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Feasibility Assessment for Reintroducing Fishers to Washington

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Washington Department of
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EXECUTIVE SUMMARY

Fishers historically occurred throughout much of the low to mid-elevation forested areas of Washington, though they were not particularly abundant. The fisher was listed as Endangered in Washington in 1998 and is likely extirpated from the state. Two major factors contributed to the decline of fishers in Washington: over-exploitation via commercial trapping, and loss, degradation and fragmentation of suitable habitat. Poisoning and predator control, and incidental capture in traps set for other species were also considered contributing factors in the decline of fishers in the state. Despite protection from legal harvest since 1934, the fisher has not recovered. Extensive surveys from 1990 to 1997 failed to detect them. Reintroduction is considered the best way to recover fishers in Washington because of the absence of nearby populations to recolonize the state. Fishers have been successfully reintroduced in 10 states and 5 provinces in North America including Oregon, Montana, Idaho and British Columbia.

This study was undertaken by the Washington Department of Fish and Wildlife, in partnership with Northwest Ecosystem Alliance, to determine the feasibility of reintroducing fishers to Washington. A team comprised of scientists with expertise in fishers, carnivores, genetics and Geographic Information System habitat analysis provided guidance in design and implementation of the study. Objectives of the study were to determine if there was an adequate amount and configuration of fisher habitat and prey in Washington, if there was a suitable source population available for reintroduction, to assess potential interspecific impacts, determine implementation and legal requirements and to identify potential stakeholders and cooperators.

A habitat assessment was conducted to determine the amount and configuration of suitable fisher habitat in Washington and to evaluate its capability to support a fisher population. In the Pacific coastal states, fishers are closely-associated with late-successional forests; large trees, snags and logs are important resting and denning sites. Suitable habitat was defined as low- and mid-elevation, late successional forest. The amount of late-successional forest needed to support a fisher population in Washington is unknown. The assessment was conducted for the Cascade Range and Olympic Peninsula based on the historical range of the fisher and the current distribution of late-successional forest.

Three potential reintroduction areas were identified from the habitat assessment: the Olympic Peninsula, Northwestern Cascades and Southwestern Cascades. The Olympic Peninsula had the largest amount of suitable fisher habitat; the largest amount of suitable habitat on public lands; the largest amount of suitable habitat in National Parks, National Monuments, and U.S. Forest Service wilderness areas; the largest land area with >50% suitable habitat; and the highest predicted carrying capacity of fishers. Within the Olympic Peninsula, the west side of the Olympic National Park was identified as the best location for a reintroduction. The Southwestern Cascades was the second best choice, and the Northwestern Cascades was ranked third.

In addition to current suitable habitat, results from forest growth modeling indicate that additional late-seral forest will become available within the next 80 years in concentrated areas on the Olympic Peninsula, particularly on the west side.

Late-successional forests support a greater richness and abundance of fisher prey species than second-growth forests. Potential reintroduction areas are landscape mosaics dominated by late-successional forests stands, and these are expected to provide a suitable prey base for a reintroduced population.

Genetic analyses indicate that fishers from British Columbia would be the most suitable source population for reintroductions in Washington. Fishers from California and Alberta would be the second and third most suitable source populations, respectively. Fishers are available from Alberta and may be available from British Columbia; however, they are not available from California due to small population size and protected status.

A fisher reintroduction in Washington is not likely to adversely affect recovery of state or federal species of concern. Although marten populations are suspected to be very low on the Olympic Peninsula and martens use similar habitats and prey species as the fisher, co-existence in other parts of their range suggests that fishers will not adversely affect marten populations. Because fishers are not protected under the Endangered Species Act, and there is no state forest practice critical habitat rule in Washington for this species, a reintroduction would not result in additional regulations for forest management practices on federal, state, or private lands, based on current statutes.

A number of cooperators and stakeholders are interested in a fisher reintroduction. Some have contributed to the assessment, including the Northwest Ecosystem Alliance, National Park Service, U.S. Forest Service and U.S. Fish and Wildlife Service. Other cooperators have offered support in implementing a reintroduction and include Northwest Trek Wildlife Park, Point Defiance Zoo, Woodland Park Zoo, and Oregon Zoo. Tribes, the Washington Department of Natural Resources, the Washington Trappers Association and Washington Forest Protection Association have been informed about the feasibility study and will be consulted with regarding a potential reintroduction.

The amount and configuration of suitable habitat, the availability of a suitable source population and the presence of a diverse prey base indicate that a fisher reintroduction is biologically feasible in Washington. National Park and National Forest lands on the west side of the Olympic Peninsula have been identified as the most suitable sites for a potential fisher reintroduction. It is recommended that a NEPA analysis be initiated for a proposed fisher reintroduction on the west side of the Olympic Peninsula on the Olympic National Park and Olympic National Forest.

INTRODUCTION

The fisher (*Martes pennanti*) has probably been extirpated from Washington and is listed as a State Endangered species (WAC 232-12-014). No known populations of fishers exist in Washington, and there are no populations close enough to Washington to naturally re-establish a population in the state (Aubry and Lewis 2003, Wild and Roessler 2004). The only means of recovery of the species in Washington is likely to be through reintroductions (Lewis and Stinson 1998). This study was undertaken to evaluate the feasibility of restoring fisher populations to Washington through reintroductions.

Translocation is a species conservation tool used to establish, reestablish, or augment a population, and has been used to restore native species. Some of the goals of translocations of rare native species are to increase genetic diversity, establish satellite populations to reduce loss of populations due to catastrophes, and increase population size quickly after successful restoration of habitat (Griffith et al. 1989).

This feasibility study was initiated in Spring 2002 by the Washington Department of Fish and Wildlife (WDFW) in partnership with the Northwest Ecosystem Alliance (NWEA). A team comprised of scientists with expertise in fishers, carnivores, genetics, and spatial habitat modeling was formed to provide guidance in the design and implementation of the feasibility study (Appendix A). The science team defined the geographic scope of the study area, developed a study design, prioritized study activities, and identified criteria to assess the feasibility of reintroduction.

Feasibility studies are recommended to determine if existing habitat, source populations, and the political and social environments are suitable for a successful reintroduction (IUCN 1995). The goal of this feasibility study is to determine if these factors are met in Washington and, if they are, to identify potential reintroduction areas.

Three previous feasibility studies provide examples of approaches for assessing feasibility that were used in this assessment. These studies were associated with a proposed reintroduction in the Great Smoky Mountains of North Carolina (Powell 1990), a 1994-98 reintroduction in Pennsylvania (Serfass et al. 1994, 2001), and a 1996-98 reintroduction in southeast British Columbia (Apps 1995; see also Fontana et al. 1999, Weir 2003). Factors evaluated included: (1) the history and status of the population; (2) an explanation of how factors that caused the decline or extirpation had been eliminated or alleviated; (3) a literature review of key elements of fisher habitat; (4) an assessment of the amount and configuration of suitable habitat and identification of potential release sites; (5) assessments of prey availability and interspecific competition; (6) identification of potential source populations; (7) an assessment of appropriate characteristics of a founding population and approaches for monitoring; (8) recognition and cooperation of stakeholders; and (9) public outreach.

OBJECTIVES

Objectives of this study are to:

- Determine if an adequate amount and configuration of suitable habitat exists in Washington to support a population of fishers,
- Determine if adequate prey exists to support a fisher population,
- Determine if there is a genetically suitable source population for reintroduction,
- Assess the potential ecological impacts of a fisher reintroduction on other species of concern,
- Identify the elements needed to implement a reintroduction program,
- Determine the legal requirements for the capture of fishers out-of-state and the release of animals in Washington, and

- Identify expected stakeholders and cooperators and discuss potential implications of a reintroduction.

A reintroduction would be deemed biologically feasible in Washington if suitable foraging, denning, and resting habitat exists in forested landscapes in amounts and spatial configurations that are likely to support a self-sustaining fisher population; and if an adequate number of fishers from a genetically suitable source population are available. Social, political, and economic factors also need to be addressed for a reintroduction to be successful, but these are not factors that determine biological feasibility.

BACKGROUND

The fisher is a medium-size mammalian carnivore and the largest member of the genus *Martes* in the family Mustelidae (Powell and Zielinski 1994). It has the body build of a stocky weasel -- a pointed face, rounded ears, a long and slender body, short legs, and a well furred tail about one-third its total length. The fur of fishers is generally dark brown, but the rump, tail, and legs are black and the head and shoulders are grizzled with gold or silver (Douglas and Strickland 1987). Males weigh about twice as much as females (adult males: 3.5-5.5 kg; adult females: 2.0-2.5 kg) and are about 20% longer than females (females: 70-95 cm; males: 90-120 cm; total length) (Douglas and Strickland 1987).

Fishers are closely associated with late-successional conifer forests at low- to mid-elevations (Buck et al. 1983, Thomas et al. 1993, Buck et al. 1994, Buskirk and Powell 1994, Jones and Garton 1994, Powell and Zielinski 1994) and require large blocks of continuous forest with high canopy closure (Powell and Zielinski 1994, Zielinski et al. 2004*a*). Habitat elements found in late-successional conifer forests that are important to fishers include large live trees, large snags, and large down logs, which are used as denning and resting sites (Buck 1982, Jones 1991, Seglund 1995, Weir and Harestad 2003, Zielinski et al. 2004*b*), as well as complex physical structure in the understory that supports abundant and diverse prey populations (Buskirk and Powell 1994, Powell and Zielinski 1994). The fisher is an opportunistic predator and its diverse diet appears to reflect seasonal changes in prey availability and vulnerability. Principal prey species during winter include snowshoe hares (*Lepus americanus*), small mammals (mice, voles, and shrews), squirrels (*Tamiasciurus* spp.), porcupines (*Erethizon dorsatum*), birds, and ungulate carrion (Strickland et al. 1982, Powell and Zielinski 1994). Birds, insects and fruit become more important during spring, summer and fall periods (Zielinski et al. 1999).

Historically, two subspecies of fishers were recognized in Washington (Goldman 1935). The subspecies *pacifica* once ranged from coastal British Columbia south to the southern Sierra Nevada and in Washington occurred from the coast to the eastern foothills of the Cascade Range. The subspecies *columbiana* formed a peninsular extension south from Canada along the Rocky Mountains into Idaho, western Montana, northeastern Washington, and into the Blue Mountains of southeastern Washington and northeastern Oregon (Hagmeier 1956, Gibilisco 1994, Lewis and Stinson 1998). Recent studies into the genetic character of fisher populations have provided some insights into population structure (Kyle et al. 2001, Drew et al. 2003, Vinkey 2003, Wisely et al. 2004), but have not resulted in any new subspecies classification.

Population Status

In 1991 the fisher was designated a WDFW Candidate species for possible listing as a State Endangered, Threatened, or Sensitive species. In accordance with the state's listing procedures (WAC 232-12-297), a status review was written for the fisher in 1998 (Lewis and Stinson 1998). Based on this review, the Washington Fish and Wildlife Commission listed the fisher as a State Endangered species in October 1998 (WAC 232-12-014).

There has been no evidence to indicate the recovery of the fisher population in Washington in recent decades, and the lack of reliable observations (i.e., those with physical evidence such as video footage, a photo, tissue sample, track-plate detection, or carcass) suggests that the species has probably been extirpated from the state. Fishers are relatively easy to trap, yet only four reports of incidental captures of fishers in Washington have occurred in recent decades: one each in 1969, 1987, 1990, and 1992 (Lewis and Stinson 1998). Most sighting reports without physical evidence are unreliable, and these have also been rare in Washington since 1990. Aubry and Houston (1992) obtained 137 trapping and sighting records of fishers from Washington from 1894 to 1991 and ranked their reliability. From 1980 to 1992, 12 sightings of fishers were judged to be highly reliable (based on a specimen, photo, or first-person trapping report); half of these were from the Olympic Peninsula. Between 1992 and 1998, an additional 14 sightings were reported, but have not been ranked using the criteria of Aubry and Houston (1992). Systematic surveys conducted between 1990 and 1997 for fishers and other forest carnivores failed to detect fishers in an extensive area of suitable habitat in Washington. WDFW and the U.S. Forest Service (USFS) surveyed approximately 1,500 sample stations in the Cascades, totaling over 17,000 camera/track plate nights, and no fishers were detected (Lewis and Stinson 1998, Aubry and Lewis 2003). Surveys in Olympic National Park in the winters of 2002-03 included over 2,000 camera-nights and produced over 1,200 animal photo detections; however, no fishers were detected (P. Happe, pers. comm.).

Factors Causing Decline

Two major factors contributed to the decline of fishers in Washington: over-exploitation via commercial trapping and loss, degradation, and fragmentation of suitable habitat. Poisoning and predator control, and incidental capture in traps set for other species were also considered contributing factors in the decline of fishers in Washington (Lewis and Stinson 1998, Aubry and Lewis 2003).

From the 1800s to early 1900s over-trapping and habitat loss were likely the most important factors that caused the decline of fisher populations. There were no trapping regulations for fishers prior to the 1930s and high pelt values (Bailey 1936, Lewis and Zielinski 1996) provided strong incentives to trap fishers (Powell and Zielinski 1994, Aubry and Lewis 2003). Fishers were most abundant in low- to mid-elevation coniferous forests that were easily accessible to trappers. This combination of factors resulted in heavy trapping pressure on fishers in the 1800s and early 1900s. Concern by biologists over declining fisher populations throughout their range in the lower 48 states prompted closure of trapping seasons in many states in the 1930s (Powell and Zielinski 1994). In Washington, the trapping season for fishers has been closed since 1934. Despite protection from commercial harvest for 70 years in Washington, it has not recovered. The timing and duration of this formal protection combined with the lack of recovery, suggest that unregulated commercial trapping had a profoundly negative impact on the fisher population in Washington. Protection from trapping occurred prior to extensive loss of habitat.

Habitat loss and degradation was another important factor in the decline of fishers in Washington. When Europeans first arrived in Washington, there were about 10 million ha (24.7 million ac) of forest. Of this, perhaps 6.1 million ha (15 million ac) were potential fisher habitat based on elevation and associated suitable forest cover types (Bolsinger et al. 1997). The exact percentage of forest in late-successional (mature and old-growth forest) condition is unknown, but it was a large proportion. Inventories in the 1930s indicated 40% was still in old-growth (Bolsinger and Waddell 1993). Logging began by clearing low elevation valleys because these areas had the largest trees, were the most accessible, and were suitable for agriculture and other development.

Logging subsequently proceeded up drainages to higher elevations. Between the early 1930s and 1992, the total area of old-growth forest in Washington was reduced by 70%, from >3.7 million ha to 1.1 million ha (Bolsinger and Waddell 1993) and about 10% of forests in Washington were converted to other uses, with greater losses occurring in western Washington (1.5 million acres, 11.2 %) than in eastern Washington (0.6 million acres, 7.5 %) (Bolsinger et al. 1997). In the Puget Sound region, extensive areas of high-volume timber were converted to urban areas, agriculture and "stump pastures" following logging (Bolsinger et al. 1997). Much of the forest in the valleys was converted to farmland, and private industry eventually acquired a large portion of the productive lower elevation timberlands. Most of the low-elevation, late-successional forest that provided suitable habitat for fishers was converted to short rotation (~50 years) plantations of even-aged Douglas-fir (*Pseudotsuga menziesii*), and is now highly fragmented (Bolsinger et al. 1997). Intensively managed tree plantations typically retain few large trees, snags and logs compared to unmanaged forests, and short-rotations prevent the development of these large structures. In addition, the removal of large amounts of coarse woody debris from intensively managed plantations greatly reduces structural diversity near the forest floor.

Use of strychnine as a predator control and harvest method may have contributed to the decline of fishers in Washington in the early 1900s (see Bailey 1930*a,b;* McIntyre 1995). Historical ranges of the gray wolf (*Canis lupus fuscus*) and the fisher overlapped extensively in the Pacific Northwest (Aubry and Lewis 2003). Poisoning and trapping of the gray wolf conducted in Washington in the early 1900s likely contributed to local extirpations of fishers (see Marshall 1992). The fisher was also sympatric with the mountain lion (*Puma concolor*) and coyote (*Canis latrans*) (Ingles 1965) that were also targets of early predator control programs. Gray wolves were essentially eliminated from the Pacific States by the 1930s (Bailey 1936, Grinnell et al. 1937, Dalquest 1948), and most predator control programs and bounties were discontinued by the 1960s. Fishers were unlikely to recover following the end of formal predator control programs due to the more significant effects of habitat loss and over-trapping (Aubry and Lewis 2003).

Mortalities from incidental captures may have also contributed to the decline of fishers in Washington (Aubry and Lewis 2003). Fishers are easily captured in traps set for other furbearers and such incidental captures can be a significant source of mortality (Lewis and Zielinski 1996). During one closed trapping season in British Columbia, the number of fishers incidentally captured exceeded the legal harvest the previous year (V. Banci, pers. comm. cited in Powell and Zielinski 1994).

Fishers may not have recovered in Washington because over-trapping reduced populations to such low levels that they could not recover due to demographic traits that include low population density, low reproductive rate, and short life span (Powell 1993). Subpopulations that became isolated by intervening areas of unsuitable habitat may have been demographically isolated and this may have accelerated their decline and prevented reoccupation of suitable habitat.

Fisher Reintroductions

The fisher is one of the most frequently and successfully reintroduced carnivores (Berg 1982, Reading and Clark 1996, Breitenmoser et al. 2001). Since the 1940s, wildlife managers have reintroduced fishers as a means of re-establishing a valuable furbearer, a natural predator of the porcupine, and a native carnivore (Berg 1982). Reintroduction efforts began in Nova Scotia in the 1940s and became commonplace in the 1950s and 1960s throughout the species' range (Berg 1982). There have been at least 31 fisher reintroductions attempted throughout their range in the U.S. and Canada from 1947 to 2003 (Table 1). Of the 31 reintroductions, 21 (68%)

were considered successful (i.e., fishers persisted $\geq$10 years following first release), 7 were considered failures (22%), 2 were not evaluated (6%), and 1 is ongoing. Reintroductions have been more successful in eastern states and provinces (79%) than in western states and provinces (58%) (Table 1).

Although fishers have not been reintroduced in Washington, reintroductions or augmentations (collectively referred to as translocations) have occurred elsewhere in the Pacific Northwest. During the late 1950s and early 1960s, fishers were translocated to Montana, Idaho and Oregon. Additional translocations occurred in the late 1970s and early 1980s in Oregon and Alberta and during the late 1980s and 1990s in Montana, Alberta, and British Columbia (Table 1). Reintroductions in Montana, Oregon, Idaho and British Columbia were successful.

Montana. After the 1920s, the lack of fisher captures by trappers indicated that fishers were rare in the state (Weckwerth and Wright 1968). Consequently, in 1959 and 1960, 36 fishers from central British Columbia were translocated to three sites in northwestern Montana (Weckwerth and Wright 1968). The Montana Fish and Game Department (MFGD) coordinated the project with the British Columbia Game Branch, British Columbia trappers and the U.S. Forest Service, and the relationships developed by MFGD expedited subsequent translocations of fishers from British Columbia to Oregon (1961) and Idaho (1962-1963) (Morse 1961).

Each fisher was ear tagged and ear tattooed (Weckwerth and Wright 1968). There were no formal efforts to monitor the success of the reintroduction. Trapping of American martens (*Martes americana*) was prohibited in the reintroduction areas for 5 years to ensure maximum protection of released fishers (Weckwerth and Wright 1968). Despite the marten season closure, 6 marked and 13 unmarked fishers were captured from 1960-1968 in traps set for other species including mink (*Mustela vison*), wolverine (*Gulo gulo*), bobcat (*Lynx rufus*), and lynx (*Lynx canadensis*). All 6 marked fishers were found dead in traps; 7 of the 13 unmarked fishers were found dead and the remaining 6 were released alive. Two other fisher mortalities were documented from 1960-1968: one was shot and the second was found dead (Weckwerth and Wright 1968). The 19 incidental captures and 2 other mortalities were the only information available to indicate the success or failure of the reintroduction. It indicated that fishers had survived up to 7 years after the releases, were successfully reproducing, had dispersed from 3-64 miles (4.8-102.4 km) from the nearest reintroduction site, and were susceptible to incidental capture and mortality in traps set for other species, despite protection from marten trapping for 5 years in the release area. Investigations into the distribution and genetic characteristics of fishers in Montana by Vinkey (2003) indicated that fishers with British Columbia genetic traits still occured in the vicinity of the release areas, and therefore, the translocation was considered successful.

Despite the persistence of fishers following the translocations, they were limited in distribution and extremely rare within their historical range in Montana (Heinemeyer 1993). Consequently, in 1989-91, 32 fishers from Minnesota (Roy 1991) and 78 fishers from Wisconsin (Heinemeyer 1993) were translocated to the Cabinet Mountains of northwestern Montana. Fifty-seven of 110 fishers (52%) were radio-tagged and monitored. Radio-tagged fishers suffered significantly high levels of mortality from predation, trapping, or research-related fatalities. Heinemeyer (1993) suspected that extended periods of captivity may have negatively affected some radio-tagged fishers from Minnesota by allowing them to become overweight and making them more susceptible to mortality. Roy (1991) reported that only 2 of 32 radio-tagged fishers that he monitored made long distance movements (>70 km) away from the reintroduction sites. Heinemeyer (1993) reported that none of 25 radio-tagged fishers she

Table 1. Fisher reintroductions in North America (modified from Roy 1991).

Release location	Source location	Year	Number released	Sex ratio M:F	Status[1]	Source
Nova Scotia	Ranch	1947-48	12	6:6	S	Benson 1959
Wisconsin	New York	1955-57	14	6:8	S	Bradle 1957
Ontario	Ontario	1956	25	Unknown	U	Berg 1982
Ontario	Ontario	1956-63	97	37:60	S	Berg 1982
Wisconsin	Minnesota, New York	1956-63	60	Unknown	S	Irvine et al. 1964
Montana	British Columbia	1959-60	36	16:20	S	Weckwerth & Wright 1968
Vermont	Maine	1959-60	35	19:16	S	Berg 1982
Oregon	British Columbia	1960	24	10:14	F	Aubry & Lewis 2003
Michigan	Minnesota	1961-63	61	Unknown	S	Irvine et al. 1964
Idaho	British Columbia	1962	39	Unknown	S	Luque 1984
Nova Scotia	Maine	1963-66	80	Unknown	S	Dodds & Martell 1971
Wisconsin	Minnesota	1966-77	60	30:30	S	Petersen et al. 1977
New Brunswick	New Brunswick	1966-69	25	10:15	S	Drew et al. 2003
West Virginia	New Hampshire	1968	23	6:10, 7 Unknown	S	Pack & Cromer 1981
Minnesota	Minnesota	1968	15	Unknown	U	Berg 1982
Maine	Maine	1972	7	Unknown	F	Berg 1982
Manitoba	Manitoba	1972-73	4	Unknown	F	Berg 1982
New York	New York	1977	43	Unknown	S	Wallace & Henry 1985
Oregon	British Columbia	1977-80	17	10:7	S	Aubry & Lewis 2003
Ontario	Ontario	1979-82	57	27:30	S	Kyle et al. 2001
Oregon	Minnesota	1981	13	8:5	S	Aubry & Lewis 2003
Alberta	Alberta	1981-83	32	16:16	F	J. Jorgenson, pers. comm.
Montana	Minnesota	1988-89	32	13:19	S	Roy 1991
Alberta	Ontario, Manitoba	1990	17	6:11	F	Proulx et al. 1994
British Columbia	British Columbia	1990-91	10	Unknown	F	R. Weir, pers. comm.
Montana	Wisconsin	1990-91	78	34:44	S	Heinemeyer 1993
British Columbia	British Columbia	1990-92	15	2:13	S	Weir 1995
Manitoba	Manitoba	1991-93	45	24:21	S	Baird & Frey 2000
Pennsylvania	New York, New Hampshire	1994-98	190	Unknown	S	Serfass 1998
British Columbia	British Columbia	1996-98	60	24:36	F	Weir et al. 2003
Tennessee	Wisconsin	2001-03	40	20:20	O	Anderson 2002

[1] S – Successful (fishers persisted for ≥10 yrs.), F – Failed, U – No evaluation, O – On-going.

monitored moved beyond 20 km from the release sites while they were monitored. The lack of long distance movements that Heinemeyer (1993) observed may have been influenced by the presence of resident fishers from Minnesota in the vicinity of the release site. Despite the high levels of mortality reported by Roy (1991) and Heinemeyer (1993), Vinkey (2003) found evidence that fishers with Minnesota and Wisconsin genetic traits occurred in northwestern Montana, and therefore these reintroductions were considered successful.

Roy (1991) made several suggestions on how to improve the success of a reintroduction which included: reintroduce fishers from nearby populations that have been exposed to similar predator complexes, release fishers earlier than January-March, release females with kits in April, use soft releases, release equal sex ratios of fishers, develop a core population and provide appropriate protection for released fishers.

Idaho. In 1962 and 1963, 39 fishers from central British Columbia were translocated to 3 locations in Idaho (Luque 1984). The fisher had apparently been extirpated or become very rare, and measures were taken to reestablish a population in Idaho in conjunction with fisher reintroduction efforts in Montana and Oregon (Williams 1962). The reintroduction was a cooperative effort among Idaho Department of Fish and Game, British Columbia trappers, British Columbia Fish and Game Branch, and the U. S. Forest Service. No formal monitoring occurred after the releases, however the trapping season was closed to protect released fishers. The incidental capture of $\geq$170 fishers by marten trappers and sightings by hunters indicated that the fisher population was persisting (Luque 1984). From 1985-88, Jones (1991) conducted an ecological study of fishers (n = 16) in Idaho, presumably the descendents of the reintroduced population. The commercial trapping season for fishers has not been opened in Idaho, where there is still concern for the status of the population. However, the persistence of the population, as evidenced by recent incidental captures and other documented observations, indicates that the reintroduction was successful. Vinkey (2003) noted that fishers from Idaho and Montana may interact demographically; consequently the Idaho fisher population may exhibit genetic traits common to Idaho, Montana, Wisconsin, Minnesota and British Columbia.

Oregon. In January and March of 1961, 24 fishers from central British Columbia were reintroduced to 2 locations in Oregon: one in the Wallowa mountains of northeastern Oregon and one in the southern Oregon Cascades (Kebbe 1961, Aubry and Lewis 2003). Fishers had declined in Oregon and the trapping season was closed in 1937 to protect the remaining population. Despite protection from commercial trapping, the fisher remained rare and restricted in its range in the state (Aubry and Lewis 2003). Severe porcupine damage to commercial timber plantations prompted the U. S. Forest Service and Weyerhaeuser Corporation to seek assistance from the Oregon Game Commission to reintroduce fishers (Aubry and Lewis 2003). The reintroduction was a cooperative effort among the Oregon Game Commission, British Columbia Trappers, British Columbia Fish and Wildlife Branch, U. S. Forest Service and Weyerhaeuser Corporation. Released fishers were protected from trapping by a closed trapping season, and trapping and poisoning of any kind were prohibited in a 625 km^2 area around the southern Oregon site for 5 years after the release (Kebbe 1961). All released fishers were eared-tagged in both ears, but no monitoring occurred after release. As a result, little or no information was available to determine if released fishers persisted (Aubry and Lewis 2003).

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 to commercial timber (Aubry and Lewis 2003). Seven timber companies, the Oregon State Wildlife Commission, and the U.S. Forest Service worked cooperatively to plan and conduct additional translocations into the southern Oregon Cascades: 17 fishers were reintroduced from British Columbia from 1977-1980 and 13 were reintroduced from northern Minnesota in 1981 (Aubry and Lewis 2003). There was no record of these fishers being ear-tagged and no post-release monitoring occurred. However, recovered specimens of fishers that were incidentally captured, road-killed or shot indicated that fishers had persisted after these releases. In addition, survey efforts in the 1990s indicated that a small population occurred on the Rogue River National Forest (Aubry and Lewis 2003). Drew et al. (2003) 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 and Lewis 2003, Aubry et al. 2004). The persistence of a small, reintroduced population (Aubry and Lewis 2003, Drew et al. 2003) in the southern Oregon Cascades indicated that the 1977-81 reintroductions were successful.

British Columbia. A number of translocations have been undertaken in British Columbia (Banci 1989, Weir 2003), but only two have been documented. With assistance from British Columbia trappers, Weir (1995) translocated and monitored 15 fishers in 1990-92 from the Chilcotin River area of west-central British Columbia to the Williams Lake area 300 km to the east in south central British Columbia. The release was considered an augmentation, as a small number of unmarked fishers were discovered during the study. Weir (1995) investigated post release movements, establishment of home ranges, and survival of translocated fishers. Some released fishers were captured during a commercial trapping season that occurred during and after the translocation (1995). Although released fishers wandered extensively and most established home ranges >15 km from their release sites, fishers persisted and reproduced in the Williams Lake area after their release (R. Weir, pers. comm.).

From 1996 to 1998, 60 fishers from central British Columbia were reintroduced to the East Kootenay area of southeast British Columbia (Fontana et al. 1999). Fishers were considered extirpated from the East Kootenay region by 1995 (Fontana et al. 1999). The goals of the reintroduction were to reestablish fishers in suitable habitat and to reestablish a link between populations in British Columbia, Alberta, Montana and Idaho.

A habitat assessment was conducted for the East Kootenay reintroduction (Apps 1995) to identify suitable habitat and rank potential reintroduction sites. Local trappers were asked not to trap near the release areas, or to use fisher excluding devices on their marten sets if they did continue to trap in the area. Translocated fishers were maintained in captivity for an average of 34 days before being released. Thirty-seven of the 60 released fishers were radio-tagged and tracked (Fontana et al. 1999). Although radio-tagged fishers moved extensively, including a number that traveled to Montana, 11 fishers established "home areas". Fontana et al. (1999) reported persistence and establishment of home ranges in the vicinity of the release sites, but they did not document reproduction in the reintroduced population.

In 2002-2003, Weir et al. (2003) did a follow-up assessment of the East Kootenay reintroduction to determine if it had been successful. The assessment included extensive hair-snagging and snow-tracking surveys, live-trapping and investigations of fishers caught by trappers. No fishers were detected during the hair snagging surveys, 1 set of confirmed and 3 sets of possible fisher tracks were found during snow tracking surveys, and 1 adult male fisher was captured during 509 trap-nights. They also reported seven incidental captures of fishers from 1997 to 2000, which indicated that reintroduced fishers had successfully reproduced. Weir et al. (2003) detected fishers on four occasions within the reintroduction area, however, the actual population size was unknown. Based on their assessment and despite having recent presence documented, this reintroduction appeared to have failed. They concluded that the quality, configuration and extent of habitat in the reintroduction area were likely insufficient to support a viable population and that factors such as incidental capture and mortality of fishers in marten traps also contributed to the apparent decline in the population.

Alberta. Loss of fishers from portions of their historical range in Alberta prompted reintroductions into southwestern Alberta (Davie 1984) and central Alberta (Proulx et al. 1994).

Thirty-two fishers were translocated from northern Alberta into the Kananaskis area of southwestern Alberta in 1981-83 (Davie 1984). All 32 fishers were radio-collared and tracked during 1981 and 1982; however, results of the monitoring efforts were not reported (Davie 1984). Five post-release mortalities were documented, including one female to predation, one male from an infection, and three females to incidental capture by registered trappers. One female was captured 15 km from its release site and the other two were captured 100 km from their release sites. The absence of incidental captures or observations of fishers in the decades following the release indicated that the reintroduction was not successful (J. Jorgenson, pers. comm.). The reason for the apparent failure was unknown.

In March and June of 1990, 17 Ontario and Manitoba fishers that had been used in a study of captive fishers were translocated to several protected parklands of central Alberta (Proulx et al. 1994). Trapping was not allowed in the protected parklands but some trapping occurred outside these areas. As part of the reintroduction, Proulx et al. (1994) tested the effect of releasing fishers at different times of the year on the success of the reintroduction. They found that fishers released in June stayed closer to the release area than those released in March. The occurrence of the mating season and the reduced vegetative cover in March may have prompted fishers to travel extensively from the release area, whereas in June fishers were not seeking mates and deciduous tree cover was much greater and provided better habitat conditions in the release area. They also found that fishers released in June selected higher quality habitats than those released in March. The establishment of home ranges and documented reproduction indicated that the reintroduction was succeeding (G. Proulx, pers. comm.). However, three years after the reintroduction, coyote snaring was allowed within the reintroduction area, and fisher mortalities as a result of snaring and pest poisoning on local farms contributed to the extirpation of the reintroduced population (G. Proulx, pers. comm.).

FOOD HABITS

Principal Prey Species

Few fisher food habits studies have occurred in western North America. Initial studies were in California (Grenfell and Fasenfest 1979) and the Pacific coastal states (Ingles 1965), and later studies were in Idaho (Jones 1991), Montana (Roy 1991), British Columbia (Weir 1995), and California (Zielinski et al. 1999) (Table 2). Ingles (1965) reported principal food items in the Pacific coastal states to include porcupines, squirrels, woodrats (*Neotoma* spp.), mice, marmots, mountain beavers (*Aplodontia rufa*), quail, and grouse. No food habits studies have previously been conducted in areas where mountain beaver and fisher populations are sympatric. However, in the early 1900s trappers on the Olympic Peninsula found mountain beaver and squirrel remains in stomachs of fishers, and scats collected along trails in summer contained huckleberries (*Vaccinium* sp.) and salal berries (*Gaultheria shallon*) (Scheffer 1995). This is the only information on food habits of fishers in Washington. Most food habits studies conducted in western North America provide information on the winter diet (Table 2). This is due to the readily available

Table 2. Percent occurrence of food items in fisher scats and gastrointestinal tracts from western North America.

	Season								
	Winter						Spring	Summer	Fall
Prey	BC[1]	MT[2]	ID[3]	ID[4]	CA[5]	CA[6]	CA[6]	CA[6]	CA[6]
Mammals									
Peromyscus maniculatus	13								
Peromyscus leucopus			14						
Peromyscus sp.		14			25	8	6	16	
Clethrionomys gapperi	19		29	6					
Unident. voles				28					
Microtus spp.	6	3				13	6	5	
Reithrodontomys megalotis					13				
Neotoma cinerea	2	7							4
Zapus princeps				6					
Marmota flaviventris			14	6					
Tamiasciurus hudsonicus	27		14	22					
Tamiasciurus douglasii						4	11	6	4
Tamius spp.		3		6				1	8
Glaucomys sabrinus	6							1	
Sciurus griseus					13	8	2	4	4
Spermophylus beecheyi							6	4	4
Spermophylus sp.				6					
Thomomys bottae							6	6	4
Thomomys sp.				6					
Castor canadensis			29	6					
Erethizon dorsatum	16	6	6						
Unident. rodents		6							
Sorex spp.	12						1	3	4
Scapanus latimanus					13		4	2	
Lepus americana	31	49	29	50					
Sylvilagus bachmani					13				
Martes pennanti	9								
Martes americana	9	7							
Martes sp.		6				8	28	15	35
Unident. Mustelids	1			6			2		
Spilogale putorius									4
Odocoileus spp. (carrion)	9	3	14	11	25				
Cervus elaphus (carrion)			29	6		25	4		
Alces alces (carrion)	16		14	11					
Unident. ungulate (carrion)			29	22					
Birds									
Galliformes	7								
Unident. birds			14	17		25	32	51	27
Reptiles							38	20	4
Insects				22	25	42	53	62	50
Fruit[7]	tr					17		8	23
Seeds				17					

[1]Weir (1995), *n* = 261 gastrointestinal tracts; [2]Roy (1991), *n* = 80 scats; [3]Jones (1991), *n* =7 gastrointestinal tracts; [4] Jones (1991), *n* = 18 scats; [5]Grenfell & Fasenfest (1979), *n* = 8 gastrointestinal tracts; [6]Zielinski et al. (1999), *n* = 201 scats; [7]*Vaccinium* spp., *Ribes* spp. or *Arctostaphylos* spp. berries.

source of carcasses provided by trappers during the legal trapping season for fishers, or collection of scats during winter releases. A single study conducted in the southern Sierra Nevada of California provides information on seasonal food habits of fishers in western North America (Zielinski et al. 1999).

Winter diet. The most important prey species in the winter diet of fishers from British Columbia (Weir 1995), Idaho (Jones 1991), and Montana (Roy 1991 (Roy 1991) were snowshoe hares, red squirrels (*Tamiasciurus hudsonicus*) and small mammals (Table 2), based on frequency occurrence of food items in scats or stomachs. Weir (1995) aggregated prey species found in fisher stomachs into nine food groups based on similarity in niches and body sizes. The top three food groups were small mammals (23.3%), followed by squirrels (15.0%), and snowshoe hares (14.5%). Porcupines occurred with greater frequency in prey remains in British Columbia (16%, Weir 1995) than in Montana (6%, Roy 1991) or Idaho (6%, Jones 1991). Ungulate carrion is also an important winter food item (Table 2). In the southern Sierra Nevada of California, important winter food items included squirrels (20.8%), cricetids (41.7%), ungulate carrion (25%), birds (25%), and insects (41.7%)(Zielinski et al. 1999). Analysis of eight fisher carcasses collected in the Trinity National Forest in northern California, included remains of false truffles (*Rhizopogon* spp.) (50%), ungulate carrion (25.0%), small mammals (12.5%), western gray squirrels (*Sciurus griseus*; 12.5%), leporids (12.5%), and beetles (25.0%) (Grenfell and Fasenfest 1979).

Spring, summer, and autumn diet. In the only study of year-round fisher food habits in the Pacific States, Zielinski et al. (1999) found little seasonal variation in the diet of fishers in the southern Sierra Nevada. The most common prey in scats during spring, summer and autumn were sciurids (15.4-24.5%), including California ground squirrel (*Spermophilus beecheyi)*, western gray squirrel, and Douglas' squirrel (*Tamiasciurus douglasii*), birds (26.9-51.0%), and insects (50.0-62.2%). Beetles (Coleoptera) and social wasps (Vespidae/Eumenidae) were the most common insects in the diet. Murids (primarily *Peromyscus* spp. and *Microtus* spp.) and reptiles were more important food items in spring and summer, comprising 15.1-26.5% and 20.4-37.7% of prey remains, respectively. Fruit became more important in the diet during fall and winter. The fact that no single family of plant or animal group occurred in more than 22% of feces attests to the diversity of the fisher diet in California. A study in the southern Oregon Cascade Range also indicates that the fisher is a dietary generalist. Prey remains collected over several years at den sites and resting sites in southern Oregon included hares, rabbits, squirrels (California ground squirrel, Douglas squirrel, northern flying squirrel [*Glaucomys sabrinus*]), woodrat, Virginia opossum (*Didelphis virginiana*), striped skunk (*Mephitis mephitis*), North American porcupine, bobcat, deer (*Odocoileus* spp.), elk (*Cervus elaphus*), birds (Stellar's jay [*Cyanocitta stelleri*], pileated woodpecker [*Dryocopus pileatus*], hairy woodpecker [*Picoides villosus*], northern flicker [*Colaptes auratus*], ruffed grouse [*Bonasa umbellus*], turkey [*Meleagris gallopavo*]), berries, and yellow jackets (Aubry and Raley 2002).

While there is little information available on the food habits of fishers in Washington (Scheffer 1995), an evaluation of the diet of bobcats in western Washington could provide insight into the diet of fishers if they are reintroduced to the state. In regions of North America where bobcats and fishers are sympatric, the diets of both species include many of the same prey species (Litvaitis 1984, Litvaitis et al. 1986, Arthur et al. 1989, Giuliano et al. 1989). Analysis of bobcat stomachs (n = 8) and scats (n = 99) collected in northern and western river drainages of the Olympic Peninsula indicates that snowshoe hares (44% of stomachs and scats) and Douglas squirrels (18%) were the most important prey; mountain beavers were rarely (1%) consumed (Schwartz and Mitchell 1945). However, Young (1958) reported that mountain beavers were the dominant food of bobcats in Washington during spring and summer periods. Since mountain beavers do not occur east of the Cascade Range in Washington (Carraway and Verts 1993), Young must have been referring to the Cascades, Olympic, or Coast Ranges. Sweeney (1978) also found that mountain beavers were the most common prey in bobcat scats (56.6%) collected primarily from the Olympic Peninsula and the Coast Range during fall and winter. Snowshoe hares were second in importance (39.5%), followed by small mammals (15.7%) and squirrels (Douglas' squirrel, northern flying squirrel; 9.2%). Black-tailed deer (*Odocoileus hemionus*) and possibly elk were less important in the diet (6.6%) (Sweeney 1978). Similarly, Knick et al. (1984) reported mountain beavers and snowshoe hares to be the primary prey of bobcats west of the Cascade Mountain crest in Washington. Mountain beavers and snowshoe hares together occurred in 68% of stomachs and accounted for 83% of the weight of all food items. The greater importance of mountain beavers in recent studies, compared to Schwartz' and Mitchell's study may be attributed to changes in availability of mountain beaver habitat (Knick et al. 1984). Mountain beavers are likely more abundant in second growth forests compared to old growth forests (Hoover 1977, Carraway and Verts 1993). Sweeney (1978) suggested that much of Schwartz and Mitchell's data on bobcat food habits apparently were collected in old-growth forests (possibly within Olympic National Park), whereas his data were collected from bobcats collected in managed forests (Olympic National Forest). Logging and burning in western Washington have increased the proportion of forests in early successional stages, the preferred habitat of mountain beavers. Mountain beaver densities may have changed as a result and provided a more abundant food resource to bobcats (Knick et al. 1984) (Knick et al. 1984).

Regional differences in bobcat diets in the Coast and Cascade Ranges of Washington and Oregon may also indicate differences in fisher diets. Sweeney (1978) compared food habits of bobcats for the Coast Range and Olympic Peninsula with the western Cascades in Washington. Bobcat stomachs from the coastal region (Coast Range and Olympic Peninsula) contained primarily mountain beavers (39.2%) and snowshoe hares (27.5%), with trace amounts of deer (2.9%). On the western slope of the Cascade Range, bobcat stomachs contained proportionally fewer mountain beavers (25%) and snowshoe hares (16.7%) and proportionally more deer (16.7%), although the sample from the Cascades was small (n = 7 stomachs). Nussbaum and Maser (1975) also compared the diet of bobcats in the Coast Range (n = 143 scats) and western Cascade Range (n = 34) in Oregon. In both the Coast Range and Cascade Range, leporids (52.5% and 70.6%, respectively) and small mammals (56.7%, 58.7%) occurred with the greatest frequencies. However, in contrast to Washington, leporids occurred with greater frequency in scats from the western Cascade Range (70.6% snowshoe hare) compared to the Coast Range (52.5%; 44.1% brush rabbit [*Sylvilagus bachmani*], 8.4% snowshoe hare). Mountain beavers occurred less frequently (8.4% and 2.9%) in this study, although they were the most common prey species in bobcat scats during spring (84.0%), summer-fall (73.7%) and winter (62.2%) months in the Coast Range of southwestern Oregon (Witmer and deCalesta 1986). In a study of the seasonal diet of bobcats in the western Cascades of Oregon, Toweill and Anthony (1988) also found snowshoe hares (30%), black-tailed deer (22%), and mountain beavers (12%) to be the most common food items in bobcat scats (n = 494). Snowshoe hares and black-tailed deer dominated the diet throughout the year, whereas mountain beavers occurred with greatest frequency in spring and summer periods. Cricetid rodents occurred in 23% of scats, varying from 9% in winter to 37% in spring. Fruit was an important food item during summer months (24%).

FISHER HABITAT CHARACTERISTICS

Within Stand-Level Structures

Den sites. In western North America, fishers select large live, decadent or dead trees for natal den sites, where females give birth to their young and nurse them until weaning at about eight weeks of age (Seglund 1995, Aubry and Raley 2002, Weir and Harestad 2003). In British Columbia, natal dens occurred in branch-hole cavities in decadent cottonwood trees (*Populus* spp.) that averaged 103 cm (41.2 in) dbh and 25.9 m (84.9 ft) in height (Weir and Harestad 2003). Den trees were the largest diameter trees in the immediate vicinity and occurred infrequently in fishers' home ranges. In California, Seglund (1995) located two natal dens belonging to the same female; one was located in a cavity of a 78 cm (31.2 in) dbh ponderosa pine (*Pinus ponderosa*) snag and the other was in a hollow lateral limb of an 88 cm (35.2 in) dbh live black oak (*Quercus kelloggi*). In southern Oregon, females used cavities in snags (n = 6) or live trees (n = 8) that averaged 93 cm (37.2 in) dbh (range 61-138 cm, 24.4-55.2 in) and 16 m (52.5 ft) in height (range 4 - 46.5 m, 13.1-152.5 ft) above the ground (Aubry and Raley 2002). Openings that provided access to hollows created by heartwood decay were mostly (57%) made by pileated woodpeckers. In Montana, Roy (1991) found a natal den in a hollow log 11 m long with a 30 cm diameter cavity. Weir's (1995) findings of fisher use of deciduous trees as natal dens were consistent with studies in eastern North America (Arthur 1987, Paragi 1990). Recent data from Oregon indicates that a variety of conifer tree species serve as dens (Aubry and Raley 2002). Availability of large den trees is likely a limiting factor for fishers in landscapes dominated by short-rotation (<50-60 years) forestry in which large snags are removed and forest succession is truncated.

Maternal dens are sites used by adult females and kits after weaning and during the period in which kits remain dependent on the adult females for food (Aubry and Raley 2002). Kits are moved from natal dens to maternal dens at about 8-10 weeks of age and are kept in maternal dens until about five months of age (late August or early September) (Paragi 1990, Seglund 1995). Maternal den structures are more variable than natal dens, and are typically closer to the ground. Adult females and kits used cavities in lower parts of live and dead trees, large (>50 cm dbh) hollow logs, mistletoe brooms, and rodent nests (Aubry and Raley 2002).

Rest sites. Fishers use rest sites between periods of activity. Rest sites are typically used for only a single resting or sleeping bout, but the same site may be used for many days when weather is severe or a large food item has been cached nearby (Powell and Zielinski 1994). Rest structures used by fishers in western North America include mistletoe and rust brooms, large lateral limbs and limb clusters in the canopies of live trees, rodent or raptor nests, cavities in snags or logs, ground burrows, or beneath piles of cull logs (Buck et al. 1983, Jones 1991, Seglund 1995, Aubry and Raley 2002, Weir and Harestad 2003, Zielinski et al. 2004*a*).

Fishers typically rest in live trees (Table 3) and the most common resting platforms are bird stick nests, large lateral limbs (Seglund 1995) or brooms (Jones 1991, Weir 1995, Aubry and Raley 2002). In the Coast Range of northwestern California, rest sites were typically located in stick nests (30%) or on large lateral limbs or limb clusters (30%), but mistletoe brooms were used infrequently (9%) (Seglund 1995). In the same area, Zielinski et al. (2004*a*) found fishers resting most frequently in cavities and broken tops of live trees (50%), followed by snags (26%), platforms (mistletoe brooms, and nests; 18%), and logs (5%). Fishers use mistletoe or rust brooms more frequently than any other type of rest site in British Columbia (Weir 1995), Idaho (Jones 1991), and Oregon (Aubry and Raley 2002). Females used witches brooms more frequently than males (Seglund 1995).

Table 3. Structures used by male and female fishers for denning and resting in western North America (adapted from Lewis and Stinson 1998).

	Trees		Snags		Ground			
Location	N	%	N	%	N	%	Total	Source
California	6	67	2	22	1	11	9	Buck 1982
California	80	63	34	27	13	10	127	Zielinski et al. 1995
California	76	67	23	20	15	13	114	Seglund 1995
Idaho	134	78	13	8	25	15	172	Jones 1991
Oregon	414	63	90	14	149	23	653	Aubry and Raley 2002
	710	66	162	15	203	19	1075	

Fishers appear to require rest sites in large diameter trees that are usually the largest and tallest within the immediate area (Buck et al. 1983, Seglund 1995, Weir 1995, Zielinski et al. 2004*a*). In British Columbia the most common rest sites were in trees that averaged 46.3 cm (18.5 in) in diameter (Weir 1995). In Idaho, fishers rested in trees that averaged 56.1 cm (22.4 in) in diameter and 16.4 m (54 ft) in height (Jones 1991). Snags and logs had a median diameter of 86.4 cm (34.5 in) and 53.3 cm (21.3 in), respectively. In the Coast Range of California, mean diameter of resting sites in live hardwood trees, live conifer trees, snags, and trees with platform structures was 87.6 cm (35.1 in), 124.7 (49.9 in), 119.0 (47.6 in), and 68.1 (27.2 in), respectively (Zielinski et al. 2004*a*). Logs averaged 95.1 cm (38.0 in) in diameter. In earlier studies in the same area, Buck et al. (1983) found rest sites in trees that averaged 114.3 cm (45.7 in) in diameter, and Seglund (1995) reported rest sites in trees and snags that averaged 105 cm (42 in) and 119 cm (47.6 in) in diameter, respectively.

Rest sites are typically in conifer trees. In Idaho, fishers rested primarily in Engelmann spruce (*Picea engelmannii*) where witches brooms were most common (Jones 1991), and in British Columbia they used hybrid white spruce (*P. engelmannii x glauca*) with rust brooms (Weir 1995). In southern Oregon, female fishers rested primarily in large, live Douglas-fir trees, and secondarily in Douglas-fir or White/grand fir (*Abies concolor/ A. grandis*) snags. Males also rested in live trees, but used western hemlock (*Tsuga heterophylla*), Douglas-fir and white/grand fir about equally; Douglas-fir snags were used secondarily (Aubry and Raley 2002). Fishers in northwestern California rested predominantly in Douglas-fir trees (Seglund 1995, Zielinski et al. 2004*a*).

Type of rest sites used varies seasonally. In the West, fishers rest predominantly in the canopies of live trees in both winter and summer (Jones 1991, Buck et al. 1994, Seglund 1995). The greater vertical layering of vegetation and greater conifer canopy cover in mature and old growth forests provide a range of cooler and moister microclimates below the forest canopy. Convective heat loss would be greater for fishers that used rest sites in the upper canopy and may prevent thermal stress for fishers during the prolonged heat and desiccation during the dry season (Zielinski et al. 2004*a*), and possibly during winter months (Jones 1991). During periods of colder temperatures, fishers typically seek out cavities. In Idaho, fishers rested more frequently in logs during winter (Jones 1991). Ground dens are used more frequently during periods of extreme cold (Arthur et al. 1989, Weir 1995). Female fishers in the Coast Range of northwestern California used snags more frequently in winter, whereas males primarily rested in the canopy of live trees during both summer and winter (Seglund 1995). Because of their smaller body size, females may require warmer micro-sites than males. Moreover, rest site selection is likely influenced by proximity to areas of high prey availability. These findings suggest that fishers select rest sites with suitable microclimate to reduce thermal stress (Jones 1991, Zielinski et al. 2004*a*).

Individual resting structures are infrequently reused (Jones 1991, Kilpatrick and Rego 1994, Seglund 1995, Zielinski et al. 2004*a*). Zielinski et al. (2004*a*) suggested that infrequent resting structure reuse indicates that fishers do not limit use of their home range to a few central locations, and instead require multiple resting structures distributed throughout their home range. Martens forage sequentially over their home range, using rest sites in snags in close proximity to foraging areas and recent kill sites (Marshall 1951). The pattern of rest site use by fishers indicates that they do the same. Zielinski et al. (2004*a*) suggested that the low reuse of rest sites may be a strategy to minimize travel time between resting locations and kill sites, which are distributed throughout the home range.

Fishers select resting structures in patches of forest characterized by greater structural complexity. In the Coast Range of northwestern California, rest sites were more structurally diverse than random sites. Rest sites were characterized by a greater number of vegetation layers, higher percentage of dead and down woody material, and a greater percentage of shrub cover than random sites (Seglund 1995). Zielinski et al. (2004*a*) characterized forest structure around rest sites in the same area. A univariate analysis revealed that rest sites contained significantly greater maximum tree dbh, greater standard deviation of dbh, small standard deviation of canopy closure, and a greater number of large conifer snags than random sites. A resource selection function included greater canopy closure, larger maximum tree size, steeper slopes, and at least one large conifer snag as significant variables in the model for the Coast Range. Rest sites were characterized by a large resting structure, were in close proximity to other large trees and occurred in areas with denser canopies. In addition, topographic position was an important factor, with rest sites located on steep slopes. Rest sites had greater structural variability (i.e., a diversity of sizes and types of structural elements) but less variable canopy cover than random sites (Zielinski et al. 2004*a*). Fishers in British Columbia also demonstrate selection for greater forest structural complexity at rest sites, particularly in stands characterized by more simplified structure (Weir and Harestad 2003). During summer months, fishers in Idaho used sites characterized by greater densities of trees >47 cm (18.5 in) dbh, snags 14-52 cm (5.5-20.5 in) dbh, and logs 14-54 cm (5.5-21.2 in) diameter than sites 50 m (15.2 ft) distant (Jones 1991). Fishers also selected more decadent patches of forest during winter, choosing sites that had greater densities of large trees (>47 cm [18.5 in] dbh), snags (24-34 cm [9.4-13.4 in] dbh and >52 cm [20.5 in] dbh), and logs (≥47 cm [18.5 in] diameter). These findings suggest that fisher rest sites are located in more structurally complex forest, typical of mature and old-growth forest conditions.

Stand-Level Characteristics

Fisher selectivity for continuous overhead cover and structural complexity at the within-stand level is also evident at the stand level for resting and foraging activities. In Idaho, fishers used forest stands with 61-80% canopy cover significantly more for resting, whereas stands with more open (21-40%) and denser (≥81%) canopy cover were used for hunting (Jones 1991). Fishers in California occurred more frequently in stands with high canopy closure. Buck (1982) reported that fisher locations were most common in forest stands with 40-70% canopy closure. In the southern Sierra Nevada, high canopy density stands (60-100%) occupied the greatest proportional area (66%) of fisher home ranges (Zielinski et al. 2004*b*). In British Columbia, fishers selected stands with a mean coniferous canopy closure between 21-60% in winter, and showed no selectivity for coniferous canopy closure during summer or autumn months (Weir 1995). During summer, fishers preferred stands with 21-40% deciduous canopy cover and avoided stands with no deciduous canopy component.

Fishers demonstrate selection for structurally complex forest stands. They may select mature closed-canopy forest because the microclimate provides warmth in winter and prevents overheating during summer (Buck 1982, Seglund 1995), and the greater structural complexity of the forest floor provides habitat for prey and winter resting structures (Weir 1995). In the Coast Range of northwestern California, fishers preferred mature, closed conifer forest, especially multi-species stands (Buck 1982). Fishers in British Columbia also demonstrated selectivity for stands with greater structural diversity, particularly greater volumes of coarse woody debris, during summer and winter months (Weir 1995, Weir and Harestad 2003). During summer, fishers in Idaho preferred mature and old-growth stands and avoided non-forest, pole-sapling, and young forest stages (Jones 1991, Jones and Garton 1994). Forested stands used in summer had more large diameter (≥34 cm [13.4 in]) trees, snags and logs compared to available habitat. During winter, fishers preferred young forest, used mature and old-growth stands in proportion to availability, and avoided nonforest and pole-sapling stands. Fishers selected stands with greater densities of 11.4-34.3 cm (4.5-13.5 in), and >62.2 cm dbh trees, greater densities of all size classes of snags, and a dense understory of Pacific yew (*Taxus brevifolia*) (Jones 1991, Jones and Garton 1994). Availability of snags was an important factor in winter site selection. Buck (1982) also found fishers using young regenerating stands in winter that had high overhead canopy cover (>80%) and vegetation between 1.5 and 3.0 m (4.9 and 9.8 ft) in height. Fishers seem to prefer more structurally complex forest for both resting and hunting, but will use stands with more simplified structure (ie, pole-sapling and young forest) for hunting. Other researchers have suggested that fishers are more selective of resting compared to foraging habitat (Arthur et al. 1989, Buskirk and Powell 1994). Although fishers in Idaho demonstrated selection for younger forests in winter, these stands were naturally regenerated following fire and contained large live trees, snags and logs; characteristic of older forests (Jones 1991).

Fishers may prefer to forage in more structurally complex forest stands because they encounter a greater abundance and diversity of prey. Douglas' squirrels are more abundant in older, more structurally complex forest stands compared to younger, managed forests (Buchanan et al. 1990). Squirrels may prefer older forest stands because these habitats provide a more abundant, perennial, and diverse source of food (e.g., conifer seed and hypogeous fungi). Older forests have a greater diversity of older age tree species, and therefore greater cone production, and greater amounts of coarse woody debris in later stages of decomposition, which is associated with greater abundance and diversity of hypogeous fungi (Buchanan et al. 1990, Carey 1991, Luoma et al. 2003). The cool, mesic conditions in older forests preserve cone caches and facilitate growth of truffles beneath well-decayed coarse woody debris that retains water during the prolonged dry summer months (Luoma et al. 2003). Small mammals may be associated with coarse woody debris for cover, nesting sites, or associations with food (McComb 2003), such as hypogeous fungi (Rhoades 1986). Southern red-backed vole (*Clethrionomys gapperi*) abundance and activity was positively correlated with coarse woody debris (Ucitel et al. 2003). While similar species of small mammals are found in naturally regenerated Douglas-fir forests in the southern Washington Cascade Range, abundance is greater in old growth than in young forests, and is likely attributed to the greater structure and productivity of the forest floor environment (West 1991). In the Western Hemlock Zone of the Olympic Peninsula, composition of small mammal communities in naturally regenerated and clearcut-regenerated young forests are similar to those found in old growth. However, old growth forests support a greater abundance and biomass of small mammals than managed forests (Carey and Johnson 1995). Many of these small mammal species exhibit numerical responses to the amount of coarse woody debris and shrub cover in the forest floor environment. Mountain beavers are also found on sites with greater availability of sword fern (*Polystichum munitum*), shrubs and ferns, greater volumes of coarse woody debris, and mesic conditions (Hacker 1991, Carraway and Verts 1993, McComb 2003). Fishers are likely to encounter mountain beavers in old growth forests with a well-developed vegetative understory and in early successional forests (Hacker 1991). Fishers may also encounter greater numbers of cavity nesting birds (e.g., woodpeckers, sapsuckers) in older forests while exploring snags as possible rest sites.

Landscape-Level Characteristics

Home range. Home range sizes have been estimated for fishers in nine radio-telemetry studies in western North America (Table 4). Home range sizes vary considerably, however the extremes in the range of values in Table 4 are likely the result of small sample sizes.

Fishers appear to be sensitive to fragmentation of their preferred habitat. In Douglas-fir forests in northwestern California, fishers were less likely to occur in stands of increasing insularity and decreasing stand area (Rosenberg and Raphael 1986). Fishers also demonstrate avoidance of nonforest cover types (Jones 1991, Roy 1991, Weir 1995). Jones (1991) suggested that management of fisher habitat at a landscape level should include a mosaic of early- and mid-successional forest seral stages to provide a diversity of prey species, and mature and old growth forest to provide key resting habitat. Patches of resting habitat should be connected by closed canopy forest to facilitate travel between patches. The proportion of each of these seral stages necessary to support fishers in a landscape is not known.

Riparian areas. Fishers are primarily associated with cool, mesic forests (Buskirk and Powell 1994) and this may explain their disproportionate use of riparian areas in more arid landscapes in some western states (see Jones 1991, Seglund 1995). Proximity to water does not appear to influence rest site selection in the cooler and moister forests in the Coast Range of the Pacific Northwest (Zielinski et al. 2004*a*).

Table 4. Mean annual home range sizes for male and female fishers as determined in nine studies in western North America.

	Males		Females		
Location	Mean annual home range (km^2)	N	Mean annual home range (km^2)	N	Study
So. Oregon Cascades	147[1]	3	25	7	Aubry and Raley 2002
So. Oregon Cascades	62[2]	4			Aubry and Raley 2002
SE British Columbia	59.1	1	27	3	Fontana et al. 1999
Central British Columbia	46.5	1	35.4	11	Weir 1995
Central British Columbia	122[3]	3			Weir 1995
Central British Columbia	73.9[4]	1			Weir 1995
Central Alberta	24.3	2	14.9	10	Badry et al. 1997
Idaho	79	6	32	4	Jones 1991
Northern California	20	3	6.8	2	Buck et al. 1983
Northern California	53.9	4	53.4	2	Dark 1997
Northern California	51	6			Truex et al. 1998
Northern California	58	2	15.0	7	Zielinski et al. 2004*b*
weighted mean $\pm$1 SD	68.2$\pm$36.5		24.95$\pm$10.9		

[1] Breeding season range size.
[2] Non-breeding season range size.
[3] Summer range size.
[4] Winter range size.

Fishers select structurally complex forest at the with-stand, stand and landscape levels. The fishers' association with large structures typically found in late-seral forest, the large home ranges of individuals, and their sensitivity to forest fragmentation suggests that landscapes comprised of large, contiguous patches of late-seral forest are more likely to support a fisher population than more fragmented landscapes containing patches of late-seral forest.

HABITAT ASSESSMENT

The objectives of the habitat assessment were: (1) to identify the amount, distribution and connectivity of fisher habitat within the study area, and (2) to determine if existing amounts and configurations of habitat could support one or more populations of fishers.

Study Area

The study area was confined to the Olympic Peninsula and Cascade Range of western Washington, where low- to mid-elevation, late-successional forests remain (Fig. 1). Although historical populations of fishers also occurred in northeastern and southeastern Washington, these areas were not included at this time because the core of the fisher's range occurred in the coniferous forests of western Washington from the eastern foothills of the Cascades to the coast (Scheffer 1938, Dalquest 1948, Lewis and Stinson 1998). In addition, the Cascade Range is an important link in the range of the Pacific fisher from British Columbia to the southern Sierra Nevada.

Methods

Identifying suitable habitat. In the absence of habitat use data for fishers in Washington, the science team used existing knowledge of fisher habitat requirements to define suitable fisher habitat as relatively dense, low to mid-elevation, late-seral coniferous forest. These forests are expected to meet the major habitat needs of fishers for denning, resting, foraging and traveling. Four variables were used to define suitable fisher habitat: % vegetative cover, % conifer cover, quadratic mean diameter, and elevation (Table 5). The vegetative cover variable was used to identify densely vegetated areas (>70% vegetative cover); fishers avoid more open cover types. Conifer cover was used to identify vegetated areas with a ≥30% conifer component. In the western part of the range, fishers are strongly associated with conifer-dominated stands and are not expected to select hardwood-dominated stands in Washington. Fishers do use hardwood trees in mixed hardwood-conifer stands, especially for denning sites. A relatively low value for conifer cover (≥30%) was used so that potentially important denning habitat was not excluded by the model. Quadratic mean diameter (QMD) is a measure of the mean diameter at breast height of trees in a stand. A QMD value of ≥20 inches was used to identify forest stands that had relatively large overstory trees representative of mature or old-growth forests. The elevation limit for suitable fisher habitat was defined as the upper limit of the Pacific silver fir zone (*Abies amabilis*). Below this elevation, snowpack is not expected to impede travel by fishers. Fishers may be less energy efficient when traveling or foraging in deep, powdery snow. The science team determined that snow levels and conditions above the Pacific silver fir zone would likely prohibit efficient travel and foraging by fishers; therefore, areas above the Pacific silver fir zone were considered less suitable for fishers. In addition, forests above the Pacific silver fir zone may not have suitable densities of large trees, snags and logs that fishers use as rest and den sites. Data to delimit the upper level of the Pacific silver fir zone were provided by J. Henderson (pers. comm.).

Mid-seral and other forests, which could be used by fishers for travel between patches of suitable habitat, were also identified. Mid-seral forests had QMD values of 10-20 inches, and could occur above or below the Pacific silver fir zone. Other forests were defined as all areas that had <70% vegetation cover. For other analyses, early seral forests (QMD values 1-10 inches) were identified.

Landcover datasets were evaluated for their utility in identifying late-seral forests. Landcover data from the Interagency Vegetation Mapping Project (IVMP; USDI Bureau of Land Management and USDA Forest Service; http://www.or.blm.gov/gis/projects/ivmp.asp) and year 2000 Landsat 7 (EROS Data Center, Sioux Falls, SD) were selected because they were the most recent landcover data available and provided complete coverage of the study area. IVMP landcover datasets were available for the Olympic Peninsula and the western and eastern Cascades. The extent of these datasets became the boundary of the study area (Fig. 1). Initial GIS modeling efforts indicated that the

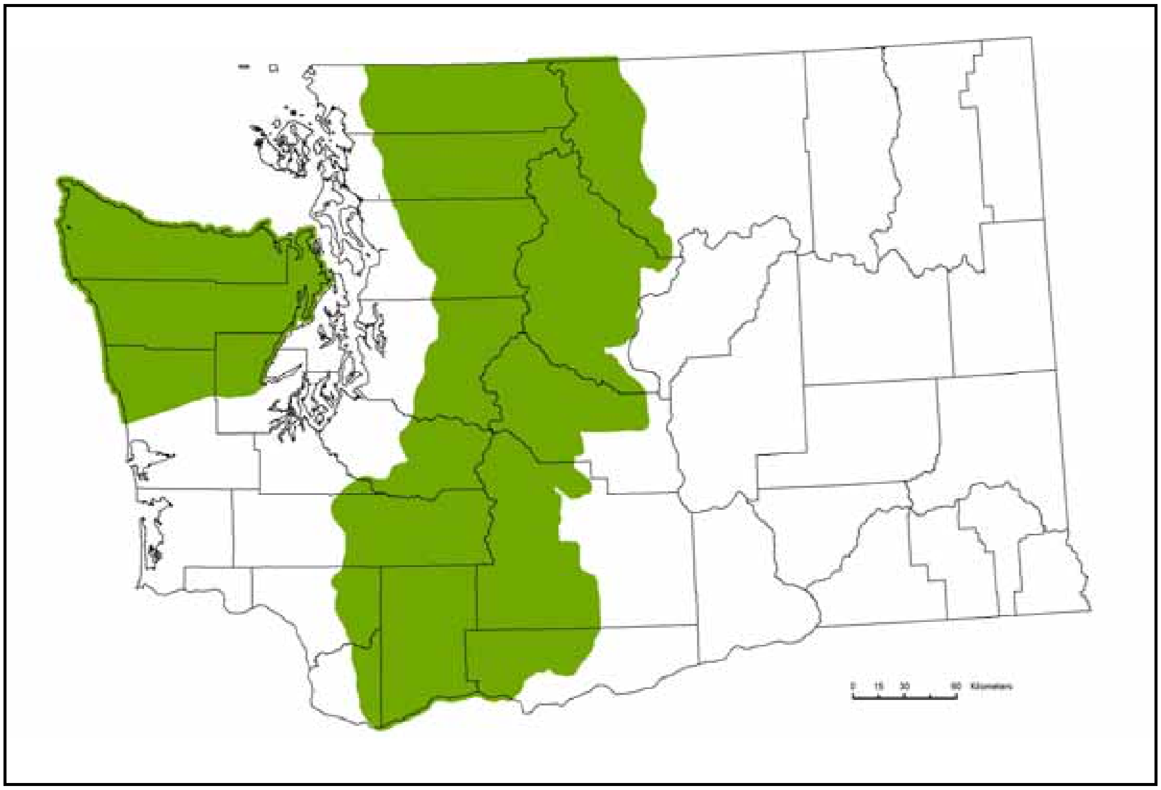

Figure 1. Study area for the habitat assessment including the Olympic Peninsula and the Cascade Range in Washington.

Table 5. Variables used to define suitable fisher habitat in western Washington.

Variable	Value
Vegetative Cover	$\geq$70%
Conifer Cover	$\geq$30%
Quadratic Mean Diameter	$\geq$20 in
Elevation	below upper level of the Pacific silver fir zone

suitable habitat model appeared to match closely with known areas of low to mid-elevation late-seral forest in the western Cascades and the Olympic Peninsula, but not in the eastern Cascades. The habitat identified in the eastern Cascades was very sparse and uniformly distributed, did not coincide well with known late-seral habitats, and did not match well with habitat identified in the western Cascades near the Cascade crest. After considering several alternative datasets to the IVMP data for the eastern Cascades, suitable habitat was identified using an interpretation of recent (year 2000) Landsat 7 data and elevation (Table 5; Jacobson et al. 2003).

Comparisons of the amount of suitable habitat available in western Washington were made using other datasets representative of late-seral forest conditions. The amount of suitable habitat identified by the fisher model was compared with the amount of late-seral forests identified by other landcover datasets on three national forests (Gifford Pinchot, Mount Baker-Snoqualmie, and Olympic) and Olympic National Park. For the Gifford Pinchot National Forest, Mount Baker-Snoqualmie National Forest and Olympic National Park, late-seral conifer forest was identified below the upper level of the Pacific silver fir zone. For the Olympic National Forest spotted owl, and marbled murrelet (*Brachyramphus marmoratus*) habitat data was used as a surrogate for late-seral forest.

Existing forest plot data on the Olympic National Park (ONP) and Olympic National Forest (ONF) was used to compare the fisher habitat model with values used to define old-growth forests. Data from forest plots located within identified suitable habitat were obtained for the Park from Pacific Meridian Resources (1996); data were also obtained from the USDA Forest Service's Current Vegetation Survey (CVS; http://www.fs.fed.us/r6/survey/) for the Olympic National Forest. These plot data provided specific measures of forest structure at a scale where fishers select habitats: the rest site and den site scale (Weir and Harestad 2003, Zielinski et al. 2004*a*). Measures of forest structure from 50 plots (27 from ONP and 23 from ONF) were used to evaluate canopy closure, the presence of a multi-storied canopy, the number of large snags and downed logs, and the presence of shrub cover, as these features are consistently found at fisher rest and den sites. Means were calculated for these structures from the 50 plots and compared to values used to define old-growth forests (Marcot et al. 1991).

Habitat connectivity. After the initial suitable fisher habitat was identified, a habitat connectivity analysis was conducted to identify areas where fishers could potentially access patches of suitable habitat due to either the proximity of patches to each other, or their proximity to other forest types that could provide travel cover and foraging habitat. The analysis was conducted in three steps. First, larger patches of suitable habitat ($\geq$10 ha) were selected by removing patches smaller than 10 ha. The 10 ha minimum patch size was considered large enough that fishers would traverse suboptimal habitat to access it (Rosenberg and Raphael 1986), whereas smaller, remote patches of suitable habitat may be used less frequently.

The second step of the analysis identified additional habitat that could potentially provide travel cover and connectivity between larger patches of suitable habitat. Mid-seral forests both above and below the Pacific silver fir zone, late-seral forest above the Pacific silver fir zone, and small patches of suitable habitat would be expected to provide connectivity among larger patches of suitable habitat. Given the size and composition of fisher home ranges in western North America, members of the science team agreed that fishers could forage in these cover types and could easily traverse 1 km of these cover types to access larger patches of suitable habitat. Thus, these cover types were identified where they occurred within 500 m of large patches of suitable fisher habitat.

The third step of the analysis identified patches of suitable habitat that were interconnected by foraging and travel cover. Fishers tend to avoid large openings but can traverse small areas of unsuitable or marginal habitat to access suitable habitat or travel cover. Members of the science team agreed that fishers could easily traverse a 200 m distance across areas of unsuitable or marginal habitat (e.g., a small opening) to access suitable habitat or other forests that provide travel cover (e.g., mid-seral forest). Step 3 of the analysis combined the original suitable habitat patches and habitats identified in step 2 into habitat blocks if they occurred within 200 m of each other.

Identifying potential reintroduction areas. The connectivity analysis identified large patches of suitable habitat that were connected by travel cover to form habitat blocks. The largest blocks of connected habitat were identified and given a 3-km buffer to create an obvious boundary, to fill in interior gaps of unclassified cover types, and to reduce edge effects during carrying-capacity analyses. The 3 km buffer was chosen because it approximates the radius (2.81 km) of a circle representing the mean home range size (25 km^2) of female fishers (Table 4). This radius was considered sufficient to eliminate edge effects in the carrying capacity analyses, as it would include habitats within a fisher home range that was centered on the edge of a block of habitat. To be consistent with the carrying capacity analyses, buffered areas were identified as potential reintroduction areas and were described with regard to total area, elevation, kilometers of major roads, road density, area within habitat concentration contours, amount of suitable habitat and other cover types, and land ownership.

A moving window analysis was used to identify areas of highly concentrated suitable habitat. A 25 km^2 moving window size was chosen because it equals the mean home range size of female fishers (Table 4) and reflects the scale at which females select habitats within a landscape. Contours with concentrations of 1-25%, 26-50%, 51-75%, and 76-100% suitable habitat were identified. The amount and configuration of suitable habitat in potential reintroduction areas could then be compared.

Carrying capacity. Two methods were used to estimate the number of female fishers that might be supported in potential reintroduction areas. The first method used PATCH (Schumaker 1998), a spatially-explicit population model, to estimate carrying capacity of each potential reintroduction area. To run the model, six data inputs were required: maximum dispersal distance; values for mean, maximum and minimum home range sizes; annual survival; and fecundity (Table 6). Habitat association scores were also required (Appendix B) in conjunction with a habitat map of each potential reintroduction area. Population size estimates could indicate whether a potential reintroduction area might support a self-sustaining population. Areas identified as source habitats (i.e. areas where recruitment exceeded mortality) could also be considered as potential reintroduction sites. Because no demographic or habitat use data exist for fishers in Washington, measures of home range size, fecundity, survival, dispersal and habitat associations were obtained from the literature, focusing whenever possible on the findings from studies conducted in western North America. A description of the habitat maps and demographic parameters incorporated into the model can be found in Appendix B.

The second method to determine carrying capacity was a simple calculation of the amount of suitable habitat in a potential reintroduction area divided by the mean home range size of female fishers (25 km^2; Table 4). This method assumes no spatial overlap in female home ranges, which is commonly reported in the literature. It also assumes that home ranges consist of 100% suitable habitat, which makes it a very conservative estimate, because home ranges are unlikely to contain 100% suitable habitat.

Ranking potential reintroduction areas. Six measures were used to compare and rank potential reintroduction areas: amount of suitable habitat, amount of suitable habitat on public land, amount of suitable habitat on conservation status 1 lands (national parks, monuments, and wilderness areas; Cassidy et al. 1997), land area with concentration of >50% suitable habitat, proximity to the nearest existing fisher population, and estimates of carrying capacity. Using rank scores of 1-3 for each criteria measure (with 1 being the best score), the potential reintroduction area with the lowest combined score was considered the most suitable as a reintroduction site.

Results

Identifying suitable habitat. A total of 901,107 ha of suitable habitat was identified by the model in three regions: the Olympic Peninsula, the west Cascades and the east Cascades (Fig. 2, Table 7). The greatest amount of suitable habitat (463,904 ha) occurred in the west Cascades, but this region was also the largest in overall size. The Olympic Peninsula was the smallest region, but had the greatest percentage of suitable habitat (24.4%). The east Cascades, although large in area, had the smallest amount of suitable habitat (184,866 ha) (Table 7). Concentrations of suitable habitat on the Olympic Peninsula occurred in the western portion, within Olympic National Park and Olympic National Forest. Suitable fisher habitat was more widely dispersed in the western and eastern Cascade regions of the study area (Fig. 2).

Table 6. Parameter values used as inputs to the PATCH model.

Input Parameter	All females	Adults	Subadults	Juveniles
Max. dispersal distance	50 km			
Home range ($\bar{x} \pm$ sd)	24.95 ± 10.9 km^2			
Min. home range[1]	14.05 km^2			
Max. home range[2]	35.85 km^2			
Annual survival ($\bar{x} \pm$ sd)		0.780 ± 0.11	0.780 ± 0.11	0.683 ± 0.22
Fecundity[3]		0.680	0.389	

[1] Minimum home range size = mean home range size (24.95) – 1 standard deviation (10.9).
[2] Maximum home range size = mean home range size (24.95) + 1 standard deviation (10.9).
[3] The number of female young produced per year, by reproductive age class.

Table 7. Amount and percentage of suitable habitat within regions of the study area.

Region	Region area (ha)	Suitable habitat (ha)	Percent of region in suitable habitat
Olympic Peninsula	1,032,123	252,337	24.4
Western Cascades	2,733,716	463,904	17.0
Eastern Cascades	2,338,752	184,866	7.9
Total	6,104,591	901,107	14.8

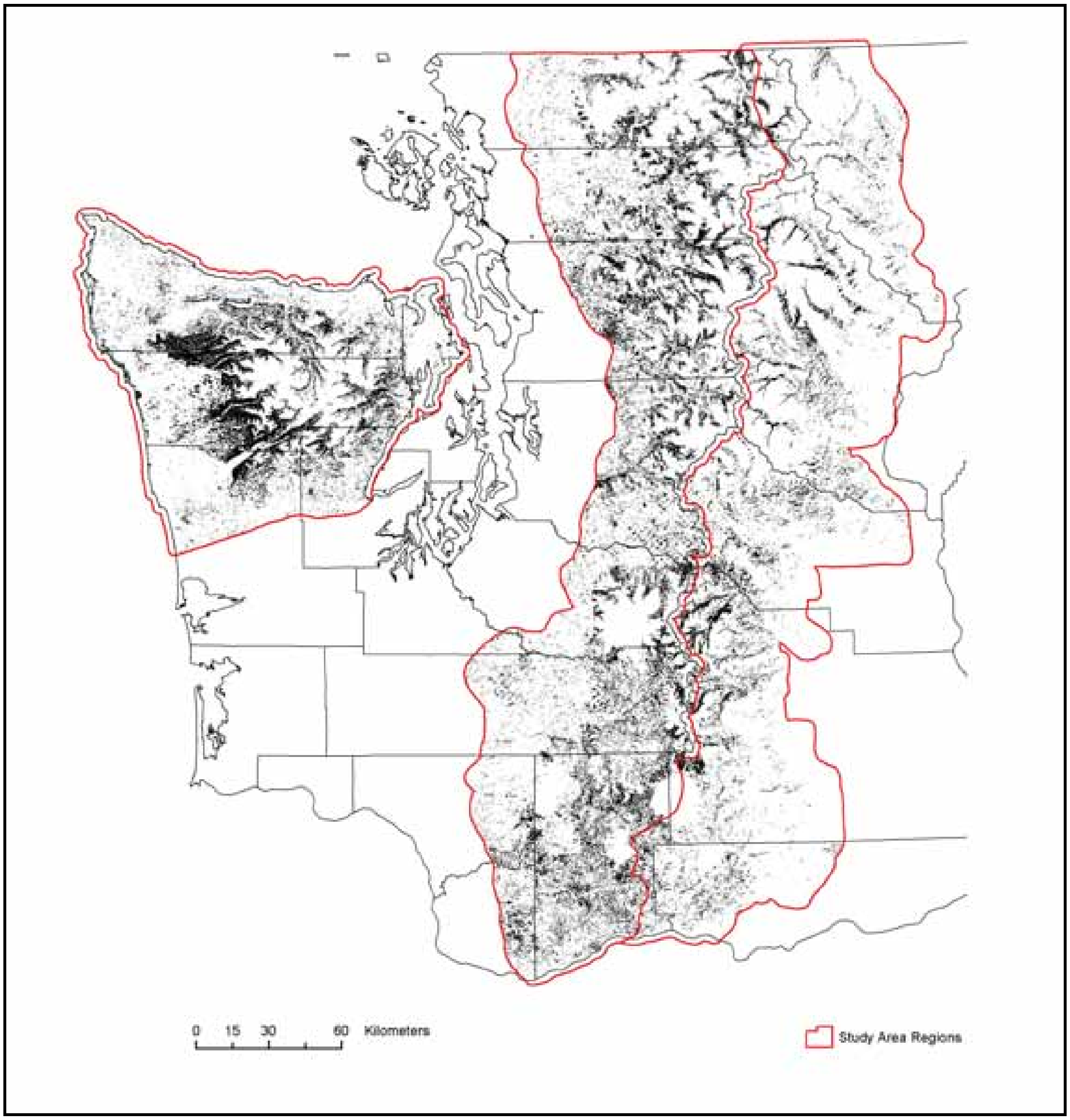

Figure 2. Suitable fisher habitat (in black) identified by a fisher habitat model in the Cascade Mountains and Olympic Peninsula of western Washington.

Comparisons between the suitable habitat model and alternative data models show close agreement on the amount of suitable habitat identified across the study area (Table 8). The spatial overlap of suitable and unsuitable habitats identified by the fisher suitable habitat model and the alternative data models ranged from 62.2 to 68.9%.

On the Olympic Peninsula mean plot values met or exceeded at least one of the three old-growth definition values for large live trees, large snags, and large downed logs (Marcot et al. 1991; Table 9). The variability of plot values is not expected to have any effect on the validity of the comparison, as individual plot values that were lower than accepted old-growth definitions may reflect values of mature forests, which are also considered suitable fisher habitat. The abundance of canopy trees in the 50-100 ft and >100 ft tall size classes suggests that canopies within the 1-ha plots were relatively dense and multi-storied (Table 9). The plot data also showed that areas identified as suitable habitat also supported significant ground cover. Forest plots located in areas identified as suitable habitat by the model contained forest structure that was consistent with accepted definitions of old-growth forests and with habitats selected by fishers as rest sites and den sites (Seglund 1995, Aubry and Raley 2002, Mazzoni 2002, Weir and Harestad 2003, Zielinski et al. 2004*a*).

Table 8. Comparison of amounts of suitable habitat identified by the suitable habitat model and the alternative data models for the Olympic Peninsula (including only the Olympic National Forest and Olympic National Park), and the Mount Baker-Snoqualmie and Gifford Pinchot National Forests, Washington.

Comparison area	Suitable habitat model (ha)	Alternative data models (ha)
Olympic Peninsula	197,762	195,575
Mount Baker-Snoqualmie National Forest	169,635	176,522
Gifford Pinchot National Forest	136,537	128,905

Table 9. Comparison of mean forest plot values for 6 fisher habitat components with 3 definitions of old-growth conifer forest (Marcot et al. 1991). Plots (1-ha) were selected on the Olympic National Park (*n* = 27) and Olympic National Forest (*n* = 23) if they occurred within suitable habitat identified by the fisher habitat model.

Habitat Component[1]		Means for Plot Data[2]		Values for Old-Growth Forest Definitions[3]		
		CVS Plots	PMR Plots	OGDTG	PNWRG	SAF
Live Trees >32” dbh	Count per ha	17		≥20	≥12	≥25
	DBH (in)	45	31	>32	≥32	>40
	Age (yrs)	299	145	>200		>200
Canopy: trees 50-100ft tall	Count per ha	14		Multi-Storied	Multi-Storied	Multi-Storied
	DBH (in)	20				
	Age (yrs)	133				
	Height (ft)	74				
Canopy: trees >100ft tall	Count per ha	21				
	DBH (in)	33				
	Age (yrs)	231				
	Height (ft)	138				
Snags ≥ 20”	Count per ha	11	20	≥10	≥5	>25
	DBH (in)	35		>20		>25
	Height (ft)	39		>15		>20
Logs ≥20” diameter	Count per ha	3		≥10		“some”
	Diameter (in)	30		≥24		>25
	Length (ft)	49		≥50		≥50
	Metric Tons		70	≥34	≥67	>45
Ground Cover ≥1ft tall	Height (ft)	3				
	Areas (%)	30	39			

[1] Plot variables identified by the Science Team as important within-stand attributes for fishers.
[2] CVS = Current Vegetation Survey plot data for the Olympic National Forest, PMR = Pacific Meridian Resources plot data for the Olympic National Park (Pacific Meridian Resources 1996).
[3] OGDTG = Old-Growth Definition Task Group, PNWRG = Pacific Northwest Regional Guide definition of the USDA Forest Service, and SAF = the Society of American Foresters.

Although the same analysis was not conducted for the Cascades, based on the Olympic Peninsula comparisons, it is assumed that the model also adequately identified suitable habitat in the Cascades.

Habitat connectivity. The habitat connectivity analysis identified mid-seral forest, late-seral forests above the Pacific silver fir zone, and small patches (<10 ha) of suitable habitat within 500 m of large patches (>10 ha) of suitable habitat (Fig. 3). Habitat patches and forests identified in Figure 3 that were within 200 m of each other were lumped into habitat blocks (Fig. 4). Three large blocks were identified: one on the Olympic Peninsula, one in the northwestern Cascades and one in the southwestern Cascades (Fig. 4). Although a portion of the southwestern Cascades block occurs on the east slope of the Cascades, there were no large blocks identified exclusively in the eastern Cascades.

Identifying potential reintroduction areas. A 3-km buffer was placed around each of the 3 largest blocks identified in Figure 4 to delineate potential reintroduction areas (Fig. 5). The remainder of the habitat assessment was restricted to these areas. All three potential reintroduction areas were dominated by suitable habitat, but also contained other cover types that are used by fishers (Table 10; Figs. 6, 7, and 8). By adding the 3 km buffer to define the potential reintroduction areas, disproportionately more private lands and cover types other than suitable habitat may have been added into these areas.

The amount of suitable habitat varied among the potential reintroduction areas, with the Olympic Peninsula having the most, followed by the Southwestern Cascades, and then the Northwestern Cascades (Table 10). However, the percent of suitable habitat was consistent among them (22-25%; Table 10). Characteristics of the potential reintroduction areas are summarized in Table 10. These areas are large in size, occur across a wide range of elevations, and are not heavily developed with major roads (e.g., state or federal highways) that could restrict fisher movements (Table 10, Fig. 9). The Olympic Peninsula area has one major road (U.S. Highway 101) and 346 km of this highway occurs around the periphery of the potential reintroduction area (Fig. 9). A relatively short distance (94 km) of U.S. Highway 2 crosses the southern third of the Northwestern Cascades area, and a slightly greater distance of U.S. Highway 12 (121 km) crosses the northern half of the Southwestern Cascades area (Table 10, Fig. 9). None of these highways are major 4-lane interstate highways and they are not expected to present significant barriers or impediments to fisher movements.

Land ownership in potential reintroduction areas is dominated by the U.S. Forest Service, National Park Service, private landowners, and Washington Department of Natural Resources, however the proportions of these ownerships varies among potential reintroduction areas (Fig. 9, Table 9).

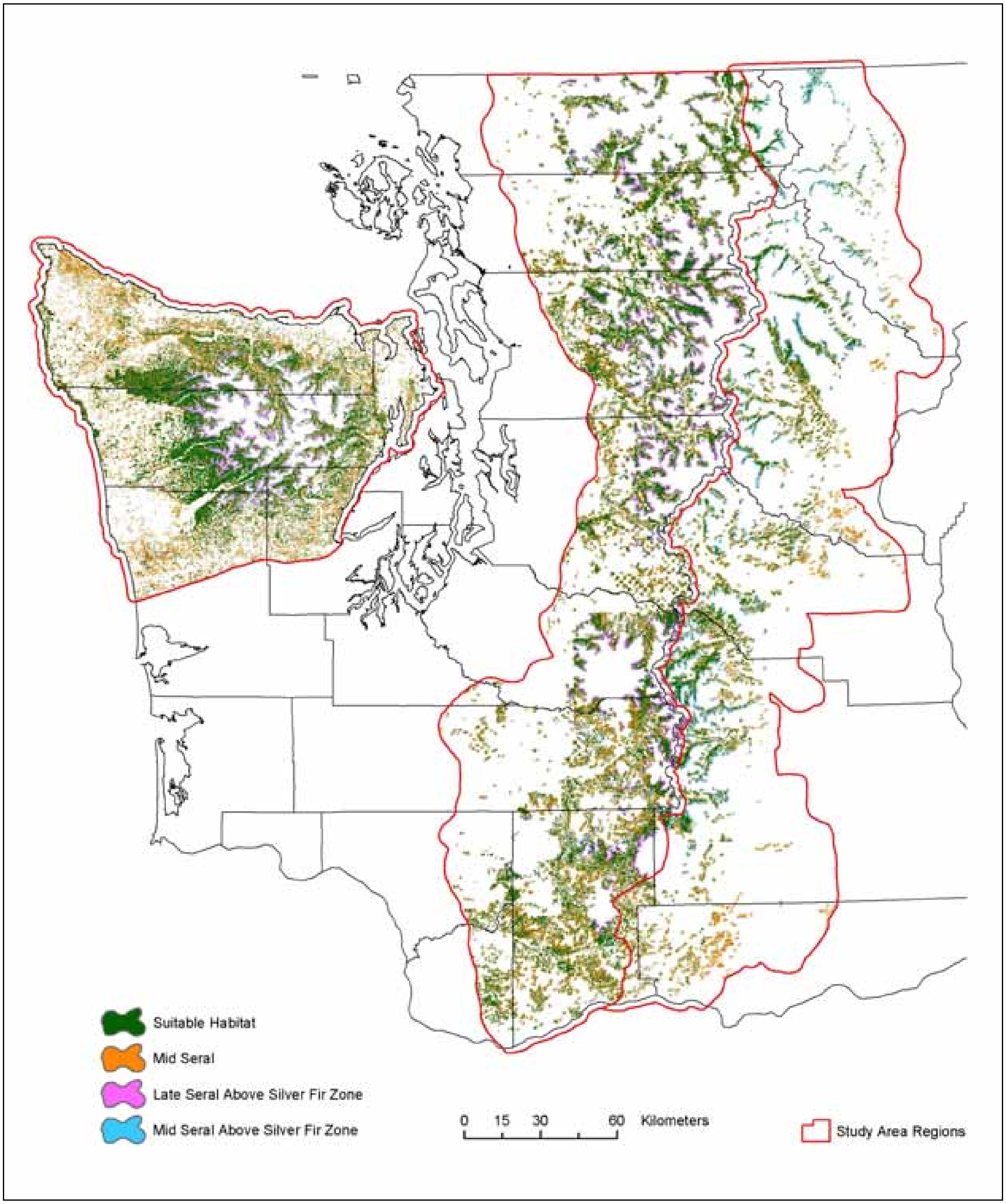

Figure 3. Potential fisher travel and foraging habitat (mid-seral, late and mid-seral above Pacific silver fir zone) within 500 m of large patches of suitable habitat in western Washington.

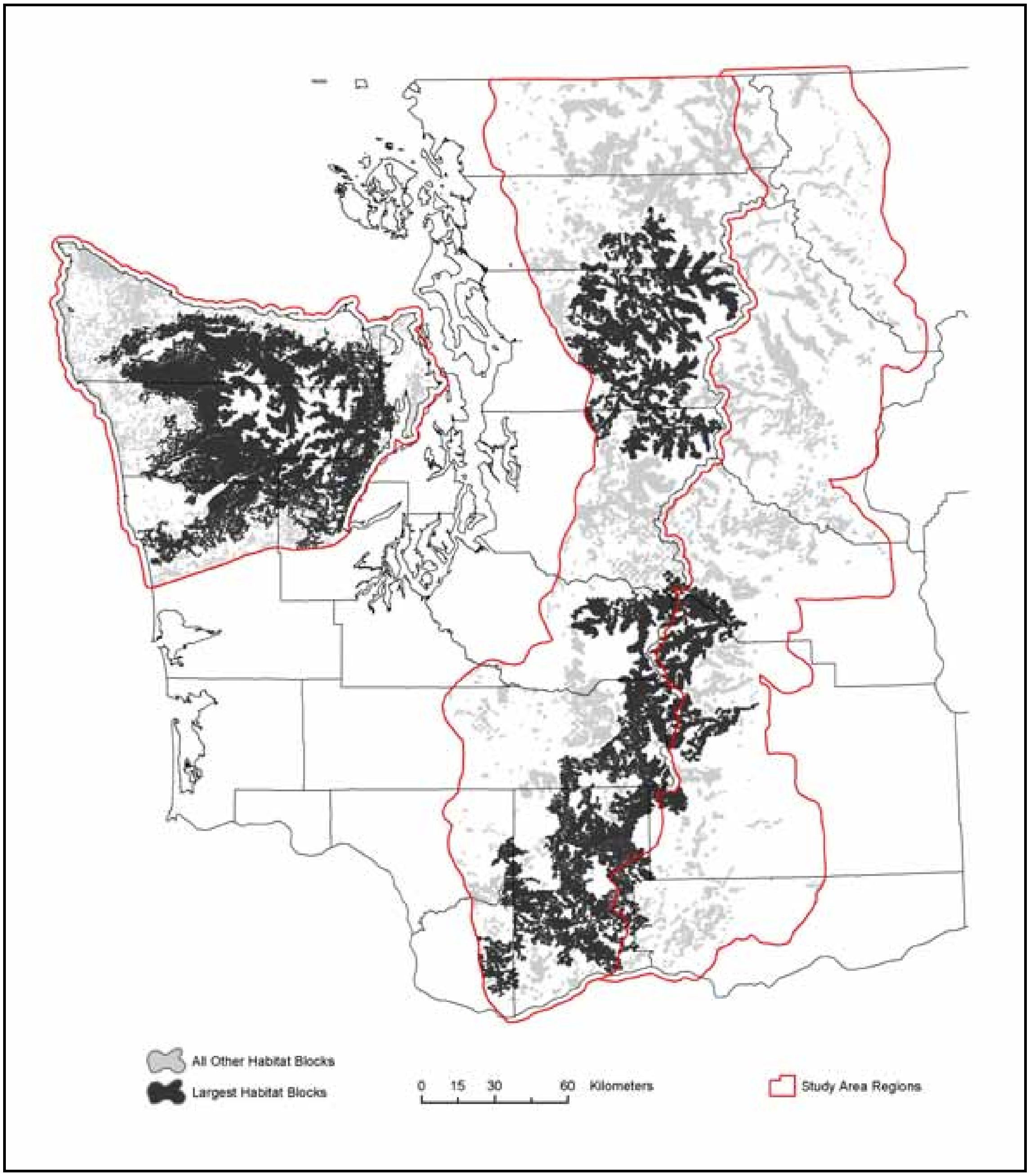

Figure 4. Largest blocks of interconnected fisher denning, resting, foraging and travel habitat in western Washington.

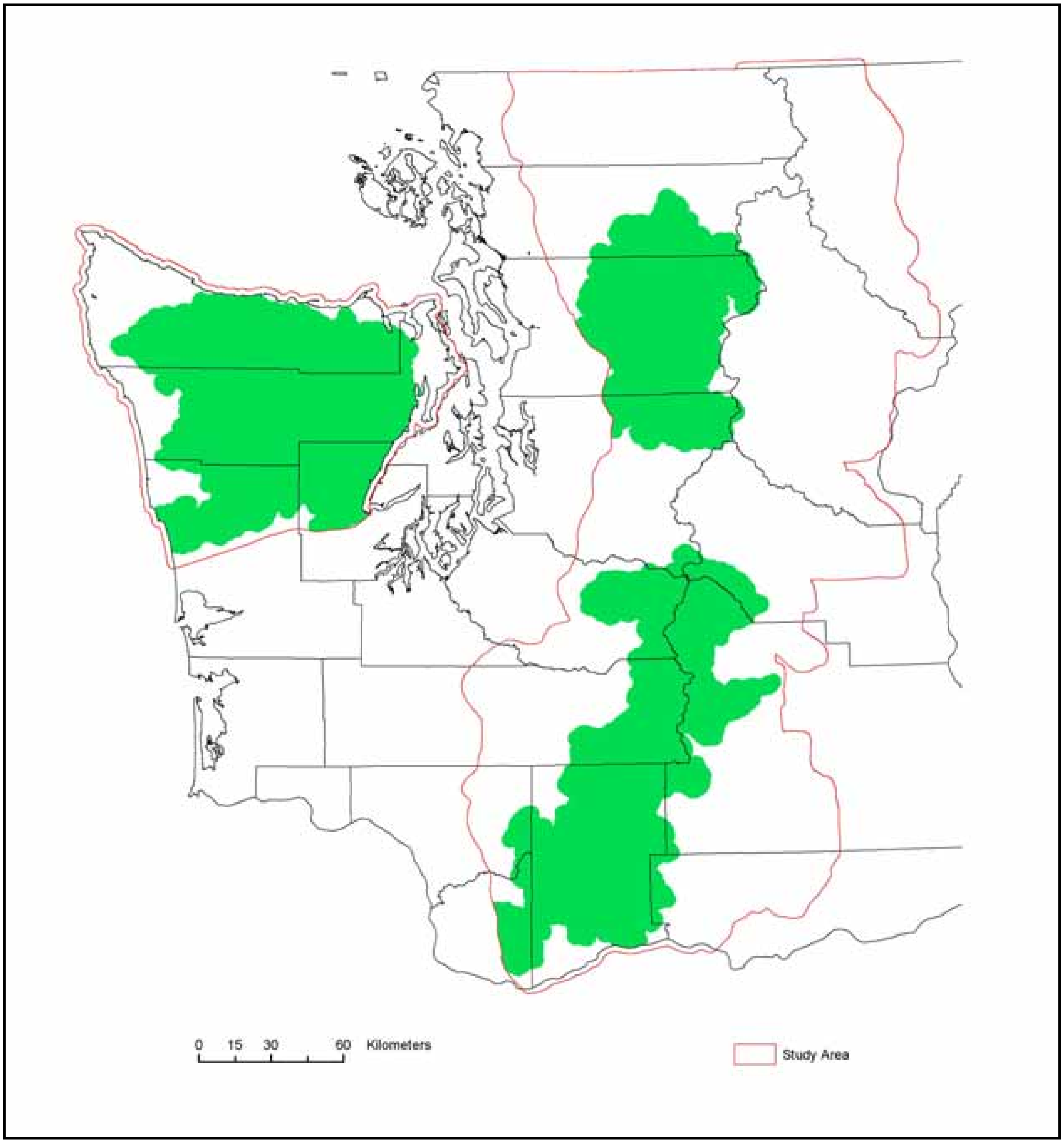

Figure 5. Potential fisher reintroduction areas in western Washington.

Table 10. Characteristics of 3 potential fisher reintroduction areas in Washington.

Characteristics	Olympic Peninsula	Northwestern Cascades	Southwestern Cascades
Area (ha)	930,496	557,807	949,640
Mean elevation in feet (range)	3942 (1-7,884)	5093 (114-10,072)	4110 (57-8,163)
Major roads (km)	346	94	121
Road density (km/km^2)[1]	1.22	1.17	1.41
Land ownership in hectares[2] (%)			
USFS	249,888 (27)	396,772 (71)	677,644 (71)
NPS	350,291 (38)		56,547 (6)
Private	163,229 (18)	93,700 (17)	127,551 (13)
WDNR	112,222 (12)	58,989 (11)	60,540 (6)
Tribal	51,418 (6)		25,717 (3)
Other	3,448 (<1)	8,346 (1)	1,641 (<1)
Forest habitat types in hectares (%)			
Suitable fisher habitat	229,376 (25)	129,722 (23)	212,496 (22)
Mid-seral below PSFZ[3]	148,362 (16)	71,829 (13)	184,685 (19)
Early-seral below PSFZ	170,428 (18)	75,752 (14)	182,380 (19)
Other forests below PSFZ	81,652 (9)	39,339 (7)	110,230 (12)
Late-seral above PSFZ	35,447 (4)	82,511 (15)	67,556 (7)
Mid-seral above PSFZ	13,967 (2)	25,769 (5)	56,274 (6)
Early-seral above PSFZ	10,889 (1)	20,752 (4)	34,300 (4)
Other forests above PSFZ	23,605 (3)	57,981 (10)	54,593 (6)

[1] Total length (km) of road categories from major highways to unimproved logging roads, divided by the total land area (km^2).
[2] USFS = U.S. Forest Service, NPS = National Park Service, WDNR = Washington Department of Natural Resources. Other lands owned by the Bureau of Land Management, the U.S. Department of Defense, U.S. Fish and Wildlife Service, Washington State Parks, Washington Department of Fish and Wildlife, counties, or cities.
[3] PSFZ = Pacific Silver Fir Zone.

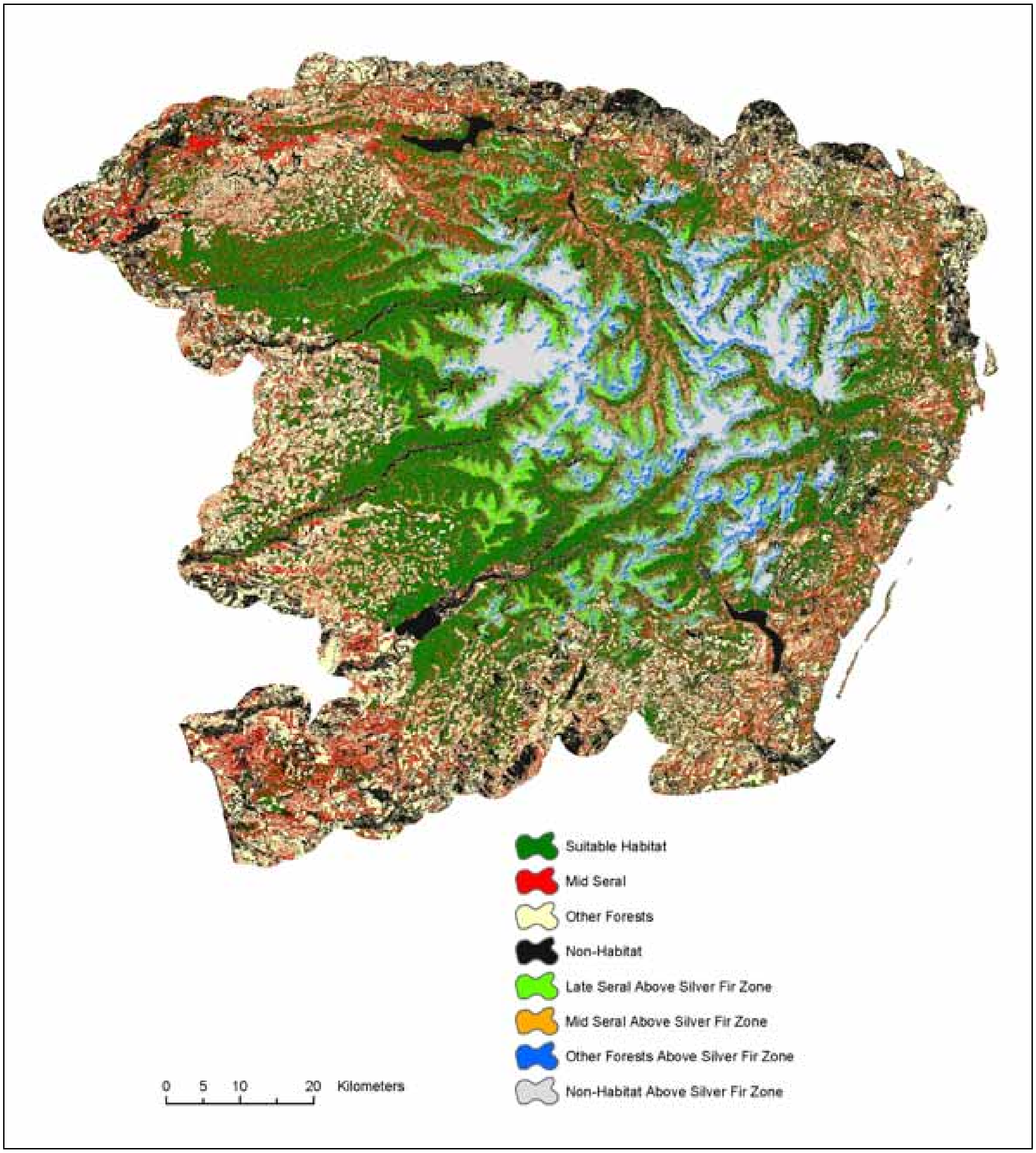

Figure 6. Suitable fisher habitat identified by a fisher habitat model and other cover types within the Olympic Peninsula potential reintroduction area in Washington.

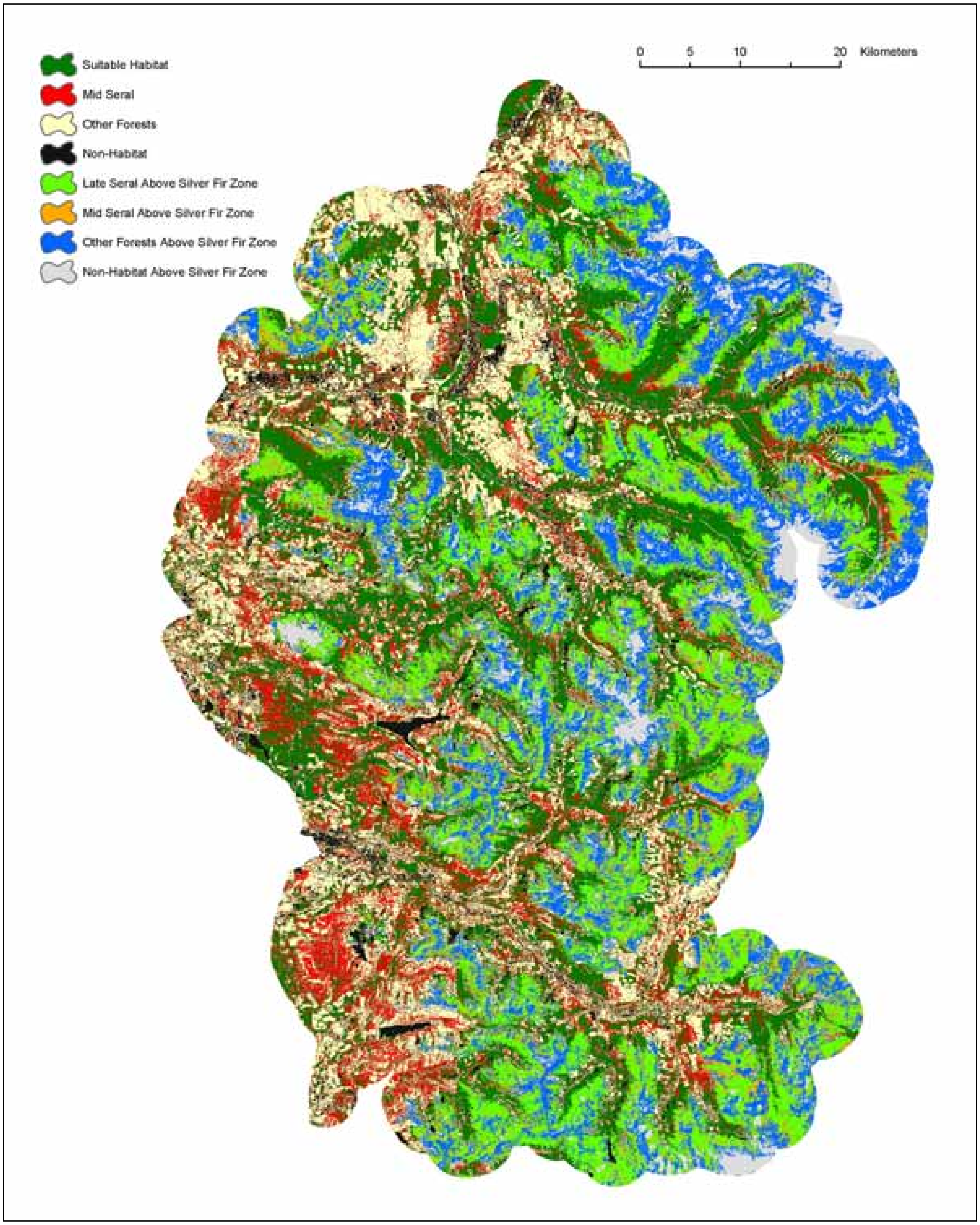

Figure 7. Suitable fisher habitat identified by a fisher habitat model and other cover types within the Northwestern Cascades potential reintroduction area in Washington.

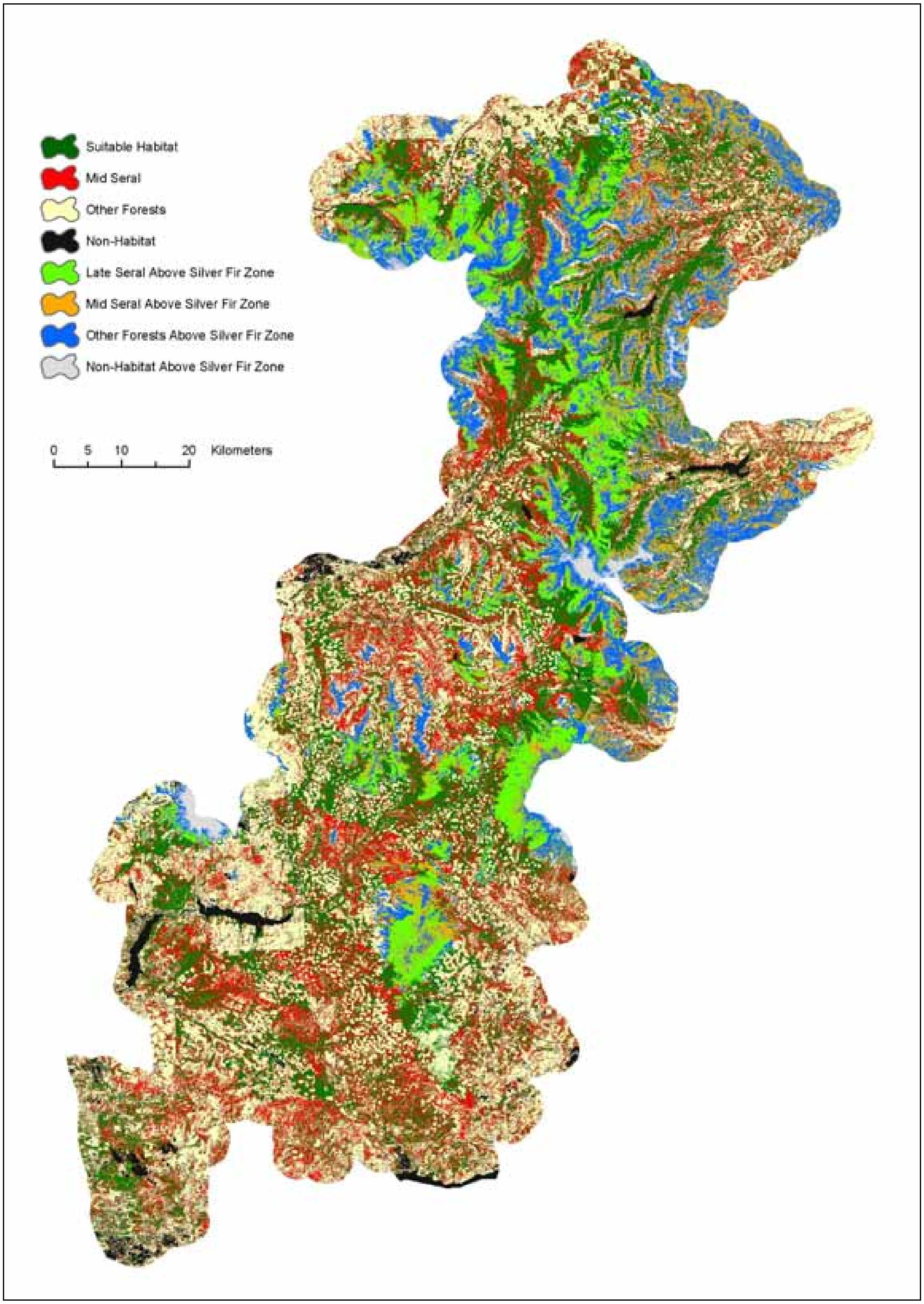

Figure 8. Suitable fisher habitat identified by a fisher habitat model and other cover types within the Southwestern Cascades potential reintroduction area in Washington.

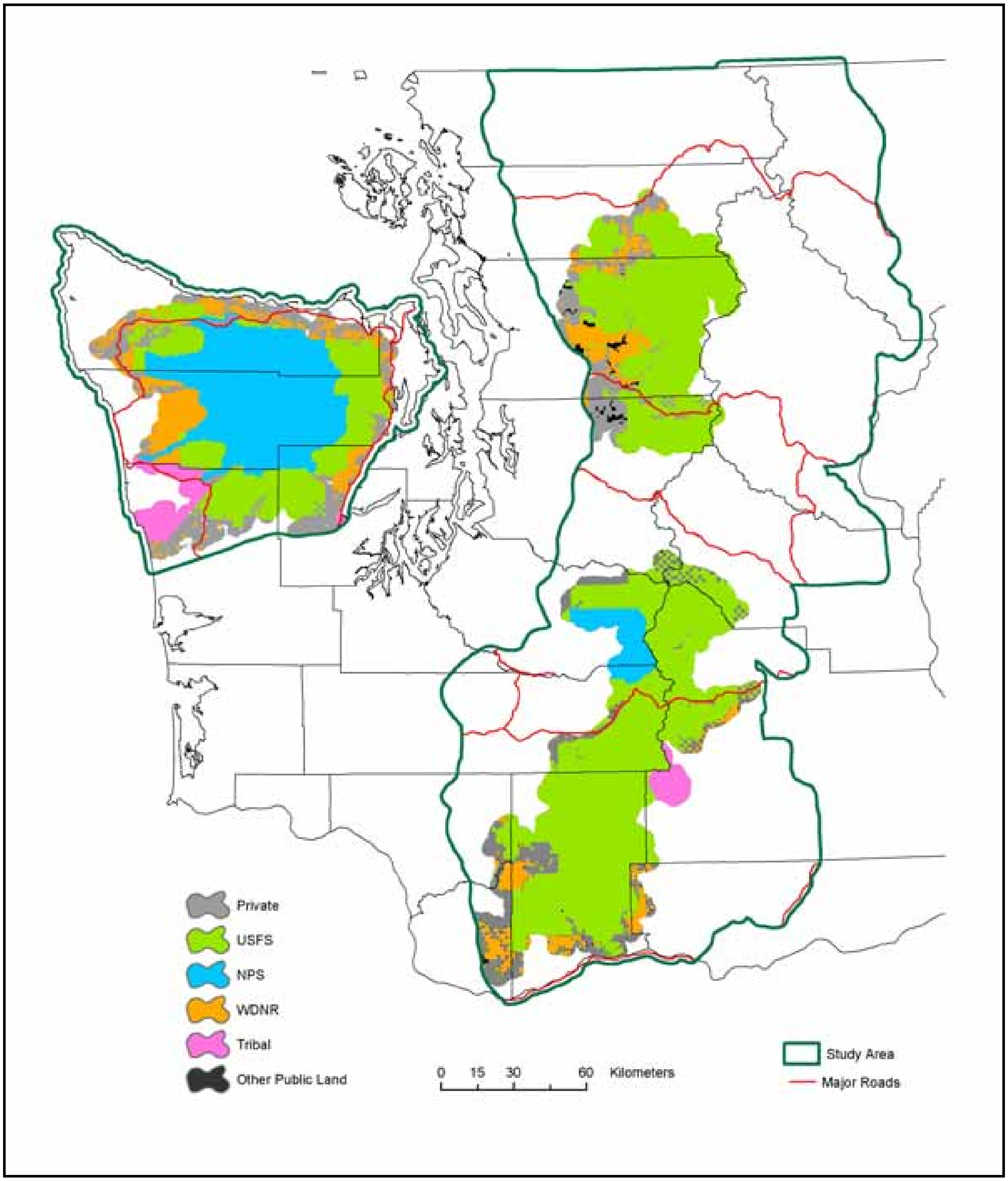

Figure 9. Land ownership and major roads within potential fisher reintroduction areas in western Washington.

Suitable fisher habitat occurs primarily on public lands (U.S. Forest Service, National Park Service and the Washington Department of Natural Resources) (Table 11). On the Olympic Peninsula, 97% of the suitable habitat is on public land (85% federal, 8% state), with 88% of federal lands occurring in the Olympic National Park and the Olympic National Forest. Eighty-seven percent of the suitable habitat in the Northwestern Cascades is on public land (74% federal, 13% state), and 91% of the suitable habitat in the Southwestern Cascades is on public land (85% federal, 6% state).

Land areas (contours) with concentrations of 1-25%, 26-50%, 51-75% and 76-100% suitable habitat were identified for each potential reintroduction area (Table 12, Figure 10). The size and distribution of the contours, especially those with greater concentrations of suitable habitat (i.e., 51-75% and 76-100%; Table 11), was different among potential reintroduction areas. The Olympic Peninsula has large blocks of highly concentrated suitable habitat, whereas both Cascade areas have a scattered distribution of small blocks of highly concentrated suitable habitat (Figure 10, Table 12). The Southwestern Cascades area has the greatest amount in 26-50% suitable habitat contours (Table 12); however, the contours appear to be distributed widely throughout the area in somewhat linear configurations (Figure 10). This configuration of contours was also seen in the Northwestern Cascades area and may be attributed to past logging of old growth forests at low elevations. In the Olympic Peninsula area, large areas of the 26-50% suitable habitat contours were located in close proximity to the large patches of highly concentrated (51-75, 76-100%) suitable habitat (Figure 10).

Table 11. Amount of suitable habitat (ha) by land ownership in potential reintroduction areas.

	Amount of suitable habitat in hectares (percent total)		
Land Ownership[1]	Olympic Peninsula	Northwestern Cascades	Southwestern Cascades
USFS	74,662 (33)	96,570 (74)	169,270 (80)
NPS	120,284 (52)		11,265 (5)
WDNR	19,208 (8)	16,727 (13)	13,559 (6)
Private	11,160 (5)	14,496 (11)	11,748 (6)
Tribal	3,830 (2)		6,321 (3)
Other	232 (<1)	1,929 (1)	333 (<1)
Total	229,376 (100)	129,722 (100)	212,496 (100)

[1] USFS = U.S. Forest Service, NPS = U.S. National Park Service, WDNR = Washington Department of Natural Resources. Other lands owned by the Bureau of Land Management, the U.S. Department of Defense, U.S. Fish and Wildlife Service, Washington State Parks, Washington Department of Fish and Wildlife, counties, or cities.

Table 12. Area (ha) within fisher suitable habitat concentration contours in potential reintroduction areas in Washington.

Potential Reintroduction Area	Concentration contours by % of suitable habitat			
	1-25%	26-50%	51-75%	76-100%
Olympic Peninsula	408,618	240,326	86,723	3,835
Northwestern Cascades	153,293	166,469	13,475	0
Southwestern Cascades	379,785	304,758	19,508	0

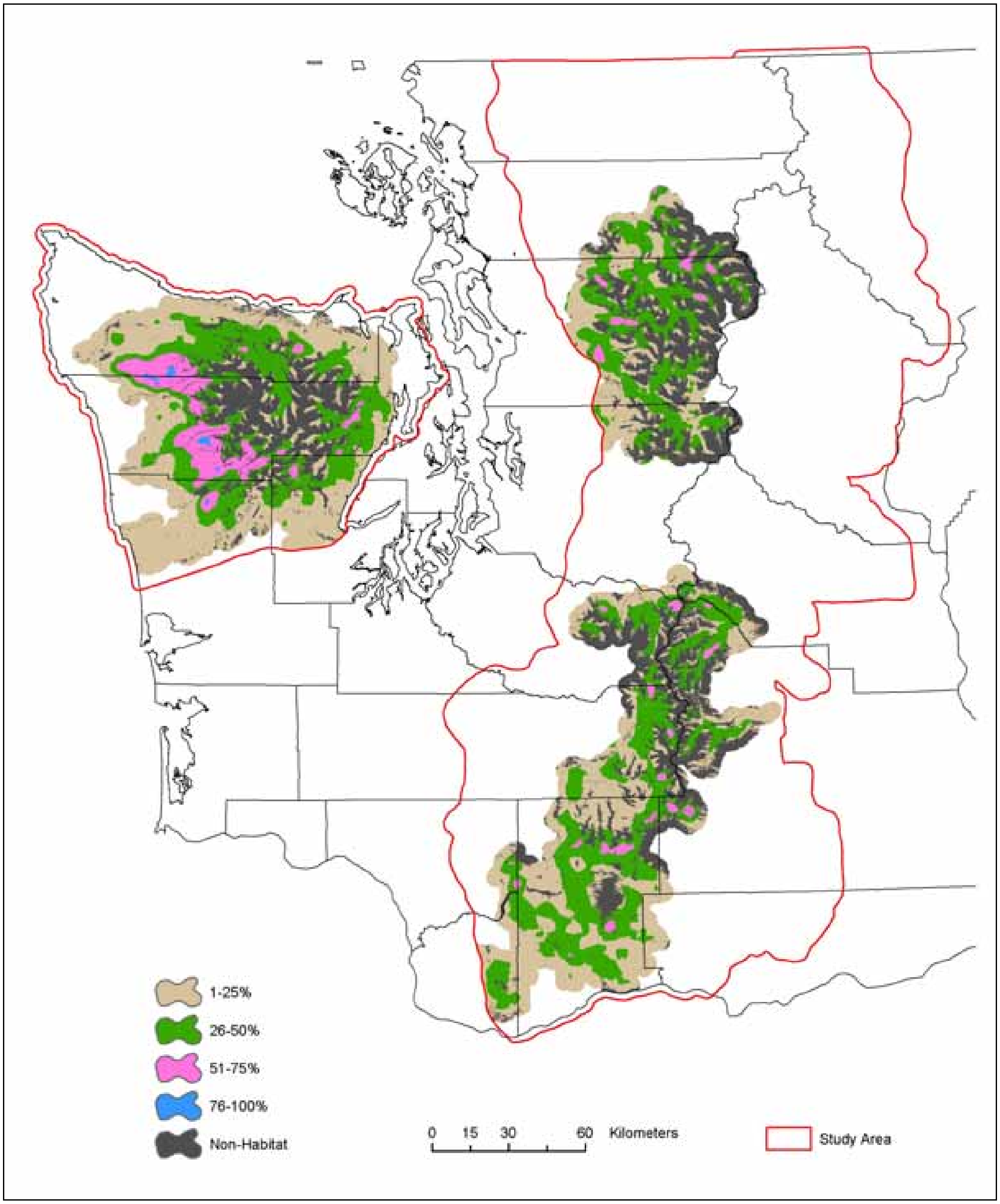

Figure 10. Percent concentrations of suitable fisher habitat within potential fisher reintroduction areas in western Washington.

Carrying capacity. PATCH simulations for each potential reintroduction area were run using 6 standard scenarios, and 3 alternative scenarios for sensitivity analyses (Table 13; Appendix B); these provided a range of estimates for each area. Model results from the 6 standard scenarios indicated that the Olympic Peninsula area could support a range from 82-102 females; the Southern Cascades from 25-60; and the Northern Cascades from 20-31 females. Increasing the number of females reintroduced from 30 to 60 to 100 resulted in increases in the number of females supported at year 20 for each area; however, the results at year 40 were relatively consistent for the Olympic Peninsula and Northern Cascades and variable for the Southwest Cascades. The median number of females supported in the Northwestern and Southwestern Cascades consistently decreased as simulations increased in length from 20 to 40 years (Table 13).

PATCH model outputs appeared to be insensitive to reducing the maximum dispersal distance from 50 km to 25 km, or increasing it from 50 km to 75 km (Table 13). In contrast, model outputs were quite sensitive to the use of lower habitat scores for suboptimal habitats. Model estimates decreased for all three areas under this scenario, however the estimates for the Cascades areas decreased by 75-100%, whereas the estimates for the Olympic Peninsula declined by <50% (Table 13).

Using the PATCH model to compare carrying capacities among potential reintroduction areas has the benefit of incorporating numerous demographic and behavioral characteristics of the fisher while evaluating the quality, quantity and spatial arrangement of habitats within each potential reintroduction area. However, it relies on a number of assumptions and data collected from other regions within the range of the fisher.

Using the area of suitable habitat within each potential reintroduction area (Table 11) and dividing that area by mean home range size for female fishers (24.95 km^2) resulted in a carrying capacity of 92 females on the Olympic Peninsula, 85 females in the Southwestern Cascades and 52 females in the Northwestern Cascades. This simple estimate of 92 females was within the range predicted by the PATCH model for the Olympic Peninsula (82-102); and resulted in predictions larger than the PATCH model for the Northern and Southwestern Cascades. Limitations of this method are that it is insensitive to the spatial distribution of suitable habitat (making it overestimate carrying capacities), and that it assumes all home ranges consist of 100% suitable habitat (making it underestimate carrying capacities).

Ranking potential reintroduction areas. Using the six measures to rank potential reintroduction areas, the Olympic Peninsula ranked first and was the most suitable potential reintroduction area (Table 14). The Southwestern Cascades area ranked second, and the Northwestern Cascades ranked third (Table 14).

Table 13. Median number of female fishers predicted by the PATCH model to be supported on potential reintroduction areas in the Olympic Peninsula, Northwestern Cascades, and Southwestern Cascades. Values were derived from 20 replicate simulations started with 30, 60 and 100 female fishers; male presence is assumed in the model.

Simulation specifications		Median number of female fishers supported		
Leslie matrices used[1]	Simulation length	Olympic Peninsula	Southwestern Cascades	Northwestern Cascades
30 Females Reintroduced				
Single Mean	20 years	82.5	36	25
	40 years	94	33.5	19.5
Six Random	20 years	81.5	35	17
	40 years	84.5	29.5	17
60 Females Reintroduced				
Single Mean	20 years	93.5	48	26.5
	40 years	92	36.5	21
Six Random	20 years	90.5	49.5	27.5
	40 years	87.5	25	21
100 Females Reintroduced				
Single Mean	20 years	98.5	59.5	31
	40 years	96	43.5	20
Six Random	20 years	102	54.5	30.5
	40 years	87	44.5	23.5
100 Females Reintroduced, additional specifications for sensitivity testing[2]				
Single Mean, 25 km maximum dispersal	20 years	101.5	57.5	32
	40 years	97.5	48	24
Single Mean, 75 km maximum dispersal	20 years	98.5	55	30
	40 years	96	40	22.5
Single Mean, low habitat scores	20 years	50.5	6	4.5
	40 years	51.5	0	0

[1] Two matrix scenarios were used in simulations. The single mean simulations were run with 1 Leslie matrix with mean values for survival and fecundity. The six random matrix simulations used four matrices of mean survival and fecundity values, one matrix with low values, and one matrix with high values; one of these six matrices was chosen at random each year of a simulation to incorporate environmental stochasticity.

[2] Three alternative simulations were run to test the sensitivity of the model to: a smaller maximum dispersal distance of 25 km, a larger maximum dispersal distance of 75 km, and lower habitat scores for suboptimal habitats.

Table 14. Ranking criteria, criteria quantities and ranks for three potential fisher reintroduction areas in Washington.

Ranking criteria	Olympic Peninsula Quantity	Olympic Peninsula Rank[1]	Southwestern Cascades Quantity	Southwestern Cascades Rank[1]	Northwestern Cascades Quantity	Northwestern Cascades Rank[1]
Suitable habitat (ha)	229,386	1	210,911	2	129,722	3
Amount of suitable habitat on public lands[2] (ha)	214,154	1	194,094	2	113,297	3
Amount of suitable habitat (ha) on Conservation status 1 lands[3]	131,030	1	50,242	2	31,986	3
Land area (ha) with a concentration of >50% suitable habitat	90,558	1	19,508	2	13,475	3
Proximity to the nearest existing population (km)[4]	~500	3	~300	2	~250	1
Carrying capacity estimate (number of females)						
PATCH model	82-102	1	25-60	2	17-31	3
Amount of suitablehabitat/ mean home range size	92	1	85	2	52	3
Rank total		9		14		19

[1] Ranks of 1 = best, 2 = second, 3 = third.
[2] Lands managed by the U.S. Forest Service, National Park Service and Washington Department of Natural Resources.
[3] Conservation status 1 lands include National Parks, National Monuments, and Wilderness areas (Cassidy et al. 1997).
[4] The nearest existing populations include a population in south central British Columbia (Weir 2003), ~250km north of the Northwestern Cascades area; and a population located in the southern Oregon Cascades (Aubry and Lewis 2003), ~300km south of the Southwestern Cascades area and ~500km south of the Olympic Peninsula area.

Discussion

The Olympic Peninsula area is the second largest potential reintroduction area (nearly 10,000 km^2) and is unique in several ways. This area has the most suitable habitat (Table 14), and contains large contiguous areas dominated by suitable habitat (Figs. 6 & 10). Much of this contiguous habitat is located in Olympic National Park, but some also occurs on Olympic National Forest and WDNR lands adjacent to the Park (Figs. 6 & 9). The Olympic Peninsula has the largest amount of suitable habitat on conservation status-1 lands (i.e., Olympic National Park and Olympic National Forest wilderness areas; Table 14); this is especially significant because these lands are protected from timber harvest and managed almost entirely to preserve or restore natural ecosystems. The reestablishment of native species is also a management objective of the National Park Service and Olympic National Park, as part of ecosystem restoration. The Olympic National Forest contains 746 km^2 of suitable habitat, and much of the Forest is managed as late-successional forest reserves, which protect and promote the development of substantial amounts of suitable habitat for fishers. Although overall forest management planning on the Olympic National Forest appears favorable to fisher recovery, the implications of implementing the Healthy Forests Restoration Act of 2003 on the National Forest remain unclear. Having the highest carrying capacity, the greatest amount of suitable habitat, and the most protection of suitable habitat, the Olympic Peninsula was the most suitable potential reintroduction area among the three considered.

The Olympic Peninsula area ranked third regarding proximity to existing fisher populations. There is little likelihood that fishers reintroduced on the Olympic Peninsula would disperse, colonize and establish a viable population in the Washington Cascades or elsewhere. However, reintroducing fishers on the Olympic Peninsula could potentially reestablish a viable population in Washington, and if successful, this population could serve as a source population for future reintroductions in the Washington Cascades.

The Southwestern Cascades area is the largest of the three potential reintroduction areas (also nearly 10,000 km^2, Fig. 9). Compared to the Olympic Peninsula, the Southwestern Cascades area has slightly less suitable habitat and slightly less suitable habitat on public lands. Less than 40% of the suitable habitat in the Southwestern Cascades is on conservation status-1 lands. The comparatively smaller amounts and higher elevations of National Park lands, and higher elevations of the USFS wilderness areas and the Mount Saint Helens National Monument explain the significant difference between the Southwestern Cascades and the Olympic Peninsula. The higher elevation of the conservation status-1 lands in the Southwestern Cascades makes many of these areas unsuitable as fisher habitat as defined by the habitat model. Suitable habitat in the Southwestern Cascades is also highly fragmented (Fig. 8), and there is less area with a high density of suitable habitat compared to the Olympic Peninsula (Fig. 10, Table 14). The amount and fragmentation of suitable habitat in the Southwestern Cascades resulted in carrying capacity estimates that were substantially lower than those estimated for the Olympic Peninsula (Table 14). When considering the habitat and carrying capacity criteria, the Southwestern Cascades was ranked second to the Olympic Peninsulas as a potential reintroduction area.

The Northwestern Cascades is the smallest of the three potential reintroduction areas, about half the size of the other two areas (Fig. 9, Table 9). It also has the highest mean elevation and highest maximum elevation (Table 9). The Northwestern Cascades is ranked third in each of the suitable habitat criteria and in both carrying capacity criteria (Table 14). This lower ranking is in part the result of the higher elevations in the area, which limited the amount of suitable habitat identified by the model (Table 14). The high elevation ridges also fragment the suitable habitat into relatively small, linear patches (Figs. 7 & 10). The small amount of suitable habitat on conservation status-1 lands (Table 14) was due to the absence of National Park lands within the area and the occurrence of USFS wilderness areas at relatively high elevations. Lower carrying capacity estimates for the Northwestern Cascades are the result of the small size of the area and the amount and fragmentation of suitable habitat. The proximity of the Northwestern Cascades to the extant population in British Columbia did not compensate for the low ranks it received for other criteria and it was ranked third as a potential reintroduction site.

The habitat model identified currently suitable fisher habitat on the Olympic Peninsula and in the Cascades. The habitat assessment, however, did not evaluate the increase in suitable habitat that is anticipated over the next 80 years in each potential reintroduction area as a result of the Northwest Forest Plan (U.S. Forest Service and USDI Bureau of Land Management 1994). This increase in habitat should be most pronounced in the Olympic Peninsula area, because the Olympic National Forest has much of its ownership designated as late-successional forest reserves (Holthausen et al. 1995). This anticipated increase in suitable habitat was not factored into carrying capacity estimates, but suggests that carrying capacity in each of the potential reintroduction areas would increase over time. Consequently, PATCH simulations that extended 20 or 40 years into the future may have underestimated carrying capacity of potential reintroduction areas.

As a spatially explicit population model, PATCH is sensitive to species-specific demographic characteristics, habitat associations, and habitat configurations. While a number of assumptions are made when using PATCH, comparisons of the carrying capacity estimates for the 3 potential reintroduction can be made because the data sources, parameters and assumptions were consistent in simulations for all three areas. Both carrying capacity estimators have their limitations, however their utility is not in their absolute value, but rather in how the individual estimates compare across potential reintroduction areas.

OLYMPIC PENINSULA

Because the Olympic Peninsula area was the highest ranked area for suitable fisher habitat, this area was the focus of additional assessments of future habitat conditions as well as potential impacts of a reintroduction on species of concern, furbearers and other species.

Future Habitat and Land Management on the Olympic Peninsula

Federal landownership. Late-seral forests in Olympic National Park are expected to provide stable amounts of suitable fisher habitat for the foreseeable future. On the Olympic National Forest, the amount of suitable fisher habitat is expected to increase as a result of forest management for spotted owls and marbled murrelets. Late-successional reserves are being managed by the Olympic National Forest to protect and enhance habitat for mature and old-growth forest associated species (Holthausen et al. 1995), including the fisher. Late-successional reserves comprise a significant proportion (66%) of the land allocation (Table 15) and the management goal for these lands is to accelerate the development of late-successional or old-growth structure. About half of the forests in late-successional reserves on the Olympic National Forest currently support late-successional forest (Holthausen et al. 1995). Entry into late-successional reserves for the purpose of timber harvest is not allowed within stands >80 years of age. Thinning (precommercial and commercial) is permitted to occur within stands less than 80 years of age, provided that the silvicultural treatment is beneficial to the creation and maintenance of late-successional forest conditions (K. O'Halloran, pers. comm.).

Holthausen et al. (1995) modeled patterns of spotted owl distribution and persistence on the Olympic Peninsula under the provisions of the Northwest Forest Plan. Growth of forest age classes was modeled for Olympic National Forest lands using maps of current owl habitat, successional stage of forests, and a model to characterize habitat succession. Late-successional reserves were the lands where regrowth of owl habitat is most likely to occur over time. Projections of forest stand development were based on the assumption that no precommercial or commercial thinning occurred in reserves, because there were no operational plans to do so at the time of the analysis. Habitat modeling projections indicated that a total of 37,300 hectares of additional late-successional habitat will be available on the Olympic National Forest within 80 years. Much of this habitat is concentrated in the northwestern corner and eastern edge of the Olympic National Forest (Fig. 11).

The increase in the amount of late-successional forest on the Olympic Peninsula is likely to have the most future benefit for fishers where it occurs adjacent or in close proximity to existing large patches of older forest. Therefore, regrowth of older forest in the northwestern and southwestern areas of the Peninsula is likely to provide the most benefit to fishers. Fishers may also be able to use smaller patches of older forest interconnected by areas of younger forest in the northeastern part of the Olympic Peninsula (Fig. 11).

Table 15. Land allocations under the Northwest Forest Plan on the Olympic National Forest.

Land allocation	Hectares (%)
Congressionally Reserved (Wilderness Areas)	36,380 (14)
Late-Successional Reserves	168,953 (66)
Administrative withdrawals[1] (24,280 ha)	
Adaptive Management Areas	50,585 (20)
Area available for ecosystem mgmt experiments (20,638 ha)	
Riparian Reserves (26,304 ha)	
Administrative withdrawals[1] (809 ha)	
Areas unsuitable for timber production (2,832 ha)	
Total	255,918

[1] Administrative Withdrawals include undeveloped and developed recreation areas, botanical areas, and select river corridors.

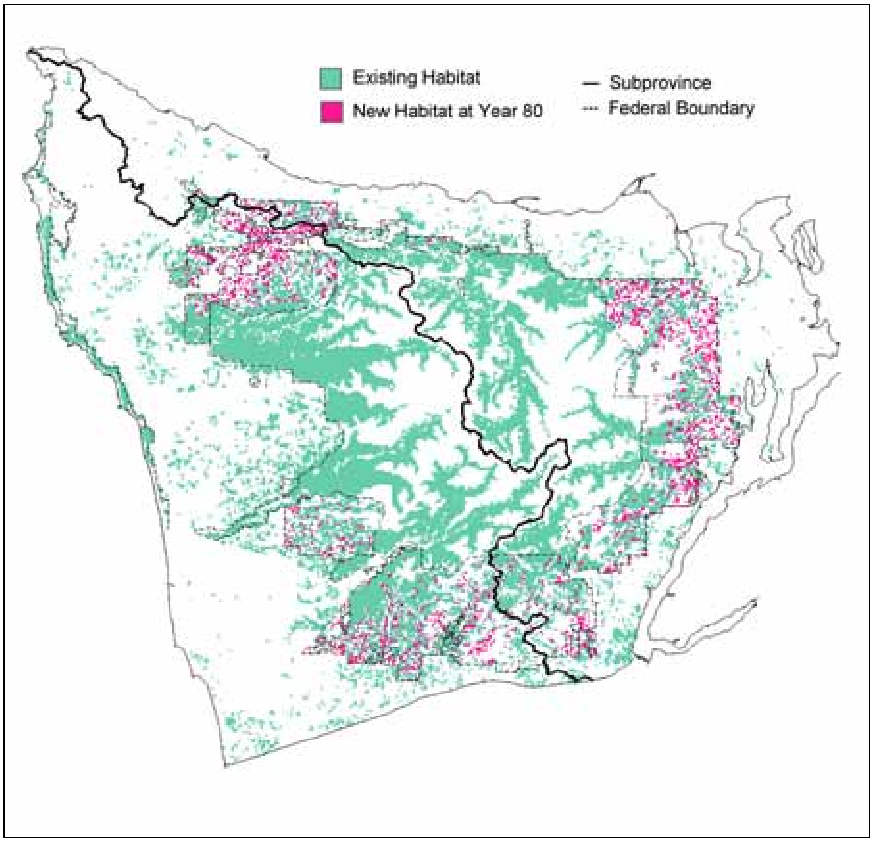

Figure 11. Existing late-successional forest habitat and projected distribution by year 2075 of new late-successional forest on the Olympic Peninsula based on forest growth modeling (source: Holthausen et al. 1995).

State and private landownership. Amounts of suitable fisher habitat are not expected to increase significantly on state and private lands on the Olympic Peninsula. Washington Department of Natural Resources developed a multi-species habitat conservation plan (HCP) to continue forest management activities on state trust lands while complying with the federal Endangered Species Act (Washington Department of Natural Resources 1997). The multi-species habitat conservation plan covers state trust lands managed by the Washington Department of Natural Resources within the range of the northern spotted owl, and the fisher is covered under this plan. Of the three planning units covered by the HCP on the Olympic Peninsula, the Olympic Experimental State Forest Unit, located in the northwest corner of the Peninsula, has the greatest potential to benefit fishers. The riparian, spotted owl and marbled murrelet conservation strategies for the experimental forest would likely result in some increase in older age stands after 80 years, which could benefit fishers. The spotted owl conservation strategy sets a minimum threshold of at least 20% of old forest in each planning unit within the experimental forest. As of September 1997, 19% of DNR's land base on the experimental forest was in stands >100 yrs old (old-forest habitat). Old-forest habitat is currently at or above the 20% threshold for several landscape planning units that are adjacent to the Olympic National Park between the Hoh and Queets Rivers. Units currently below old forest habitat threshold levels (20%) are predicted to attain threshold levels over the next 20-80 years. Harvest activities planned for any landscape unit must maintain the old-forest habitat threshold of 20% (Washington Dept. Natural Resources 1997).

The long-term conservation strategy for marbled murrelets on DNR-managed lands has not yet been completed, but in the interim, harvest is deferred for most potential murrelet habitat. The strategy will likely entail preservation of some murrelet nesting habitat and will increase the amount of late successional forest available to other old-forest dependent species, including the fisher. In addition, forest stand management will increasingly focus on retention of elements of existing stands to promote diversity within each stand and development of spotted owl habitat at earlier ages than might be achieved without retention. Large structural elements to be retained include live trees (5 retained/acre harvested) and snags (3 retained/acre harvested) with attributes, such as cavities, important to wildlife. These structural features are likely to be beneficial to fishers as natal and maternal dens and resting sites. It's unknown whether fishers will be able to use these landscapes in which older forest patches are concentrated along riparian areas and unstable slopes within a matrix of younger, managed forest.

In May, 2001 the Forest and Fish Law was enacted in Washington to provide for the protection of aquatic resources (namely listed salmon stocks) and to ensure compliance with the Endangered Species Act and Clean Water Act on private and state forest lands. The only aspect of this law that may be beneficial to fishers is a 50-foot buffer zone adjacent to the water of fish-bearing rivers and "shorelines of the state" (WAC 222-16-030). No timber harvest or construction is permitted within the zone along shorelines or along segments of natural waters that contain fish habitat. These narrow strips of forest may function as travel corridors and foraging areas for fishers if they develop into mature age classes with structural complexity near the forest floor. Over time, large live trees, snags and logs with cavities are likely to occur within the strips and may be suitable as dens and resting sites for fishers. However, the no-entry 50-ft buffer has a limited extent on non-fish bearing rivers with a maximum upstream distance of 500 ft. Therefore, narrow strips of older, more structurally complex forests on state and private lands will be limited largely to fish-bearing rivers and shorelines at lower elevations. It's unknown whether fishers would use riparian areas in intensively managed forests lacking these buffers. Upland areas on intensively managed tree farms are unlikely to provide functional fisher habitat.

Potential Prey

Based on fisher food habit studies conducted in Idaho, Montana, and Oregon, anecdotal reports of prey remains in fishers trapped in Washington, and information on bobcat food habit studies in western Washington, it appears that mountain beavers, snowshoe hares, Douglas' squirrels, birds, small mammals, and ungulate carrion may be important foods of fishers if they were reintroduced to Washington. Mountain beavers and snowshoe hares are likely to be the most important prey species. They are likely to be more abundant in managed second growth forests on the Olympic National Forest and in riparian areas, talus slopes and recently burned areas of Olympic National Park. Small mammals and Douglas' squirrels are likely to be secondary in importance. Birds may be important during spring, summer, and autumn periods (Zielinski et al. 1999, Aubry and Raley 2002), and fruits are likely to be more important during autumn and winter periods (Scheffer 1995, Zielinski et al. 1999). These and other potential prey species that occur in low- and mid-elevation coniferous forests on the Olympic Peninsula are likely to provide a reintroduced fisher population with a diverse prey base.

Impacts to Species of Concern

Northern spotted owl. The northern spotted owl is a state endangered species and a federal threatened species in Washington. Fishers use similar forest types and prey on some of the same species as northern spotted owls. Thus, there is potential for fishers to compete with spotted owls on the Olympic Peninsula. Northern spotted owls nest, roost, and forage in late seral forests characterized by high canopy cover and complex structure (Forsman et al. 1984, Gutiérrez et al. 1995, Hershey et al. 1998) and prey on northern flying squirrels, bushy-tailed woodrats (*Neotoma cinerea*), mice (*Peromyscus* spp.), voles (*Microtus/Clethrionomys* spp.), and snowshoe hares (Gutiérrez et al. 1995). In the northern portion of the owl's range, northern flying squirrels are the most common prey in the diet (32% freq. of occurrence), with red tree voles (*Phenacomys longicaudus*; 12%), mice (10%) and voles (10%) occurring less frequently, and lagomorphs occurring rarely (4%) in the diet (Gutiérrez et al. 1995). In contrast, fishers prey on northern flying squirrels infrequently, and deer mice occur at similar frequencies in the diets of both spotted owls and fishers. However, this is not likely to result in competition because fishers can switch to a number of alternate small mammalian prey (e.g., voles, shrews, chipmunks) should deer mice populations decline. Forests on the Olympic Peninsula support a diverse small mammal prey base, and while there are some prey species in common, most of these species are only rarely used by either fishers or spotted owls. The greater flexibility in use of small mammals by fishers suggests little opportunity for competition.

There is potential for fishers to prey on spotted owls. Although predators of spotted owls are primarily forest raptors, fishers have been documented resting in spotted owl nest trees (Gutiérrez et al. 1995). Pre-fledged young are likely at greater risk of fisher predation than adults. Prematurely fledged young may also be vulnerable to fisher predation when on the ground. However, the widely dispersed nest sites of owls throughout the Olympic Peninsula and the low population densities of fishers would make encounters between the two species unlikely. Fishers may prey on barred owls (*Strix varia*) (Mazur and James 2000), a known competitor of the spotted owl in Washington. For these reasons, reintroducing fishers to the Olympic Peninsula is not expected to adversely affect spotted owl populations to the degree that it would impede recovery.

Marbled murrelet. The marbled murrelet is a state and federal threatened species in Washington. In Washington, marbled murrelets nest in old-growth forests and mature forests with old growth structures (large trees with large limbs or platforms suitable for nesting) within 80 km of the Pacific Ocean (Nelson 1997). Fishers are associated with the same forest types as murrelets, therefore there is potential for fishers to climb trees and predate murrelet nests during the nesting period. Although predation is the main cause of marbled murrelet nest failure (57% in WA, OR, and CA), nest predators are primarily corvids and great horned owls (*Bubo virginianus*) (Nelson and Hamer 1995). Mammalian species have not been reported to predate marbled murrelet nests, but fishers are considered a potential predator (Nelson and Hamer 1995). Arrival and departure of adults at nests during periods of low light, cryptic coloration of adults and chicks, and general lack of movement while at the nest, suggest that detections of murrelet nest sites by fishers is unlikely. Fledging birds may become grounded during initial flights from the nest to the ocean and therefore could become vulnerable to fishers. However, fishers have large home ranges and encounters with grounded murrelets would be rare. Therefore, reintroducing fishers to the Olympic Peninsula is not considered to have potential detrimental impacts on murrelet populations that would impede recovery.

Northern goshawk. The northern goshawk (*Accipiter gentilis*) is a state candidate species and as such could become state-listed as endangered, threatened, or sensitive. Although formerly designated a Category II species of concern by the U.S. Fish and Wildlife Service, that designation was eliminated in 1996. In response to a 1991 petition to list the northern goshawk in the western United States, the Service found in 1997 that the petition was not warranted (U.S. Fish and Wildlife Service 1998).

Fishers use similar forest types and prey on many of the same species as northern goshawks. In the western United States, northern goshawks prefer to nest in stands with dense canopy closure (60-90%) and late-seral forest structures (i.e., trees >53 cm dbh) (Daw and DeStefano 2001, Squires and Reynolds 1997). Goshawks also appear to be sensitive to forest fragmentation, preferring to nest in large continuous tracts of forest and seldom using isolated tracts (Squires and Reynolds 1997). Foraging sites occur in moderately dense, mature forests (Squires and Reynolds 1997). Both goshawks and fishers eat many of the same prey species (e.g., ground and tree squirrels, snowshoe hares, grouse, corvids, woodpeckers, and large passerines), and diet composition changes seasonally, presumably in response to abundance or availability of prey (Squires and Reynolds 1997). Thus, it appears that when food is limiting there is the potential for competition to occur between these two forest predators. However, the diverse diet and demonstrated ability of both species to switch to alternate prey, suggests that competition between the two would be minimal.

Goshawks could be vulnerable to predation by fishers. A study in northeastern Wisconsin found that following fisher reintroduction to the state, fisher predation on goshawks had resulted in reduced nest success and increased adult mortality, and had led to unstable goshawk populations (Erdman et al. 1998). From 1971 to 1981 goshawk nest success was 94% and failures of active nests occurred in only 3 of the 11 years there were active nests. However, nest success decreased to 64% during the period from 1982 to 1992, and active nests failed in every year. The significant decline in goshawk productivity during the period from 1982-1992 was attributed to greater nest failure in northern Wisconsin where fisher populations were recovering (Erdman et al. 1998). Turnover rates of adult females at nests doubled to 40% and adult female goshawks killed by fishers were recovered at nests. Factors that may have contributed to fisher predation of goshawk nests and adults in Wisconsin included researcher activity near nests, reduced nest concealment, and concentration of fishers and goshawks in fragments of suitable habitat. While researchers did not climb nest trees to band young, researchers may have left human “scent trails” that may have directed fishers to nest sites. A recent defoliation event, caused by forest tent caterpillars, reduced deciduous cover around nests, and may have made nests more visible to fishers from the ground. Also, clear-cut logging of large tracts of forest in the landscape where goshawks nested could have concentrated both fishers and goshawks into remaining fragments of suitable forest, increasing the number of encounters between fishers and goshawks. With the exception of the Wisconsin study, there have been no records in the literature of fishers killing goshawks. Other mustelids have been documented killing nestlings (e.g., wolverine) and adults close to the ground (e.g., marten) (Squires and Reynolds 1997). Therefore, it is difficult to determine if researcher activity, defoliation events and logging in the Wisconsin study could have biased nest predation results. Since remains of accipiters have not been documented in fisher diets (Powell 1993), fishers occur at low population densities, and both goshawks and fishers coexist in similar habitats in the West, fisher predation on goshawks nests and adults is likely to be a rare occurrence. Therefore, reintroducing fishers to the Olympic Peninsula is unlikely to have a significant adverse impact on goshawk populations.

Impacts to Furbearers

American marten. The American marten is classified as a furbearer in Washington and appears to be rare on the Olympic Peninsula. Surveys for martens conducted between 1991-1992 along roads, primarily within Olympic National Forest (Jones and Raphael 1991, Sheets 1993), recorded only two marten detections on the east side of the Olympic Peninsula, and no detections on the west side (Zielinski et al. 2001). Surveys conducted in the Park in 2000-2002 also failed to detect martens (P. Happe, pers. comm.).

Fishers and martens utilize similar habitats and prey species and both forest carnivores historically occurred sympatrically on the Olympic Peninsula (Lewis and Stinson 1998, Zielinski et al. 2001). The mechanisms for coexistence of American martens and fishers, sympatric throughout much of their ranges, remain largely unknown (Martin 1994). Martens are closely associated with late-seral, mesic coniferous forest with complex physical structure near the ground (Jones and Raphael 1990, Buskirk and Ruggiero 1994); forests also used by fishers (Powell and Zielinski 1994, Lewis and Stinson 1998). Moreover, fishers and martens utilize many of the same prey species including voles, tree squirrels, and hares (Clem 1977, Raine 1987, Buskirk and Ruggiero 1994, Martin 1994, Powell and Zielinski 1994). Thus, these species overlap in two dimensions of the niche, namely forest cover and food habits, and therefore have the potential to compete. Studies of winter food habits have tried to elucidate the degree of overlap of sympatric fishers and martens. In Ontario, Clem (1977) found extensive overlap in the winter diet of martens and fishers, a period when competition is most likely to occur due to a reduced diversity and abundance of prey and suitable cover. Raine (1987) evaluated dietary overlap between sympatric martens and fishers in Manitoba, and found that both species relied heavily on snowshoe hares. While these studies demonstrated significant overlap in diet during winter, there was no indication that food was limiting (Raine 1987).

Partitioning in other niche dimensions, such as habitat use, spatial patterns, and activity patterns may allow these two species to coexist. In some areas of the West, martens occur at higher elevations than fishers, minimizing the potential for competitive interaction (Raine 1987, Powell and Zielinski 1994). In comparison to the fisher, the marten engages in more arboreal and subnivean activity and eats smaller prey (Raine 1987, Buskirk and Ruggiero 1994). Martens are small enough to specialize in hunting small mammals, such as red-backed voles and other small mammals beneath the snow. In contrast, fishers eat larger prey, such as snowshoe hares, and make exclusive use of porcupines (Clem 1977, Buskirk and Ruggiero 1994, Martin 1994). The winter period may be the most likely time for competition to occur, but it may be of such short duration that it does not result in competitive exclusion of martens by fishers (Clem 1977). Martens occur sympatrically with other mustelid species and competitive interactions among them have not been reported

(Buskirk and Ruggiero 1994). However, marten remains have been found in small amounts (5%) in fisher feces where the species are sympatric (Raine 1987). Thus, it's difficult to determine what potential impact, if any, a reintroduction of fishers to the Olympic Peninsula could have on the resident marten population because there is no information on food habits or spatial partitioning of habitat for these two species in Washington. The sympatry of fishers and martens in much of their range suggests that fishers may not have an adverse affect on marten populations. Given that martens are rare on the Olympic Peninsula, reintroducing fishers could potentially adversely affect martens. However, historical coexistence of these species suggests a reintroduction would have a limited impact on marten populations.

Bobcat. Bobcats and fishers were sympatric in western Washington, and probably utilized some of the same prey species (Table 2, Sweeney 1978, Knick et al. 1984). Reintroduction of the fisher has the potential to adversely impact bobcat populations through competition. Fishers were reintroduced to Wisconsin during the 1950s and 1960s, and their populations expanded in an area that supported a bobcat population. Gilbert and Keith (2001) evaluated the evidence for consumptive, territorial, and encounter competition between bobcats and fishers during 1991-95 in Wisconsin. Bobcats did not alter their diets in the presence of fishers, but fisher diets contained a higher proportion of small mammals and less deer when bobcats were common, suggesting bobcats interfered with fisher consumption of deer. There was no evidence of territorial competition between the species based on overlapping home ranges of fishers and bobcats. Encounter competition or predation by fishers was inferred based on an increase in bobcat kitten mortality and reduction in bobcat population growth. However, competition between bobcats and fishers was weak and Gilbert and Keith speculated that stable coexistence between the species occurred (Gilbert and Keith 2001). These findings suggest that if fishers were reintroduced to Washington, it may result in a similar stable coexistence between populations of bobcats and fishers.

In summary, a fisher reintroduction to Washington is not likely to adversely influence recovery of State or Federal species of concern or bobcat and marten populations. Fishers hunt predominantly by scent and hearing and therefore detection of active nests of murrelets, goshawks and spotted owls in the upper tree canopy are probably rare. In addition, individual fishers travel over a large home range in search of prey, making encounters between these species a rare occurrence. Although marten populations are thought to be very low on the Peninsula, co-existence of these two species where they are sympatric in other parts of their range suggests that fishers will not adversely affect marten populations. Following a reintroduction of fishers to the southern Cascade Range in Oregon, there is no evidence that fishers are adversely affecting spotted owls, marbled murrelets, northern goshawks, or bobcats in the Oregon study area (K. Aubry, pers. comm.).

Impacts to Other Species

Fishers were reintroduced in many states and provinces in North America during the late 1950s and 1960s to control porcupines that were damaging commercial timber (Powell and Zielinski 1994) and resulted in some success in reducing porcupine populations. Reestablishment of fishers on the Olympic Peninsula could reduce both porcupine and mountain beaver populations.

Potential Predators

Potential predators of fisher include the red fox (*Vulpes vulpes*), coyote, bobcat, lynx, mountain lion, wolverines, other fishers, and golden eagles (*Aquila chrysaetos*). Because snowshoe hares are an important prey species in the diet of bobcat, lynx and fishers in certain areas, competitive interactions between these forest predators may result in predation of fishers. Competition with these species does not appear to be a limiting factor in regions where fishers have been reintroduced (Powell and Zielinski 1994, Gilbert and Keith 2001). Predation of fishers can be high when individuals are translocated to regions that contain a different predator community than that of the source population. Fishers translocated from the Great Lakes States to northwestern Montana experienced a high rate of predation (9 of 32; 28%) by mountain lions, coyotes, wolverines, lynx, and golden eagles (Roy 1991). Roy speculated that fishers were possibly naive and unwary of the new predator community in Montana because these predator species are absent or rare in the Great Lakes States. Increased movements of fishers associated with breeding activity following release in winter may also have made them more vulnerable to predation. Predation by mountain lions, coyotes, lynx, wolverines, and golden eagles on fishers may have also been the result of competition for resources, such as snowshoe hares (Roy 1991). Fishers could become more vulnerable to predation when deep snow at higher elevations concentrates some forest predators at lower elevations. Neither lynx nor wolverines occur on the Olympic Peninsula, where the most suitable reintroduction area was identified and golden eagles, which occur in small number on the Peninsula, would not be expected to be a threat to fishers. It is unknown how land-management activities in forests surrounding Olympic National Park may have altered the local predator complex, compared to when indigenous fishers inhabited the area historically, and what potential effect this could have on fisher recovery.

SOURCE POPULATION

One of the goals of the feasibility study is to identify the most appropriate source population of fishers to reintroduce into Washington. Two primary factors influence the choice of a suitable source population for reintroduction: genetic similarity to the historical extirpated population and conservation status of the potential source population. The most suitable source populations from a genetic standpoint, may not be available for translocations because of small population size and protected status. For source populations that are genetically suitable and not afforded protected status, the potential effect of removing animals from the source population also needs to be assessed.

Three studies have looked at the genetic relationships among historical Washington fisher specimens and fishers in potential source areas in British Columbia, Alberta and California (Kyle et al. 2001, Drew et al. 2003, Warheit 2004). No research has been published that comprehensively addresses the historical genetic character of fishers in Washington. Drew et al. (2003) provide the only genetic data specific to fishers from Washington. They showed that mtDNA control region Haplotype 4 and possibly Haplotype 1 were present in historical museum skins of fishers from Washington (n = 3). Using mtDNA control sequences, Drew et al. (2003) demonstrated a close genetic association between native British Columbia and California populations. Modern specimens from these populations shared some of the same haplotypes with historical museum specimens from Washington State. Kyle et al. (2001), using microsatellite markers, showed that genetic distances were relatively small between western Alberta and British Columbia populations, but they did not examine populations in the western States.

Fishers in Idaho, Oregon, and California are not available for translocation to Washington because of their protected status, but they may be available from British Columbia or Alberta. In British Columbia fishers are a red-listed species, indicating a high level of conservation concern. However, they are not listed under British Columbia's Wildlife Act, and are classified as a furbearer with a closed season. Alberta has indicated that they have sufficient numbers of fishers to provide for a reintroduction in Washington (B. Treichel, pers. comm.).

Because Alberta is a potential source of fishers for reintroduction into Washington, it was important to confirm the close relationship between Alberta and British Columbia fishers (and indirectly between Alberta and Washington fishers). WDFW obtained 48 tissue samples from fishers trapped in Alberta and sequenced a 309 basepair section of the control region following the methods of Drew et al. (2003) and Vinkey (2003) (Warheit 2004). The purpose of this analysis was to: (1) determine whether western Alberta fishers share haplotypes with modern specimens from other populations in the Pacific states (Drew et al. 2003) and with museum specimens from Washington (Haplotype 4), and (2) evaluate regional population relationships. Alberta data were combined and analyzed with the data from Drew et al. (2003) and Vinkey (2003) using a series of phylogenetic and population statistical packages (Warheit 2004). Alberta samples were divided into a western and eastern population, following Kyle et al. (2001). Three museum skin samples from Washington (Drew et al. 2003) represented the pre-extirpation Washington sample.

Haplotype frequencies provide information on common ancestry in fisher populations (Appendix C). Haplotype 1 appears to be ancestral to many haplotypes that occur in populations in Canada, the Great Lakes States, and the Pacific coastal states (Warheit 2004). Therefore, its possible occurrence in Washington fishers does not provide much information about relationships among populations that share this haplotype. Haplotype 4, found in the Washington sample, is restricted to British Columbia and one historical sample from California. Haplotype 4 is closely related to Haplotype 6, and the former appears to be ancestral to the latter. Haplotype 6 is restricted to British Columbia and the western Alberta population. Although limited, these data and a population tree (Fig. 12) indicate that prior to their extirpation, Washington fishers were most closely related to British Columbia fishers (Warheit 2004). The presence of Haplotype 4 in both Washington and British Columbia populations and the close relationship between Haplotypes 4 and 6 suggest this relationship.

Relationships between Washington fishers and populations in California and western Alberta are more distant (Warheit 2004). A close association between California and Washington fishers can be implied by: (1) geography, (2) the close genetic distance between Haplotype 1 and Haplotype 4, and (3) the historical sample that Drew et al. (2003) identified as being Haplotype 4. Drew et al. (2003) also determined a close genetic association between British Columbia and California fishers. In his analysis, Warheit found that fishers in western Alberta and Washington shared no common Haplotypes, although this analysis was based on only three specimens from Washington. This suggests that genetic similarities are closer between Washington, California and British Columbia fishers than between Washington and western Alberta fishers. The connection between Washington and western Alberta fishers is suggested by the presence of Haplotype 6 in western Alberta, which appears closely related and possibly derived from Haplotype 4, one of two haplotypes found in historical specimens from Washington. The two individuals that had Haplotype 6 were trapped in two Alberta Fur Management Units that border British Columbia.

The clustering of western Alberta and Wisconsin fishers suggested by Figure 12 may be an artifact for three reasons. First, the Wisconsin fisher sample size was small and came from a single county. From this small sample size there were only 2 haplotypes represented. In other eastern areas, where sample sizes were greater (Minnesota, New Brunswick), a greater number of haplotypes were found. Second, the high frequency of Haplotype 1 (Appendix C), considered to be ancestral to the 11 others in the dataset, in both Wisconsin fishers and the western clade (Washington, California, British Columbia) contributed to the grouping.

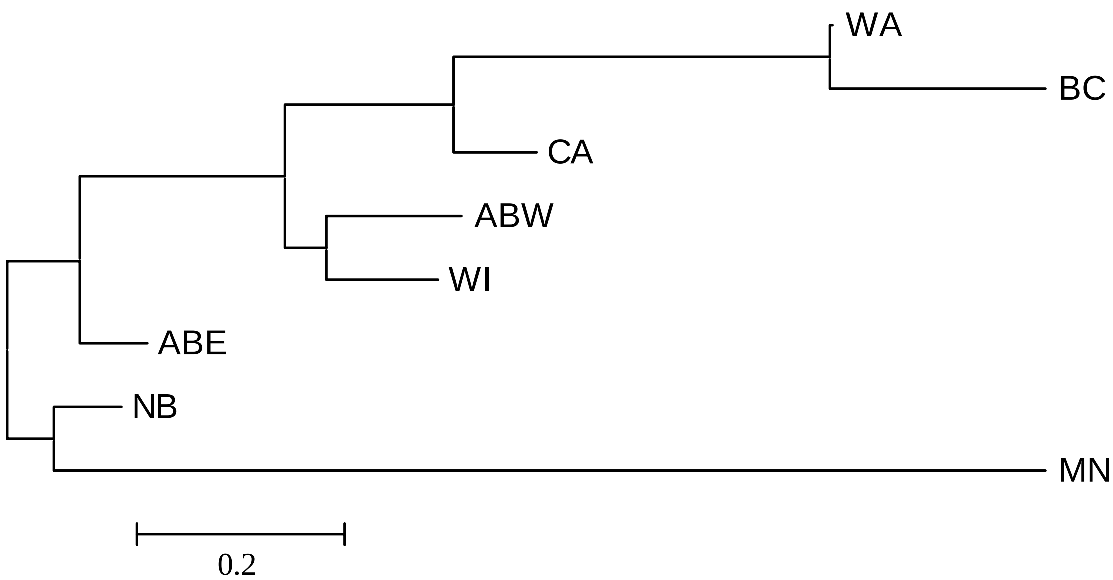

Figure 12. Minimum evolution population tree for fishers in Washington, California, British Columbia, Alberta East and West, Wisconsin, Minnesota, and New Brunswick. Tree derived from Tamura and Nei pairwise distances calculated in Arlequin. Tree developed in MEGA. The tree is unrooted. (Source: Warheit 2004).

However, relationships based strictly on an ancestral haplotype may result in artificial grouping of taxa. Third, the second haplotype found in Wisconsin fishers (Haplotype 5) is restricted in distribution to the eastern region (Minnesota, Wisconsin, and New Brunswick) and is closely related to Haplotype 8, also restricted to the eastern region (New Brunswick). Since Haplotypes 5 and 8 appear to be derived from Haplotype 7, the most common haplotype in western Alberta fishers, this also contributed to the clustering of western Alberta and Wisconsin fishers. Additional samples from other areas in Wisconsin may identify additional haplotypes that would place Wisconsin fishers closer to other states and provinces from the eastern region (New Brunswick, Minnesota). Therefore, regardless of the relationships implied by Figure 12, Wisconsin fishers may have little genetic similarity to Washington fishers (Warheit 2004).

Warheit (2004) and Drew et al. (2003) concluded that the best source population for fisher reintroductions in Washington is British Columbia. The next best source population would be from the western-most regions of Alberta or California (Warheit 2004). Because fishers are protected in California, western Alberta appears to be the second choice after British Columbia as a source of fishers for a reintroduction to Washington.

REINTRODUCTION REQUIREMENTS

Release Strategy

Previous reintroductions of fishers throughout their range provide insights into successful reintroduction approaches. The majority of documented fisher reintroductions have been successful; however, the specific strategies of each reintroduction vary and not all aspects of a reintroduction were well documented. Analysis of specific aspects of both successful and unsuccessful reintroductions would be useful for developing an approach for a successful reintroduction of fishers in Washington. If it is determined to be feasible to reintroduce fishers, an implementation plan will be developed in conjunction with public review of the proposal. While this plan would include the specifics of the reintroduction proposal, some general release strategy components are included below.

Number of animals to release. The number of animals to release is among the most important aspects of any reintroduction (Griffith et al. 1989, Breitenmoser et al. 2001). Reintroducing enough animals is important for maintaining heterozygosity and to guard against demographic and environmental stochasticity. For conserving endangered species populations, an effective population size (N_e) of $\geq$50 individuals is considered necessary for species survival, whereas an N_e of $\geq$500 is considered necessary to allow for continuing adaptation of a species (Shaffer 1987). An N_e of 500 can translate to an actual population size that is 2 to 10 times greater (i.e., 1000-5000, depending upon the species; Franklin and Frankham 1998), and this large number of founders presents logistical difficulties for reintroduction efforts that aim to promote long-term survival of a species. Only two carnivore reintroductions released 500 or more individuals (Breitemoser et al. 2001).

Successful fisher reintroductions in North America released significantly more fishers than unsuccessful ones (means of 51 vs. 22, respectively; P=0.011, 2-sample t-test)(Table 1). Slough (1994) recommended releasing $\geq$50 marten to preserve allelic diversity and avoid inbreeding depression. Powell (1990) recommended releasing 30-40 individuals among three release sites as part of a reintroduction program proposed in Great Smoky Mountains National Park. In a review of reintroductions across taxa, Griffiths et al. (1989) found that multiple-year reintroductions were frequently more successful than single-year reintroductions. Home range establishment and reproduction may be more successful if animals are released in consecutive years (Griffiths et al. 1989, Heinemeyer 1993). Based on recommendations from previous reintroduction efforts and PATCH model simulation results (Table 13), a release of $\geq$60 fishers over multiple years could be an initial target.

Sex ratio. Among successful reintroductions where the sex ratio was known, 60% were female-biased, 27% were male-biased, and 13% were even. Similarly, among unsuccessful reintroductions, 75% were female-biased and 25% were even. Berg (1982) recommended even or female-biased sex ratios for reintroductions of fishers, martens and otters. Powell (1990) also recommended a female-biased (3:2) sex ratio.

Age structure. Where feasible, adults (>1 year old) are preferred over juveniles in the founding population because of their higher survival rates and sexual maturity that together should result in greater population growth.

Time of year. For most reintroductions, fishers were obtained from a source population during the commercial trapping season. As trapping seasons usually occur in the fall and winter, when fisher pelts are prime, releases frequently occurred in the late fall, winter or early spring. A winter and early-spring release places fishers at a disadvantage due to relatively low food availability, possible impediments to travel (e.g., deep snow), and a short period of time to establish territories and find a den site before the birthing and breeding period in March-April. Fishers have delayed implantation, and breeding occurs shortly after the young are born. An alternative release strategy involves a late-summer/early fall release. This would give fishers the opportunity to become familiar with their new surroundings during a more favorable season and a period of greater prey abundance. Individuals would have more time to establish home ranges and locate potential mates and denning sites. However, implementing this strategy could be a logistical challenge because it would require a special trapping period prior to the commercial trapping season, and could result in lower trapping success since it would occur during a period of greater prey abundance. A second release strategy involves winter captures. Males and non-reproductive females would be hard-released in male/female pairs during the breeding season in March or early April. Releasing individuals in early spring may reduce movements due to the greater availability of prey and onset of the breeding season. Pregnant females, on the other hand, would be maintained in captivity until their kits were able to run at 12 weeks, and would subsequently be released as a social unit in late summer (Proulx et al. 1994, Fontana et al. 1999). Releasing females and their kits as a social unit may limit dispersal and result in establishment of home ranges sooner at release sites (Proulx et al. 1994).

Capture, transport and release. The handling period between capture and release of animals should be limited to minimize stress and changes in behavior (Heinemeyer 1993). Fishers are typically live-trapped in baited cage-type traps (e.g., Tomahawk Live Trap Co., model number 211a; 1.0 x 0.3 x 0.3 m) with an enclosed "cubby" box attached to the end of the trap. Following capture, fishers can be transported to a holding facility in the field close to the capture site. Animals can be shut inside the attached "cubby" box, which minimizes visual and noise disturbances and keeps them from injuring themselves. At the holding facility, animals may be transferred to individual holding cages (0.75 x 0.75 x 2.0 m) that contain a den box and straw. Food (0.5 kg of road-killed deer and/or commercial feed provided daily) and water are provided. Fishers should be kept in a building at ambient temperature (Heinemeyer 1993), noise minimized, and cages covered to reduce anxiety of fishers. During the holding period individual animals may be anesthetized. The animal can then be physically examined, anatomical measurements and blood taken, a lower premolar may be extracted to estimate age, hairs follicles or other tissue taken for DNA profiling, passive integrated transponders (i.e., pit tag) placed subcutaneously, vaccinations for canine and feline distemper administered, photographs taken of ventral spotting for individual identification, and ear tags and a radio-collar fitted and attached (Roy 1991, Heinemeyer 1993). When enough animals have been captured, they may be transferred to travel boxes and transported by vehicle or aircraft to release sites (see Roy 1991, Heinemeyer 1993).

Release sites should be selected for suitable denning, resting, and foraging habitat, proximity to water and isolation from human activity. Two methods have been used to release fishers at reintroduction sites: soft releases and hard releases. In a soft release animals are held at a release site for several days prior to release to encourage them to reside near the site. In a hard release individuals are released without a holding period. Soft releases have been used for marten (Davis 1983), however few fisher reintroductions have used soft releases. Providing food in the vicinity of the release site to encourage residency has been used with both hard and soft releases (Roy 1991, Heinemeyer 1993, Weir 1995). Heinemeyer (1993) evaluated soft vs. hard release methods, but did not find one method to be more successful than the other. Despite using a soft release in a 1990-1992 reintroduction in central British Columbia, Weir (1995) reported that it did not provide enough incentive to keep fishers from dispersing from the release sites. There is no indication that either hard or soft releases influence success or failure of fisher reintroductions. Proulx et al. (1994) suggested that the time of year fishers are released may better explain the tendency for released fishers to remain near or travel from the release area. June-released fishers were less likely to leave the general release area than fishers released in March.

Monitoring

The earliest fisher reintroductions typically used sightings and captures as evidence of persistence and reintroduction success. In later reintroductions, fishers were ear-tagged to enable identification of recaptured individuals. More recently, radio-tagging has been used to monitor movements, assess survival, and determine reproductive success of released fishers. Monitoring efforts have also involved surveys (e.g., snow tracking, track-plate boxes, camera stations) and benefited from reports of incidental captures, sightings, and road-kills.

Monitoring would be an important aspect of a Washington reintroduction program. Fishers released in the state would be monitored in a multi-year effort to evaluate: (1) persistence in the release area, (2) survival, and (3) reproduction. Monitoring efforts would likely involve radio-telemetry, genetic sampling techniques, and track-plate and camera stations. These efforts would also provide the opportunity to conduct research into the habitat use, demography and behavior of released fishers. The details and logistics of a monitoring and research plan for a reintroduced population would be outlined in an implementation plan.

STAKEHOLDERS AND COOPERATORS

A number of federal and state agencies, private companies, and non-governmental organizations would be stakeholders and cooperators in a potential reintroduction. Several of these agencies and organizations have already been involved with or have been updated on the progress of this feasibility study. These include the Washington Department of Natural Resources, National Park Service, U.S. Forest Service, U.S. Fish and Wildlife Service, Northwest Ecosystem Alliance, Washington Trappers Association, Washington Forest Protection Association, and several zoos. These stakeholders and cooperators could be involved directly or indirectly in affecting a potential reintroduction through their own mandates and missions. Conversely a fisher reintroduction could have implications for these entities. Some of these effects and implications are outlined here.

Federal

The U.S. Forest Service has listed fishers as a sensitive species on all National Forests in Washington. Reestablishing native fauna to National Parks is part of the mission statement of the National Park Service, and reintroducing fishers to Olympic National Park is consistent with this goal. Staff members from Olympic National Forest and the Olympic National Park have been involved in this feasibility study and have been investigating how their agencies might be involved or contribute to a possible reintroduction. The U.S. Fish and Wildlife Service would likely be a participant in a reintroduction involving a federal candidate species.

Tribal

As sovereign nations and adjacent landowners, tribes would be interested in a proposed reintroduction and would be consulted.

State

Washington Department of Natural Resources is an adjacent landowner on the Olympic Peninsula and has developed a multi-species HCP that includes the fisher. DNR is aware of this feasibility study and will be consulted with on the proposed reintroduction to determine possible partnerships, interests and concerns.

Private

Trappers. The Washington Trappers Association has been kept informed about the feasibility study and has discussed how the Association might work with WDFW to implement a possible reintroduction.

Timber industry. Representatives of the Washington Forest Protection Association have an interest in the feasibility study and have been informed about the development of the study.

Non-Governmental Organizations

The Northwest Ecosystem Alliance (NWEA) and WDFW are partners in the feasibility study and organizations such as NWEA, Woodland Park Zoo, Northwest Trek Wildlife Park, and the Oregon Zoo have expressed interest and offered support for a possible reintroduction.

LEGAL REQUIREMENTS

Based on current statutes, a reintroduction of fishers in Washington is not expected to result in changes to forest management practices on federal, state, or private forestlands.

Federal

U.S. Fish and Wildlife Service. Currently, fishers are not listed as endangered or threatened by the U.S. Fish and Wildlife Service and therefore receive no federal protection. However, the Center for Biological Diversity and the Sierra Nevada Forest Protection Campaign (representing 19 organizations and one individual) petitioned the Service to list a discrete population segment of the fisher in its West Coast range, including Washington, Oregon, and California, as endangered pursuant to the federal Endangered Species Act and to designate critical habitat. On July 12, 2003 the U.S. Fish and Wildlife Service (FWS) announced in the Federal Register that the petition presented "substantial information that the West Coast population of the fisher may be a distinct population segment for which listing may be warranted" (Jones 2003). On April 8, 2004 the FWS published its finding in the Federal Register that the petitioned action was warranted, but precluded by higher priority actions. Upon publication of the 12-month petition finding, the West Coast DPS of the fisher was added to the list of federal candidate species (Wild and Roessler 2004).

NEPA. If a reintroduction were proposed on National Park lands, the proposed action would need to follow requirements outlined in the National Environmental Policy Act (NEPA). If it occurred on National Forest lands, no NEPA analysis would be required, since it would be a state, rather than a federal action (K. O'Halloran, U.S. Forest Service, pers. comm.).

State

Although fishers are State-listed as endangered in Washington there is currently no critical habitat rule (WAC 222-16-080) for the fisher under the State Forest Practices Act (RCW 76.09) and thus no restrictions of forest practices activities in fisher habitat. WDFW does not anticipate asking the Forest Practices Board for a critical habitat rule, because little is known about fisher habitat requirements in Washington, and if a reintroduction occurs it is anticipated to occur on federal land. If fishers are reintroduced, WDFW would encourage the protection of den sites if they become known through research and monitoring. Currently, there is no commercial take of fishers allowed in Washington. Trapping for fishers was closed in 1934 and has never been re-opened. Under its' status as a state endangered species it cannot be legally trapped or killed.

Passage of Initiative 713 by Washington voters in 2000 banned the use of body-gripping traps to capture furbearers by licensed trappers, prohibited the sale of commercially valuable furbearer pelts that were obtained by body-gripping traps, and directed that a permit system be utilized to capture only animals involved in nuisance or damage activity on private land (Koenings et al. 2003). Furbearers may be captured using live traps. As a result of these trapping restrictions, total furbearer harvest in Washington has declined by 80%, and trapper numbers have declined by 60% (Koenings et al. 2003). Legislative proposals seeking to amend Initiative 713 are being developed. If I-713 is overturned and a commercial harvest of furbearers were reinstated, the closure on fisher harvest would remain in place because of its' protected status. However, fishers could be subject to incidental take in traps set for other legally harvested furbearers.

State Environmental Policy Act requirements for a fisher reintroduction would be met by the fulfillment of a NEPA process by the Olympic National Park (RCW 43.21C).

PERMITTING

Diseases & Parasites

Fishers are susceptible to a number of diseases and parasites, but they do not appear to be significant sources of mortality in fisher populations. Diseases of fishers include trichinosis, toxoplasmosis, leptospirosis, Aleutian disease (Strickland et al. 1982), rabies, canine and feline distemper, and plague (Wild and Roessler 2004). Fishers in British Columbia were infected with coccidia and callicivirus (Fontana et al. 1999; R. D. Weir, unpubl. data). Many internal and external parasites have been reported for fishers (Douglas and Strickland 1987).

If a source population of fishers is identified in either Canada or the U. S., permits would be needed to authorize transport of animals and to document that animals were healthy and free of diseases harmful to domestic livestock in Washington. Agencies from both Canada and the U.S., at state/provincial and federal levels, have requirements that need to be met in accordance with the transfer of fishers across borders.

Federal

The Canadian government does not require any federal permits for exporting fishers, and the U.S. government does not require disease testing fishers or health certificates to transfer fishers from Canada into the United States. However, if rabies has been documented in any wild mammal in the order Carnivora from the state of origin in the 12 months prior to translocation, fishers would not be allowed entry into Washington. The U.S. Fish and Wildlife Service (FWS) does require that fishers be inspected by a FWS inspector at the international border to ensure humane transport of animals and that the animals be "declared" several days prior to entry into Washington. A completed Declaration Form (3-177) would be needed and FWS notified 48 hours in advance of transport by vehicle through the Seattle area or 72 hours if transport is via another port of entry in Washington. The FWS inspector would be contacted in advance to coordinate inspection at the port of entry. Since fishers are not federally protected, no additional federal permits (i.e., CITES, section 10 permit for "take") would be required. U.S. Customs requires completion of a Declaration Permit. Arrangements would need to be made with the Customs agent if a border crossing would occur outside normal business hours and coordination (and payment of a fee) with a bonding agent may be necessary at the border.

State and Provincial

The Washington Department of Agriculture (WDOA) requires completion of a health certificate and issuance of a permit prior to entry of fishers into Washington. A licensed, accredited veterinarian in the province or state where fishers are collected completes the health certificate or Certificate of Veterinary Inspection. After completion of the health certificate, the provincial or state veterinarian would need to contact WDOA to obtain a permit number (and record on the health certificate) prior to transfer of fishers into Washington. This permit is recorded at the Olympia office of WDOA and enforcement is informed of the transfer. Fishers could not enter Washington if exposed to tuberculosis or if a diagnosis of rabies had been documented in the province of origin during the previous 12 months. British Columbia and Alberta require a "Possession Permit", signed by the regional biologist or Director of Wildlife, for transport of fishers within the province and out of Canada. If fishers were obtained from British Columbia, the provincial veterinarian would take blood samples of all fishers as a future reference health archive prior to transport out of the province.

CONCLUSIONS AND RECOMMENDATIONS

Three potential reintroduction areas were identified in Washington that had large blocks of suitable, well-connected fisher habitat: the Olympic Peninsula, Southwestern Cascades and Northwestern Cascades. The Olympic Peninsula was identified as the most suitable area for reintroduction because it had the greatest amount of suitable habitat, greatest amount of suitable habitat on public lands and conservation status one lands, and the highest potential fisher carrying capacity. Although geographically isolated, the Olympic Peninsula appears capable of supporting the greatest number of fishers and could serve as a source population for subsequent reintroductions in the Cascade Range. Monitoring and research of a reintroduced population on the Olympic Peninsula would provide information on fisher habitat use and survival that could help indicate if conditions in the Cascades were suitable for additional reintroductions.

Fishers are likely to encounter sufficient densities of prey species in forest cover types and seral stages that occur at low- to mid-elevations on the Olympic Peninsula. These prey species are likely to include pine squirrels, small mammals, mountain beavers, snowshoe hares, and birds. Late-successional dominated landscapes found within Olympic National Park and Olympic National Forest should provide a stable and predictable prey base to sustain a fisher population.

A source population exists for a potential reintroduction in Washington. British Columbia is the most suitable source population for reintroductions in Washington. If animals are not available from British Columbia due to concerns about their status, then fishers from western Alberta would be the next best source population. Western Alberta has sufficient numbers of fishers available to support a reintroduction to Washington.

A fisher reintroduction is not expected to have an adverse affect on populations of species of concern, such as spotted owls, marbled murrelets and northern goshawks, or on furbearer species such as the coastal population of American marten and bobcat. Given the low density and wide-ranging habits of fishers, interactions between fishers and these species are expected to be rare occurrences.

Release strategies could include either a late summer/early fall capture and release or a winter capture and release of males and non-reproductive females in early spring and adult females and kits in late summer. Animals would be monitored following release to evaluate demographic characteristics (e.g., survival, reproduction, and dispersal), persistence in the release area and habitat use.

Because fishers are not protected at the federal level and there is no state forest practice rule in Washington for this species, a reintroduction is unlikely to result in changes to forest management practices on federal, state, or private lands, based on current statutes.

The public land base comprised of Olympic National Park and Olympic National Forest has a high concentration of contiguous suitable fisher habitat, which should provide a diverse prey base for fishers and support a viable reintroduced population. Based on the findings of the feasibility study, it is believed that a reintroduction of fishers is biologically feasible in Washington and it is recommended that a NEPA analysis be initiated for a proposed reintroduction in the western Olympic Peninsula on the Olympic National Park and Olympic National Forest.

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Appendix A. Members of fisher science team and agency affiliation.

Name	Team Expertise	Work Affiliation
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Appendix B. Data inputs and simulation specifications used with the PATCH model to estimate the carrying capacity of potential reintroduction areas.

The carrying capacity of potential reintroduction areas was estimated using the spatially-explicit population model PATCH (Schumaker 1998). Demographic simulations conducted with PATCH require a habitat map and species-specific values for habitat associations, territory size, dispersal distance, survival, and fecundity. There are no data from Washington for estimates of these parameters, therefore model inputs were obtained from studies elsewhere in the range of the fisher. Because values obtained for fishers in eastern and mid-western North America may poorly represent a Washington fisher population, emphasis was placed on using values from studies done in western North America whenever possible.

HABITAT ASSOCIATIONS

In PATCH, the user incorporates a habitat map (a bitmap file). The user also provides scores for each cover type in the habitat map (e.g., late-seral forest, early seral forest) to indicate the relative value of the cover type as habitat to the species of interest. IVMP raster data was used to identify cover types in each of the potential reintroduction areas. Six cover type classes were identified: late-seral forest, mid-seral forest, early-seral forest, other forests, cleared areas, and non-habitat (Table 1). Four of the six forested cover types were defined using the same variables used to define suitable habitat in Table 3 of the Habitat Assessment section. The number of cover types was increased to 12 in the final map to distinguish cover type pixels that occurred above the upper limit of the Pacific silver fir zone. This distinction was made to lower the value of habitats at higher elevations, because fisher mobility and foraging success may be lower in areas with deep snow (Raine 1983, Krohn et al. 1995). The habitat maps produced for the PATCH analysis are very similar in appearance to Figures 6, 7, and 8 in the Habitat Assessment section of this report.

Habitat associated scores were based on a scale of 0-10, with 0 being given to those cover types considered non-habitat (e.g., water, urban) and 10 being given to cover types considered optimal habitat (Table 1). Habitat scores were given to 6 cover types, both above and below the Pacific silver fir zone (Table 1). Because fishers select late-seral forests at the stand level, and use structural elements typically found in late-seral stands as den and rest sites, the late-seral forest cover type was considered optimal habitat. Mid-successional (mid-seral) forests are also used by fishers, because these forests can provide important foraging and traveling habitat, and may provide den and rest sites. Jones and Garton (1994) and Fontana et al. (1999) reported seasonal selection for mid-seral conifer habitats in the winter, and also proportional use or avoidance in other seasons. Mid-seral forests were considered intermediate in their importance to fishers. Early-successional (early seral) forest stands are used by fishers but tend to be avoided relative to their availability in a landscape (Buck 1982, Jones and Garton 1994, Fontana et al. 1999). Early seral forests were considered relatively low in their importance to fishers and were assigned low scores. A set of low habitat scores was used to test the sensitivity of the model to lower values for suboptimal habitats (Table1).

TERRITORY/HOME RANGE

PATCH can use either territory or home range size. Home range sizes have been estimated for female fishers in eight radio-telemetry studies in western North America (Table 2), therefore home range size is used in place of territory size. The mean home range size plus or minus 1 standard deviation was used to select a minimum (14 km^2), mean (25 km^2) and maximum (36 km^2) home range size to use in PATCH analyses.

Table 1. Variables and values used to define 6 cover type classes within potential fisher reintroduction areas and habitat scores used for cover types above and below the upper limit of the Pacific silver fir zone (PSFZ) in Washington.

Forest cover types[1]	Model variables			Habitat scores		Low habitat scores	
	Vegetation cover (%)	Conifer cover (%)	QMD[2] (inches)	Below PSFZ	Above PSFZ	Below PSFZ	Above PSFZ
Late-seral	>70	>30	20+	10	6	10	4
Mid-seral	>70	>30	10-19.9	6	4	4	2
Early-seral	>70	>30	0-9.9	3	2	2	1
Other	<70			4	2	2	1
Cleared	<70	<30		2	1	1	0
Non-habitat[3]				0	0	0	0

[1] From Interagency Vegetation mapping Project vegetation data.
[2] QMD = Quadratic Mean Diameter.
[3] Areas classified as urban, agricultural, water, barren, or snow.

Table 2. Mean annual home range sizes for female fishers as determined in eight studies in western North America.

Location	Mean annual home range size (km^2)	N	Study
Southern Oregon Cascades	25	7	Aubry and Raley 2002
Southeast British Columbia	27	3	Fontana et al. 1999
Central British Columbia	35.4	11	Weir 1995
Central Alberta	14.9	10	Badry et al. 1997
Idaho	32	4	Jones 1991
Northern California	6.8	2	Buck et al. 1983
Northern California	53.4	2	Dark 1997
Northern California	15.0	7	Zielinski et al. 2004*b*
weighted mean ($\pm$ 1 SD)	24.95 $\pm$ 10.9		

DISPERSAL

Dispersal is incorporated into the PATCH model by providing a measure of the maximum number of steps (i.e., number of diameters of a hexagon equal to the size of a weighted mean home range) that an individual could take in a single year. This equates to an average maximum dispersal distance. PATCH allows individuals of all three age classes to move, and randomly chooses the number of steps an individual will take in a given year. Individuals can move in a number of ways, however we chose the recommended movement pattern called a "directed walk" where an animal may move randomly through a landscape but will tend to move toward unoccupied hexagons with suitable habitat. Both the variability in the distance and the direction of annual movements of individuals provides stochasticity to a simulation. We used a weighted mean home range size of 25 km^2 and a maximum dispersal distance of 50 km for female fishers, or approximately 10 home range diameters. We also allowed dispersing females to move from one to 10 hexagons in a single year.

Few data are available from the literature for maximum dispersal distances and none was available from the Pacific Northwest. We chose 50 km based on an average of the maximum dispersal distance (63 km; Table 3), the weighted mean dispersal distance (28 km; Table 3) and the reported maximum dispersal distance of 55 km for a male fisher in southern Oregon (Aubry and Raley 2002). Given the limited data, a sensitivity analysis was conducted using PATCH to test the effect of a smaller maximum dispersal distance (25 km; 5 home range diameters) and a larger maximum dispersal distance (75 km; 15 home range diameters) on model outputs across potential reintroduction areas.

SURVIVAL, FECUNDITY, AND LESLIE MATRICES

Leslie matrices are used to project populations into the future based upon a starting female population size, age-specific female survival, and age-specific fecundity. Demographic simulations in PATCH require at least one Leslie matrix. PATCH can also draw a single matrix randomly from a group of matrices in each year of a simulation to incorporate environmental stochasticity.

Survival

Survival values are required for each age class used in the matrix. Four estimates of adult/yearling female survival and two estimates of juvenile survival were reported in the literature (Table 4). The values from Connecticut and Maine were derived from fisher populations that were subjected to trapping mortality, whereas a Washington population would not be subjected to this source of mortality. York (1996) indicated that the Connecticut population was subjected to light trapping pressure. Using data for fisher survival reported for Maine (Krohn et al. 1994), we were able to remove trapping mortalities from the data and recalculate survival rates that may better represent an un-trapped population (Table 4).

Table 3. Mean and maximum dispersal distances of female fishers as determined in three studies.

Location	Maximum dispersal distance (km)	Mean dispersal distance (km)	N	Study
Oregon		6	4	Aubry & Raley 2002
Maine	18.9	11.3	5	Arthur et al. 1993
Massachusetts	107	37	19	York 1996
Mean	63	18.1		
Weighted mean		28		

Table 4. Age-specific survival rates for female fishers taken from the literature. Adult and yearling female survival was estimated annually based on radio-telemetry data; juvenile rates were generated using both mark-recapture and radio-telemetry data.

	Survival rate		
Location	Adults/ Yearlings (*n*)	Juveniles (*n*)	Source
Southern Sierra Nevada, California	0.612 (24)	-	Truex et al. 1998
Northwestern Coast, California	0.838 (22)	-	Truex et al. 1998
Connecticut	0.90 (23)	0.84 (33)	York 1996
Maine	0.78 (19)	0.36 (16)	Krohn et al. 1994[1]
Weighted mean	0.780	0.683	
Weighted SD	0.110	0.224	

[1] Survival estimates for Krohn et al. (1994) were derived by extending survival rates from the non-trapping interval through the trapping season (39 day interval) to include 364 days and to exclude mortalities that resulted from trapping.

Fecundity

Fecundity values for the Leslie matrix are derived by multiplying the mean annual survival rate for an age class by the percent of females producing young in that age class, the weighted mean litter size for that age class, and the proportion of females in the litter (assumed 0.5). Juveniles cannot be impregnated. Yearlings can be impregnated at approximately 1 year of age, but do not give birth until they are 2-year olds. This is a consequence of delayed implantation, which characterizes reproduction in mustelids.

Percent of females producing young. The percent of females that produce young each year can vary among years and study areas (Table 5). Few studies have data that differentiate between the percent of adults (≥3 years old) and subadults (2 year olds) that produce young each year. Consequently most estimates are provided for all females that are ≥ 2 years old.

Mean litter size. Litter size data are available for both wild fishers (Table 6) and captive fishers. The size of the litters and the accuracy of the counts are greater in captive litters, as litter sizes can be determined easily and counts can be done soon after parturition, prior to neonatal mortality. Conversely, it is easy to underestimate wild litter sizes due to the difficulty of counting kits in the wild and the potential for kit mortality prior to a count. For this analysis only values obtained from wild litters were used.

Table 5. Estimates of the annual percentage of female fishers producing young; adults are ≥3 years old, and all females include adults and 2-year olds.

	Percent females producing young (*n*)			
Location	All females	Adults	2-yr olds	Study
Oregon	0.594			K. Aubry, pers. comm.
British Columbia	0.68			R. Weir, pers. comm.
NW California	1.00 (14)	1.00 (14)		J. Thompson, pers. comm.
Connecticut	0.68 (28)	0.69	0.57	York 1996
Midwest/Northeast	0.55 (31)			York and Fuller 1993
Maine	0.60 (25)	0.65 (21)	0.25 (4)	Paragi 1990
Mean	0.684	0.78	0.41	
SD	0.163	0.19	0.22	

Table 6. Mean litter sizes of wild fisher litters for adults (≥3 years old), 2-yr olds, and all females.

	Mean litter size (*n*)			
Location	All females	Adults	2-yr olds	Study
Oregon	1.78 (9)	-	-	K. Aubry, unpubl. data
Connecticut	2.83 (19)	2.91 (11)	2.75 (8)	York (1996)
Montana	3 (1)	-	-	Roy (1991)
Maine	2.1 (12)	2.12 (9)	2 (1)	Paragi (1990)
Weighted mean	2.43	2.55	2.67	
Weighted SD	0.59	0.39	0.24	

Leslie Matrices

Survival and fecundity values are input as Leslie matrices in PATCH, while the other values are input directly. Given the limited data that exist for these vital rates, we developed matrices with only three age classes: adult, subadult and juvenile (Table 7). Three matrices were developed: a matrix with mean values from the literature (Table 8), a second with the mean values minus one half standard deviation (Table 9), and a third with the mean values plus one half stand deviation (Table 10).

Table 7. An example of a 3-age-class Leslie matrix. Age-class specific fecundity = mean annual survival rate * mean percent of females producing young * mean litter size * proportion of females in the litter (assumed 0.5).

Juvenile fecundity	Subadult fecundity	Adult fecundity
Juvenile survival	0	0
0	Subadult survival	Adult survival

Table 8. Leslie matrix with mean fecundity and survival values from the literature and unpublished research. Matrix entries are taken from Tables 4-6, except as noted. The finite rate of change (i.e., lambda) of a population characterized by this matrix is 1.110.

0	0.389 (=0.780*0.41*2.43 *0.5)[1]	0.680 (=0.780*0.684*2.55*0.5)
0.683	0	0
0	0.750[2]	0.750[2]

[1] The mean litter size of 2.43 for all females was used for subadult females instead of the 2.67 kits per litter value, as subadult litter sizes are not expected to exceed those of adults.
[2] Survival values for all subadults and adults were reduced to 0.750 (from 0.780) so that survival rates did not exceed 1.00 when they were scaled up to correspond to optimal habitats (see Schumaker 1998).

Table 9. Leslie matrix with low fecundity and survival values as derived by subtracting one half standard deviation from values for mean annual survival, percentage of females producing young, and mean litter size taken from the literature (Tables 4-6). The finite rate of change (i.e., lambda) of a population characterized by this matrix is 0.980.

0	0.233 (=0.725*0.30*2.14*0.5)	0.515 (=0.725*0.602*2.36*0.5)
0.568	0	0
0	0.725	0.725

Table 10. Leslie matrix with high fecundity and survival values as derived by adding one half standard deviation to values for mean annual survival, percentage of females producing young, and mean litter size taken from the literature (Tables 4-6). The finite rate of change (i.e., lambda) of a population characterized by this matrix is 1.222.

0	0.592 (=0.835*0.520*2.73*0.5)	0.883 (=0.835*0.766*2.76*0.5)
0.750[1]	0	0
0	0.750[1]	0.750[1]

[1] Survival values for all three age classes were reduced to 0.750 (from 0.835 for subadults and adults, and from 0.792 for juveniles) so that survival rates did not exceed 1.00 when they were scaled up to correspond to optimal habitats (see Schumaker 1998). Note however that the original survival rate (0.835) was used in the fecundity calculations for adults and subadults.

Two scenarios were used when incorporating Leslie matrices into PATCH simulations. The first scenario used only the matrix with mean values (Table 8). The second scenario involved using six matrices in a simulation, where PATCH randomly chose one matrix for each year of a simulation. The six matrices included four with mean values (Table 8), one with low values (Table 9) and one with high values (Table 10). This frequency of matrices was designed to reflect the frequency of occurrence of environmental stochasticity following a normal distribution. Simulations run with the six random matrices exhibited greater variability in population estimates.

A score of 7.5 was assigned to each matrix for all PATCH simulations. This score indicates that the values observed in Tables (matrices) 8, 9, and 10 are used for hexagons that have a habitat score of 7.5. However, when hexagons have higher habitat scores (i.e., greater amount of suitable habitat; >7.5) the matrix values are increased linearly to correspond to the higher quality habitat. Conversely, the matrix values are reduced linearly for hexagons with lower quality habitats (i.e., lower scores; <7.5).

The Poptools add-in for Microsoft Excel (http://www.cse.csiro.au/poptools/) was used to evaluate the sensitivity of population outcomes to specific values in the matrices. Using the matrix elasticity analysis, all models (low, mean, and high value matrices) were found most sensitive to adult female survival, followed by juvenile survival, and then equally by subadult survival and adult female fecundity. The greater sensitivity for adult female survival was most pronounced in the low- and mean-value matrices and less pronounced in the high-value matrix. Lamberson et al. (2000) conducted a similar sensitivity analysis and found that lambda values were particularly sensitive to adult female survival and the percent of adult females producing young.

DISCUSSION

While these data are useful for conducting population simulations in PATCH, much of the source information is from outside the Pacific Northwest and its' applicability to Washington is unknown and may be somewhat limited. It is suspected that the small sample sizes of demographic parameter estimates resulted in some unsupportable conclusions (e.g., mean subadult litter sizes exceeding those of adults). It was documented when judgement calls were made and alternative values for estimates were used. A Leslie matrix that used mean values from the literature supported substantial population growth (lambda = 1.110), ~11% growth per year. Lambda values from other studies were substantially lower except under the most optimistic scenarios (York and Fuller 1993, Lamberson et al. 2000). Given the substantially positive lambda values for the mean and high value matrices (1.110 and 1.222, respectively) as well as the relatively high lambda value for the low value matrix (0.98), caution should be used when considering the results. While the larger lambda values used in PATCH simulations may be encouraging from a demographic standpoint, it does not address the issues of habitat sufficiency or the sufficiency of genetic variability in a founding population that could greatly affect the success of a reintroduction.

Appendix C. Distribution of control region haplotypes among localities (source: Warheit 2004).

Haplotype[A]	CA	CA[B]	WA[B]	BC	Abwest[D]	Abeast[D]	MN	WI	NB
MP01	32	1	1[C]	2	0	0	3	10	0
MP02	6	0	0	0	0	0	0	0	0
MP03	0	0	0	0	1	8	0	0	17
MP04	0	1	2	10	0	0	0	0	0
MP05	0	0	0	0	0	0	1	6	4
MP06	0	0	0	17	2	0	0	0	0
MP07	0	0	0	12	16	3	1	0	5
MP08	0	0	0	0	0	0	0	0	7
MP09	0	0	0	20	0	0	0	0	0
MP10	0	0	0	0	0	0	25	0	0
MP11	0	0	0	3	11	7	1	0	0
TOTAL	38	2	3	64	30	18	31	16	33

[A] Data for all localities, except Alberta (Abwest and Abeast) are from Drew et al. (2003) and Vinkey (2003). Haplotypes were defined by Drew et al. (2003) and identified sequentially (1-11) with a MP (*Martes pennanti*) prefix.
[B] Historical samples from museum skins (see Drew et al. 2003).
[C] Identified as either Haplotype MP01 or MP04.
[D] ABwest (Alberta Fur Management Zones 332, 346, 347, 351, 356, 358, 360, 442, 446, 520, 525, 534, 536, 542, 544). ABeast (Alberta Fur Management Zones 502, 503, 512, 515).