel widely in their home ranges and rarely reuse resting

structures (Zielinski *et al.* 2004a, pp. 481–482; Lofroth *et al.* 2010, pp. 57, 72). Female fishers with dens show stronger site fidelity, but still may use five or more den sites throughout a season (Paragi *et al.* 1996a, p. 80). This characteristic may make fishers more resilient to fire. However, because they are less vagile (able to disperse) than spotted owls, fishers may be more sensitive to barriers to dispersal created by large patches of stand replacing fire.

1.2 Emergency Fire Suppression Activities

Some fire suppression activities may affect fisher habitat. These include backburning (intentional burning to control the progression of wildfire), construction of fuel breaks (removal of all flammable material down to mineral soil), and removal of snags or other large trees. Some fire suppression activities occur on a relatively small spatial scale, while others occur over much larger areas. In regard to emergency suppression, Backer *et al.* (2004, p. 937) state: “[t]he ecological impacts of fire-suppression activities can be significant and may surpass the impacts of the fire itself.”

Backburning has effects similar to those of wildfire, but in some cases, backburning may produce patches of high severity fire even when the wildfire itself is burning at low and moderate severity (Backer *et al.* 2004, p. 944). Wide fuel breaks may remove long, linear strips of fisher habitat. There have been isolated cases of widespread large tree removal for fire personnel safety. Fire suppression techniques that focus on the removal of snags may diminish the distribution, abundance, and recruitment of fisher den and rest sites across the landscape (Naney *et al.* 2012, pp. 29–37). In addition, exotic plants and animals, both terrestrial and aquatic, may be transferred from site to site within fires and across large geographic areas when crews travel

from one state to another (Backer *et al.* 2004, p. 940), which may have indirect effects on vegetation and prey communities in the post-fire landscape.

1.3 Post-Fire Management Activities

Salvage logging (harvest of dead or soon to be dead trees with commercial value) occurs on the vast majority of private timberlands in the analysis area, and also occurs on Federal lands. Smaller fires are also salvage logged, but the number of these operations is difficult to estimate. This type of harvest can lead to increased erosion and sedimentation; damage to soils and nutrient-cycling processes; removal of snags and live trees; decreased regeneration of trees; shortened duration of early-successional ecosystems; increased spread of weeds from vehicles;

damage to recolonizing vegetation; reduction in hiding-cover and downed woody material for fisher prey; increased short-term and medium-term fire risk; and alterations of patterns of landscape heterogeneity (Service 2011, p. III-48). Moreover, these activities reduce the ecosystem benefit of disturbance from fire in diversifying and rejuvenating landscapes (Lindenmayer *et al.* 2004, p. 1303). The recent threat assessment for fishers also acknowledged that modification of forest structure from fire was greater when followed by post-fire salvage logging (Naney *et al.*, 2012, page 31). Establishment of conifer plantations after salvage logging has been linked to higher severity in future fires (Perry *et al.* 2011, p. 709). As there are so few studies of fisher use of burned landscapes, it is difficult to separate out the effect of post-fire salvage logging from the effects of fire. We do have indications that fishers are able to use some salvage logged post-fire landscapes a decade or more post-fire at least for foraging or dispersal (see *Fisher use of burned landscapes* above).

Hazard tree reduction projects post-fire also have the potential to reduce large live trees and snags that pose a threat to human safety and also may be suitable for fisher den or rest sites in a post fire landscape. Some form of hazard tree treatment occurs after the vast majority of fires unless they occur in wilderness areas. Areas with especially dense road networks or near wildland urban interface are the most heavily impacted. There are no data specific to the potential effects on fisher from such operations.

1.4 Fuels Reduction Treatments

New information since Service (2014)

There have been few studies of the effects to fishers from mechanical thinning of forests as a means of reducing the risk of severe wildfire. Garner (2013, entire) reported that fishers may tolerate fuels reduction treatments provided they focus on the reduction of surface and ladder fuels, and care is taken to maintain both canopy cover and sufficient abundance of forest structures, such as large diameter defective and standing dead trees, most likely to provide suitable rest and den sites. Fisher home ranges included larger proportions of treated areas than are found on the landscape as a whole, but when selecting microsites within their home ranges, fishers tended to avoid using sites within 200 meters of a mechanically thinned area (Garner 2013, p. ii).

The results of Sweitzer *et al.* (2016, entire) suggest some similar effects. These researchers report a modest reduction in local habitat use by fishers after disturbance from restorative fuel

reduction (Sweitzer *et al.* 2016, p. 218). Fishers did not completely cease to use those areas, however; the resulting persistence rate was 0.67, and a female was observed denning in such an area (Sweitzer *et al.* 2016, p. 219). They suggested fishers may have shifted to foraging in adjacent forest habitat with less disturbance on a temporary basis, and most likely would resume using areas that had undergone restorative fuel reduction within a few years (Sweitzer *et al.* 2016, p. 220). This study also found fishers using previously burned areas, including areas that had been subjected to managed burns (see *Fisher use of burned landscapes*, above).

1.5 Stressors Related to Wildfire in Each of the Analysis Area Sub-regions

Sierra Nevada

There is evidence of increasing fire severity in yellow pine-mixed conifer forests (Miller and Safford 2012, p. 46; but see Mallek *et al.* 2013, p. 15), which comprise the majority of fisher habitat in the Sierra Nevada. This finding has been challenged by Hanson and Odion (2015), but other studies also report that fires in low and mid-elevation forests in the Sierra Nevada and southern Cascades are burning at higher severities at present as opposed to historically (Mallek *et al.* 2013, p. 1; see also Safford *et al.* 2015, entire). Mallek *et al.* (2013 and references therein, p. 17) suggest that large and severe fires in the absence of strategic forest management approaches could reduce habitat quality and population size for fishers in the southern Sierra Nevada. Because fisher habitat in this region occurs in a narrow band running north to south, fires burning at high severity within fisher habitat have the potential to disrupt north-south connectivity of habitat within the Sierra Nevada (Figures 14 and 15).

The estimate given in Appendix C (Tables 25a and 25b) shows the amount of habitat likely to be lost to fire, but does not estimate the effects of the population fragmentation that would result if connectivity is lost between the northern and southern ends of the area occupied by the SSN population of fishers. If habitat connectivity is lost to the north of the area currently used by the SSN population, this loss could prevent the population from expanding (see the Examples: 2013 Fire Season section below). In addition, if forests burned at high severity in this region are replaced by chaparral or grasslands (see above, and Climate Change section), such a change would represent a permanent loss of habitat. Low- or mixed-severity fire, on the other hand, may play an integral role in maintaining mixed conifer-hardwood forest suitable for fisher (Shatford *et al.* 2007, pp. 144–145; Donato *et al.* 2009, p. 142; Halofsky *et al.* 2011, p. 14; Baker 2014, p. 26; Cocking *et al.* 2014, pp. 94, 102–104).

Northern California – Southwestern Oregon

The fire regime in Northern California and Southwestern Oregon is historically extremely variable, as is the forest composition within this region. In forests with a large hardwood or

redwood component, post-fire stump sprouting may speed the recovery of fisher habitat (Skinner *et al.* 2006, p. 184; Skinner and Taylor 2006, p. 210; Stuart and Stephens 2006, pp. 159–160). However, fisher habitat is highly fragmented in many parts of this sub-region (see Figure 2), and temporary losses of habitat may impede dispersal and increase fragmentation of the resident

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fisher population (Rustigian-Romsos 2013, pers. comm.).

Western Oregon Cascades

Most of the Western Oregon Cascades have a historical fire return interval of 25–200 years, and some higher elevation areas as well as the northernmost portion of the sub-region have fire return intervals longer than 200 years. Most of the Western Oregon Cascades contain large blocks of contiguous habitat.

Eastern Oregon Cascades

As in the Sierra Nevada and Coastal Oregon, high quality habitat in this region occurs mainly in a narrow band, with a few scattered outlying fragments of high quality habitat. Fires burning through this band of habitat have the potential to decrease habitat connectivity.

Coastal Oregon

The historical fire-return interval in Coastal Oregon is relatively long, greater than 200 years. Historically, most fires here have burned at high intensity. Fisher habitat in Coastal Oregon occurs in a narrow strip, similar to the band of fisher habitat in Sierra Nevada, but is more fragmented. Severe fires that remove fisher habitat in Coastal Oregon have the potential to further disrupt habitat connectivity.

Western Washington Cascades

The Western Washington Cascades historically experienced fire even less frequently than Coastal Oregon or Washington, and as in those areas, fires were most often high-severity stand-replacing fires. The total area burned in this region is projected to increase over the long-term, though this extent will still be relatively small compared with the area burned in other sub-regions (Littell *et al.* 2010, pp. 14–15). High quality fisher habitat is relatively sparse and fragmented in this sub-region (see Figure 2).

Eastern Washington Cascades

Our habitat model for the Eastern Washington Cascades (see Figure 2) and Lewis and Hayes (2004, p. 20) shows that little high quality habitat available is in this sub-region and that the intermediate habitat is fragmented. High-severity fire occurring in this sub-region is likely to further reduce habitat availability and connectivity.

Coastal Washington

The southern portion of the Coastal Washington sub-region is very similar to Coastal Oregon in both fire regime and the spatial arrangement of fisher habitat. The Olympic peninsula has more diversity in fire regimes, and in a recent threat assessment, some fisher experts rated the threat of wildfire as a greater concern in Coastal Washington (Naney *et al.* 2012, pp. 24–25). However, there is a larger block of contiguous fisher habitat on the Olympic peninsula, and habitat

connectivity is unlikely to be problematic there unless fires become extremely large, severe, and widespread in the future.

Examples: 2013 Fire Season

During the 2013 fire season, at least 25 fires of 2 km² (500 ac) or greater burned at least partly within high-quality or intermediate fisher habitat within the analysis area. The majority of the fires were in the Sierra Nevada and in the NCSO areas, but several fires also burned in the Eastern Oregon Cascades and Eastern Washington Cascades, and one fire complex (including at least two fires) burned habitat in the Western Oregon Cascades near the boundary with the NCSO sub-region. Fire perimeters (USDI GS 2013) are shown in Figure 13, and areas burned within high-quality and intermediate habitat are shown in Table 4. The figure and calculations for the table used fire perimeters current as of September 11, 2013.

The Rim Fire is particularly noteworthy both for its large size and for its location, which was just to the north of the current range of the SSN population (Figure 14). The Rim Fire perimeter covered approximately 655 km² (253 mi²) of high-quality fisher habitat and 114 km² (44 mi²) of intermediate habitat. The amount of fisher habitat burned in the Rim Fire is greater than the amount of fisher habitat burned in the entire Sierra Nevada sub-region during 2008, the year with the most extensive fires in the Sierra Nevada, when 564 km² (218 mi²) of high quality and 187

km² (72 mi²) of intermediate habitat burned. If the fire burned at mainly low severity within fisher habitat, the effects may be minimal. However, if the fire burned large patches at high severity, the habitat currently occupied by the SSN population may be disconnected from habitat to the north. The population may thus be unable to expand northward or to shift its range northward as many species are expected to do in response to climate change. The effect of the Rim Fire on fisher habitat requires further analysis when all fisher habitat relative to post-fire data are available.

A fire need not be as large as the Rim Fire to disrupt habitat connectivity in the Sierra Nevada if it burns at high severity in a location with already limited habitat connectivity (Figure 15). As an example, the location of the Aspen Fire highlights this possibility, as it occurred at the north end of a narrow isthmus connecting two larger blocks of high quality habitat. Because both the size and severity of fire may be increasing within fisher habitat in the Sierra Nevada, this risk is likely to increase in the future.

In the other regions, the amount of fisher habitat burned during the 2013 fire season is consistent with the amount burned during fire seasons between 1984 and 2011. In each sub-region where fires burned during 2013, the area of fisher habitat burned fell between the median and the maximum area burned per year between 1984 and 2011. Coastal Washington, Coastal Oregon, and the Western Washington Cascades did not have any major fires within fisher habitat during 2013, as was also the case during most years between 1984 and 2011.

Figure 13. Fire perimeters within the analysis area for fire season 2013.

Figure 14. Inset depicts perimeter of the 2013 Rim fire as of 11 September 2013 in the Sierra Nevada. Hatch marks southeast of fire perimeter depict current distribution of the Southern Sierra Nevada fisher Population.

Figure 15. Sierra Nevada sub-region depicting 2013 fire perimeters as of 10 September 2013 to exemplify that the location of a fire may have impacts on habitat connectivity.

Table 4. Area (sq. km) of fisher habitat within fire perimeters during the 2013 fire season

| Fire name | High quality | habitat (sq. km) | Intermediate habitat (sq. km) |
|---|--------------|------------------|-------------------------------|
| Total area burned (sq. km) | | | |
| All sub-regions total | 1075 | 531 | 1605 |
| Sierra Nevada | 840 | 151 | 991 |
| Rim Fire* | 655 | 114 | 768 |
| American Fire | 89 | 20 | 109 |
| Aspen Fire* | 78 | 9 | 86 |
| Fish Fire* | 8 | 0 | 8 |
| Power Fire | 4 | 0 | 4 |
| Kyburz Fire | 2 | 0 | 2 |
| Shirley Complex | 2 | 0 | 2 |
| Hough Complex | 2 | 0 | 2 |
| Panther Fire | 0 | 8 | 8 |
| Northern California-Southwestern Oregon | | | |
| | 217 | 246 | 463 |
| Douglas Complex | 138 | 68 | 205 |
| Whiskey Complex | 20 | 22 | 42 |
| Salmon Complex | 20 | 25 | 44 |
| Corral Fire* | 18 | 24 | 42 |
| Big Windy Complex* | 12 | 50 | 62 |
| Butler Fire* | 4 | 36 | 41 |
| Panther Fire | 3 | 15 | 18 |
| Dance Fire | 2 | 0 | 2 |
| Labrador Fire* | 0 | 7 | 7 |
| Western Oregon Cascades | 2 | 9 | 11 |
| Whiskey Complex | 2 | 9 | 11 |
| Eastern Oregon Cascades | | | |
| | 16 | 87 | 103 |
| Government Flats Complex | | | |
| | 10 | 22 | 32 |
| Green Ridge Fire | 6 | 0 | 6 |
| Sunnyside Turnoff Fire | 0 | 65 | 65 |
| Eastern Washington Cascades | 0 | 38 | 38 |
| Mile Marker 28 Fire | 0 | 27 | 27 |
| Eagle Fire | 0 | 5 | 5 |
| Mile Fire | 0 | 3 | 3 |
| Moore Point Fire | 0 | 3 | 3 |

*Fire not contained as of 9/11/2013; final area burned may vary from area given here.

New information since Service (2014)

Results from the first 20 years of monitoring within the area covered by the NWFP include the entire fisher analysis area with the exception of the Sierra Nevada region and the eastern portions of the Eastern Oregon and Eastern Washington Cascades regions. On Federal lands, 6 percent of the older forests (classified as old-growth structural index of 80 or more (“OGSI-80”)) were lost between 1993 and 2012 (this loss was offset by 3.1 percent increase due to ingrowth during that time period, for a total net change on Federal lands of -2.9 percent). Of that 6 percent loss, the majority (4.2 percent) was attributable to wildfire. However, the loss and ingrowth of OGSI-80 is variable by region. On private lands, 23.2 percent of OGSI-80 was lost in total (offset by 11.6 percent ingrowth, for a total net change on private lands of -11.7 percent), with 0.7 percent of

that loss attributable to wildfire (most of the loss on private lands was due to timber harvest). In total, there was a net loss of OGSi-80 in the NWFP area of 5.9 percent. An estimated 573,900 ac of OGSi-80 were lost to wildfire between the years 1993 and 2012 on Federal and non-Federal lands within the NWFP area combined (Davis *et al.* 2015, pp. 27–28); see Table 6 in the *Vegetation Management* section for details). Similar data were not available for the southern Sierra Nevada area.

1.6 Conservation Measures that May Reduce Impacts of Fire Effects

The increasing frequency and magnitude of wildfires is recognized as a problem on both Federal and private lands throughout the western United States. As a result, both State and Federal agencies have developed and are implementing aggressive fire risk reduction programs. For example, in California the California Fire Safe Council provides wildfire prevention grants for hazardous fuels reduction on non-Federal lands, and the State Department of Forestry and Fire Protection (CAL FIRE) offers several such grant opportunities aimed at reducing the threat of wildfire effects and offers technical assistance to non-Federal landowners to design and implement fuels reduction projects. CAL FIRE additionally carries out a variety of fuels reduction projects in the State of California

(http://calfire.ca.gov/resource_mgt/resource_mgt_EPRP_FuelsTreatment). The Oregon Department of Forestry and Washington Department of Natural Resources have similar programs dedicated to funding and technical assistance for fuels reduction projects.

The National Fire Plan, developed in 2000 by the US Department of Agriculture and the Department of the Interior (followed by the Healthy Forests Initiative of 2002), is aimed largely at reducing hazardous fuels through prescribed burns and other treatments on Federal lands. Most National Forests in the analysis area have many such projects underway; the Klamath National Forest, for example, has multiple fuels reduction projects under review or in various stages of implementation, as well as fire recovery projects

(<http://www.fs.usda.gov/projects/klamath/landmanagement/projects>). The BLM, National Resources Conservation Service (NRCS), and other agencies similarly have fuels reduction projects planned or underway within the analysis area; examples include the Hellgate Recreation Area Hazardous Fuel Reduction Project and Ashland Forest All-Lands Restoration Project (<http://www.blm.gov/or/resources/recreation/rogue/rogue-haz-fuel.php>; <http://www.nrcs.usda.gov/wps/portal/nrcs/detail/or/programs/financial/eqip/?cid=nrcseprd355456>.)

All of these efforts are aimed at reducing the frequency, size, and severity of future wildfires within the analysis area. However, there are no published studies that evaluate whether implementation of these fuel reduction projects offset negative effects of this stressor on fishers within the analysis area.

1.7 Wildfire Conclusion

Wildfire is a natural ecological process that occurs with varying frequency and intensity throughout the range of the West Coast DPS of fisher. There are some indications that wildfire may be increasing in terms of frequency, magnitude, and severity and these projected increases are greater in California and southern Oregon than areas further north. Whether fires may be increasing in severity, is subject to continuing debate. Studies on the effects of wildfire on fisher, although limited, demonstrate a variety of both positive and negative consequences, depending on the size, severity and landscape position of the fire. If the severity and extent of the fire is such that substantial areas of canopy and large trees are lost, it may take decades for the area to support reproduction. If the fire severity is low or mixed, important habitat elements can be both created and removed within a home range, such that the burned habitat may continue to support both fisher foraging and reproduction. The degree to which fire may affect fisher populations is unknown, but all indications are that the population response would be specific to the landscape location, size, and intensity of the fire. Within the analysis area there are areas of suitable but unoccupied habitat which may or may not be accessible by extant fisher populations due to location (outside the current known distribution) or existing forested and non-forested landscape patterns. Much of the unoccupied suitable habitat occurs in the northern portion of the DPS with long fire return intervals. Based on our analysis, we consider wildfire to be a medium-level impact to fisher. The best available data indicate that the stressor is impacting habitat within the area currently occupied by populations of fisher. Therefore, we consider wildfire to be a medium-level impact to fishers currently and in the future.

2.0 Forest Insects and Tree Diseases

In most cases, the usual pattern of localized outbreaks and low density of tree-consuming insects and trees diseases are beneficial, providing structures conducive to rest and den site use by fishers or their prey. However, large area-wide epidemics of forest disease and insect outbreaks may displace fishers if canopy cover is lost and salvage and thinning prescriptions in response to outbreaks degrade the habitat (Naney *et al.* 2012, p. 36). In addressing outbreaks of the mountain pine beetle and other insects in British Columbia, Weir and Corbould (2008, pp. 161–162; 2010, pp. 408–409) state that reduction in overhead cover may be detrimental to fishers and that wide-scale salvage operation may substantially reduce the availability and suitability of remaining forests for fishers. For example, sudden oak death (*Phytophthora ramorum*) in southwestern Oregon and northwestern California could be a stressor if it spreads into areas and causes tree mortality in primary tree species used for fisher den and rest sites or tree species

used as primary food sources for fisher prey. Insects and diseases that degrade habitat are not, by themselves, a significant stressor for fishers or their habitat. However, insect and tree disease outbreaks are also intricately related to wildfire and climate change. Synergies that increase the severities of these stressors are common. For example, trees damaged by wildfire or stressed by

drought may be more susceptible to larger-scale outbreaks of forest insect pests and tree diseases. We evaluated those synergies in the *Climate Change* section of this report, as well as in the *Cumulative and Synergistic Effects of Stressors* section of *Stressors Related to Other Natural or Manmade Factors Affecting its Continued Existence*.

Overall, based on our current analysis, the best available information indicates there are no current outbreaks of insect or tree disease that are significantly impacting populations or the west coast DPS rangewide. Additionally, though there is potential for future impacts if an outbreak occurs, the best available information does not indicate a high likelihood of a population or rangewide impact in the future should an outbreak occur. Thus, impacts associated with forest insects and tree diseases are considered to be a low-level impact to fishers currently and in the future.

3.0 Climate Change

Our analyses include consideration of ongoing and projected changes in climate. The terms “climate” and “climate change” are defined by the Intergovernmental Panel on Climate Change (IPCC). The term “climate” refers to the mean and variability of different types of weather conditions over time, with 30 years being a typical period for such measurements, although shorter or longer periods also may be used (IPCC 2013, p. 1450). The term “climate change” thus refers to a change in the mean or variability of one or more measures of climate (for example, temperature or precipitation) that persists for an extended period, typically decades or longer, whether the change is due to natural variability, human activity, or both (IPCC 2013, p. 1450).

Scientific measurements spanning several decades demonstrate that changes in climate are occurring and that the rate of change has been faster since the 1950s. Examples include warming of the global climate system, substantial increases in precipitation in some regions of the world, and decreases in precipitation in other regions. (For these and other examples, see IPCC 2007, p. 30; and Solomon *et al.* 2007, pp. 35–54, 82–85.) The IPCC characterizes warming of the global climate system as “unequivocal” (IPCC 2013, p. 4), and reports that human influence has been detected in warming of the atmosphere and ocean, changes in the global water cycle, reductions in snow and ice, global mean sea level rise, and changes in some climate extremes (IPCC 2013,

p. 17, Figure SPM.6, Table SPM.1; see also). Results of scientific analyses presented by the IPCC show that most of the observed increase in global average temperature since the mid-twentieth century cannot be explained by natural variability in climate, and it is “extremely likely” (defined by the IPCC as 95 to 100 percent probability) that this change is due to the observed increase in greenhouse gas (GHG) concentrations in the atmosphere as a result of human activities and other anthropogenic forcings (IPCC 2013, p. 17 and Figure SPM.6; Solomon *et al.* 2007, pp. 21–35). Further confirmation of the role of GHGs comes from analyses by Huber and Knutti (2011, p. 4), who concluded that it is extremely likely that approximately 75 percent of global warming since 1950 has been caused by human activities.

Scientists use a variety of climate models, which include consideration of natural processes and variability, as well as various scenarios of potential levels and timing of GHG emissions, to evaluate the causes of changes already observed and to project future changes in temperature and other climate conditions (for example, Meehl *et al.* 2007, entire; Ganguly *et al.* 2009, pp. 15555,

15558; Prinn *et al.* 2011, pp. 527, 529). All combinations of models and emissions scenarios yield very similar projections of increases in the most common measure of climate change, average global surface temperature (commonly known as global warming), until about 2035 or mid-century (for example, IPCC 2013, pp. 955–956, 1037; IPCC 2014, p. 57). Although projections of the magnitude and rate of warming differ after about mid-century, the overall trajectory of all the projections is one of increased global warming through the end of this century, even for the projections based on scenarios that assume that GHG emissions will stabilize or decline. Thus, there is strong scientific support for projections that warming will continue through the twenty-first century, and that the magnitude and rate of change will be influenced substantially by the extent of GHG emissions (Meehl *et al.* 2007, pp. 760–764 and 797–811; Ganguly *et al.* 2009, pp. 15555–15558; Prinn *et al.* 2011, pp. 527, 529; IPCC 2013, pp. 44–45. See IPCC 2013, entire, for other global projections of climate-related changes, such as frequency of heat waves and changes in precipitation). Long-term predictions of climate change effects vary depending upon alternative emissions scenarios, which in turn vary over a wide range depending on both socioeconomic development and climate policy; thus, uncertainty increases in the predicted magnitude of potential effects after mid-century (IPCC 2013, pp. 1035–1040; IPCC 2014, p. 56). For this reason, as described in the section **Classification of Stressors –Timing (Immediacy) of the Stressor**, for the purposes of making reliable predictions about both the direct and indirect effects of climate change on the West Coast DPS of fisher, we conclude that the near-term predictions supported by wide agreement across both models and emissions scenarios provide the most reasonable scientific basis for our evaluation. We estimated approximately 40 years as a reasonable period of time for reliably forecasting such

effects.

Various changes in climate may have direct or indirect effects on species. These effects may be positive, neutral, or negative, and they may change over time depending on the species and other relevant considerations, such as interactions of climate with other variables (for example, habitat fragmentation) (IPCC 2007, pp. 8–14, 18–19). Identifying likely effects often involves aspects of climate change vulnerability analysis. Vulnerability refers to the degree to which a species (or system) is susceptible to, and unable to cope with, adverse effects of climate change, including climate variability and extremes. Vulnerability is a function of the type, magnitude, and rate of climate change and variation to which a species is exposed, its sensitivity, and its adaptive capacity (IPCC 2007, p. 89; see also Glick *et al.* 2011, pp. 19–22). There is no single method for conducting such analyses that applies to all situations (Glick *et al.* 2011, p. 3). We use our expert judgment and appropriate analytical approaches to weigh relevant information, including uncertainty, in our consideration of various aspects of climate change.

Global climate projections are informative, and in some cases, the only or the best scientific information available for us to use. However, projected changes in climate and related impacts can vary substantially across and within different regions of the world (for example, IPCC 2007, pp. 8–12). For example, in analyzing the potential effects of climate change on tree mortality, Allen *et al.* (2015, p. 22) specifically warn that “many forest responses will be site- and region-specific, so it is important to be cautious about overgeneralizing.” We therefore use “downscaled” projections when they are available, and have been developed through appropriate scientific procedures, because such projections provide higher resolution information that is more relevant to spatial scales used for analyses of a given species (see Glick *et al.* 2011, pp. 58–

61, for a discussion of downscaling). With regard to our analysis for the West Coast range of the fisher, downscaled projections are available in some cases, as are some regional climate models, which provide higher resolution projections using a modeling approach that differs from downscaling.

Most reports discussing downscaled or regional projections of climate change for California and the Pacific Northwest use a suite of climate models along with two different emissions scenarios. The exact suite of models and scenarios varies among reports, but the climate models generally encompass a range of sensitivities to climate scenarios, and the emissions scenarios usually include a lower-emissions scenario along with a medium to high-emissions scenario. The differences between higher- and lower-emissions scenarios are minimal in the next few decades, but become increasingly pronounced after the mid-twenty-first century, thereby leading to

greater uncertainty in projections beyond that timeframe (Cayan *et al.* 2009, p. 7; Mote and Salathé 2010, p. 39). However, the current emissions trajectory is higher than any of the emissions scenarios used in climate projections for California and the Pacific Northwest (Hansen *et al.* 2013, pp. 1–2). Therefore, the projections we discuss here may underestimate the potential effects of climate change. We note that although these projections are downscaled from the global projections, they do not capture the variation that occurs on the much finer local scale at which fishers select and use their environment.

3.1

Temperature

Historical records show increases in temperature throughout the analysis area over the last century. Weather stations in the Pacific Northwest showed a warming trend of approximately 0.8 degrees Celsius (oC) (1.4 degrees Fahrenheit [oF]) per century during the period from 1920–2000 (Mote *et al.* 2010, p. 17). All but two years since 1998 have had temperatures above the 20th century average (Mote *et al.* 2013, p. 28). In the Columbia Basin, which covers large portions of the analysis area in Washington and Oregon, average temperatures rose by 1 oC (1.8 oF) between 1950 and 2006 (Littell *et al.* 2011, pp. 9–11). In California, average temperatures rose by 0.36 oC to 0.92 oC (0.65 oF to 1.7 oF) between 1950 and 1999, with several datasets showing no recent temperature change in the vicinity of Mount Shasta, but relatively large amounts of warming in the Sierra Nevada (Bonfils *et al.* 2008, p. S49 and Fig. 1).

All simulations project a larger increase in temperature across the analysis area over the twenty-first century than occurred during the twentieth century. Projections for temperature increases across the analysis area range from 1 oC to 3 oC (1.8 oF to 5.4 oF) by mid-century and from 2 oC to 5.8 oC (3.6 oF to 10.4 oF) by late in the twenty-first century (Mote *et al.* 2013, p. 34; Pierce *et al.* 2013b, p. 844; Cayan *et al.* 2012, p. 4; Halofsky *et al.* 2011, p. 14; Mote and Salathé 2010, p. 41; Hayhoe *et al.* 2004, p. 12423). Some higher-emissions scenarios were not analyzed in these studies and would likely result in greater warming outside the range reported above (Mote and Salathé 2010, p. 41). Summer temperatures are projected to increase more than winter temperatures (Pierce *et al.* 2013b, p. 845; Cayan *et al.* 2012, p. 8; Mote and Salathé 2010, pp. 41–42; Salathé *et al.* 2010, pp. 65–66; Barr *et al.* 2010a, p. 8; Koopman *et al.* 2010, p. 8; see Table 5).