Post-Release Movements, Survival, and Resource Selection of Fishers (*Pekania pennanti*) Translocated to the Olympic Peninsula of Washington

Jeffrey C. Lewis

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Reading Committee:

Stephen West, Chair

Kurt J. Jenkins

Keith B. Aubry

Program Authorized to Offer Degree:
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University of Washington

Abstract

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Jeffrey C. Lewis

Chair of Supervisory Committee:

Emeritus Professor Stephen D. West

School of Environmental and Forest Sciences

The fisher (*Pekania pennanti*) is a mid-sized carnivore in the weasel family (Mustelidae) that occurred throughout the temperate and boreal forest ecosystems of North America. Because of the extremely high value of the fisher's pelt in the 1800s and early 1900s, fisher populations were overexploited and extirpated throughout much of the southern portion of their historical range (southern Canada and northern U.S.). Translocations have been successful at restoring fishers throughout much of the vacant portions of the historical range, however little information from these translocations is available in the published literature. As part of the fisher recovery process in Washington state, we translocated 90 fishers (50 F, 40 M) from central British Columbia to the Olympic Peninsula. We radio-collared each fisher to allow investigations of post-release movements, survival, and resource selection of a large sample of translocated fishers. Fishers moved extensively after their release and the extent of these movements was best explained by sex and the month when movements occurred. Similarly, sex was the most influential factor in explaining the distances between release sites and established home ranges of 48 fishers (27 F, 21 M) that established home ranges. Mean distance (±SE) from a release site to an established home range was 44.5 ± 6.4 km for males and 30.1 ± 3.6 km for females; however, the mean number of days from release until home range establishment was similar for the sexes. Twenty-six of 27 females established home ranges from December to October of their first year and the distribution of establishment dates did not differ from a uniform distribution ($\chi^2 = 7.00$, df = 10, $P = 0.725$). Conversely, 19 of 21 males established home ranges between April

and July of their first year and this distribution of establishment dates differed from a uniform distribution ($\chi^2 = 46.571$, df = 10, $P < 0.001$). Mean home range sizes for males (128.3 km^2) and females (63.5 km^2) were among the largest reported for the species. Eighteen of the 27 females (67%) and 8 of 21 males (38%) established home ranges within the Olympic Fisher Recovery Area (*i.e.,* Olympic National Park and Olympic National Forest).

The survival of the founders was most strongly influenced by the year in which they were released (cohort), season of the year, sex, and age; duration of captivity, weight, release-date and number of intact canines did not influence survival. The importance of influential variables was best explained by the lower survival rates of fishers released in year 2 (cohort), the lower survival rates of fishers during the breeding season (season), the higher survival rates of males (sex) and the higher survival rates of juveniles (age). Point estimates for survival rates were highest for juvenile males (range of annual survival for the 3 cohorts was 0.65-0.94), followed by adult males (0.69-0.91), juvenile females (0.46-0.89) and adult females (0.37-0.86); this finding is inconsistent with what is commonly found in established fisher populations, where adult females often have the highest survival rates and juveniles have the lowest. Cause of mortality was determined for 24 of 35 recovered fishers; predation was the leading cause of mortality (40%) followed by vehicle strikes (20%), drowning (6%) and incidental capture (3%). Predation and vehicle strikes appeared to be more common as causes of mortality in our study than in most other studies of fisher mortality.

The selection of home ranges differed between the sexes as well as the degree of selectivity. Most (79%) females established home ranges within the unmanaged forest landscapes of the Olympic Fisher Recovery Area. A resource selection model that included the percentage of forest in the intermediate-size tree class and mean elevation was clearly the best among the 17 models that we evaluated, indicating substantial selection by females. Because most females established home ranges within unmanaged forest landscapes in Olympic National Park, they selected unmanaged forest in the intermediate-size tree class within landscapes dominated by unmanaged forest in the intermediate-size and large tree classes. Most (56%) males established home ranges on managed forest landscapes and exhibited a selection for home ranges with smaller overstory trees and a greater percentage of forest in the small tree class than was available within the study area; however all 17 models of resource selection received some level support by the data, indicating weaker selection by males for the variables we included in our

models. The amount of natural open space was the factor that best distinguished male and female use of home range core areas, with females using core areas with substantially less natural open area than males.

This work represents the first investigations of survival and resource selection of a translocated fisher population, and the second investigation of post-release movements of a translocated fisher population. It also provides insights into the factors that can influence translocation success. Our findings should be particularly useful for managers that are conducting a feasibility assessment or developing an implementation plan for a fisher translocation.

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INTRODUCTION

The fisher (*Pekania pennanti*) is a mid-sized (2-6 kg) carnivore in the weasel family (Mustelidae) that once occurred throughout much of the temperate and boreal forests of Canada and the northern U.S. (Powell 1993, Lewis *et al.* 2012). Because of the fisher's valuable pelt, which sold for as much as $350 (Seton 1926, Bailey 1936, Grinnell *et al.* 1937, Dalquest 1948), fisher populations were overexploited in the 1800s and early 1900s (Powell 1993, Lewis and Zielinski 1996, Lewis *et al.* 2012). Despite protection from trapping in many states and provinces beginning in the early 1900s, many fisher populations in southern Canada and the northern U.S. were extirpated (Lewis *et al.* 2012).

Beginning the 1940s, concerted efforts were made to reestablish fishers within vacant portions of their historical range to restore a valuable furbearer, an important predator of porcupines (*Erethizon dorsatum*) and a member of the native carnivore community (Berg 1982, Powell 1993). Once self-sustaining populations were reestablished through reintroductions, effective trapping regulations could support robust fisher populations *and* allow trappers to take a harvestable surplus. Most of the earliest reintroductions occurred in the central and eastern portions of North America and these efforts were initiated well before the existence of Endangered Species Act or the listing of the fisher as a candidate species (U.S. Fish and Wildlife Service 2004). However, unlike many species, the fisher was valued by a significant cross-section of the public (*e.g.*, sportsmen, the timber industry) and its restoration in many locations was prompted by economic concerns as well as the restoration of ecological integrity.

There have been at least 35 fisher translocation projects within the historical range of the fisher (*i.e.*, reintroductions and augmentations) since the 1940s; most were successful, however a much greater proportion of translocation failures occurred in western North America (Lewis *et al.* 2012). These 35 translocations presented opportunities for biologists and managers to evaluate the translocation process and to report their findings in the literature, however little information about these efforts has been published (Lewis *et al.* 2012). Consequently there is a significant lack of information available for past translocations to inform future translocation efforts.

Given the expensive, time-consuming, and complicated nature of carnivore translocations, this lack of information only adds to the challenges and uncertainties that accompany many translocations.

In Washington, the fisher had been extirpated since the mid-1900s despite the closure of the fisher trapping season in 1934, the year that the Washington Department of Fish and Wildlife was established (Lewis and Stinson 1998); the last known specimen was a fisher that was trapped on the Olympic Peninsula in 1969 (Lewis and Stinson 1998). Because fishers are easily trapped (Powell 1993), it was common for them to be incidentally captured in traps set for other species (Lewis and Zielinski 1996). The high value of their pelt also made fishers vulnerable to illegal harvests (Lewis and Stinson 1998). The extirpation of the fisher in Washington prompted the Washington Fish and Wildlife Commission to list the fisher as a state endangered species in 1998 (Hayes and Lewis 2006). In 2004, the U.S. Fish and Wildlife Service (2004) found that the fisher's West Coast distinct population segment (*i.e.*, western Washington, western Oregon, and California) was warranted for listing as threatened or endangered, but its listing was precluded by higher priority listing actions. The fisher is currently the subject of a federal status review to determine if it should be listed as threatened or endangered in all or part of its West Coast range (U.S. Fish and Wildlife Service 2013).

As part of the fisher status review conducted in Washington (Lewis and Stinson 1998), WDFW determined that fisher recovery in the state would require reintroductions. Given the uncertainty that a reintroduction would be successful, WDFW conducted a feasibility assessment for reintroductions within the historical range of fishers in western Washington (Lewis and Hayes 2004). Lewis and Hayes (2004) found that reintroductions could be successful on the Olympic Peninsula and in the Cascade Range and that the Olympic Peninsula was the most suitable location for the first reintroduction. Following the development of a recovery plan for the fisher in Washington (Hayes and Lewis 2006), a reintroduction implementation plan (Lewis 2006), and a National Environmental Policy Act analysis (National Park Service 2007), WDFW and the National Park Service (Olympic National Park) initiated a fisher reintroduction in Olympic National Park and the larger Olympic Peninsula in the fall of 2007. The first fishers were captured in central British Columbia in December of 2007, and on 27 January 2008 we released the first group of fishers in Washington, 8 females and 5 males.

We saw the reintroduction of 90 fishers (50 F, 40 M) to the Olympic Peninsula from 2008 to 2010 as a unique opportunity to improve our understanding of the behaviors and fates of released fishers by investigating the post-release movements, survival, and resource selection of a large number of radio-collared fishers. An investigation of this type and magnitude had not been attempted before. We hoped that it would help us increase the likelihood of success of the Olympic fisher reintroduction project and those that came after.

Post-Release Movements

Little is known about how fishers move once they've been released. The nature and magnitude of post-release movements could have profound consequences on the success of a translocation effort, and yet only one study has been published that addressed these movements (Proulx *et al.* 1994). The success of a translocation can be associated with the percentage of the released population that occupies a recovery area, where the likelihoods of surviving, locating a suitable home range, and locating a mate, are expected to be the greatest (Miller *et al.* 1999, Stamps and Swaisgood 2007, Spinola *et al.* 2008). We released all 90 fishers within Olympic National Park. Olympic National Park and Olympic National Forest comprise the Olympic Fisher Recovery Area (Hayes and Lewis 2006), where fishers were expected to find expansive areas of high-quality habitat suitable for a home range (Lewis 2006). However, because fishers are likely to move extensively after being released, our goal was to investigate factors (*i.e.*, sex, age, release-year cohort, and release date) that could influence the post-release movements, in the event that these factors could be manipulated to minimize the movements of released fishers (*i.e.*, risk) and increase their opportunity to contribute to translocation success (*i.e.*, establish a home range in or near the recovery area).

Survival

Despite the critical importance of some minimum level of survival to achieve translocation success, there is no published or unpublished information on the survival of translocated fishers. The importance of founder survival cannot be overstated, and monitoring survival (or conversely, mortality) is important during the translocation process to allow mid-course adjustments to increase survival (*e.g.*, avoiding release sites in areas with high incidences of mortality or high levels of emigration). With unlimited resources, managers could continue to release founder individuals until a self-sustaining population had become established, however most managers have limited resources and must use the best available information to conduct a translocation with a high likelihood of success. In the absence of information on survival, we chose to release ~100 fishers within a high-quality reintroduction area over three years, and we were fortunate to obtain a founder population with a slightly female-biased sex-ratio (50 F, 40 M). Our research goal was to identify factors that were related to survival, especially those factors that managers could manipulate. With a better understanding of the influence of factors such as sex, age, weight, number of intact canines, the duration of captivity, the timing of release, or the presence of conspecifics in a release area, managers can compose a founder population and implement a translocation program that maximize founder survival.

Resource Selection

The fisher population in Washington was extirpated before any field studies could investigate the ecological relationships of fishers within the state. Consequently, as part of the reintroduction feasibility assessment, Lewis and Hayes (2004) used published information on the habitat associations of fishers from other states and provinces to identify suitable habitats and reintroduction areas in Washington. Based on the literature, they identified late-seral forests at low and mid-elevation as high quality habitat, especially for females. When identifying suitable reintroduction areas, Lewis and Hayes (2004) identified a large area dominated by high-quality habitat within the Olympic Peninsula, which represented the highest quality reintroduction area within the state. This reintroduction area was similar in extent to the Olympic Fisher Recovery Area identified in the Washington State Fisher Recovery Plan (Hayes and Lewis 2006). These findings prompted the choice of the Olympic Peninsula as the first area to restore fishers in Washington.

We assumed that fishers would find the Olympic reintroduction area suitable and that it would support a self-sustaining fisher population even if lands outside this area did not appreciably support this population. Our hope was that the main limiting factor for establishing fishers on the Olympic Peninsula was the lack of a founder population, and our goal was to provide that population. Recent studies (Weir and Corbould 2010, Sauder and Rachlow 2014) and reviews (Lofroth *et al.* 2011 and Raley *et al.* 2012) on fisher habitat selection characterized managed forest landscapes in western North America that support populations of fishers. In some portions of their range fishers may only require landscapes with closed-canopy forests that have sufficient large woody structures available for rest and den sites (Raley *et al*. 2012). If, however, fishers were closely tied to unmanaged landscapes dominated by late-seral forests to meet their needs in Washington (*e.g.*, Olympic National Park, Olympic National Forest), large portions of their historical range in the state would likely be unoccupied (*e.g.*, southwest Washington). An investigation into the selection of home ranges established by translocated fishers would be instrumental in identifying the habitats and landscapes necessary to recover fishers within their historical range in Washington.

Similar to studies of survival of translocated fishers, there is no published information on the resource selection for translocated fishers. We translocated 90 fishers from the managed sub-boreal forests of central British Columbia and released them on protected federal lands within a temperate forest ecosystem. While our founder individuals were released into an unfamiliar environment and may have been inefficient at exploiting it, they provided us with an opportunity to evaluate home range selection by a large number of fishers, and to evaluate home range selection by sex. Furthermore this research would provide a baseline for future research to determine if the selection of home ranges differs between founders and their descendants.

LITERATURE CITED

Bailey, V.O. 1936. The mammals and life zones of Oregon. North American Fauna 55:1-416.

Berg, W.E. 1982. Reintroduction of fisher, pine marten and river otter. Pages 159-175 in G.C. Sanderson, editor. Midwest furbearer management. North Central Section of The Wildlife Society, Bloomington, Illinois.

Dalquest, W.W. 1948. Mammals of Washington. University of Kansas, Lawrence, Kansas.

Grinnell, J., J.S. Dixon and J.M. Linsdale. 1937. Fur-Bearing Mammals of California: Their Natural History, Systematic Status and Relations to Man. Volume 1. University of California, Berkeley.

Hayes, G.E. and J.C. Lewis. 2006. Washington State recovery plan for the fisher. Washington Department of Fish and Wildlife, Olympia, Washington. 69 pp. (at: http://wdfw.wa.gov/wlm/diversty/soc/fisher/).

Lewis, J.C. 2006. Implementation plan for reintroducing fishers (*Martes pennanti*) to Olympic National Park. Washington Department of Fish and Wildlife, Olympia.

Lewis, J.C. and G.E. Hayes. 2004. Feasibility assessment for reintroducing fishers to Washington. Washington Department of Fish and Wildlife, Olympia. 70 pp. (at: http://wdfw.wa.gov/wlm/diversty/soc/fisher/)

Lewis, J.C., R.A. Powell and W. J. Zielinski. 2012. Carnivore translocations and conservation: insights from population models and field data for fishers (Martes pennanti). PLoS ONE 7(3): e32726.

Lewis, J.C. and D. W. Stinson. 1998. Washington State status report for the fisher. Washington Department of Fish and Wildlife, Olympia, Washington.

Lewis, J.C. and W.J. Zielinski. 1996. Historical harvest and incidental capture of fishers in California. Northwest Science 70:291-297.

Lofroth, E.C., C.M. Raley, J.M. Higley, R.L. Truex, J.S. Yaeger, J.C. Lewis, P.J. Happe, L.L. Finley, R.H. Naney, L.J. Hale, A.L. Krause, S.A. Livingston, A.M. Myers and R.N. Brown. 2010. Volume I - Conservation Assessment for Fisher (*Martes pennanti*) in South-central British Columbia, Western Washington, Western Oregon, and California. USDI Bureau of Land Management, Portland, OR.

Lofroth, E.C., J.M. Higley, R.H. Naney, C.M. Raley, J.S. Yaeger, S.A. Livingston and R.L. Truex. 2011. Conservation Assessment for Fisher (*Martes pennanti*) in South-central British Columbia, Western Washington, Western Oregon, and California – Volume II: Key findings from fisher habitat studies in British Columbia, Montana, Idaho, Oregon, and California. USDI Bureau of Land Management, Denver, CO.

Miller, B., K. Ralls, R.P. Reading, J.M. Scott and J. Estes. 1999. Biological and technical considerations of carnivore translocation: a review. Animal Conservation 2:59-68.

Powell, R.A. 1993. The fisher: Life history, ecology, and behavior, 2nd edition. University of Minnesota Press, Minneapolis, Minnesota.

Proulx, G.A., J. Kolenosky, M. Badry, M.K. Drescher, K. Seidel and P.J. Cole. 1994. Post-release movements of translocated fishers. Pages 197-203 in S.W. Buskirk, A. Harestad, M. Raphael and R.A. Powell, editors. Martens, Sables, and Fishers: Biology and Conservation. Cornell University Press, Ithaca, New York.

Raley, C.M., E.C. Lofroth, R.L. Truex, J.S. Yaeger and J.M. Higley. 2012. Habitat ecology of fishers in western North America: a new synthesis. Pages 231-254 in K.B. Aubry, W.J. Zielinski, M.G. Raphael, G. Proulx and S.W. Buskirk, editors. Biology and Conservation of Martens, Sables, and Fishers: A New Synthesis. Cornell University Press, Ithaca, New York.

Sauder, J. and J.L. Rachlow. 2014. Both forest composition and configuration influence landscape-scale habitat selection by fishers (*Pekania pennanti*) in the Rocky Mountains of Idaho. Forest Ecology and Management 314:75-84.

Seton, E.T. 1926. Lives of Game Animals. Vol. 2. Doubleday, Doran & Co., New York.

Spinola, R.M., T.L. Serfass and R.P. Brooks. 2008. Survival and post-release movements of river otters translocated to western New York. Northeastern Naturalist 15(1):13-24.

Stamps, J.A. and R.R. Swaisgood. 2007. Someplace like home: experience, selection, and conservation biology. Applied Animal Behaviour Science 102:392-409.

U.S. Fish and Wildlife Service. 2004. Endangered and threatened wildlife and plants: 12-month finding for a petition to list the west coast distinct population segment of the fisher (*Martes pennanti*). Federal Register 69(68):18770-18792.

U.S. Fish and Wildlife Service. 2013. Endangered and Threatened Wildlife and Plants; Status Review of the West Coast Distinct Population Segment of the Fisher as Endangered or Threatened. Federal Register 78:16828-16830.

Weir, R.D. and F.B. Corbould. 2010. Factors affecting landscape occupancy by fishers in north-central British Columbia. Journal of Wildlife Management 74(3):405–410.

CHAPTER 1

Post-Release Movements of Fishers Translocated to the Olympic Peninsula of Washington

JEFFREY C. LEWIS, *University of Washington, School of Environmental and Forest Sciences, Seattle, WA 98195 and Washington Department of Fish and Wildlife, 600 Capitol Way N, Olympia, WA 98501*

PATRICIA J. HAPPE, *Olympic National Park, 600 E. Park Ave., Port Angeles, WA 98362*

KURT J. JENKINS, *U.S. Geological Survey, Forest and Rangeland Ecosystem Science Center, 600 E. Park Ave., Port Angeles, WA 98362*

DAVID J. MANSON, *Olympic National Park, 600 E. Park Ave., Port Angeles, WA 98362*

INTRODUCTION

Translocations are among the tools commonly used to protect species at risk of extinction and include reintroductions, augmentations, introductions and assisted colonizations (IUCN 1987, Neilson 1988, Seddon 2010). In the future, translocations are expected to be used more frequently as the persistence of many species relies on perpetual conservation efforts (Scott *et al.* 2010) and as species are affected by climate change (Thomas *et al.* 2004, Schwartz *et al.* 2006, Lawler *et al.* 2009, Schwartz and Martin 2013). Although translocations have been used extensively, many have failed to restore self-sustaining populations (Griffith *et al.* 1989, Wolf *et al.*1996, Fischer and Lindenmayer 2000). To address the uncertain success of translocations, wildlife managers have been strongly encouraged to incorporate active monitoring programs as part of a translocation project (IUCN 1987, 1995, 2012). Survival, reproduction, and home range establishment are often monitored as indicators of founder population performance and reintroduction success (Armstrong and Seddon 2008), and the movements of released individuals influence each of these measures, and therefore, translocation success (Van Heezik *et al.* 2009, Mihoub *et al.* 2011, Bodinof *et al.* 2012).

Following translocation, released individuals must move throughout a foreign environment to find cover and food, avoid predators, locate potential mates, and find a suitable home range. These movements can place an individual at risk, and this risk is likely to increase with post-release movements of greater distance or duration. Managers are particularly concerned with movements that result in individuals leaving the reintroduction area, where they are likely to encounter less suitable habitat, fewer potential mates, or a greater number of predators and hazards (Miller *et al.* 1999, Stamps and Swaisgood 2007, Spinola *et al.* 2008). Movements outside the reintroduction area may limit the contributions that these individuals can make to translocation success (Armstrong and Seddon 2008, Van Heezik *et al.* 2009, Ryckman *et al.* 2010).

Many factors may influence the post-release movements of founder individuals, including the sex and age of the individual, the date it was released, and whether it was released into an area that contained conspecifics (Sjoasen 1997, Van Heezik *et al.* 2009, Ryckman *et al.* 2010, Mihoub *et al.* 2011). These four factors can be controlled by wildlife managers, and if these factors are related to reduced movements (in distance or duration), greater fidelity to a reintroduction area, or more rapid home-range establishment, managers could use this information to design translocation efforts that have a greater likelihood of success. We hypothesized that some or all of these factors would influence the movements of fishers (*Pekania pennanti*) that we translocated to the Olympic Peninsula to recover the species in Washington State.

The fisher is a wide-ranging, mid-sized carnivore (2-6 kg) of the weasel family (Mustelidae) that occurs only in the boreal and temperate forests of North America (Powell 1993). It is also among the most successfully translocated carnivores (Reading and Clark 1996, Breitenmoser *et al.* 2001, Powell *et al.* 2012). However, despite the occurrence of at least 37 fisher translocations since 1940 (Lewis *et al.* 2012), little information about fisher translocations has been published, and only one published study monitored the post-release movements of radio-collared fishers (Proulx *et al.* 1994).

Because of the high value of its pelt in the late 1800s and early 1900s (up to $350 per pelt; Seton 1926, Bailey 1936, Grinnell *et al.* 1937, Dalquest 1948), the fisher was extirpated in much of the southern portion of its range (*i.e.,* the northern United States and southern Canada; Lewis *et al.* 2012), largely as a result of over-exploitation (Powell 1993). Translocations have been important for restoring the fisher throughout much of this area (Berg 1982, Powell 1993, Aubry and Lewis 2003, Lewis *et al.* 2012). In 1998, the fisher was listed as an endangered species in Washington state (Hayes and Lewis 2006), and it is currently a candidate for federal listing as threatened or endangered throughout its West Coast range (western Washington, western

Oregon, and California; U.S. Fish and Wildlife Service 2004); a 12-month status review of the fisher is currently underway (U.S. Fish and Wildlife Service 2013).

Data from previous fisher translocations are largely anecdotal; however, they suggest that post-release movements of translocated fishers can vary in distance, duration, and outcome (Weckworth and Wright 1968, Roy 1991, Heinemeyer 1993, Proulx *et al.* 1994, Weir 1995, Fontana *et al.* 1999, Serfass *et al.* 2001). In established populations, the movement patterns of fishers differ between the sexes (Powell 1993, Lofroth *et al.* 2010). For example, males use larger home ranges than females (Arthur *et al.* 1989, Powell 1994, Zielinski *et al.* 2004, Weir *et al.* 2009) and adult males frequently move extensively during the breeding season (Leonard 1986, Arthur *et al.* 1989, Aubry *et al.* 2004, Weir and Corbould 2007). Consequently, we predicted that translocated males would move greater distances following release than females (Proulx *et al.* 1994, Fontana *et al.* 1999).

Movements and home range-establishment patterns are also likely to vary by age-class. An unknown proportion of the adult females released are pregnant (due to delayed implantation) and will need to establish home ranges prior to the birthing season (March and April), which may be shortly after their release. Because adult males often move extensively during the breeding season, home range establishment by adult males may be delayed by the breeding season and occur later than for juvenile males.

The duration of post-release movements and the timing of home range establishment may also vary in relation to release date. While managers can control the date when fishers are released, the initiation of the breeding and denning seasons are fixed (~1 March and ~1 April, respectively; Powell 1993, Lofroth *et al.* 2010). The establishment of a home range may be delayed for adult males that are released shortly before the beginning of the breeding season, because they will have had little time to locate a suitable home range before their attention turns to locating reproductive females. The establishment of a home range by females is likely to make them more detectable by males, and increase the likelihood of female reproductive success. However, home range establishment may be hurried for females that are released shortly before the breeding season, especially for those that are pregnant.

Fisher translocations often involve the release of individuals during 2 or more years (Lewis *et al.* 2012), whereby each group of fishers released annually represents a different release-year cohort. In year 1 of a reintroduction, fishers will be released into areas where no resident fishers occur and where no environmental cues of fisher presence exist, other than their own. In subsequent years, fishers could be released in or near areas that are already occupied by resident fishers that were released in previous years. The presence of resident fishers is likely to influence the behavior of newly released fishers as they seek mates, interact with competitors, and establish home ranges. Conspecific attraction has been reported for many species, even territorial species (Stamps 1988, Smith and Peacock 1990, but see Sjoasen 1997), which suggests that fishers may be drawn to areas where previous release cohorts have become established and may initiate home ranges sooner than those in the first release-cohort.

From 2008 to 2010, we translocated 90 fishers (50 F, 40 M) from central British Columbia to Olympic National Park to reestablish a self-sustaining population in Washington State. In this study our goals were to describe the post-release movements of fishers and determine how their movements are influenced by factors that managers can control to increase the likelihood of translocation success. Specifically, our objectives were to 1) characterize the post-release movements of translocated fishers and evaluate how these movements may be influenced by sex, age, release-year cohort, and month, 2) describe home range establishment and evaluate factors (*i.e.*, sex, age, release date, release-year cohort) that may influence either the timing of establishment or the distance from a release site to an established home range, 3) estimate the size and location of home ranges and home range core-areas for fishers, and 4) characterize exploratory forays made by fishers outside their home range.

STUDY AREA

The study area (14,412 km^2 [5,564 mi^2]) included most of the Olympic Peninsula in western Washington, which is bordered by the Pacific Ocean to the west, the Strait of Juan de Fuca to the north, and Puget Sound to the east (Figure 1.1). The study area was delineated by the areal extent of all telemetry locations obtained for all fishers released on the Olympic Peninsula (with the exception of an exploratory movement outside the study area by 1 male). The center of the Peninsula is dominated by the Olympic Mountains as well as glaciated headwaters and steep drainages that radiate outward in all directions. Elevations range from sea level to 2,415 m

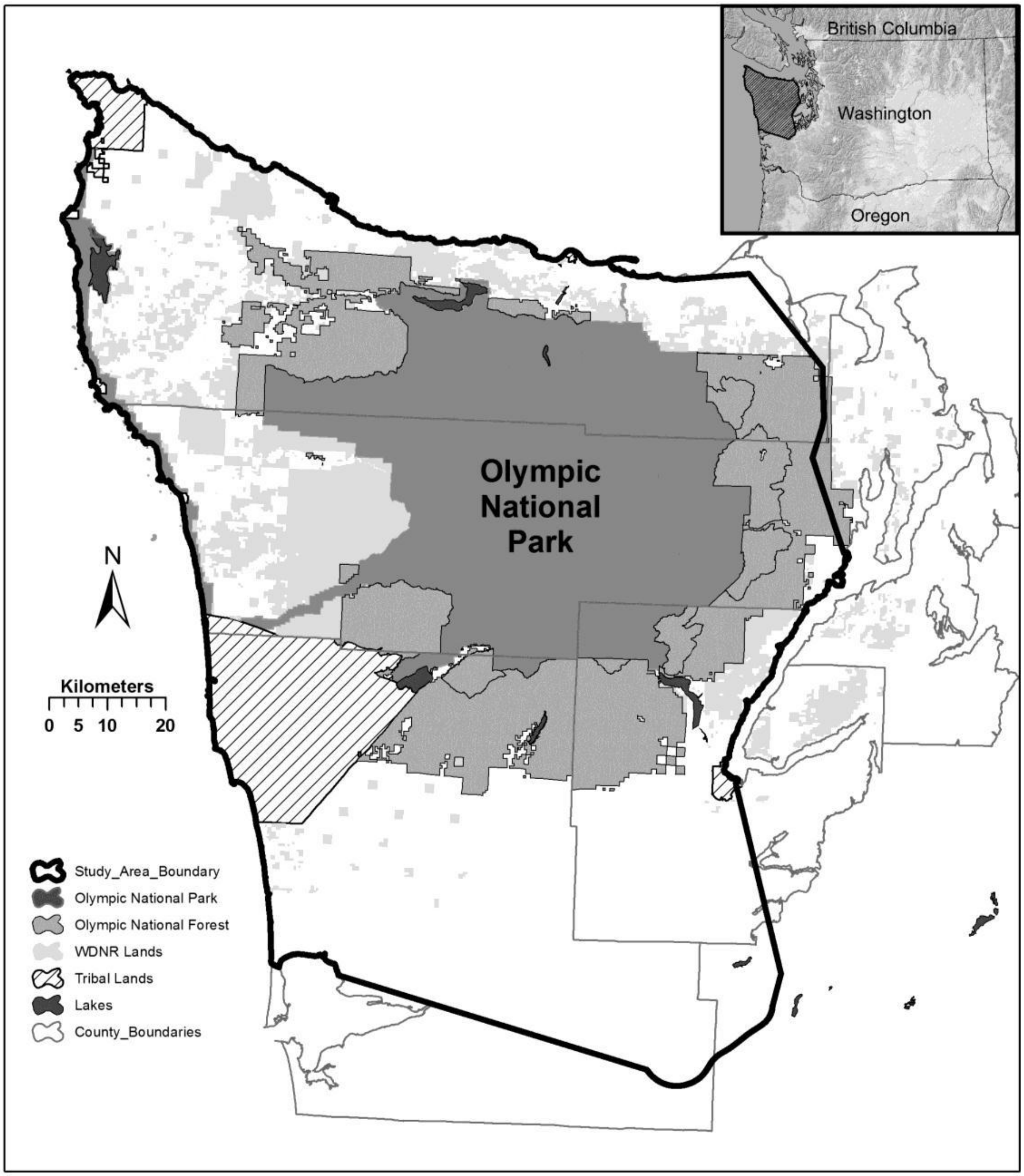

Figure 1.1. The study area of the Olympic fisher reintroduction project on the Olympic Peninsula, Washington, 2008-2011.

(7,923 ft) at the top of Mt. Olympus near the center of Olympic National Park. The mountainous center of the Peninsula slopes to a pronounced coastal plain to the west and smaller plains to the north and east. The Olympic Peninsula has a temperate maritime climate characterized by warm summers and cool winters (Peel *et al.* 2007). The one exception to this pattern is the northeastern corner of the Peninsula, which is in the rain shadow of the Olympic Mountains, and has a more Mediterranean climate characterized by warm and dry summers and cool winters (Peel *et al.* 2007). Annual precipitation ranges from 315 to 500 cm on the west slope of the Olympic Mountains; whereas annual precipitation is typically <40 cm in the northeastern corner of the Olympic Peninsula (National Park Service 1998). Eighty percent of annual precipitation on the Olympic Peninsula falls from October through March. Most winter precipitation falls as rain at elevations below 305 m (1,000 ft) and as snow above 760 m (2,500 ft).

The moist climate and broad range of elevations profoundly affect vegetation patterns throughout the Peninsula. Conifer forests dominate the Olympic Peninsula and include forests in the Sitka spruce zone (generally <180 m elevation but can be as high as 600 m in the western valleys of the Peninsula), western hemlock zone (150-600 m on the west side, 0-1,200 m elevation on the east side), Pacific silver fir zone (*Abies amabilis*; mid-elevations), mountain hemlock zone (*Tsuga mertensiana*; generally >1,070 m elevation) and the subalpine fir zone (*Abies lasiocarpa*; >1,200 m elevation; Franklin and Dyrness 1988, Houston and Schreiner 1994). Forests in the Sitka spruce zone on the western Olympic Peninsula are often referred to as temperate rainforests, and are characterized by large-diameter trees, epiphytic plants, and dense understory vegetation. Hardwoods (red alder [*Alnus rubra*], bigleaf maple [*Acer macrophyllum*] and black cottonwood [*Populus trichocarpa*]) are common in riparian forests along major rivers (Houston and Schreiner 1994, National Park Service 2005).

Olympic National Park includes the mountainous center of the Peninsula as well as mid- and low-elevation forested river drainages. The Park interior contains about 284,955 ha (704,139 ac) of forest and almost all (96%) of which is managed as wilderness. Olympic National Park contains one of the largest areas of contiguous temperate rainforest in North America. Much of the Park's circumference is bounded by Olympic National Forest, which encompasses mountainous terrain in the middle portions of many of the drainages that originate in the Park. Olympic National Park and Olympic National Forest make up the Olympic Recovery Area as outlined in the Washington State Fisher Recovery Plan (Hayes and Lewis 2006). Lower elevation lands outside the Olympic National Forest boundary are owned and managed by the Washington Department of Natural Resources (WDNR), Native American tribes, private timber companies and other private landowners, counties, and local municipalities.

Because of the drier climate in the eastern and northeastern portions of the Olympic Peninsula, wildfires in this area have had a more pronounced role in forest succession than in other areas of the Peninsula (Henderson *et al.* 1989). This history includes numerous fires since the 1850s that did not exceed 4,000 ha (9884 ac) in size and were widely distributed across this portion of the Peninsula. Smaller fires created mid-elevation landscape mosaics that are dominated by unmanaged mid- and late-seral forests in Olympic National Park and Olympic National Forest. A very large fire or series of fires (*i.e.,* 1.2-4.0 million ha [3-10 million ac]) occurred around 1701 within a 15-20 km band of forest along the northern and eastern periphery of the Olympic Peninsula. Remnant stands from the 1701 fire occur throughout the eastern portions and much of the northern portions of Olympic National Forest (Henderson *et al.* 1989). Whereas fires have been prevalent historically in the eastern and northeastern portion of the Olympic Peninsula, large fires have been much less frequent in the southern and western Olympic Mountains. Other disturbances including wind-throw and landslides have also altered forest structure and the forest landscape mosaic throughout the study area (Henderson *et al.* 1989).

Forest lands have been managed intensively on Olympic National Forest and WDNR lands since the 1950s; however, both ownerships have retained areas of unmanaged conifer forests. As part of the Northwest Forest Plan (USDA Forest Service and USDI Bureau of Land Management 1994), much of Olympic National Forest is now managed as late-successional forest reserves, where older forests are protected and younger forests are managed to accelerate the development of older forest characteristics. Washington Department of Natural Resources' lands on the Olympic Peninsula are managed under the guidance of a Habitat Conservation Plan (Washington Department of Natural Resources 1997), which targets the retention of older forests for northern spotted owls (*Strix occidentalis caurina*) and marbled murrelets (*Brachyramphus marmoratus*) in portions of the western Olympic Peninsula. Private and tribal lands occur at lower elevations at the periphery of National Forest and WDNR lands; these lands are dominated by second-growth forests as a result of intensive timber management practices.

METHODS

We released 90 fishers (50 F, 40 M) at 21 sites in Olympic National Park from 27 January 2008 to 20 February 2010 (Table 1.1) (Lewis and Happe 2008, Lewis *et al.* 2011). Fishers were released throughout road-accessible portions of the park to promote their occupancy in landscapes dominated by late-successional conifer forests. All fishers were immediately released upon arrival at their release site; there was no acclimation period prior to release, as has been the case for some fisher reintroductions (*i.e.*, soft releases; Roy 1991, Heinemeyer 1993, Weir 1995). Of the 90 fishers released, 82 (50 F, 32 M) were equipped with a 40-g VHF radio-collar (model MI-2 with mortality sensor, Holohil Systems Ltd., Carp, Ontario, Canada; http://www.holohil.com). Of the remaining eight fishers, five males (each weighing >4.5 kg) were equipped with a 120-g Argos satellite collar (Kiwisat 202, Sirtrack Ltd, Havelock North, New Zealand; http://www.sirtrack.com) and three males were equipped with a 41-g VHF transmitter that was surgically implanted in their abdominal cavity (model IMP/310/L with mortality sensor, Telonics Inc., Mesa, AZ, USA; http://www.telonics.com). Animal handling procedures met or exceeded the guidelines of the American Society of Mammalogists for the use of wild mammals for research (Sikes and Gannon *et al.* 2011) and were approved by Washington Department of Fish and Wildlife's Animal Care and Use Committee.

Radio-Telemetry

Given the remote nature and the broad geographic extent of the study area (Figure 1.1), we located fishers using aerial radio-telemetry methods. Telemetry flights originated from airports in Port Angeles, Shelton, and Olympia, Washington. We attempted to locate each fisher at least once per week; however, inclement weather prevented flights during some weeks, and the large size and rugged nature of the study area, the short transmission distance of the radio-collars, and the extensive movements of many fishers, made it impossible to locate each fisher every week. During the breeding season (~1 March to ~30 June), we also prioritized efforts to locate female fishers more frequently in an attempt to detect reproductive activity. We located fishers by scanning the frequencies of radio-collar transmitters until a signal was detected. After a transmitter was detected, the pilot used standard aerial-telemetry methods to home in on radio-collared fishers and locate them as precisely as possible. Once we isolated the fisher to a specific location, we recorded that location with a global positioning system (GPS) and gave it an

Table 1.1. Number of fishers released on the Olympic Peninsula from 27 January 2008 to 20 February 2010, by release-year cohort, release dates, sex and age-class.

	Fisher age-class[a]		
Release-year cohort 1 27 Jan 2008 2 Mar 2008	Juveniles	3	5
	Adults	9	1
	Total (18)	12	6
Release-year cohort 2 21 Dec 2008 17 Jan 2009 23 Feb 2009	Juveniles	6	11
	Adults	14	0
	Total (31)	20	11
Release-year cohort 3 24 Dec 2009 21 Jan 2010 20 Feb 2010	Juveniles	7	16
	Adults	11	7
	Total (41)	18	23
	Grand Total (90)	**50**	**40**

[a] For females, juveniles were <1 year old, and adults were ≥1 year old. For males, juveniles were <2 years old, and adults were ≥2 years old.

accuracy rating based on the flight biologist's judgment of signal strength, altitude of the aircraft, and time spent locating the fisher. An accuracy rating of 1.5 was given to locations with an estimated error <500 m, a rating of 2 for locations with an estimated error <1 km, a rating of 3 for locations with an estimated error <5 km, and a rating of 4 for locations with an estimated error >5 km. A rating of 1 was given to ground locations where the location was known (*e.g.*, carcass found, visual observation, fisher occupying a known den) or considered to have an error ≤100 m. We used standard ground-telemetry techniques to locate fishers at dens and rest sites, and to recover dead fishers.

We evaluated the accuracy of aerial locations by placing test collars at 31 known locations within the study area that were representative of the range of topographic and vegetation conditions occupied by fishers. Because the locations of test collars were unknown to our pilots, we obtained unbiased estimates of location errors by having our pilots estimate test collar locations with the same intensity of search effort they used when locating a fisher. Locations given an accuracy rating of 1.5 had a mean location error of 281 ± 47 m (± SE) and locations given an accuracy rating of 2 had a mean location error of 420 ± 117 m. Thus, the accuracy of test-collar locations was generally greater than indicated by the accuracy rating.

For analyses of post-release movements and home range estimation, we used only locations with accuracy ratings ≤3; <5% of the locations used in our analyses had an accuracy rating of 3. To avoid temporal autocorrelation, we used locations collected ≥24 hr apart. Second, we limited the number of ground locations that were included in each individual's data set to 1 during each 7-day period to minimize any bias associated with a greater number of ground locations in more accessible areas. An exception was made for male M011 because the area he inhabited was entirely accessible by vehicle and his movements were consistently monitored from the ground at routine intervals by the staff of the Makah Tribal Forestry Department.

Post-Release Movements

To investigate post-release movements of fishers, we measured the distances between consecutive telemetry locations for each fisher that were ≤31 days apart. We characterized movement distances by month, sex, age-class, and release-year cohort. Because there could be more than one movement for each fisher per month, we calculated a mean movement distance for each individual in each month (if >1 distance was available) and then calculated monthly mean movements for each sex and age-class (*e.g.*, mean movement distance in March 2008 for adult females released in Year 1). Females <1-year old at the time of release were considered juveniles, whereas those ≥1-year old were considered adults because females can become pregnant at 1 year of age (Powell 1993). In contrast, because males do not become effective breeders until they are >2-years old, males that were <2-years old at the time of release were considered juveniles and those ≥2-years old were considered adults (Powell 1993).

We developed models to evaluate the influences of sex, age, release-year cohort, and month (independent variables) on post-release movement distances during the 22-month period following release (from December to September of their second year). Because our data included successive movements by individual fishers, we used mixed general linear models for repeated-measures (using PROC-Mixed in SAS; SAS 9.3; www.sas.com) to evaluate factors associated with fisher movements. We then ranked resulting models using Akaike's Information Criteria (AIC; Burnham and Anderson 2002).

Home Range Establishment

After release, fishers generally explore their new environment prior to establishing a home range (Heinemeyer 1993, Proulx *et al.* 1994, Weir 1995). The nature of this exploration may vary considerably among individuals, but once a translocated fisher finds a suitable location, its movements become more localized and shorter in distance as it establishes a home range. Consequently, we determined the timing of home range establishment by examining the movement patterns of individual fishers. To identify a change in movement distances associated with home range establishment, we calculated the mean-squared distance (MSD) between consecutive locations for each fisher, following the methods of Weir (1995) and Weir *et al.* (2009). We calculated MSD by squaring the six individual distances between the first seven locations (*e.g.*, distance from location 1 to 2, 2 to 3,…6 to 7) then averaging those six distances. We repeated this calculation for all locations in each fisher's dataset in increments of seven locations (*i.e.*, distances between locations 1-7, 2-8, 3-9, etc.) until a MSD value was calculated for all groupings of six consecutive movement distances. Because each distance was squared, one large movement (among the six) significantly increased MSD. To estimate the date when home range establishment began, we plotted MSD (y axis) against date (x axis); we used a minimum of 10 locations collected for ≥2 consecutive months to indicate home range establishment.

We estimated the distance from each fisher's release site to its newly established home range by measuring the distance from the release site to the geographic center of the first 10 home range locations. We also determined the number of days between each fisher's release date and the estimated date of home range establishment. We evaluated univariate relationships among the predictor variables Age-Class, Sex, Release-Year Cohort, and Release Date with the dependent variables Distance to an Established Home Range and Days to Home Range Establishment before including them in a multivariate model. If we found significant univariate relationships, we used a multivariate analysis with PROC GLM in SAS to investigate the influence of Sex, Age-Class, Release- Year Cohort, and Number of Days from Release Date to the Breeding Season (1 March) on Distance to an Established Home Range and Days until Home Range Establishment. We identified parsimonious models of factors related to home range establishment by ranking candidate models with Akaike's Information Criteria for small samples (AIC_c; Burnham and Anderson 2002). We used a chi-square goodness-of-fit test (Zar 1999) to determine if the timing (*i.e.*, month) of home range establishment for males and females exhibited a uniform temporal distribution.

Home Range Size

We estimated home range sizes for fishers with ≥20 locations collected over 6 or more consecutive months following the estimated date of home range establishment. To allow meaningful comparisons with other studies, we used two methods that have commonly been used to estimate the configuration and size of fisher home ranges in previous studies. We used the fixed-kernel method with smoothing parameters selected by least-squares cross validation (Worton 1989) to estimate 95% utilization distribution (UD) contours. Similarly, we delineated home range core-areas by generating 50% UD contours with the fixed-kernel method. We also estimated home ranges using the 100% minimum convex polygon (MCP) method. We used Home Range Tools for Arc GIS (Rodgers *et al.* 2007) to delineate home ranges and program Animal Space Use (Horne and Garton 2009) to calculate smoothing parameters.

Fishers are known to make exploratory forays outside their home ranges, and it is common for males to move long distances during the breeding season to search for reproductive females (Leonard 1986, Arthur *et al.* 1989, Aubry *et al.* 2004, Weir and Corbould 2007). We defined exploratory forays as locations >2 interquartile ranges from the median x or y location coordinates when they existed as isolated kernels (containing ≤ 2 locations) separated from the primary home range distribution. Because such forays occur outside home range boundaries, we eliminated all locations associated with such exploratory forays from our home range estimates. We also measured the distances from outlier points to the perimeter of the home range to evaluate the seasonal patterns and magnitude of exploratory movements among radio-collared fishers.

RESULTS

Post-Release Movements

The month of the year and the fisher's sex were the most important factors associated with the movement distances of fishers during the 22 months after release (Table 1.2). The highest ranking model, which included Month, Sex, and the Month*Sex interaction term, received 100% support as the best model ($w_i = 1.0$; Table 1.2). No models that included Age-Class or Release-Year Cohort were competitive. Mean movement distances varied substantially across months, with movements generally greatest during the breeding season (March to June; Figure 1.2). Although movements of males and females exhibited similar seasonal patterns, movement distances of males tended to be greater than those of females during the breeding season.

From December to February in their first year after release, translocated fishers generally moved 5,000-10,000 m between successive locations (Figure 1.2). During the breeding season (1 March to 30 June), mean movement distances were the longest for males and commonly exceeded 10,000 m. In July and August of their first year after release, movement distances were large for females, and during the remainder of the non-breeding season (mid-summer, fall and winter) movement distances for both sexes were the smallest (<5,000 m). The exceptions to these smaller non-breeding season movements were those of fishers from release-year cohort 3, which were strongly influenced by large movements made by a juvenile male in September of year 1, and a juvenile female in November-December of year 1 (Figure 1.2). In year 2,

Table 1.2. Models examining the effects of Sex, Age-Class, Release-Year Cohort, and Month on the movement distances of reintroduced fishers for 22 months after release. Distances were based on 1,866 movements by 88 fishers (48 F, 40 M). Models were ranked using AIC.

Model	**Number of parameters**	**AIC**	**ΔAIC**	**w_i**
Month + Sex + Month*Sex	45	38900.3	0.0	1
Month + Age-Class + Month*Age-Class	45	38959.6	59.3	<0.0001
Month + Cohort + Sex + Age-Class	27	39303.5	403.2	<0.0001
Month + Sex + Age-Class + Sex*Age-Class	26	39312.5	412.2	<0.0001
Month + Sex + Age-Class	25	39335.0	434.7	<0.0001
Month + Cohort	25	39339.7	439.4	<0.0001
Month + Sex	24	39349.9	449.6	<0.0001
Month + Age-Class	24	39356.8	456.5	<0.0001
Month	23	39371.8	471.5	<0.0001
Cohort + Sex + Age-Class	6	39804.8	904.5	<0.0001
Sex + Age-Class + Sex*Age-Class	5	39806.8	906.5	<0.0001
Cohort + Sex	5	39822.7	922.4	<0.0001
Cohort + Age-Class	5	39826.2	925.9	<0.0001
Sex + Age-Class	4	39836.9	936.6	<0.0001
Cohort	4	39840.9	940.6	<0.0001
Sex	3	39854.2	953.9	<0.0001
Age-Class	3	39859.1	958.8	<0.0001

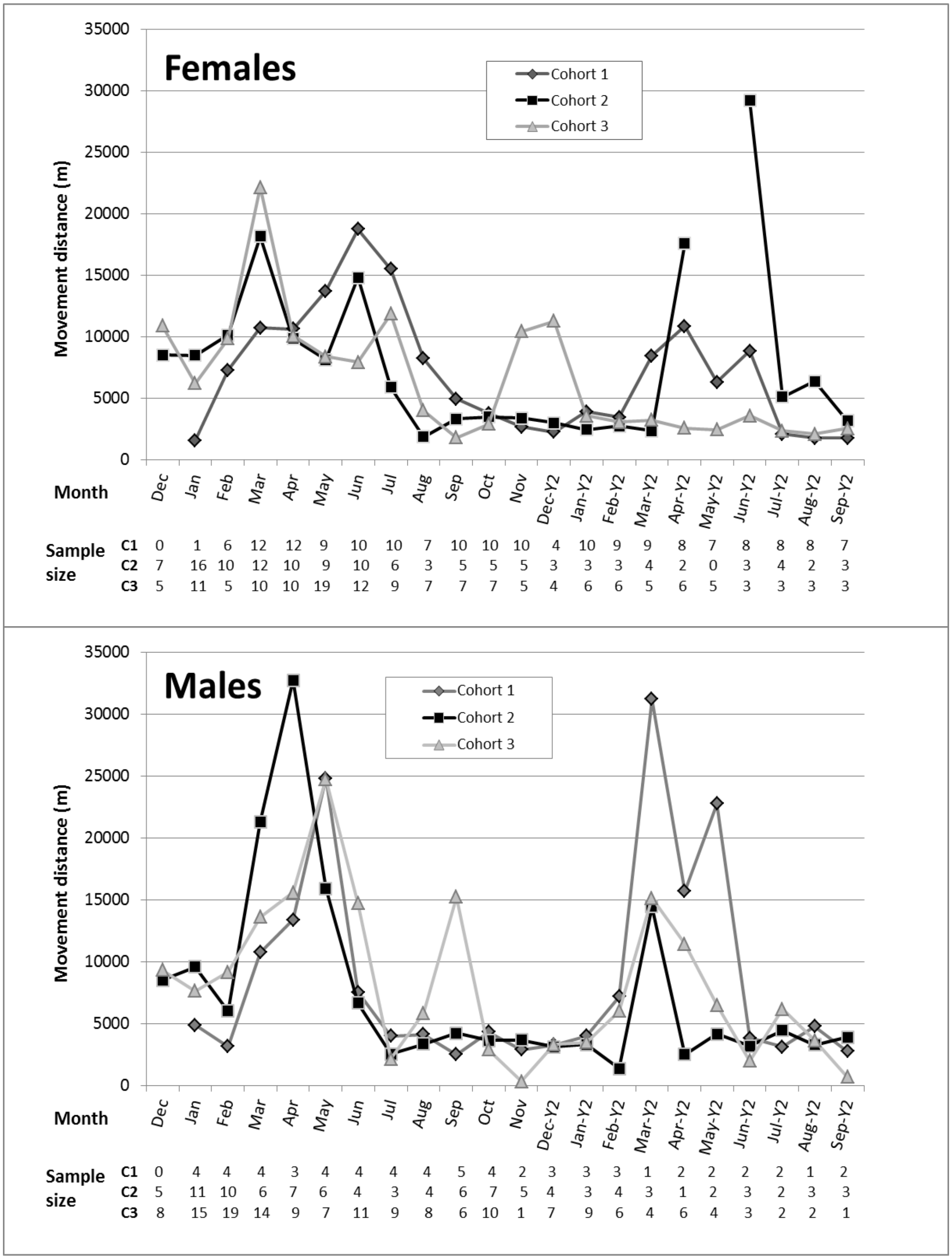

Figure 1.2. Mean post-release movement distances by release-year cohort for female (top) and male (bottom) fishers released on the Olympic Peninsula, Washington, from 2008 to 2010. To allow comparison, we plotted movement distances for each release-year cohort to coincide temporally over a common span of 22 months. Sample sizes for each monthly mean-value are listed below the corresponding month in each graph.

movement distances peaked from March until May for males and from March until June for females from release-year cohorts 1 and 2; these larger movements coincided with the breeding season, as they did in year 1 (Figure 1.2).

Home Range Establishment

Among the 90 fishers released from 2008 to 2010, 48 (27 F, 21 M) met our criteria for home range establishment (Table 1. 3). For the remaining 42 fishers, our data were insufficient to meet the criteria because 1) contact with a fisher was intermittent or lost (radio-collar failure or possible emigration from the study area), 2) the fisher died, 3) a mortality signal was detected but the fisher's status as alive or dead could not be determined, or 4) an individual did not appear to establish a home range while we monitored it (Table 1.3). Males that established a home range moved widely away from release sites and 38% (8 of 21) of males established ranges within the Olympic Recovery Area (Olympic National Park and Olympic National Forest), whereas females moved less widely than males and 67% (18 of 27) of females established home ranges within the Olympic Recovery Area (Figure 1.3).

Movement distances to home ranges and the timing of home range establishment were highly variable. Fishers established home ranges at distances from <3 to >100 km from their release sites (Table 1.4, Figures 1.3 and 1.4), and mean distances were greater for males than for females. Furthermore, the timing of home range establishment ranged from just 2 days to >1 year after release (Table 1.4). Among the nine candidate models we constructed to investigate the factors that influence Distance to an Established Home Range, we found the greatest support for models that included Sex and Release Date, Sex and Age, and Sex alone (Table 1.5). We found little support ($\Delta AIC_c > 2.0$) for models that did not include Sex as a covariate and for one model that included Sex and Cohort (Table 1.5). Because there was only one adult male that established a home range, the effect of Age was largely based on differences in movements between adult females (≥1 year-olds) and those of juveniles of both sexes (males <2 years old; females <1 year old; Table 1.1). Males tended to establish home ranges that were farther from their release sites than females (Table 1.4). The inclusion of Release Date with Sex in the highest ranking model ($w_i = 0.27$) was due to a weak negative relationship between the Distance to an Established Home Range for males and Release Date (Figure 1.5). The model that included Age with Sex was ranked second ($\Delta AIC_c = 0.23$; $w_i = 0.24$) and reflects a weak positive

Table 1.3. Outcome of post-release movements among 90 radio-collared fishers released from 27 January 2008 to 20 February 2010 on the Olympic Peninsula, Washington, by release-year cohort and sex.

	Females				**Males**			
Outcome	**Cohort 1**	**Cohort 2**	**Cohort 3**	**All Females**	**Cohort 1**	**Cohort 2 (n=11)**	**Cohort 3 (n=23)**	**All Males**
Home range established[a]	10 (83%)	7 (35%)	10 (59%)	27 (54%)	5 (83%)	6 (55%)	10 (43%)	2 (53%)
Insufficient data to indicate home range establishment[b]	2 (17%)	13 (65%)	7 (41%)	23 (46%)	1 (17%)	5 (45%)	13 (57%)	1 (47%)

[a] Home range establishment was identified for 48 fishers (27 F, 21 M) that occupied a localized area based on >10 locations collected over >2 consecutive months.

[b] Data were insufficient because 1) contact was intermittent or lost, 2) a mortality occurred (remains were recovered), or 3) a mortality signal was detected but the fisher's status as alive or dead could not be determined, or 4) an individual did not appear to establish a home range while we monitored it.

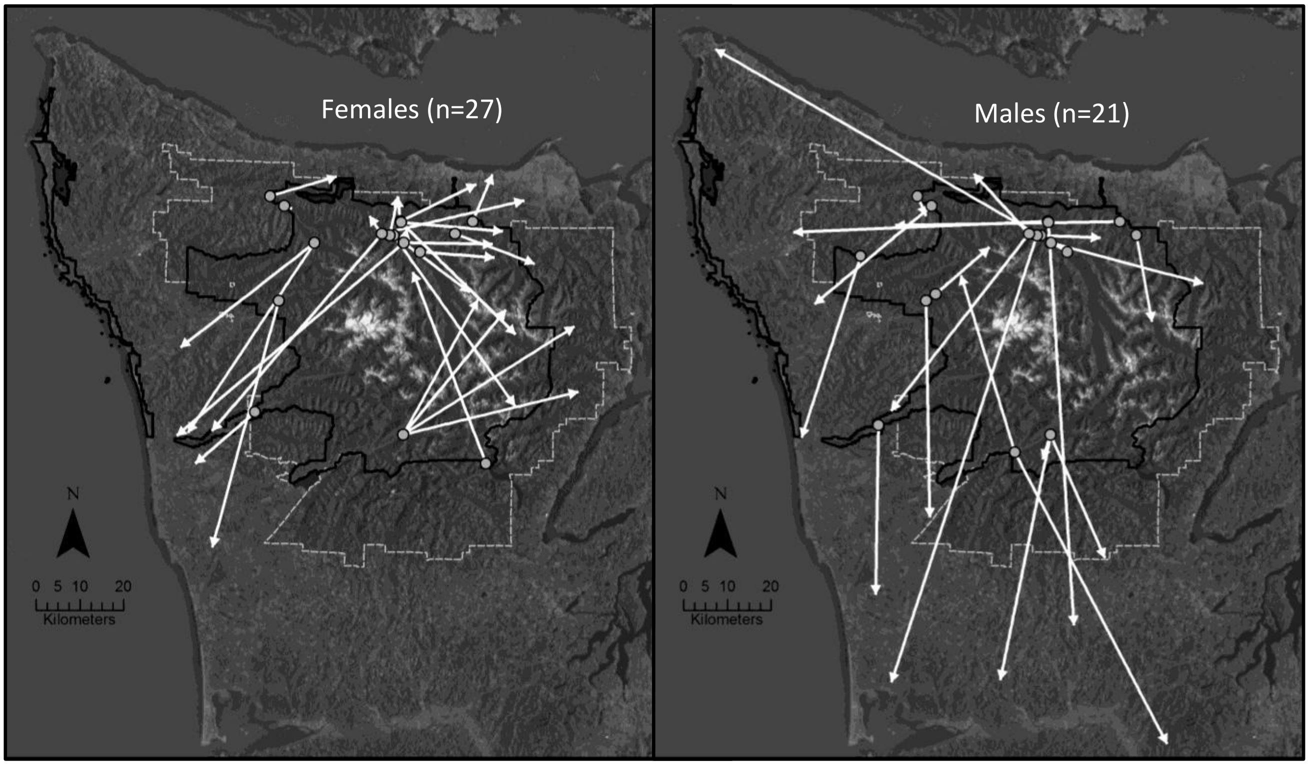

Figure 1.3. Vectors illustrating straight-line distances from release sites to the center of a home range for 27 females and 21 males that established home ranges on the Olympic Peninsula, Washington, 2008-2011. Olympic National Park is delineated by the black line; Olympic National Forest is adjacent to the Park and is delineated by the dashed white line. Together, they comprise the Olympic Fisher Recovery Area.

Table 1.4. Mean distance from the release site to the center of a home range and mean number of days from the release date to the home range establishment date for 48 fishers (27 F, 21 M) that established a home range on the Olympic Peninsula, 2008-2011.

	Distance to HR center (km)			**Days to HR establishment date**		
Population segment	**Mean**	**95%CI**	**Range**	**Mean**	**95%CI**	**Range**
Juvenile females (n = 11)	28.9	17.9-40.7	3.1-60.4	142.7	85.3-200.0	6-270
Adult females (n = 16)	30.9	21.9-39.8	4.1-66.4	143.1	96.4-189.7	2-383
All females (n = 27)	30.1	23.0-37.1	3.1-66.4	142.9	107.4-178.3	2-383
Juvenile males (n = 20)	43.8	30.7-56.8	2.6-107.9	137.1	112.8-161.3	8-272
Adult males (n = 1)	58.4	—	—	155	—	—
All males (n = 21)	44.5	32.0-56.9	2.6-107.9	137.9	114.7-161.0	8-272

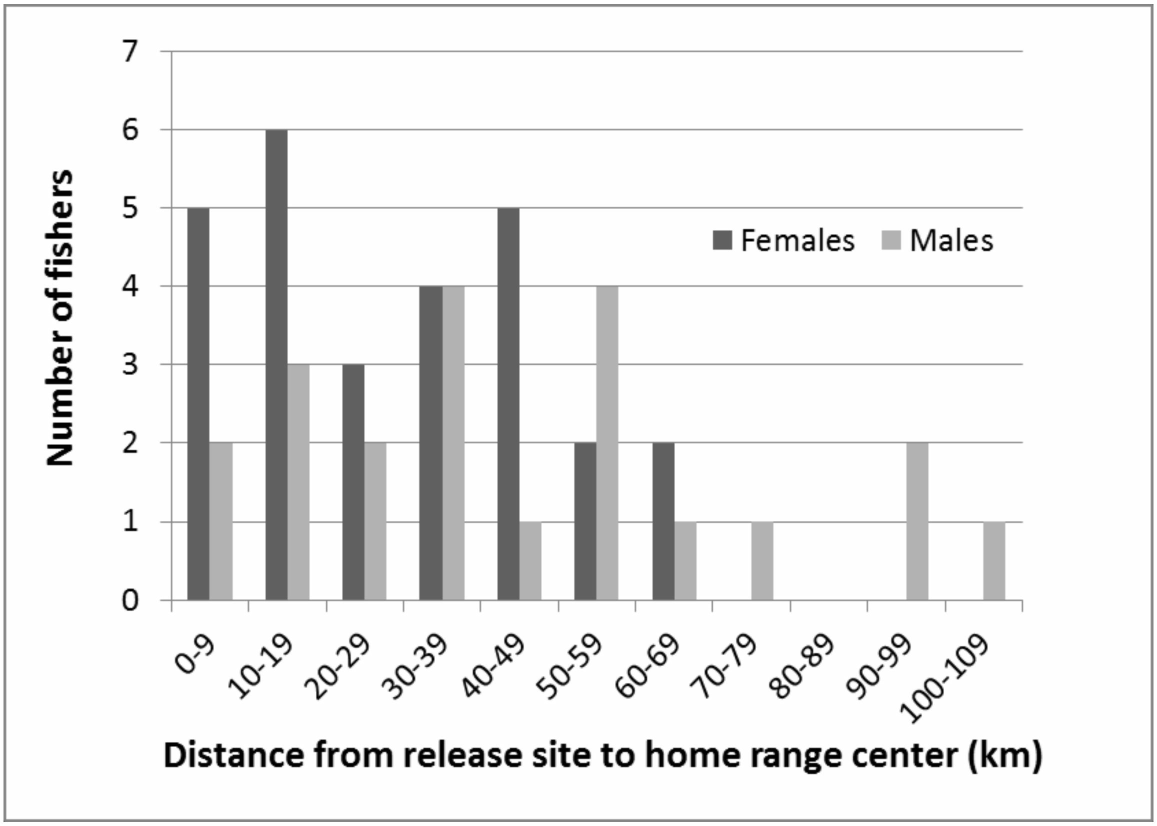

Figure 1.4. Distance from release site to home range center for 27 females and 21 males that established a home range on the Olympic Peninsula, Washington, 2008-2011.

Table 1.5. Models examining the influence of Sex, Age, Release-Year Cohort and days until the breeding season (Release Date) on the distance from a fisher's release site to an established home range. Models were ranked using AIC_c.

Model	**K**	**AIC_c**	**ΔAIC_c**	**w_i**
Sex + Release Date	4	971.72	0.00	0.27
Sex + Age	4	971.94	0.23	0.24
Sex	3	972.04	0.33	0.23
Release Date	3	973.75	2.04	0.10
Sex + Cohort	5	974.75	3.03	0.06
Age	3	975.47	3.75	0.04
Age + Release Date	4	975.66	3.94	0.04
Cohort	4	977.50	5.78	0.02
Age + Cohort	5	979.43	7.71	0.01

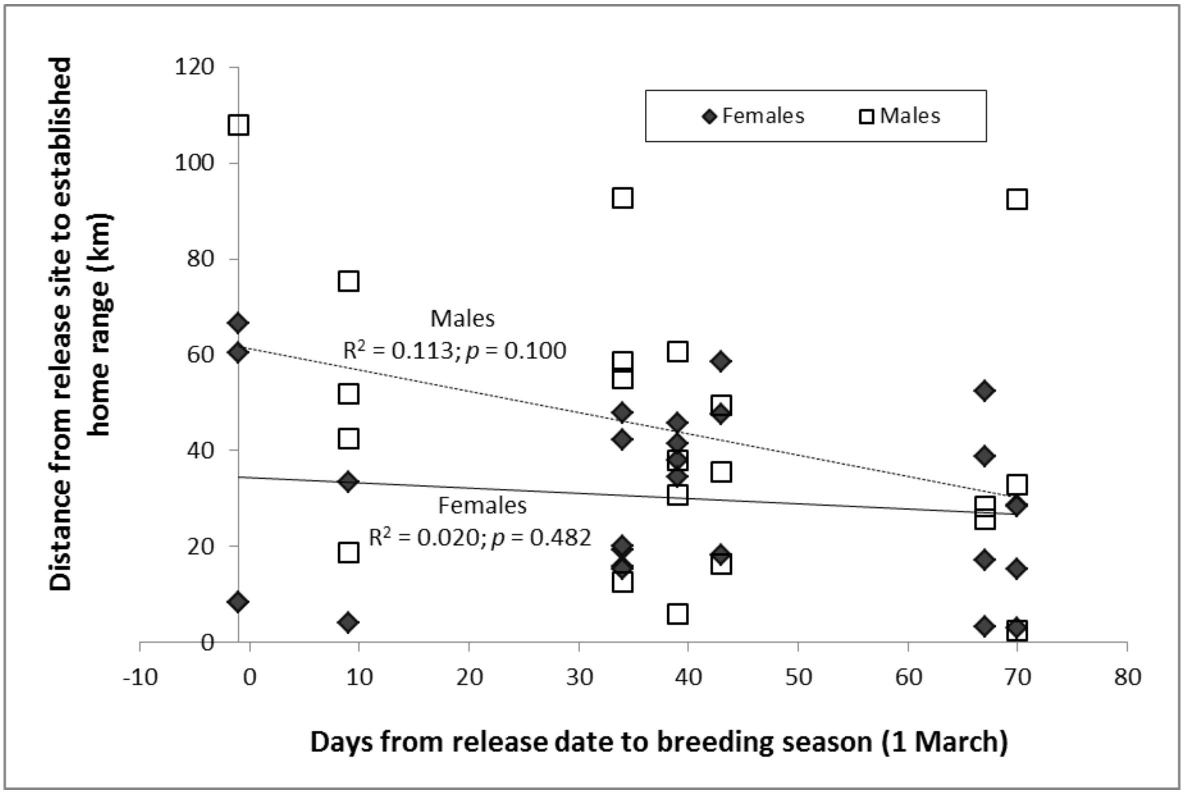

Figure 1.5. Distance from release site to the center of an established home range in relation to the number of days from a release date until the beginning of the breeding season (1 March) for 27 female and 21 male fishers released on the Olympic Peninsula, Washington, 2008-2010. Least-squares linear regression lines were plotted to indicate relationships by sex.

relationship between age and distance from a release site to an established home range for males (Figure 1.6). This relationship largely reflects the small difference in distances we found for males released before they were 1-year old (*n* = 12) with those of males released as 1-year olds (*n* = 8) (Figure 1.6). There was no apparent relationship between Release Date or Age with Distance to an Established Home Range for females (Figures 1.5 and 1.6).

Forty-seven of the 48 fishers that established a home range did so by October, ≤11 months after being released (Figure 1.7); the exception was one female that established a home range 383 days (~13 months) after being released. The temporal distribution of home range establishment during the first 240 days after release was not significantly different from a uniform distribution for females ($\chi^2 = 7.00$, df = 10, $P = 0.725$), but was non-uniform for males ($\chi^2 = 46.571$, df = 10, $P < 0.001$; Figure 1.7). Nineteen of 21 males (90%) that established a home range did so from April to July (Figure 1.7). Thirty-five (73%) of the 48 fishers (of both sexes) that established a home range did so in the 5-month period from April to August following release, whereas 45 (93%) established a home range from February to October (Figure 1.7).

Home Range Size

We had sufficient data to estimate home range sizes for 29 (19F, 10M) of the 90 fishers that were released from 2008 to 2010 (Table 1.6, Figure 1.8). Females home ranges were estimated based on an average of 40.1 ± 3.6 (SE) telemetry locations (range: 20-70), whereas the home ranges of males were based on an average of 30.1 ± 3.3 locations (range: 23-57). Mean home range size for males was larger than for females based on both 95% UD contours (128.2 km^2 vs. 63.4 km^2, respectively; one-tailed $t = 2.82$, df = 12, $p = 0.007$) and the 100% MCP method (79.6 km^2 vs 50.8 km^2, respectively; one-tailed $t = 2.208$, df = 27, $p = 0.017$; Table 1.6, Figure 1.8). Similarly, the mean size of home range core-areas was larger for males than for females based on 50% UD contours (32.8 km^2 vs. 15.3 km^2, respectively; one-tailed $t = 2.985$, df = 12, $p = 0.006$) (Table 1.6). Both male and female fishers moved beyond the boundaries of their home ranges (*i.e.*, forays). We obtained 10 locations for males >10 km from their home ranges, and 14 such locations of females (Figure 1.9). Examination of these locations indicated five forays made by four males (ages 2-4 years), and seven forays made by six females (ages 2-6 years) (Appendix 1.1). Locations that were >10 km outside an individual's home range were obtained only during

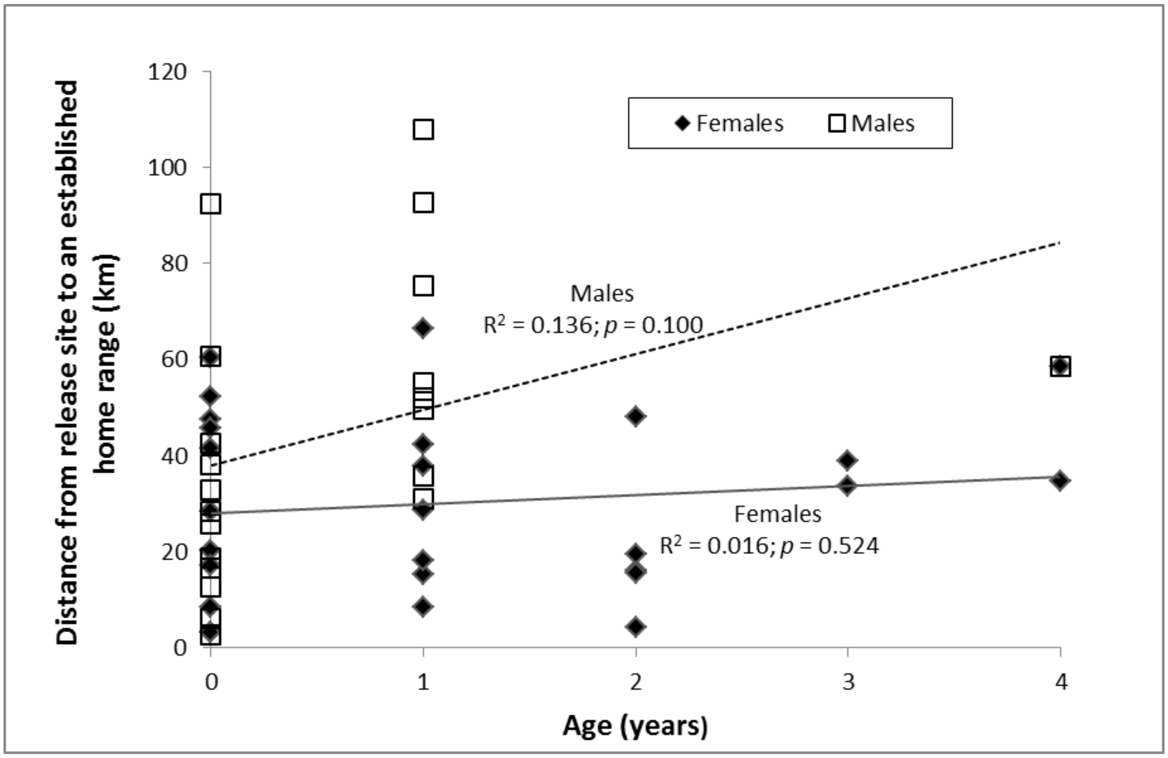

Figure 1.6. Distance from release site to an established home range in relation to sex and age of 27 female and 21 male fishers that established home ranges on the Olympic Peninsula, Washington, 2008-2011. Least-squares linear regression lines were plotted to indicate relationships by sex.

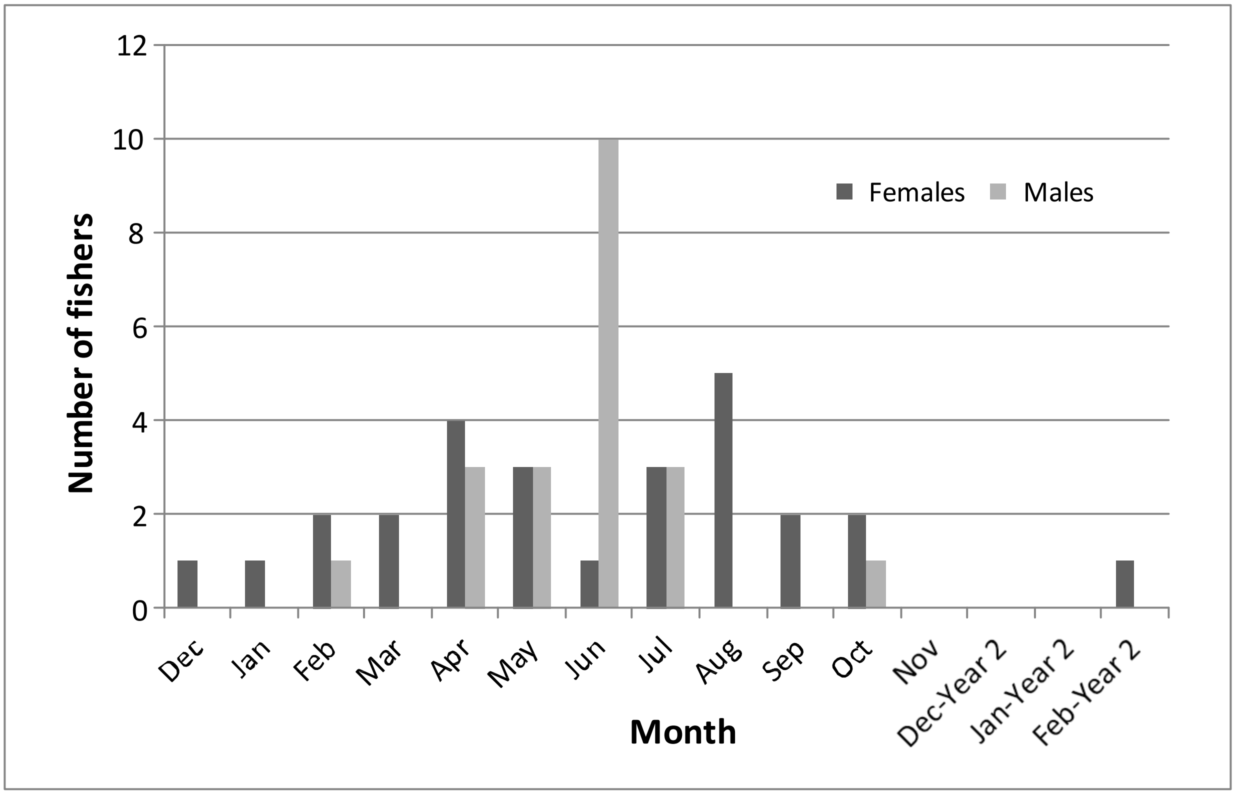

Figure 1.7. The month of establishment for 27 female and 21 male fishers that established home ranges on the Olympic Peninsula, Washington, 2008-2011. Dates of home range establishment were constrained to months following their release; fishers were released in December (n=30), January (n=38), February (n=15) and March (n=7).

Table 1.6. Estimates of home range and core area sizes (km^2) for fishers that established home ranges on the Olympic Peninsula, Washington, 2008-2011; sizes were estimated for fishers with ≥20 home range locations obtained over ≥6 consecutive months.

	Males (n = 10)			Females (n = 19)		
Estimator	**Mean**	**95%CI**	**Range**	**Mean**	**95%CI**	**Range**
100% MCP[a]	79.6	65.2-93.9	38.5 - 188.1	50.8	33.9-67.6	3.2 - 126.6
95% UD[b]	128.3	86.8-169.7	49.9 - 238.7	63.5	45.8-81.1	2.9 - 136.7
50% UD[b] (core area)	32.8	22.1-43.4	12.1 - 63.8	15.3	10.8-19.7	0.6 - 39.1

[a] MCP = Minimum convex polygon estimate of home range

[b] UD = Utilization distribution estimate of a home range (95%) or core area (50%) using a fixed-kernel estimating procedure.

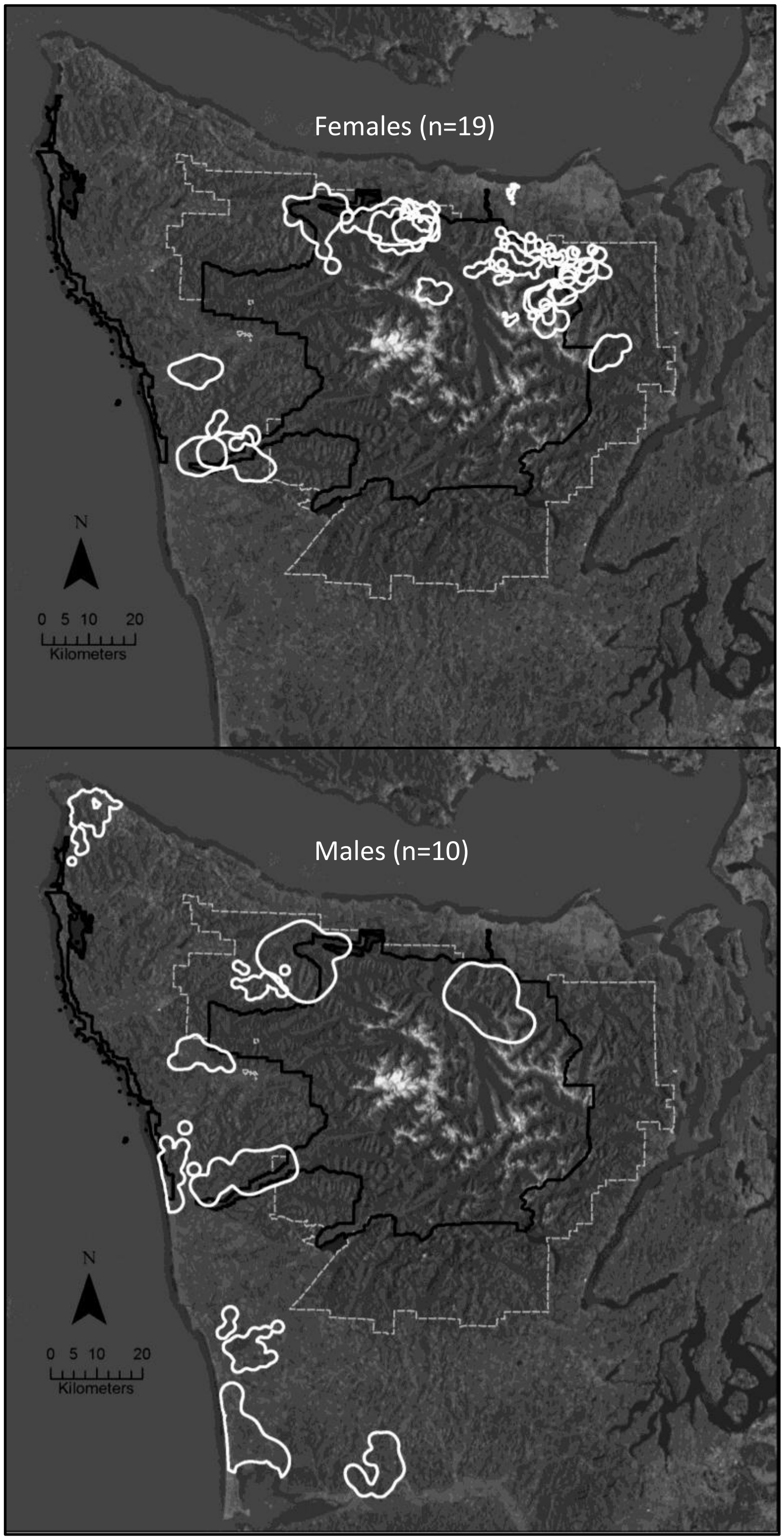

Figure 1.8. Estimated home ranges (95% fixed-kernel contours) for female and male fishers on the Olympic Peninsula, Washington, 2008-2011. Olympic National Park is delineated by the black line, whereas Olympic National Forest is adjacent to the Park and is delineated by the dashed white line.

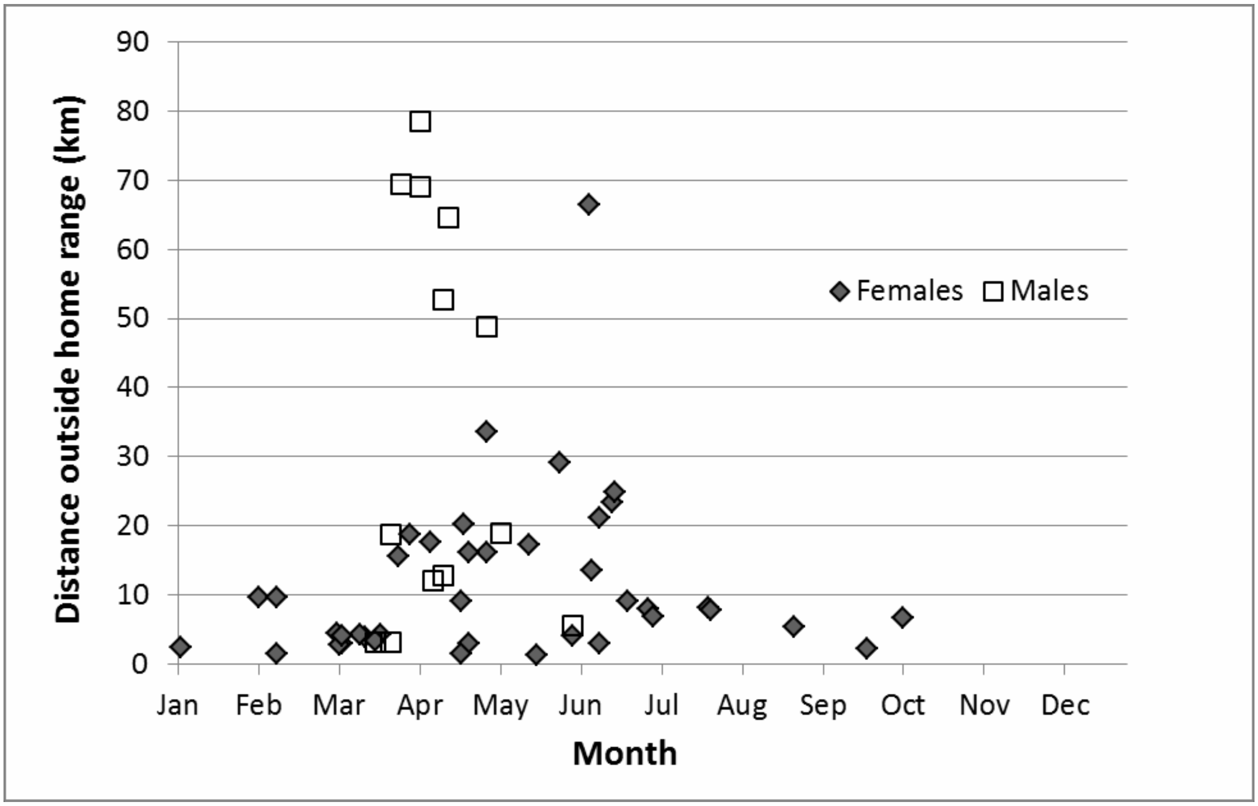

Figure 1.9. The date and distance of locations detected outside fisher home ranges. Data include 12 locations for 6 of 10 males with estimated home ranges, and 39 locations for 10 of 19 females with estimated home ranges.

the breeding season (~1 March to ~30 June) for both males and females (Figure 1.9). Male forays appeared to be concentrated during the breeding season, whereas females also made shorter forays outside the breeding season (≤10 km; Figure 1.9). The maximum number of days during which a foray occurred ranged from 26 to 130 days for males and from 12 to 100 days for females (Appendix 1.1).

DISCUSSION

Translocated male and female fishers moved extensively after being released on the Olympic Peninsula, and many moved substantial distances from their release sites. Post-release movements were strongly influenced by sex and the month in which movements occurred. Sex also had a strong influence on the distances fishers moved from a release site to an established home range, and on fisher home range sizes. Although release date and age may have had a weak influence on the distances that males moved from a release site before establishing a home range, these factors did not appear to influence the movements of females. Moreover, the year that a fisher was released (Release-Year Cohort) did not influence the distance to an established home range for males or females. Females showed a greater tendency to stay within the Olympic Recovery Area than males, as indicated by the locations of established home ranges. The limited spatial overlap of male and female home ranges that we observed is consistent with a population that is not yet at carrying capacity. Despite this limited overlap, males traversed large portions of the study area to locate females during the breeding season.

We predicted that male movements would be greater than those of females, which was supported by several of our findings, including the longer movement distances for males between relocations during the breeding season, the longer movement distances for males between their release sites and their home ranges, and the larger size of male home ranges.

In established populations, male fishers may move extensively during the breeding season as they seek reproductive females (Leonard 1986, Arthur *et al.* 1989, Aubry *et al.* 2004, Weir and Corbould 2007). The extensive breeding-season movements we observed for recently translocated males explains why the month of the year influenced post-release movements by male fishers. Previous studies of established populations have not documented similar long-distance movements by females outside their home range during the breeding season as we did.

Mating typically occurs within a female's home range, where they are located by breeding males (Powell 1993, Lofroth *et al.* 2010). We speculate that the peak in female movements during their first breeding season (shortly after being released) reflected extensive searching for mates by females that had not yet established a home range. The peak in female movements during the second breeding season was the result of six females from release cohorts 1 and 2 that left established home ranges, presumably to find a mate. Breeding season movements by females with established home ranges had not been reported before and suggest that there were too few adult males within the study area to locate each reproductive female (only 1 adult male was released in year 1 and no adult males were released in year 2). We documented reproduction for only one of these six females during the study (F004; Appendix 1.1), which indicated that mating opportunities were likely missed (Lewis *et al.* 2011).

The propensity for males to move greater distances after being released may explain the greater distances between release sites and home ranges for males. However, sex-specific differences in resource selection provide an alternative explanation for these differences. We released all 90 fishers within Olympic National Park; however, most fishers were released relatively close to the Park's boundary, due to limited vehicular access to the Park's interior. The shorter distances we observed for females between their release sites and home ranges was likely related to female selection of unmanaged mid-seral forests at mid-elevations (Chapter 3), which were found almost exclusively in Olympic National Park and Olympic National Forest and near many of the release sites. The greater distances moved by males was likely related to their selection of landscapes with smaller overstory trees and greater amounts of forest in the small tree class (Chapter 3), which are common on non-federal lands and located farther from release sites.

The timing of home range establishment differed between the sexes as indicated by the relatively uniform distribution of establishment dates during the 11-month period following release (December to October) for females and the clumped distribution of establishment dates for males (Figure 1.7). According to Buskirk and Powell (1994) and Raley *et al.* (2012), female fishers seek home ranges that give them access to the highest quality and quantity of essential resources, whereas males seek both essential resources and reproductive females. We hypothesize that many females were unable to locate a high-quality home range shortly after being released. Therefore, the variability in dates of home range establishment we observed for females was the result of variability in the search time necessary for females to locate a suitable home range. Because males appear to be less selective for habitat than females (Chapter 3), we hypothesize that males require less time to locate a suitable home range and therefore many males established home ranges following the breeding season, after having spent the previous months seeking reproductive females.

We predicted that fishers would move greater distances if they were released shortly before the beginning of the breeding season. The data provided weak support for this prediction as indicated by the inclusion of Release Date in the highest ranking model. The inclusion of Release Date in the top model was the result of a weak relationship between Release Date and Distance to an Established Home Range for males. Proulx *et al.* (1994) believed that the movement distances they found for both male and female fishers released in March were substantially greater than the movement distances of fishers released in June because they occurred during the breeding season. Our movement data indicated that breeding season movements for female fishers appeared to extend into June, so we included June within the breeding season. Regardless of this inconsistency in breeding-season chronologies, there appears to be some support for our prediction that both the release date and the onset of the breeding season may influence post-release movements by males.

Home ranges of males were larger than those of females, consistent with expectations based on body size (Harestad and Bunnell 1979) and the results of previous studies. Home ranges of translocated fishers on the Olympic Peninsula were among the largest known, but were generally consistent with the home range sizes of fishers studied in British Columbia (Table 1.7). Because habitat selection by translocated individuals is likely to be influenced by their natal environment (Stamps and Swaisgood 2007), translocated individuals may be inefficient at exploiting foreign environments and this inefficiency may require translocated individuals to use larger home ranges (Weilenmann *et al.* 2010, Russell *et al.* 2010). Fishers that we translocated from British Columbia to Olympic National Park in Washington were obtained from landscapes dominated by sub-boreal spruce-pine forests and were released into an environment dominated by temperate coniferous forests. These differences in the composition and structure of habitat conditions in their new environment are likely to present many challenges to translocated fishers, including differences in the nature and accessibility of available prey.

Table 1.7. Home range size estimates for fishers in western North America.

			Home range size (km^2)	
Study	**Location**	**Method**[a]	**Males (n)**	**Females (n)**
MCP Estimates				
Weir *et al.* 2009	Central British Columbia	100% MCP	144.2 (3)	52.6 (11)
This study	Olympic Peninsula, WA	100% MCP	79.6 (10)	50.8 (19)
Yaeger 2005	Northwestern California	100% MCP	38.3 (9)	23.5 (7)
Zielinski *et al.* 2004	Northwestern California	100% MCP	58.1 (2)	15 (7)
Buck *et al.* 1983	Northwestern California	100% MCP	19.8 (5)	4.2 (4)
Mazzoni 2002	Southern Sierra Nevada	100% MCP	21.9 (4)	11.9 (7)
Zielinski *et al.* 2004	Southern Sierra Nevada	100% MCP	30.0 (4)	5.3 (8)
Badry *et al.* 1997	Central Alberta	95% MCP	24.3 (2)	14.9 (5)
Aubry & Raley 2006	Southern Cascades, Oregon	95% MCP	62.0 (4)	25 (7)
Other Estimates				
Weir *et al.* 2009	Central British Columbia	95% FK	161.3 (3)	37.9 (11)
This study	Olympic Peninsula, WA	95% FK	128.3 (10)	63.5 (19)
Heinemeyer 1993	Northwestern Montana	90% AK	85.2 (2)	27.5 (7)
Jones 1991	North-central Idaho	90% HM	78.6 (6)	40.5 (4)

[a] Methods included: Minimum Convex Polygon (MCP); Fixed Kernel (FK); Adaptive Kernel (AK); and Harmonic Mean (HM).

Alternatively, the large home range sizes we observed may indicate that the quality, quantity, and distribution of resources and hazards on the Olympic Peninsula require fishers to use large home ranges. Home range sizes may also be influenced by population densities that are below the density at carrying capacity. Further research on fisher home ranges on the Olympic Peninsula is needed to determine whether mean home range sizes become smaller and more consistent with the home range patterns of fishers in established populations in western North America, where home range sizes increases linearly with latitude (Lofroth *et al.* 2010).

MANAGEMENT IMPLICATIONS

During reintroduction efforts, fishers of both sexes move extensively after release. Efforts to release fishers in the interior of a reintroduction/recovery area where habitat conditions are believed to be best for fishers may help promote home range establishment by exposing fishers to greater amounts of high-quality habitat as they explore their new environment. Releasing fishers at fewer and more-centralized release sites may accomplish this and may also alert fishers to the presence of other released individuals, especially potential mates. A greater awareness of potential mates may result in fewer females making potentially risky breeding-season forays from established home ranges to locate males.

We found that the establishment of home ranges by females occurred in all but 1 month of the year and, although releasing more females during the fall (November and December) may not change the broad temporal distribution of home range establishment, it should enable more females (and males) to establish home ranges prior to the breeding season. Managers may have a number of options when working with trappers (*e.g.*, specific requests for fall captures, greater financial incentives) to obtain, and subsequently release, a greater percentage of individuals during the fall months.

LITERATURE CITED

Armstrong, D.P. and P.J. Seddon. 2008. Directions in reintroduction biology. Trends in Ecology & Evolution 23: 20–25.

Aubry, K.B. and J.C. Lewis. 2003. Extirpation and reintroduction of fishers (Martes pennanti) in Oregon: implications for their conservation in the Pacific states. Biological Conservation 114:79-90.

Aubry, K.B., S.M. Wisely, C.M. Raley and S.W. Buskirk. 2004. Zoogeography, spacing patterns, and dispersal in fishers: insights gained from combining field and genetic data. Pages 187-206 in D.J. Harrison, A.K. Fuller, and G. Proulx, editors. Marten and fishers (Martes) in human-altered environments: an international perspective. Springer, New York, New York.

Aubry, K.B. and C.M. Raley. 2006. Ecological characteristics of fishers (Martes pennanti) in the southern Oregon Cascade Range, update: July 2006. U.S. Forest Service, Pacific Northwest Research Station, Olympia, Washington. 31 pp.

Arthur, S.M., W.B. Krohn and J.R. Gilbert. 1989. Home range characteristics of adult fishers. Journal of Wildlife Management 53:680-688.

Badry, M.J., G. Proulx and P.M. Woodard. 1997. Home-range and habitat use by fishers translated to the aspen parkland of Alberta. Pp. 233-251 in G. Proulx, H.N. Bryant and P.M. Woodard, editors. Martes: taxonomy, ecology, techniques, and management. Provincial Museum of Alberta, Edmonton, Alberta, Canada.

Bailey, V. O. 1936. The Mammals and Life Zones of Oregon. North American Fauna 55:1-416.

Berg, W. E. 1982. Reintroduction of fisher, pine marten and river otter. Pages 159-175 in G. C. Sanderson, editor. Midwest furbearer management. North Central Section of The Wildlife Society, Bloomington, Illinois.

Bodinof, C.M., J.T. Briggler, R.E. Junge, J. Beringer, M.D. Wanner, C.D. Schuette, J. Ettling, R.A. Gitzen and J.J. Millspaugh. 2012. Postrelease movements of captive-reared Ozark hellbenders (Cryptobranchus alleganiensis bishopi). Herpetologica 68(2): 160-173.

Breitenmoser, U., C. Breitenmoser-Wursten, L.W. Carbyn, and S.M. Funk. 2001. Assessment of carnivore reintroductions. Pages 240-281 in J.L. Gittleman, S.M. Funk, D.W. Macdonald, and R.K. Wayne, editors. Carnivore Conservation, Cambridge University Press, New York, New York, USA.

Buck, S., C. Mullis, and A. Mossman. 1983. Final Report: Corral Bottom-Hayfork Bally fisher study. Arcata, CA: Humboldt State University and USDA Forest Service.

Burnham, K.P. and D.R. Anderson. 2002. Model selection and inference: a practical information-theoretic approach. Springer-Verlag, New York.

Buskirk, S.W. and R.A. Powell. 1994. Habitat ecology of fishers and American martens. Pages 283-296 in S.W. Buskirk, A.S. Harestad, M.G. Raphael and R.A. Powell, editors. Martens, sables, and fishers: biology and conservation. Cornell University Press, Ithaca, New York.

Dalquest, W. W. 1948. Mammals of Washington. University of Kansas, Lawrence, Kansas.

Davis, M.H. 1983. Post-release movements of introduced marten. Journal of Wildlife Management 47:59-66.

Fischer, J. and D.B. Lindenmayer. 2000. An assessment of the published results of animal relocations. Biological Conservation 96(1):1-11.

Fontana, A.J., I.E. Teske, K. Pritchard and M. Evans. 1999. East Kootenay fisher reintroduction program, 1996- Ministry of Environment, Lands, and Parks, Cranbrook, British Columbia.

Franklin, J.F. and C.T. Dyrness. 1988. Natural vegetation of Oregon and Washington. Oregon State University Press, Corvallis.

Griffith, B., J.M. Scott, J.W. Carpenter and C. Reed. 1989. Translocation as a species conservation tool: status and strategy. Science 245:477-480.

Grinnell, J., J.S. Dixon, J.M. Linsdale. 1937. Fur-bearing mammals of California: their natural history, systematic status and relations to man. Volume 1. University of California, Berkeley.

Harestad, A.S. and F.L. Bunnell. 1979. Home range and body weight - a reevaluation. Ecology 60:389-402.

Hayes, G.E. and J.C. Lewis. 2006. Washington State recovery plan for the fisher. Washington Department of Fish and Wildlife, Olympia, Washington. 69 pp. (at: http://wdfw.wa.gov/wlm/diversty/soc/fisher/).

Heinemeyer, K.S. 1993. Temporal dynamics in the movements, habitat use, activity, and spacing of reintroduced fishers in northwestern Montana. MS Thesis, University of Montana, Missoula.

Henderson, J.A., D.H. Peter, R.D. Lesher and D.C. Shaw. 1989. Forested plant associations of the Olympic National Forest. U.S. Forest Service Ecological Technical Paper 001-88, Portland, Oreg. 502 pp.

Horne, J.S. and E.O. Garton. 2009. Animal Space Use 1.3 <http://www.cnr.uidaho.edu/population_ecology/animal_space_use>.

Houston, D.B. and E.G. Schreiner. 1994. The Olympic Mountains and Environs. Pages 14-26 (Chapter 2) in Houston, D.B., E.G. Schreiner, and B.B. Morehead, editors. Mountain Goats in Olympic National Park: biology and management of an introduced species. Scientific Monograph NPS/NROLYM/NRSM-94/25. USDI National Park Service. Port Angeles, Washington.

IUCN. 1987. IUCN position statement on translocation of living organisms: introductions, reintroductions, and re-stocking. Available at: www.iucn.org/themes/ssc/pubs/policy/transe.

IUCN. 1995. IUCN/SSC Guidelines for re-introductions. Forty-first meeting of the IUCN Council, Gland, Switzerland. May 1995, 6 pp. Available at: www.iucn.org/themes/ssc/pubs/policy/reinte.

IUCN. 2012. IUCN guidelines for reintroductions and other conservation translocations. Available at: http://www.issg.org/pdf/publications/Translocation-Guidelines-2012.pdf

Jones, J.L. 1991. Habitat use of fisher in northcentral Idaho. M.S. thesis. University of Idaho. Moscow.

Kinley, T.A. and N.J. Newhouse. 2008. Ecology and translocation-aided recovery of an endangered badger population. Journal of Wildlife Management 72(1):113-122.

Lawler, J.J., S.L. Shafer, D. White, P. Kareiva, E.P. Maurer, A.R. Blaustein and P. J. Bartlein. 2009. Projected climate induced faunal change in the Western Hemisphere. Ecology 90(3):588-597.

Leonard, R.D. 1986. Aspects of reproduction of the fisher, Martes pennanti in Manitoba. Canadian Field-Naturalist 100(1):32-44.

Lewis, J.C. and P.J. Happe. 2008. Olympic Fisher Reintroduction Project: 2008 Progress Report. Washington Department of Fish and Wildlife, Olympia. (at: http://wdfw.wa.gov/wlm/diversty/soc/fisher/)

Lewis, J.C., P.J. Happe, K.J. Jenkins and D.J. Manson. 2011. Olympic Fisher Reintroduction Project: 2010 Progress Report. Washington Department of Fish and Wildlife, Olympia.

Lewis, J.C., R.A. Powell and W.J. Zielinski. 2012. Carnivore translocations and conservation: Insights from Population Models and Field Data for Fishers (Martes pennanti). PLoS ONE 7(3): e32726.

Lewis, J.C. and W.J. Zielinski. 1996. Historical harvest and incidental capture of fishers in California. Northwest Science 70:291-297.

Lofroth, E.C., C.M. Raley, J.M. Higley, R.L. Truex, J.S. Yaeger, J.C. Lewis, P.J. Happe, L.L. Finley, R.H. Naney, L.J. Hale, A.L. Krause, S.A. Livingston, A.M. Myers and R.N. Brown. 2010. Volume I - Conservation Assessment for Fisher (Martes pennanti) in South-central British Columbia, Western Washington, Western Oregon, and California. USDI Bureau of Land Management, Portland, OR.

Mazzoni, A.K. 2002. Habitat use by fishers (Martes pennanti) in the southern Sierra Nevada, California. MS Thesis, California State University, Fresno, CA.

Mihoub, J.B., A. Robert, P. LeGouar and F. Sarrazin. 2011. Post-release dispersal in animal translocations: social attraction and the “vacuum effect”. PLoS ONE 6(12):e27453.

Miller, B., K. Ralls, R.P. Reading, J.M. Scott and J. Estes. 1999. Biological and technical considerations of carnivore translocation: a review. Animal Conservation 2:59-68.

National Park Service. 1998. Strategic Plan for Olympic National Park 1998-2002. Olympic National Park, Port Angeles, Washington.

National Park Service. 2005. Fire Management Plan and Environmental Assessment. Olympic National Park, Port Angeles, Washington.

Neilson, L. 1988. Definitions, considerations and guidelines for translocation of wild animals. Pages 12-52 in L. Neilson and R.D. Brown, editors. Translocation of wild animals. Wisconsin Humane Society and Caesar Kleberg Wildlife Research Institute, Milwaukee, WI. 333 p.

Peel, M.C., B.L. Finlayson and T.A. McMahon. 2007. Updated world map of the Köppen-Geiger climate classification, Hydrology and Earth Systems Sciences 11:1633-1644.

Powell, R.A. 1993. The Fisher: Life History, Ecology, and Behavior. Second edition. University of Minnesota Press, Minneapolis, Minnesota.

Powell, R.A. 1994. Structure and spacing of Martes populations. Pages 101–121 in S.W. Buskirk, A.S. Harestad, M.G. Raphael and R.A. Powell, editors. Martens, Sables, and Fishers: Biology and Conservation. Cornell University Press, Ithaca, New York, USA.

Powell, R.A., J.C. Lewis, B.G. Slough, S.M. Brainerd, N.R. Jordan, A.V. Abramov, V. Monakov, P.A. Zollner and T. Murakami. 2012. Evaluating translocations of martens, sables, and fishers: testing model predictions with field data. Pages 93-137, in K.B. Aubry, W.J. Zielinski, M.G. Raphael, G. Proulx, and S.W. Buskirk, editors. Biology and Conservation of Martens, Sables, and Fishers: A New Synthesis. Cornell Univ. Press. Ithaca, NY.

Proulx, G.A., J. Kolenosky, M. Badry, M.K. Drescher, K. Seidel and P.J. Cole. 1994. Post-release movements of translocated fishers. Pages 197-203 in S.W. Buskirk, A. Harestad, M. Raphael and R.A. Powell, editors. Martens, Sables, and Fishers: Biology and Conservation. Cornell University Press, Ithaca, New York.

Raley, C.M., E.C. Lofroth, R.L. Truex, J.S. Yaeger and J.M. Higley. 2012. Habitat ecology of fishers in western North America: a new synthesis. Pages 231-254 in K.B. Aubry, W.J. Zielinski, M.G. Raphael, G. Proulx and S.W. Buskirk, editors. Biology and Conservation of Martens, Sables, and Fishers: A New Synthesis. Cornell Univ. Press. Ithaca, NY.

Reading, R.P. and T.W. Clark. 1996. Carnivore reintroductions: an interdisciplinary examination. Pages 296-336 in J.L Gittleman, editor. Carnivore Behavior, Ecology, and Evolution. Vol. 2. Cornell University Press, Ithaca, New York, USA.

Rodgers, A.R., A.P. Carr, H.L. Beyer, L. Smith and J.G. Kie. 2007. HRT: Home Range Tools for ArcGIS. Version 1.1. Ontario Ministry of Natural Resources, Centre for Northern Forest Ecosystem Research, Thunder Bay, Ontario, Canada.

Roy, K.D. 1991. Ecology of reintroduced fishers in the Cabinet Mountains of northwest Montana. MS Thesis, University of Montana, Missoula.

Russell, J.C., A.J.C. McMorland and J.W.B. MacKay. 2010. Exploratory behavior of colonizing rats in a novel environment. Animal Behaviour 79:159-164.

Ryckman, M.J., R.C. Rosatte, T. McIntosh, J. Hamr and D. Jenkins. 2010. Postrelease dispersal of reintroduced elk (Cervus elaphus) in Ontario, Canada. Restoration Ecology 18(2):173-180.

Schwartz, M.W., L.R. Iverson, A.M. Prasad, S.N. Matthews and R.J. O'Connor. 2006. Predicting extinctions as a result of climate change. Ecology 87:1611–1615.

Schwartz, M.W. and T.G. Martin. 2013. Translocation of imperiled species under changing climates. Annals of the New York Academy of Sciences 1286:15-28.

Scott, J.M., D.D. Goble, A.M. Haines, J.A. Wiens and M.C. Neel. 2010. Conservation-reliant species and the future of conservation. Conservation Letters 3:91–97.

Seddon, P.J. 2010. From reintroduction to assisted colonization: moving along the conservation translocation spectrum. Restoration Ecology 18(6):796-802.

Seton, E.T. 1926. Lives of Game Animals. Vol. 2. Doubleday, Doran & Co., New York.

Serfass, T.L., R.P. Brooks and W.M. Tzilkowski. 2001. Fisher reintroduction in Pennsylvania: Final report of the Pennsylvania fisher reintroduction project. Frostburg State University, Frostburg, Maryland.

Sikes, R.S., W.L. Gannon, and the Animal Care and Use Committee of the American Society of Mammalogists. 2011. Guidelines of the American Society of Mammalogists for the use of wild mammals in research. Journal of Mammalogy 92(1):235-253.

Sjoasen, T. 1997. Movements and establishment of reintroduced European otters Lutra lutra. Journal of Applied Ecology 34:1070-1080.

Smith, A. and M.M. Peacock. 1990. Conspecific attraction and the determination of metapopulation colonization rates. Conservation Biology 4:320–323.

Stamps, J.A. 1988. Conspecific attraction and aggregation in territorial species. American Naturalist 131:329–347.

Stamps, J.A. and R.R. Swaisgood. 2007. Someplace like home: experience, selection, and conservation biology. Applied Animal Behaviour Science 102:392-409.

Spinola, R.M., T.L. Serfass and R.P. Brooks. 2008. Survival and post-release movements of river otters translocated to western New York. Northeastern Naturalist 15(1):13-24.

Thomas, C.D., A. Cameron, R.E. Green, M. Bakkenes, L.J. Beaumont, Y.C. Collingham, B.F.N. Erasmus, M.F. de Siqueira, A. Grainger, L. Hannah, L. Hughes, B. Huntley, A.S. van Jaarsveld, G.F. Midgley, L. Miles, M.A. Ortega-Huerta, Resources. 1997. Habitat Conservation Plan. Washington Department of Natural Resources, Olympia A.T. Peterson, O.L. Phillips and S.E. Williams. 2004. Extinction risk from climate change. Nature 427:145–148.

USDA Forest Service and USDI Bureau of Land Management. 1994. Record of decision for amendments to Forest Service and Bureau of Land Management planning documents within the range of the northern spotted owl. USDA Forest Service and USDI Bureau of Land Management, Portland, Oregon.

U.S. Fish and Wildlife Service. 2004. Endangered and threatened wildlife and plants: 12- month finding for a petition to list the west coast distinct population segment of the fisher (Martes pennanti). Federal Register 69(68):18770-18792.

U.S. Fish and Wildlife Service. 2013. Endangered and threatened wildlife and plants; status review of the West Coast distinct population segment of the fisher as endangered or threatened. Federal Register 78(53):16828-16829.

Van Heezik, Y., R.F. Maloney and P.J. Seddon. 2009. Movements of translocated captive-bred and released critically endangered kaki (black stilts) *Himantopus novaezelandiae* and the values of long-term post-release monitoring. Oryx 43(4):639-647.

Washington Department of Natural Resources. 1997. Habitat conservation plan. Washington Department of Natural Resources, Olympia.

Weckwerth, R.P. and P.L. Wright. 1968. Results of transplanting fishers in Montana. Journal of Wildlife Management 32:977-980.

Weilenmann, M., M. Gusset, D.R. Mills, T. Gabanapelo and M. Schiess-Meier. 2010. Is translocation of stock-raiding leopards into a protected area with resident conspecifics an effective management tool? Wildlife Research 37(8):702-707.

Weir, R.D. 1995. Diet, spatial organization, and habitat relationships of fishers in south-central British Columbia. MS Thesis, Simon Fraser University, Burnaby, British Columbia, Canada.

Weir, R.D. and F.B. Corbould. 2007. Factors affecting diurnal activity of fishers in north-central British Columbia. Journal of Mammalogy, 88(6):1508–1514.

Weir, R.D., A.S. Harestad and F.B. Corbould. 2009. Home ranges and spatial organization of fishers in central British Columbia. Canadian Field-Naturalist 123(1):126-132.

Wolf, C.M., T. Garland, Jr. and B. Griffith. 1996. Predictors of avian and mammalian translocation success: reanalysis with phylogenetically independent contrasts. Biological Conservation 86:243-255.

Worton, B.J. 1989. Optimal smoothing parameters for multivariate fixed and adaptive kernel methods. Journal of Statistical Computation and Simulation 32:45-57.

Yaeger, J.S. 2005. Habitat at fisher resting sites in the Klamath Province of northern California. MS Thesis, Humboldt State University, Arcata, California, USA.

Zar, J.H. 1999. Biostatistical Analysis, 4th ed. Prentice Hall, Upper Saddle River, NJ. 662 pp.

Zielinski, W.J., R.L. Truex, G.A. Schmidt, F.V. Schlexer, K.N. Schmidt and R.H. Barrett. 2004. Home range characteristics of fishers in California. Journal of Mammalogy 85(4):649-657.

Appendix 1.1. Breeding season forays >10 km from a home range boundary for 10 of 29 fishers with estimated home ranges on the Olympic Peninsula, Washington, 2008-2011. The breeding season for fishers is ~1 March to ~30 June.

Fisher	**Foray occurred between (span in days)**	**Number of locations outside home range**	**Most distant location from home range edge (km)**	**Age during foray**	**Release year cohort**
F003	12 Jun 08 – 23 Jun 08 (12)	2	24.8	4	1
F003	2 Feb 09 – 30 Mar 09 (56)	3	18.7	5	1
F004	13 Mar 09 – 8 May 09 (56)	3	20.3	4	1
F016	6 Apr 09 – 15 Jul 09 (100)	3	33.4	3	1
F018	26 Mar 09 – 15 Jun 09 (81)	5	17.6	3	1
F018	22 Mar 10 – 11 Jun 10 (81)	1	21.1	4	1
F024	5 Apr 10 – 7 Jul 10 (93)	3	66.4	2	2
F043	12 Apr 10 – 11 Jun 10 (60)	1	16.1	6	2
M011	27 Feb 09 – 27 May 09 (89)	7	78.5	3	1
M011	28 Jan 10 – 7 Jun 10 (130)	1	53.5	4	1
M014	24 Feb 09 – 8 May 09 (73)	1	78.7	3	1
M077	25 Feb 11 – 29 Apr 11 (63)	2	12.8	2	3
M093	17 Mar 11 – 12 Apr 11 (26)	1	18.8	3	3

CHAPTER 2

Survival of Fishers Translocated to the Olympic Peninsula of Washington

JEFFREY C. LEWIS, *University of Washington, School of Environmental and Forest Sciences, Seattle, WA 98195 and Washington Department of Fish and Wildlife, 600 Capitol Way N, Olympia, WA 98501*

KURT J. JENKINS, *U.S. Geological Survey, Forest and Rangeland Ecosystem Science Center, 600 E. Park Ave., Port Angeles, WA 98362*

PATRICIA J. HAPPE, *Olympic National Park, 600 E. Park Ave., Port Angeles, WA 98362*

DAVID J. MANSON, *Olympic National Park, 600 E. Park Ave., Port Angeles, WA 98362*

PAUL C. GRIFFIN, *U.S. Geological Survey, Forest and Rangeland Ecosystem Science Center, 600 E. Park Ave., Port Angeles, WA 98362*

INTRODUCTION

The goal for most conservation translocations is the reestablishment of a self-sustaining population (Scott and Carpenter 1987, Griffith *et al.* 1989, Seddon 2010). However, despite large investments of time, money, collaboration and expertise during translocation projects, many fail to meet this goal (Griffith *et al.* 1989, Yalden 1993, Wolf *et al.* 1996, Miller *et al.*1999, Fischer and Lindenmayer 2000, Breitenmoser *et al.* 2001). Given the prevalence of translocations to support species at risk, surprisingly little is known about the factors that influence translocation success for many species. However, this information could be critically important when resources (*e.g.*, funding, source populations) are limited or when a species is in imminent danger of extinction.

A minimum level of survival of translocated individuals is necessary for any translocation to succeed (Armstrong and Seddon 2007); however, uncertainty about founder survival is inherent in most translocation projects. In the absence of information on survival, managers may be able to offset low survival rates by releasing a large number of individuals or by continuing releases until evidence of success is apparent. However, managers almost always have limited resources for translocations and must rely on the best available information to conduct translocations in ways that will maximize the likelihood of success. Unfortunately, for many species, there is little information to guide managers in ways to maximize survival of individuals in a founding population when implementing a translocation.

In the contiguous United States (US), reintroductions have played a key role in restoring populations of the fisher (*Pekania pennanti*), a mid-sized carnivore of the family Mustelidae that occurs only in the temperate and boreal forests of North America. Because of the extremely high price paid for their pelts in the late 1800s and early 1900s (up to ~$350 US/pelt; Seton 1926, Bailey 1936, Grinnell *et al.* 1937, Dalquest 1948), the fisher was overexploited and extirpated from much of the southern portion of its range in the northern US and southern Canada (Strickland *et al.* 1982, Powell 1993, Lewis *et al.* 2012). Since the mid-1900s, fisher reintroductions have been successful at restoring self-sustaining fisher populations in many vacant portions of their historical range in mid-western and eastern North America (Berg 1982, Powell 1993, Lewis *et al.* 2012), however, reintroductions in western North America have been less successful.

In the Pacific states, the fisher currently occupies <25% of its historical range (Lewis *et al.* 2012), and the species was apparently extirpated in Washington state by the mid-1900s. By the year 2000, Oregon was the only state on the Pacific coast that had attempted fisher reintroductions; however, only one of three attempts was successful at reestablishing a population (Aubry and Lewis 2003). The fisher is federally listed as a candidate for threatened or endangered status in the Pacific states (*i.e.*, the fisher’s West Coast Distinct Population Segment; U.S. Fish and Wildlife Service 2004), and is currently the subject of a federal status review that may result in its listing as threatened or endangered (U.S. Fish and Wildlife Service 2013). It has been listed as a state endangered species in Washington since 1998; the Fisher Recovery Plan for the state indicated that reintroductions in both the Olympic Mountains and Cascade Range would be required to achieve recovery goals (Hayes and Lewis 2006).

The fisher has been successfully reestablished in 25 (71%) of the 35 documented translocation attempts within their historical range (Lewis *et al.* 2012); it is one of the most successfully translocated carnivores (Breitenmoser *et al.* 2001, Powell *et al.* 2012). Despite this level of success, little empirical information has been produced from these efforts to help inform future translocation programs, and no information on the survival of translocated fishers has been reported. Knowledge of survival would be important for determining the number and composition of individuals in a founder population that would result in a high likelihood of success; this is especially critical because of the importance of female survival for population growth and stability (Buskirk *et al.* 2012). Moreover, the nature of the translocation process may also influence fisher survival rates. With basic information on the influence of the translocation process, new strategies could be devised to improve fisher survival and increase the likelihood of translocation success.

As a first step toward the recovery of fishers in Washington state (Hayes and Lewis 2006), a partnership of state, provincial and federal agencies, and private organizations translocated 90 fishers from central British Columbia to Olympic National Park in northwestern Washington state from 2008 to 2010 (Lewis *et al.* 2011). Our research goals were to determine the sex- and age-specific survival rates of this founder population of 90 fishers and to identify factors that managers can manipulate to increase the likelihood of translocation success by increasing survival rates.

Previous studies of established fisher populations indicated that survival rates of females tended to be greater than males (Krohn *et al.* 1994, Koen *et al.* 2007), rates for adults tended to be higher than juveniles (Paragi *et al.* 1994, Krohn *et al.* 1994, Koen *et al.* 2007), and rates tended to be higher where fishers were not commercially trapped (Paragi *et al.* 1994, Krohn *et al.* 1994, York 1996, Belant 2007, Jordan 2007, Koen *et al.* 2007). Lower male survival has been attributed to a greater vulnerability of males to trapping mortality and other hazards (*e.g.*, vehicle traffic, predators) because of their larger home ranges and long-distance movements during the breeding season (Douglas and Strickland 1987, Powell 1993). Males may also be killed by other males when competing for reproductive females during the breeding season. Lower survival rates for juveniles may be associated with the long-distance movements they make when dispersing. Such movements are likely to put juveniles at greater risk of trapping mortality, predation, and vehicle collisions.

We found no data in the literature on variation in survival rates for fishers during different times of the year (*i.e.*, among seasons). However, fisher behaviors (*e.g.*, extensive breeding season movements, kit-rearing, repeated use of den sites, dispersal) and vulnerability to mortality will vary seasonally. Because seasonal differences in behavior may influence survival probabilities, we hypothesized that survival rates would vary among seasons that were biologically meaningful for fishers. For translocated fishers, we considered meaningful seasons to include: 1) an orientation season, which was the period time from release until the beginning of the breeding season (*i.e.,* December until 28 February; only in year 1); 2) the breeding season (1 March to 30 June), 3) the kit-rearing season (1 July to 30 September), and 4) the fall-winter season (1 October to 28 February). This chronology was based on movement data collected in our study area (see Lewis *et al.* 2008, 2010, 2011, Chapter 1) and chronologies provided by Powell (1993) and Lofroth *et al.* (2010).

Most fisher translocations involve the release of fishers over a period of ≥2 years, which results in the release of two or more release-year cohorts (individuals released in the same year). Fishers released in year 1 of a reintroduction (*i.e.,* release-year cohort 1) are placed in a reintroduction area(s) that is unoccupied by fishers and contains no cues to indicate the presence of fishers other than members of their own cohort. The absence of resident fishers means that environmental cues of habitat suitability or the presence of potential mates, as well as intraspecific competition with residents, are lacking. However, we hypothesize that the presence of resident fishers from a previous cohort may prompt translocated fishers to establish a home range (Stamps 1988, Smith and Peacock 1990), and therefore increase the likelihood of survival among the cohorts released in years 2 and 3. Given that carrying capacity is not expected to be reached for many years after translocations begin, there may have been little competition for suitable home ranges in years 2 and 3.

Many aspects of the translocation process are likely to influence the survival of translocated animals. Similarly, there are aspects of fisher biology and natural history that may affect the survival of translocated fishers. For example, translocated fishers may vary in 1) physical condition at the time of their release, 2) the duration of time they were kept in captivity prior to their release, and 3) the season of their release. Each of these factors may influence survival of translocated fishers, yet the extent of their importance is poorly understood.

As a carnivore, the fisher relies on its four canine teeth to capture and kill prey. Breakage of canine teeth during the translocation process is not uncommon (Arthur 1988, Heinemeyer 1993, Serfass *et al.* 2001, Koen *et al.* 2007); however, translocated fishers with broken canines may be less proficient at capturing prey, resulting in reduced survival probabilities. As members of the Mustelidae, fishers maintain a delicate balance between energetically demanding foraging behaviors and getting enough to eat (Powell 1979). Individuals with relatively low body weight or those that lose weight during the translocation process may have poorer body condition and be at greater risk of predation or starvation than animals with above-average body weight.

As energetic and wide-ranging carnivores (Powell 1993, Lofroth *et al.* 2010), fishers may be stressed by extended periods of captivity. The translocation process includes a number of stages (*e.g.*, capture, transport, housing, handling, release). Each stage can be stressful for translocated individuals (Hartup *et al.* 2005) and individuals that spend a greater amount of time in captivity are exposed to a greater amount of stress (Teixeira *et al.* 2007, Franceschini *et al.* 2008, Dickens *et al.* 2010). If stress related to the translocation process has an important influence on survival rates of translocated fishers, survival probability would be greater for fishers held for only a short period of time following capture than for those held for a long period. Similarly, because the stress of the translocation process occurs during an individual's first year and because those that survived year-1 will have gained experience and familiarity with their new environment, we predict that survival rates of translocated fishers will be higher in year-2 than in the year of their release (Larkin *et al.* 2003, McCorquodale *et al.* 2013).

Fishers are often released in the fall and winter months because they are typically obtained for translocation from licensed trappers during a commercial trapping season (*i.e.*, typically in the late fall and early winter months; Lewis *et al.* 2012). This places individuals in a foreign environment at various intervals before the breeding season, which begins on approximately 1 March (Powell 1993). Consequently, fishers that are released just before (February) or at the beginning (March) of the breeding season could behave differently and have different survival probabilities than those released during the fall (November or December).

This investigation was motivated by the recognition that moderate-to-high survival rates are likely to be essential for translocation success. With credible estimates of survival for translocated fishers and an understanding of factors that influence survival, managers may be able to design translocation programs that have a greater likelihood of success. In this paper, our goals were to estimate survival rates for a translocated population of fishers and identify factors that had the greatest effect on survival. Our specific objectives were to: 1) identify intrinsic factors (*e.g.*, sex, age, weight, body condition), extrinsic factors (*i.e.*, season) and translocation-process factors (*e.g.*, release-year cohort, release date, duration of captivity) that influence the survival of fishers released on the Olympic Peninsula, 2) estimate sex- and age-specific survival rates for the founder population, 3) determine the causes of mortality among recovered fishers, and 4) identify ways in which survival rates and causes of mortality differ between this translocated population and established fisher populations that have been studied in other regions.

STUDY AREA

The study area (14,412 km^2 [5,564 mi^2]) includes most of the Olympic Peninsula in western Washington, which is bordered by the Pacific Ocean to the west, the Strait of Juan de Fuca to the north, and Puget Sound to the east (Figure 2.1). The study area was delineated by the areal extent of all telemetry locations obtained for all fishers released on the Olympic Peninsula (with the exception of an exploratory movement outside the study area by 1 male). The center of the Peninsula is dominated by the Olympic Mountains as well as glaciated headwaters and steep drainages that radiate outward in all directions. Elevations range from sea level to 2,415 m (7,923 ft) at the top of Mt. Olympus near the center of Olympic National Park. The mountainous center of the Peninsula slopes to a pronounced coastal plain to the west and smaller plains to the north and east. The Olympic Peninsula has a temperate maritime climate characterized by warm summers and cool winters (Peel *et al.* 2007). The one exception to this pattern is the northeastern corner of the Peninsula, which is in the rain shadow of the Olympic Mountains, and has a more Mediterranean climate characterized by warm and dry summers and cool winters (Peel *et al.* 2007). Annual precipitation ranges from 315 to 500 cm on the west slope of the Olympic Mountains; whereas annual precipitation is typically <40 cm in the northeastern corner of the Olympic Peninsula (National Park Service 1998). Eighty percent of annual precipitation on the Olympic Peninsula falls from October through March. Most winter precipitation falls as rain at elevations below 305 m (1,000 ft) and as snow above 760 m (2,500 ft).

The moist climate and broad range of elevations profoundly affect vegetation patterns throughout the Peninsula. Conifer forests dominate the Olympic Peninsula and include forests in the Sitka spruce zone (generally <180 m elevation but can be as high as 600 m in the western valleys of the Peninsula), western hemlock zone (150-600 m on the west side, 0-1,200 m elevation on the east side), Pacific silver fir zone (*Abies amabilis*; mid-elevations), mountain

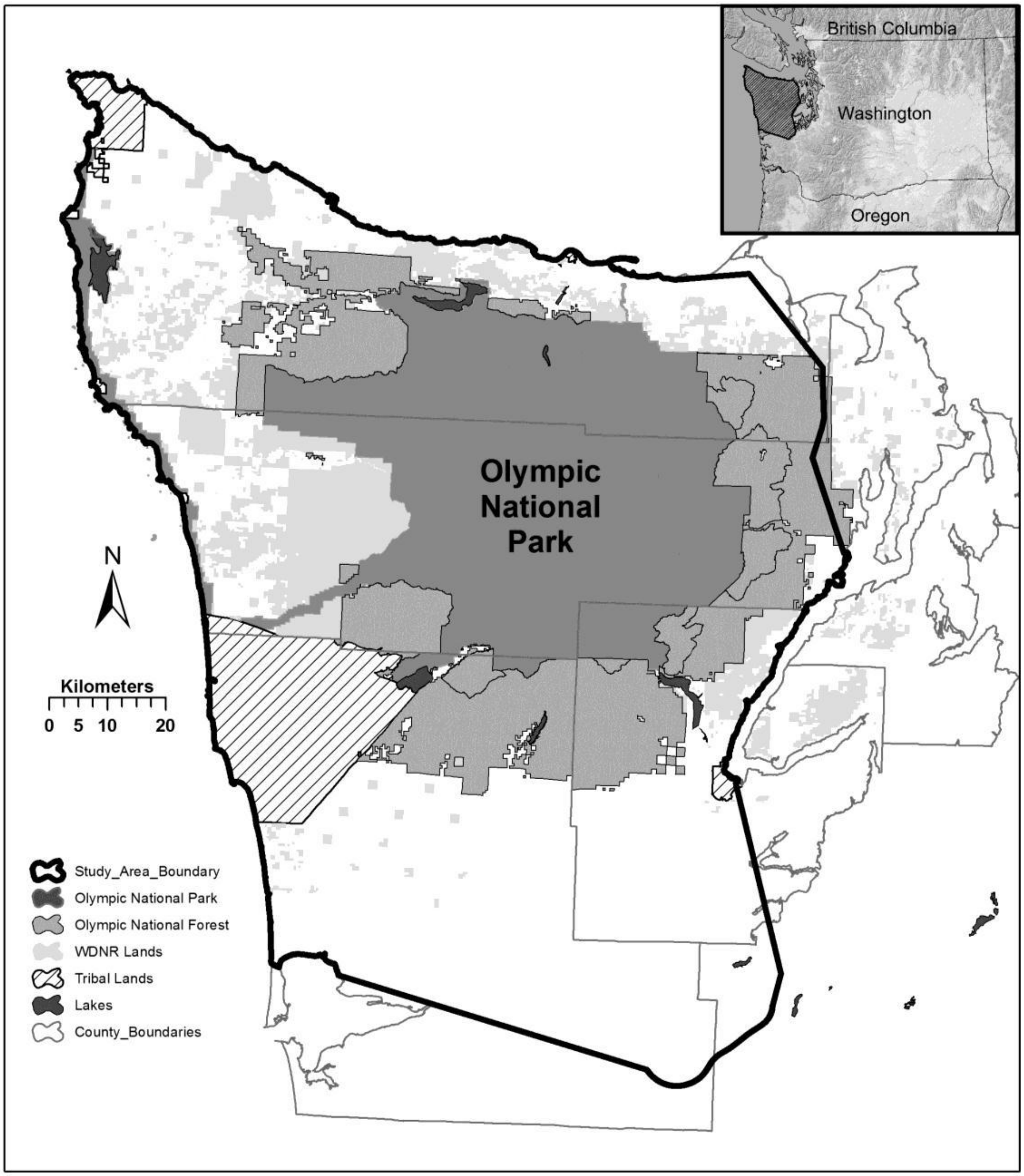

Figure 2.1. The study area of the Olympic fisher reintroduction project on the Olympic Peninsula, Washington, 2008-2011.

hemlock zone (*Tsuga mertensiana*; generally >1,070 m elevation) and the subalpine fir zone (*Abies lasiocarpa*; >1,200 m elevation; Franklin and Dyrness 1988, Houston and Schreiner 1994). Forests in the Sitka spruce zone on the western Olympic Peninsula are often referred to as temperate rainforests, and are characterized by large-diameter trees, epiphytic plants, and dense understory vegetation. Hardwoods (red alder [*Alnus rubra*], bigleaf maple [*Acer macrophyllum*] and black cottonwood [*Populus trichocarpa*]) are common in riparian forests along major rivers (Houston and Schreiner 1994, National Park Service 2005).

Olympic National Park includes the mountainous center of the Peninsula as well as mid- and low-elevation forested river drainages. The Park interior contains about 284,955 ha (704,139 ac) of forest and almost all (96%) of which is managed as wilderness. Olympic National Park contains one of the largest areas of contiguous temperate rainforest in North America. Much of the Park's circumference is bounded by Olympic National Forest, which encompasses mountainous terrain in the middle portions of many of the drainages that originate in the Park. Olympic National Park and Olympic National Forest make up the Olympic Recovery Area as outlined in the Washington State Fisher Recovery Plan (Hayes and Lewis 2006). Lower elevation lands outside the Olympic National Forest boundary are owned and managed by the Washington Department of Natural Resources (WDNR), Native American tribes, private timber companies and other private landowners, counties, and local municipalities.

Because of the drier climate in the eastern and northeastern portions of the Olympic Peninsula, wildfires in this area have had a more pronounced role in forest succession than in other areas of the Peninsula (Henderson *et al.* 1989). This history includes numerous fires since the 1850s that did not exceed 4,000 ha (9884 ac) in size and were widely distributed across this portion of the Peninsula. Smaller fires created mid-elevation landscape mosaics that are dominated by unmanaged mid- and late-seral forests in Olympic National Park and Olympic National Forest. A very large fire or series of fires (*i.e.*, 1.2-4.0 million ha [3-10 million ac]) occurred around 1701 within a 15-20 km band of forest along the northern and eastern periphery of the Olympic Peninsula. Remnant stands from the 1701 fire occur throughout the eastern portions and much of the northern portions of Olympic National Forest (Henderson *et al.* 1989). Whereas fires have been prevalent historically in the eastern and northeastern portion of the Olympic Peninsula, large fires have been much less frequent in the southern and western Olympic Mountains. Other disturbances including wind-throw and landslides have also altered forest structure and the forest landscape mosaic throughout the study area (Henderson *et al.* 1989).

Forest lands have been managed intensively on Olympic National Forest and WDNR lands since the 1950s; however, both ownerships have retained areas of unmanaged conifer forests. As part of the Northwest Forest Plan (USDA Forest Service and USDI Bureau of Land Management 1994), much of Olympic National Forest is now managed as late-successional forest reserves, where older forests are protected and younger forests are managed to accelerate the development of older forest characteristics. Washington Department of Natural Resources' lands on the Olympic Peninsula are managed under the guidance of a Habitat Conservation Plan (Washington Department of Natural Resources 1997), which targets the retention of older forests for northern spotted owls (*Strix occidentalis caurina*) and marbled murrelets (*Brachyramphus marmoratus*) in portions of the western Olympic Peninsula. Private and tribal lands occur at lower elevations at the periphery of National Forest and WDNR lands; these lands are dominated by second-growth forests as a result of intensive timber management practices.

METHODS

Capture, Handling and Translocation

Contractors were hired to coordinate fisher trapping activities with British Columbia trappers, provide a facility for housing captive fishers, care for captive fishers, and assist with preparing fishers for translocation. These contractors instructed and assisted participating British Columbia trappers, obtained captured fishers from trappers, and facilitated payments to trappers for fishers that were suitable for translocation. Trapping seasons for fishers in British Columbia began in November and continued through February. Trappers were provided box traps and specialized boxes for transporting fishers safely (Lewis 2006).

Contractors retrieved fishers from trappers, transported them to the captive facility, and provided an individual housing unit for each fisher. They also provided straw bedding, a litter box, water *ad libitum*, and a diet that promoted weight gain (raw meat from recovered deer [*Odocoileus spp*] beaver [*Castor canadensis*], snowshoe hare [*Lepus americanus*], and red squirrel [*Tamiasciurus hudsonicus*] carcasses, eggs, and hamburger). The captive facility had the capacity to hold 20 fishers. Consequently, when the number of captive fishers reached 12, project biologists from

Washington made arrangements to travel to British Columbia, inspect and prepare fishers at the captive facility, and transport fishers to Washington before the capacity of the facility was exceeded. While at the captive facility, each fisher was chemically immobilized and given a health inspection by a licensed British Columbia veterinarian. Of the 99 fishers captured, 90 (50 F, 40 M) were certified by the veterinarian as suitable for translocation to Washington. Each certified fisher was vaccinated for rabies (IMRAB-3; Merial: http://us.merial.com/) and canine distemper (Purevax; Merial: http://us.merial.com/), treated with ivermectin and Dronsit (Bayer AG; http://www.bayer.com/) for parasites, equipped with a radio-transmitter and a pit-tag, measured, weighed, and photographed. One first premolar was extracted to age each fisher, and blood and tissue samples were taken for evaluation of exposure to diseases and genetic analyses, respectively. We also assessed the condition of each fisher's teeth; eight (7 F, 1 M) of the 90 certified fishers had only two canines, whereas the remaining 82 fishers had three or four. Among the 90 certified fishers, total time in captivity ranged from 2 to 55 days (mean: 21 ± 12 [SD] days). Five fishers (3 F, 2 M) that received special medical treatment (*e.g.*, surgery to remove an impacted canine, treatment for a wound), were kept in captivity for 32 to 55 days (mean: 42 ± 9 [SD] days).

Eighty-two (50 F, 32 M) of the 90 fishers were equipped with a 40-g VHF radio-collar (model MI-2 with mortality sensor, Holohil Systems Ltd, Carp, Ontario, Canada; http://www.holohil.com). Of the remaining eight fishers, five males (each weighing >4.5 kg) were equipped with a 120-g Argos satellite collar (Kiwisat 202, Sirtrack Ltd., Havelock North, New Zealand; http://www.sirtrack.com) and three males were equipped with a 41-g VHF transmitter (model IMP/310/L with mortality sensor, Telonics Inc., Mesa, AZ, USA; http://www.telonics.com) that was surgically implanted in their abdominal cavity. For the 85 fishers with VHF transmitters (collar or implant), the normal radio-signal of 42 beats/min indicated that the fisher was alive; whereas a mortality signal of 72 beats/min indicated that the transmitter had been motionless for more than 6 hours (*i.e.*, the individual had died, or the collar came off while it was still alive).

Fishers were transported from central British Columbia to Port Angeles, Washington in 1 day, housed overnight in their transport boxes, and released the following morning at pre-determined release sites in Olympic National Park. We released fishers at 21 sites within the Park from 27

January 2008 to 20 February 2010 (Table 2.1; see summaries by Lewis and Happe 2008 and Lewis *et al.* 2011). Release sites were distributed throughout most of the Park to facilitate fisher occupancy of a number of large landscapes dominated by late-successional conifer forest and to maintain even sex ratios in areas where one sex was missing or poorly represented (Lewis and Hayes 2004, Lewis 2006, Lewis *et al.* 2010, 2011). All fishers were immediately released upon arrival at their release site; there was no acclimation period prior to release, as has been the case for some fisher reintroductions (*i.e.,* soft releases; Roy 1991, Heinemeyer 1993, Weir 1995). Animal handling procedures met or exceeded the guidelines of the American Society of Mammalogists for the use of wild mammals for research (Sikes and Gannon *et al.* 2011), and were approved by the Washington Department of Fish and Wildlife's Animal Care and Use Committee.

Radio-Telemetry

Because of the remote nature and broad geographic extent of the study area (Figure 2.1), we used aerial telemetry to monitor the survival status (alive, dead, or censored) of radio-collared fishers. We attempted to locate each fisher and determine its survival status at least once per week; however, inclement weather, the large size and rugged nature of the study area, the short transmission distance of the radio-collars, and the extensive movements of many fishers, made it impossible to locate all fishers each week. Consequently, we pooled survival status data into monthly intervals. During the breeding season (~1 March to ~30 June), we prioritized telemetry efforts on female fishers to detect evidence of reproduction. Upon discovering a mortality signal, we attempted to locate the collar using standard ground-based telemetry techniques to determine if the fisher had died. The remains of dead fishers were recovered and necropsies were conducted by pathologists from Colorado State University (Veterinary Diagnostics Laboratory) or the University of California at Davis (Veterinary Genetics Laboratory) to determine the cause of mortality. For the five males with Argos satellite collars (which have no mortality signal), we used the extent of the fisher's movements and the precision of their locations (provided by the satellite telemetry service) to determine if an individual was alive or dead.

Table 2.1. Number of fishers released on the Olympic Peninsula, Washington, from 27 January 2008 to 20 February 2010, by release-year cohort, release dates, sex and age-class.

	Fisher age-class[a]		
Release-year cohort 1 27 Jan 2008 2 Mar 2008	Juveniles	3	5
	Adults	9	1
	Total (18)	12	6
Release-year cohort 2 21 Dec 2008 17 Jan 2009 23 Feb 2009	Juveniles	6	11
	Adults	14	0
	Total (31)	20	11
Release-year cohort 3 24 Dec 2009 21 Jan 2010 20 Feb 2010	Juveniles	7	16
	Adults	11	7
	Total (41)	18	23
	Grand Total (90)	**50**	**40**

[a] For females, juveniles were <1 year old, and adults were ≥1 year old. For males, juveniles were <2 years old, and adults were ≥2 years old.

Survival Analyses

We examined the survival status of radio-collared fishers over a period of 48 months (January 2008–December 2011). We evaluated survival status data for the 24 months following the release of fishers in each release-cohort; for time periods exceeding 24 months, sample sizes became too small to generate reliable estimates. For each month, an individual fisher could have one of three survival-status designations: alive, dead, or censored. A fisher was censored from survival analyses for any month that its status could not be determined (*i.e.*, not found); some fishers were only censored temporarily because they were subsequently relocated alive or dead.

We structured known-fate models in Program MARK (White and Burnham 1999) to estimate survival rates and examine their relationships to intrinsic (*i.e.*, sex, age, weight, number of broken canines), extrinsic (season), and reintroduction-process factors (*i.e.*, release cohort, duration of captivity, release date). The known-fate model in MARK allows for staggered entry (Pollock *et al.* 1989) of fishers into the population in different months (*e.g.*, Dec 2008, Jan 2009, Feb 2009) and among years, as was the case for fishers released in Olympic National Park from 2008 to 2010 in three release cohorts (Table 2.2). We structured known-fate models that included individual covariates such as sex, age, weight, number of days in captivity, release date, and the presence of ≥2 broken canines (Table 2.2). We used a 1-month interval as the time step in survival analyses.

We constructed two sets of candidate models based on *a priori* biological hypotheses for explaining the influence of intrinsic, extrinsic, and translocation-process factors on fisher survival. The first set included models that incorporated the main effects of release-cohort, season, age, and sex (Table 2.2). To examine the influence of release cohort on survival rates, we treated the three release cohorts as separate groups within the analysis. We constructed models in which we assumed that survival rates were constant over time, and others in which survival rates varied by season. In models that included a seasonal influence on survival rate, we grouped the months of the year into four seasons that we believed were biologically meaningful to fishers (Table 2.2). The age of each fisher was determined by laboratory analysis of the cementum annuli (Arthur *et al.* 1992) in the first premolar we extracted prior to release. Fisher ages ranged from 0 to 5 years; in candidate models that included a linear effect of Age, we incremented each fisher's age by 1 for the second year after that animal's release.

Table 2.2. Intrinsic, extrinsic and reintroduction-process variables incorporated into models of fisher survival, and grouped as main and additional effects.

Variable	**Description**
Main effects	
Release-year cohort	Release-year cohort 1 (released from January to March 2008) Release-year cohort 2 (released from December 2008 to February 2009) Release-year cohort 3 (released from December 2009 to February 2010)
Season	Orientation season (from release to 28 Feb; year 1 only) Breeding season (1 Mar-30 Jun), Kit-rearing season (1 Jul-30 Sep) Fall-winter season (1 Oct-28 Feb)
Age	Estimated age in years at the time of release. Age was increased by 1 year at the start of an individual's second year after release.
Sex	Male or female
Additional effects	
Duration of captivity	Number of days from the capture date in British Columbia until the release date for each fisher
Release date	Timing of release, as indicated by the number of days prior to the beginning of the breeding season (1 March) that the release occurred.
Standardized weight	Weight of each individual fisher relative to sex-specific means and standard deviations; computed as (fisher weight-mean weight)/standard deviation of weight
Canines	≤1 or ≥2 broken canines

In a subset of models that included both the effects of Season and Age, we also included the interaction between Season and Age (Season*Age). In models that included Sex, the effect was considered to be constant across all seasons. In a subset of models that included both the effects of Release-Year Cohort and Sex, we also included the interaction between Release-Year Cohort and Sex (Release-Year Cohort*Sex in model names).

We used Akaike's Information Criteria adjusted for small sample size (AIC_c) to assess the fit of each candidate model within each of the two candidate model sets (Burnham and Anderson 2002). We used the difference in AIC_c scores (ΔAIC_c) and Akaike's weights (w_i) to inform our selection of a best model or a group of highly supported models. In the first set of candidate models, we considered those with ΔAIC_c scores <2 to be highly supported by the data (Burnham and Anderson 2002), and we used these as base models in our second set of candidate models.

In the second set of candidate models, we evaluated improvement in model fit by adding an additional parameter to each of the base models. The additional factors we considered were related to the translocation process or to the condition of each translocated fisher, and included Duration of Captivity, Release Date, Standardized Weight, and Canines (Table 2.2). Models that included Standardized Weight included a linear effect of standardized body weight in the logit link function; this reflected the assumption that any effect linked to a fisher's weight at the time of release persisted throughout the next 2 years. Similarly, models with the binary effect of broken canine teeth (Canines in model names) or the linear effect of the proximity of the release date to the onset of the breeding season ("Release Date" in model names) include that effect for two years after release.

We used the model-averaging function in program MARK (White and Burnham 1999) to obtain a weighted average of survival estimates from highly supported models from our second candidate set, and used these estimates to illustrate the effects of influential covariates. To estimate seasonal and annual survival rates, we calculated the product of monthly survival rates to produce a cumulative survival rate for a season or an entire year. We used the delta method (Seber 1982, Cooch and White 2013) to calculate variance and 95% CIs for cumulative survival rates.

RESULTS

Survival Models

The first candidate model set comprised 21 models that included only main-effect variables (Release-Year Cohort, Season, Sex, Age; Table 2.3). Among these 21 models, three had ΔAIC_c scores <2 and collectively represented 75% of the total model weights (w_i) (Table 2.3). Consequently, there was no single best model; rather, there were three models that were highly supported by the data. Release-Year Cohort and Season were included in all three highly supported models, whereas Sex and Age were only included in two (Table 2.3). There was only weak support for candidate models that included the interactions between Season and Age, Release-Year Cohort and Age, or Release-Year Cohort and Sex.

The second candidate model set included the three base models (highest ranking models from the first candidate model set; Table 2.3) as well as models constructed by adding a single additional-effect variable to each base model. Within the second candidate set, no single model stood out as the best model; rather, eight of the 15 models were well supported by the data (*i.e.,* ΔAIC_c <2; Table 2.4). The three base models were among the eight highly supported models in the second candidate set. Sex was included in six of the eight models, and Age was included in five. When Duration of Captivity was added to base models 1, 2, and 3, it resulted in small improvements in model support, by AIC_c 0.21, 0.39,and 0.45, respectively (Table 2.4). In comparison to the three base models, evidence ratios (ratio of model weights [w_i]; Table 2.4) associated with Duration of Captivity were 0.14 /0.13 (1.08), 0.11/0.09 (1.22), and 0.08/0.06 (1.33), respectively (average =1.21), which also indicated only limited support for the effect of the Duration of Captivity. When we excluded data for five fishers that were kept in captivity for a longer period of time to allow for medical treatment and recovery, the addition of Duration of Captivity to the base models actually reduced model support. Release Date was included in two of the eight top-ranked models, but inclusion of Release Date diminished AIC_c values by 0.90, 1.13, and 1.03, compared to base models 1, 2, and 3, respectively. The evidence ratios in favor of base models 1, 2, and 3 over comparable models that included Release Date was 1.62, 1.80, and 1.5 (average = 1.64). Similarly, inclusion of Canines or Standardized Weight reduced model support compared to the base models (Table 2.4).

Table 2.3. Models in candidate set 1 that examined the influence of Release-Year Cohort, Season, Sex and Age (main effects) on survival of fishers released on the Olympic Peninsula, Washington, 2008-2011. Each model was characterized by Akaike's Information Criteria for small samples (AIC_c), the difference between model AIC_c scores (ΔAIC_c), and Akaike's weights (w_i) (Burnham and Anderson 2002).

Model	**Number of parameters**	**AIC_c**	**ΔAIC_c**	**w_i**
Release-Year Cohort + Season + Sex	7	297.70	0.00	0.34
Release-Year Cohort + Season + Sex + Age	8	298.34	0.65	0.24
Release-Year Cohort + Season + Age	7	299.03	1.33	0.17
Release-Year Cohort + Season	6	300.84	3.15	0.07
Release-Year Cohort + Season + Sex + Age + Season*Age	12	301.02	3.33	0.06
Release-Year Cohort +Sex	4	301.40	3.70	0.05
Release-Year Cohort + Age	4	302.24	4.55	0.03
Release-Year Cohort	3	305.07	7.37	0.01
Release-Year Cohort + Age + Cohort*Age	7	306.13	8.43	0.00
Release-Year Cohort + Sex + Cohort*Sex	7	306.15	8.45	0.00
Season + Age	5	308.99	11.30	0.00
Season + Sex	5	309.26	11.56	0.00
Season + Age + Sex	6	309.49	11.80	0.00
Age	2	310.02	12.32	0.00
Season	4	310.22	12.52	0.00
Season + Age + Season*Age	9	310.86	13.16	0.00
Sex	2	311.69	14.00	0.00
Null	1	313.13	15.43	0.00

Table 2.4. Models in candidate set 2 that examined the influence of adding four covariates (Duration of captivity, Release date, Canines, and Standardized weight), individually , to base models from candidate set 1 to assess improvement in model fit. Each model was characterized by Akaike's Information Criteria for small samples (AIC_c), the difference between model AIC_c scores (ΔAIC_c), and Akaike's weights (w_i; Burnham and Anderson 2002).

Model	**Number of parameter**	**AIC_c**	**ΔAIC_c**	**w_i**
Release-Year Cohort + Season + Sex + Duration of Captivity	8	297.48	0.00	0.14
Release-Year Cohort + Season + Sex - base model 1	7	297.70	0.21	0.13
Release-Year Cohort + Season + Sex + Age + Duration of Captivity	9	297.96	0.47	0.11
Release-Year Cohort + Season + Sex + Age - base model 2	8	298.34	0.86	0.09
Release-Year Cohort + Season + Age + Duration of Captivity	8	298.58	1.10	0.08
Release-Year Cohort + Season + Sex + Release Date	8	298.60	1.11	0.08
Release-Year Cohort + Season + Age - base model 3	7	299.03	1.55	0.06
Release-Year Cohort + Season + Sex + Age + Release Date	9	299.48	1.99	0.05
Release-Year Cohort + Season + Sex + Canines	8	299.49	2.01	0.05
Release-Year Cohort + Season + Sex + Standardized Weight	8	299.70	2.21	0.05
Release-Year Cohort + Season + Age + Release date	8	300.07	2.58	0.04
Release-Year Cohort + Season + Sex + Age + Standardized Weight	9	300.34	2.85	0.03
Release-Year Cohort + Season + Sex + Age + Canines	9	300.34	2.86	0.03
Release-Year Cohort + Season + Age + Canines	8	301.04	3.55	0.02
Release-Year Cohort + Season + Age + Standardized Weight	8	301.06	3.57	0.02

Survival Rates

Survival rates were variable both within and among cohorts, seasons, sexes, and ages of fishers. Although confidence intervals overlapped, survival tended to be greatest for fishers released in cohort 1 and lowest for those released in cohort 2 (Figure 2.2, Table 2.5). Survival tended to be least during the breeding season for both males and females, and tended to be greater for males than females (Figure 2.3, Table 2.5). We found no appreciable difference in survival rates among age classes within each of the cohorts and seasonal periods, but point estimates of survival were consistently lowest in the oldest age class and highest for juveniles (Figure 2.2, Table 2.5). Similarly, there was no discernable difference in survival rates of fishers between the first and second year following release (Figures 2.2 and 2.3, Table 2.5).

Causes of Mortality

We recovered the carcasses of 35 of the 90 fishers we released and monitored from January 2008 to December 2011. We could determine the cause of mortality of 24 of the 35 recovered fishers (68.5%; Table 2.6); predation was the leading cause of death, followed by vehicle strikes. Because we could not always recover fishers soon after death, we were unable to determine the cause of mortality for the remaining 11 fishers due to the effects of decomposition or scavenging. Females made up 74% of documented mortalities (26 of 35; Table 2.6) but represented only 56% of the founder population (Table 2.1). The percent of documented female mortalities differed substantially among release-year cohorts and the percent of documented mortalities was more than twice as high for females than for males (Table 2.6). These findings are consistent with Sex and Cohort being important variables in the highest ranking models (Tables 2.3 and 2.4).

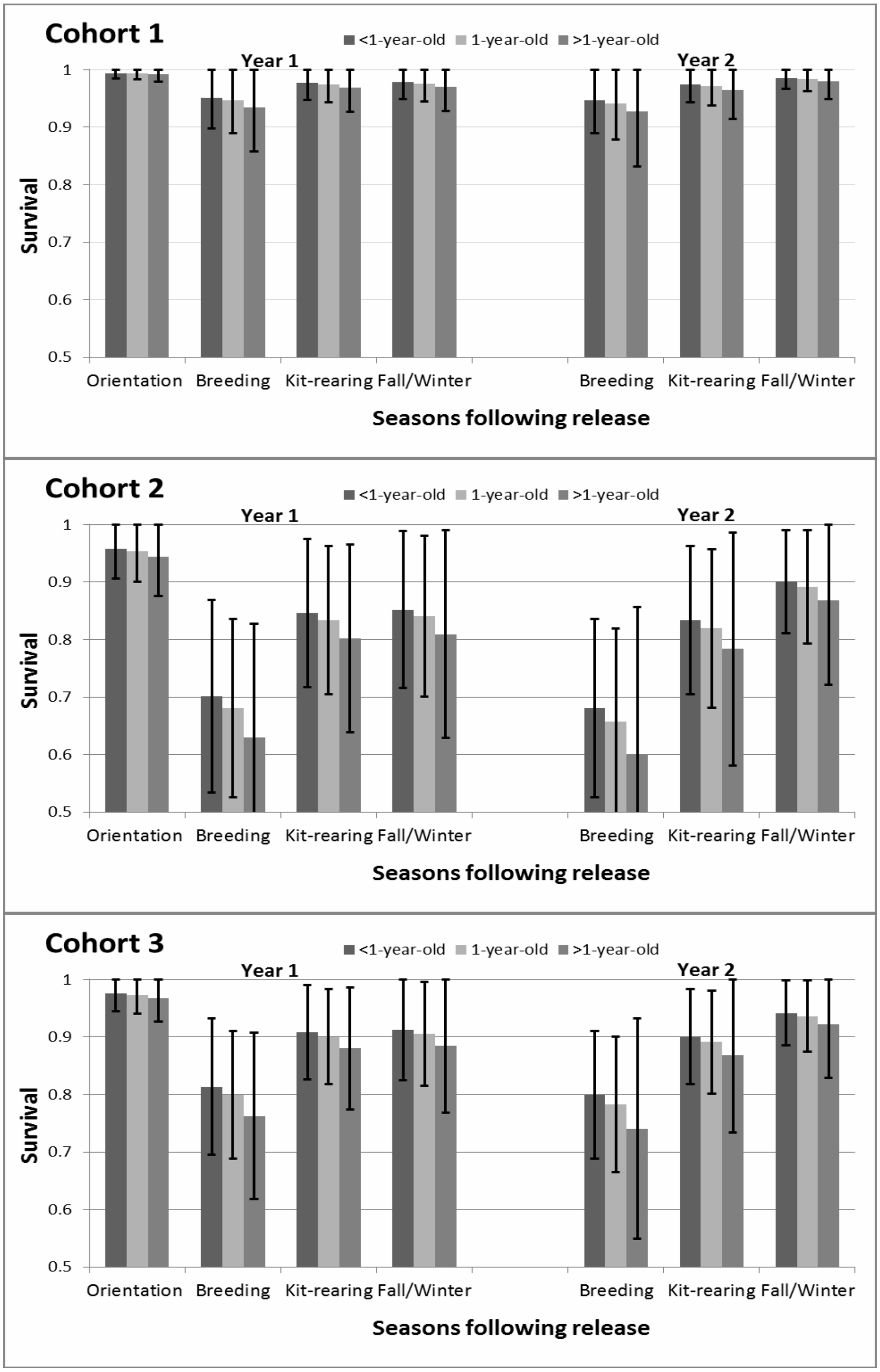

Figure 2.2. Age-specific fisher survival by season and release-year cohort for fishers released and monitored on the Olympic Peninsula, Washington, 2008-2011. Survival was based on the model-averaged estimates of the eight best models (*i.e.,* those with ΔAIC_c scores <2; Table 4). Error bars indicate the 95% confidence interval for each estimate.

Table 2.5. First and second year survival rate estimates (S and associated 95% CIs) by release-year cohort for sex- and age-specific segments of the founding population of fishers released and monitored on the Olympic Peninsula, Washington, 2008-2011.

	Cohort 1		Cohort 2		Cohort 3	
Population segment[a]	$S_{(\text{Year 1})}$	95% CI	$S_{(\text{Year 1})}$	95% CI	$S_{(\text{Year 1})}$	95% CI
Juvenile females	0.89	0.77-1.00	0.46	0.21-0.70	0.62	0.39-0.85
Adult females	0.84	0.70-0.99	0.34	0.15-0.52	0.51	0.31-0.71
All females	0.86	0.73-0.99	0.37	0.19-0.56	0.55	0.35-0.74
Juvenile males	0.94	0.87-1.00	0.65	0.45-0.86	0.77	0.62-0.92
Adult males	0.91	0.80-1.00	—[b]	—[b]	0.69	0.47-0.92
All males	0.93	0.86-1.00	0.65	0.45-0.86	0.76	0.60-0.92
	$S_{(\text{Year 2})}$	95% CI	$S_{(\text{Year 2})}$	95% CI	$S_{(\text{Year 2})}$	95% CI
Juvenile females	0.87	0.74-1.00	0.40	0.19-0.62	0.58	0.37-0.79
Adult females	0.82	0.64-1.00	0.28	0.07-0.49	0.45	0.22-0.68
All females	0.84	0.68-0.99	0.32	0.13-0.51	0.49	0.29-0.70
Juvenile males	0.93	0.84-1.00	0.61	0.38-0.84	0.74	0.57-0.91
Adult males	0.90	0.76-1.00	0.50	0.15-0.85	0.65	0.37-0.93
All males	0.92	0.83-1.00	0.59	0.34-0.83	0.72	0.54-0.91

[a] Survival rates were generated from the highest ranking base model that included sex and age (S[cohort + season + sex + age]). An average age of 0 years was used to estimate survival for juvenile females; an average age of 2.5 years for adult females; 1.70 for all females; 0.375 for juvenile males; 2.875 for adult males and 0.88 for all males.

[b] Rates for adult males (≥2 years of age) were not included because no adult males were released in cohort 2. We calculated year-2 survival estimates for adults from cohort 2 because males released as 1-year-olds in year 1 were considered adults in year 2.

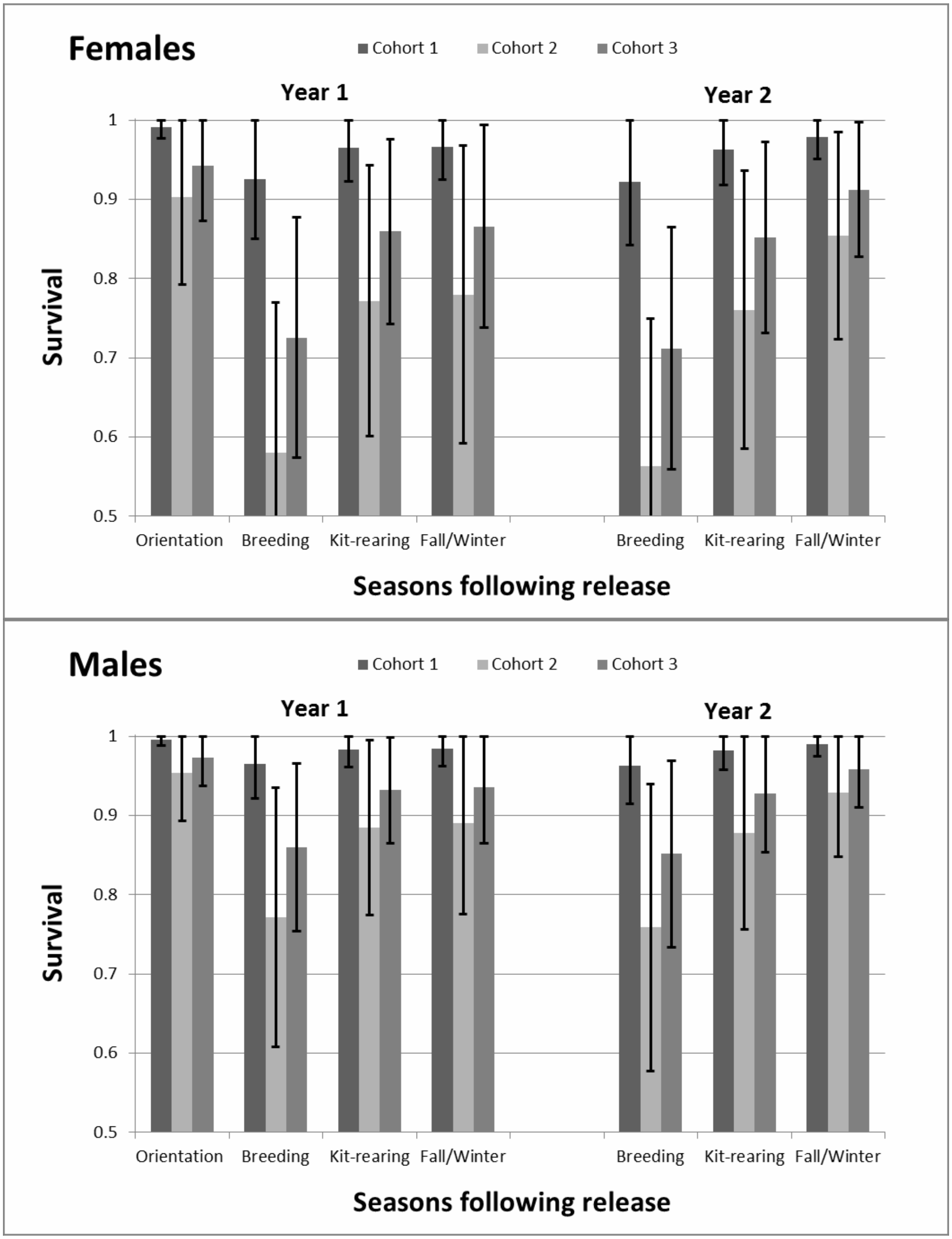

Figure 2.3. Seasonal fisher survival by release-year cohort and sex for years 1 and 2 following release on the Olympic Peninsula, Washington, 2008-2011. Survival rates are based on the model-averaged estimates of the eight top models (i.e., those with $\Delta AIC_c < 2$; Table 4). An average age of 1.33 years was used in each model that included the age covariate. Error bars indicate the 95% confidence interval for each estimate.

Table 2.6. Cause of mortality of reintroduced fishers recovered from January 2008 to December 2011 on the Olympic Peninsula, Washington**.**

	Females				**Males**				**All (%)**
Cause of mortality	**Year 1**	**Year 2**	**Year 3**	**All years**	**Year 1**	**Year 2**	**Year 3**	**All years**	**All years**
Predation	1	3	7	11	1		2	3	14 (40.0)
Vehicle strike	1	3	1	5		2		2	7 (20.0)
Unknown	1	4		5			2	2	7 (20.0)
Unknown (possible predation)		2		2		1	1	2	4 (11.4)
Drowning		1	1	2					2 (5.7)
Trapping related[a]		1		1					1 (2.9)
Total mortalities	3	14	9	26	1	3	5	9	35 (100.00)
% of cohort[b]	25%	70%	50%	52%	17%	27%	22%	23%	39%

[a] This female died from injuries she suffered after being caught in a leg-hold trap ~14 months after release.

[b] The number of mortalities for a release cohort divided by the number of fishers in a release cohort *100, by sex (see Table 1 for numbers of females and males released in each cohort).

DISCUSSION

The main effects of Release-Year Cohort, Season, Sex, and Age clearly influenced the survival of reintroduced fishers on the Olympic Peninsula, whereas the additional effects of Duration of Captivity, Release Date, Weight, and the number of broken canines (Canines) did not have a strong influence on survival rates. Previous studies of fisher survival have only been conducted on established fisher populations, and did not investigate the influence of intrinsic or extrinsic factors on fisher survival; however, several studies estimated sex- and age-specific survival rates (Table 2.7).

Release cohort had the greatest influence on fisher survival rates, due to lower survival rates in the first 2 years after release for release-year cohort 2 compared to release-year cohort 1. Although a number of factors may have contributed to the variation in survival rates among the 3 release-year cohorts (*e.g.*, significant variations in winter weather patterns, varying reactions to vaccinations and/or immobilization drugs, exposure to a disease or toxin), we can only speculate about the potential effects of these factors on the observed variation in survival rates. The survival rates for fishers in release-year cohort 1 were comparable to the highest survival rates reported for established fisher populations that were either subjected to light trapping pressure (Michigan [Belant 2007]; Massachusetts [York 1996]) or no trapping pressure (California [Jordan 2007]; Table 2.7). Conversely, survival rates for female fishers in release-year cohort 2 were lower than those reported for any established population, whereas survival rates for males (juveniles only) were comparable to reported estimates (Table 2.7). Point estimates for females in release-year cohort 3 tended to be lower than most reported rates, whereas estimates for males in release-year cohort 3 were comparable to reported estimates.

Our data did not support the hypothesis that stress associated with the translocation influenced survival rates. There was very little or no support for survival models that included an effect of the number of days in captivity. Nor was there any evidence that survival rates increased from year 1 to year 2 for any cohort.

The effects of season on survival rates appeared to be driven primarily by lower survival rates during the breeding season for release-year cohorts 2 and 3, particularly for females (Figure 2.2). Females that had not established a home range before or during the breeding season in year 1 were

Table 2.7. Estimated survival rates for established fisher populations in North America (modified from Buskirk et al. 2012).

Study	**Location**	**Population segment**	**Survival interval**	**Survival rate (95% CI)**
Belant 2007[a,b]	Michigan	All fishers	Annual	0.89 (0.50-0.99)
Jordan 2007[c]	Sierra Nevada, CA	Males	Annual	0.88 (0.54-0.98)
		Females	Annual	0.88 (0.59-0.97)
Koen *et al.* 2007[a,b]	Ontario	Males >9 months old	Annual	0.33 (0.18-0.60)
		Females >9 months old	Annual	0.63 (0.47-0.86)
		Adult males	Annual	0.45 (0.24-0.83)
		Adult females	Annual	0.81 (0.72-0.91)
		Males >9 months old	2-year	0.15 (0.04-0.50)
		Females >9 months old	2-year	0.51 (0.34-0.78)
Weir & Corbould 2008[a]	British Columbia	Females	Annual	0.88 (n/a)
		All fishers	Annual	0.71 (n/a)
Aubry & Raley 2006[b,c]	Oregon Cascades	Males	Annual	0.85 (n/a)
		Females	Annual	0.78 (n/a)
Truex *et al.* 1998[c]	California			
	So. Sierra Nevada	Males	Annual	0.73 (n/a)
	So. Sierra Nevada	Females	Annual	0.61 (n/a)
	Eastern Klamath	Males	Annual	0.86 (n/a)
	Eastern Klamath	Females	Annual	0.73 (n/a)
	North Coast	Males	Annual	0.84 (n/a)
	North Coast	Females	Annual	0.84 (n/a)
York 1996[a]	Massachusetts	Adult males	Annual	0.77 (0.63-0.95)
		Adult females	Annual	0.90 (0.80-1.00)
		Juvenile males	Annual	0.77 (0.46-1.00)
		Juvenile females	Annual	0.84 (0.65-1.00)
Krohn *et al.* 1994[a,b]	Maine	Adult males	Trapping season (39 d)	0.57 (0.42-0.78)
		Adult females	Trapping season (39 d)	0.79 (0.64-0.97)
		Juveniles (both sexes)	Trapping season (39 d)	0.38 (0.25-0.57)
		Adults (both sexes)	Non-trapping season	0.89 (0.81-0.99)
		Juveniles (both sexes	Non-trapping season	0.72 (0.53-0.99)
Paragi *et al.* 1994[a,b]	Maine	Adult females	Annual	0.65 (0.50-0.86)
		Juveniles (both sexes)	Annual	0.27 (0.14-0.50)

[a] Fishers were commercially trapped in the study area

[b] Fishers were translocated previously near this study area

[c] Fishers were not commercially trapped

probably at greater risk of mortality because they were still seeking a suitable home range, and as suggested by the movements of many females (Chapter 1), they may have been seeking a mate as well.

The lower survival estimates we found for males during the breeding season are consistent with their extensive breeding-season movements and the fighting that can occur between males during the breeding season (Douglas and Strickland 1987; Chapter 1). Atypical breeding season movements by females and the typical breeding-season movements of males are likely to increase the risk of mortality, especially when those movements occur in a novel environment (Chapter 1).

Previous studies indicated that survival rates generally vary with sex and age-class; adult females often had the highest survival rates, adult males typically had lower survival rates, and juveniles had the lowest survival rates in the population (Table 2.7). In our study, we also found that sex and age-class influenced fisher survival rates; however, they had the opposite effect in our study. The point estimates for survival rates in this population were highest for juvenile male fishers, lower for adult males and juvenile females, and lowest for adult females. We believe that there are three primary explanations for these results, all of which are related to fishers being released into an unfamiliar environment. First, the survival of juvenile fishers was likely higher because juveniles are adapted to disperse and to successfully establish themselves in unfamiliar environments (Greenwood 1980, Waser 1996, Chapter 1). The propensity for juveniles to traverse and exploit foreign environments when dispersing from their natal range helps to explain the differences between juvenile survival rates and those of adult females, which are adapted to occupy and exploit an established home range that can support successful reproduction. Second, among adults, males are more likely to disperse and to disperse greater distances than females (Greenwood 1980, Waser 1996); males are also adapted to move extensively during the breeding season through familiar and unfamiliar environments while seeking and competing for reproductive females (Leonard 1986, Arthur *et al.* 1989, Aubry *et al.* 2004, Weir and Corbould 2007). Third, the smaller size of female fishers (adults and juveniles) makes them more vulnerable than males to predation by mid-sized carnivores (*e.g.*, bobcats and coyotes; Wengert *et al.* 2013), which likely contributes to the lower survival rates for adult and juvenile females. Although no data are available from previous fisher translocations for comparison, studies associated with translocations of the Arabian oryx (*Oryx leucoryx*; Stanley-Price 1989), the golden lion tamarin (*Leontopithecus rosalia*;

Kleinman *et al.* 1991) and the North-Island saddleback (*Philesturnus rufusater*; Armstrong *et al.* 2005) indicated that juveniles and younger individuals had higher initial survival rates than adults.

Predation made up a large percentage of the mortalities in our study and was among the most common sources of mortality in other fisher translocations (Table 2.8). Predation was also a significant source of mortality in three studies of established fisher populations in California (Table 2.8). Wengert *et al.* (2013) reported predation as the source of mortality for 62 of 101 fisher carcasses recovered from throughout the fisher's range in California. Predation in California and in this study appears to be among the most substantial source of mortality for fishers, yet predation has not been reported to be a significant source of fisher mortalities previously (Powell 1993). However, recent advances in our ability to identify causes of mortality, and in the forensic identification of predators have greatly improved our ability to identify predation events and the predators that were responsible (see Wengert 2013, Wengert *et al.* 2013).

Vehicle strikes were also an important mortality source for fishers in our study, but this was not the case for other translocations. Five of the seven vehicle strikes occurred on the only highway in our study area (US Highway 101), which varies in width from two to four lanes. While other paved roads may present risks to fishers, as indicated by two females that were killed on paved roads with reduced speed limits, highways appear to be a significant threat to fisher survival. This threat was likely exacerbated in our study area because Highway 101 surrounds the interior of the study area and is in close proximity to large expanses of high-quality fisher habitat in Olympic National Park and Olympic National Forest.

Mortality from trapping in our study area was limited to a single female that was incidentally captured in a leg-hold trap. In Washington state, the use of body-gripping traps on non-tribal lands has been prohibited since 2000 (Initiative 713; see Regulatory Code of Washington 77.15.194), and this prohibition likely limited the risk of trapping-related mortality. In other fisher translocation projects and population studies, however, trapping was a substantial cause of fisher mortality (Table 2.8).

Table 2.8. Cause and number (%) of fisher mortalities from recent studies in North America (modified from Lofroth et al. 2010).

			Anthropogenic				Natural		
Study	Location	No. mortalities	Trapping	Poaching	Vehicle Strike	Other[a]	Predation	Other[b]	Unknown
<u>Translocation studies</u>									
Roy 1991	Western Montana	14	3				9		2
Heinemeyer 1993	Western Montana	14	3			4	2		5
Weir 1995	Central British Columbia	5	2					2	1
Fontana *et al.* 1999	SE British Columbia	8	2				3		3
Serfass *et al.* 2001	Pennsylvania	3	1		1		1		
	total	44	11 (25)		1 (2)	4 (9)	15 (34)	2 (5)	11 (25)
<u>Other studies</u>									
Buck *et al.* 1983	NW California	7					4		3
Krohn *et al.* 1994	Maine	50	40	3	2	1	2	2	1
Truex *et al.* 1998	California (3 study areas)	24		2	2	1	9		10
Higley *et al.* 1998, Higley & Mathews 2006	NW California	12	2				8	1	1
Aubry & Raley 2006	SW Oregon	9		1		1	2	4	1
York 1996	Massachusetts	12	4	1	3		2	1	1
Koen *et al.* 2007	Ontario	28	9		2	1	2	6	8
Weir & Corbould 2008	Central British Columbia	9	3			1	1	3	1
	total	151	58 (38)	7 (5)	9 (6)	5 (3)	30 (20)	17 (11)	26 (17)
<u>This Study</u>	Olympic Peninsula, WA	35	1 (3)		7 (20)		14 (40)	2 (6)	11 (31)

[a] Includes mortalities associated with handling by researchers, radio-collars, predation by domestic dogs, being trapped in cisterns or railroad cars.

[b] Includes drowning, emaciation/starvation, infections, disease, accidents, and freezing.

IMPLICATIONS FOR FUTURE REINTRODUCTIONS

Population simulations conducted by Lewis *et al.* (2012) indicated that the likelihood of reintroduction success was nearly the same regardless of how many years (1-5) a founder population of 80 or 100 fishers was released. Our results, however, indicate that survival is highly stochastic, and multiple years of releases may be beneficial in protecting against random effects on survival. Obtaining 80 or more fishers in a single year may be difficult to accomplish, and many reintroduction projects would have to be conducted over multiple years to obtain the target founder population size. By conducting a fisher reintroduction over ≥ 2 years, managers can limit the adverse effects of low-survival years (*e.g.*, year 2 in our study) by spreading the risk of a low-survival event over a greater period of time.

The lower survival rates we observed for females suggest that a founder population that is biased toward females (55-65% females) could compensate for low adult female survival and also result in greater reproduction and recruitment than a population with an even or male-biased sex-ratio.

Given the limited resources that are available for most reintroductions, managers should release a founder population that is structured to improve reproduction and survival rates and, ultimately, reintroduction success. Despite the lower survival rates we found for adult females, a founder population that comprises a disproportionately high percentage of adult females (or simply a large number of them; *e.g.*, 30-40) would still be justified because of their importance for reproduction (Sarrazin and Legendre 2000). The greater survival rates we found for juvenile females indicates their value as founder individuals because they can mate shortly after being released, they are more likely than adult females to survive their first year, and they may give birth to their first litter in year 2.

In established fisher populations, Buskirk *et al.* (2012), found that population growth (lambda) was most strongly influenced by the survival rates of juvenile females and older adult females. In a translocated population, the survival rates for females may be even more influential, but their effects on lambda are unknown in the absence of data on fecundity. In a translocated population that is stable or growing, lower survival rates for adult females may be offset by the greater survival rates we observed for juvenile females.

As the dominant breeders in a fisher population, adult males are essential for reintroduction success and have only slightly lower survival rates than juvenile males. Lewis *et al.* (2012) argued that the importance of the number of male fishers released as a predictor of reintroduction success was due to the greater number of large adult males (*i.e.*, >2 years-old and >4.5 kg) in translocation projects that released a greater number of males. Given their relatively high survival rates and their importance for reproduction, managers can structure a founder population that includes a disproportionately large percentage of large adult males (*e.g.*, 15% of the male population) to increase the likelihood of success.

To minimize the mortality rate of founders, reintroduction or recovery areas should, whenever possible, contain few highways, especially those that surround or bisect such areas; release sites should also be located well away from highways (*e.g.*, >15 km). Given the importance of predation as a cause of mortality, additional research is needed to identify the most important predators of reintroduced fishers, and evaluate actions that facilitate fisher reestablishment in relation to predation risks. Because fishers are susceptible to incidental capture in traps set for other species (Powell 1993, Lewis and Zielinski 1996), the prohibition of body-gripping traps for capturing furbearers in Washington may have minimized trapping-related mortalities in our study area. The protection provided by this prohibition is likely to facilitate population growth on the Olympic Peninsula and minimize trapping-related mortalities in the Cascade Range, where future reintroductions are planned (Lewis 2013).

LITERATURE CITED

Armstrong, D.P. and P.J. Seddon. 2007. Directions in reintroduction biology. Trends in Ecology & Evolution 23:20-25.

Armstrong, D.P., R.S. Davidson, J.K. Perrott, J. Roygard and L. Buchanan. 2005. Density-dependent population growth in a reintroduced population of North-Island sadddlebacks. Journal of Animal Ecology 74:160-170.

Arthur, S.M. 1988. An evaluation of techniques for capturing and radiocollaring fishers. Wildlife Society Bulletin 16(4):417-421.

Arthur, S.M., R.A. Cross, T.F. Paragi and W.B. Krohn. 1992. Precision and utility of cementum annuli for estimating ages of fishers. Wildlife Society Bulletin 20:402-405.

Arthur, S.M., T.F. Paragi and W.B. Krohn. 1993. Dispersal of juvenile fishers in Maine. Journal of Wildlife Management 57(4):868-874.

Aubry, K.B and J.C. Lewis. 2003. Extirpation and reintroduction of fishers (*Martes pennanti*) in Oregon: implications for their conservation in the Pacific states. Biological Conservation 114:79-90.

Aubry, K.B., S.M. Wisely, C.M. Raley and S.W. Buskirk. 2004. Zoogeography, spacing patterns, and dispersal in fishers: insights gained from combining field and genetic data. Pages 187-206 in D.J. Harrison, A.K. Fuller, and G. Proulx, editors. Marten and fishers (*Martes*) in human-altered environments: an international perspective. Springer, New York, New York.

Aubry, K.B. and C.M. Raley. 2006. Ecological characteristics of fishers (*Martes pennanti*) in the southern Oregon Cascade Range, update: July 2006. U.S. Forest Service, Pacific Northwest Research Station, Olympia, Washington. 31 pp.

Bailey, V.O. 1936. The mammals and life zones of Oregon. North American Fauna, No.55. 416 pp.

Belant, J.L. 2007. Human-caused mortality and population trends of American marten and fisher in a U.S. National Park. Natural Areas Journal 27:155-160.

Berg, W.E. 1982. Reintroduction of fisher, pine marten and river otter. Pages 159-175 in G. C. Sanderson, editor. Midwest furbearer management. North Central Section of The Wildlife Society, Bloomington, Illinois.

Breitenmoser, U., C. Breitenmoser-Wursten, L.W. Carbyn and S.M. Funk. 2001. Assessment of carnivore reintroductions. Pages 240-281 in J.L. Gittleman, S.M. Funk, D.W. Macdonald and R.K. Wayne, editors. Carnivore Conservation, Cambridge University Press, New York, New York.

Buck, S., C. Mullis and A. Mossman. 1983. Final report: Corral Bottom-Hayfork Bally fisher study. Arcata, CA: Humboldt State University and USDA Forest Service.

Burnham, K.P. and D.R. Anderson. 2002. Model selection and inference: a practical information-theoretic approach. Springer-Verlag, New York.

Buskirk, S.W., J. Bowman and J. H. Gilbert. 2012. Population biology and matrix demographic modeling of American martens and fishers. Pages 77-92 in K.B. Aubry, W.J. Zielinski, M.G. Raphael, G. Proulx and S.W. Buskirk, editors. Biology and Conservation of Martens, Sables, and Fishers: A New Synthesis. Cornell Univ. Press. Ithaca, NY.

Cooch, E.G. and G.C. White. 2013. Program Mark: 'A gentle introduction', 11th ed. http://www.phidot.org/software/mark/docs/book/.

Dalquest, W.W. 1948. Mammals of Washington. Univ. of Kansas Publications. Lawrence, Kansas. 444 pp.

Dickens, M.J., D.J. Delehanty and L.M. Romero. 2010. Stress: an inevitable component of animal translocation. Biological Conservation 143:1329-1341.

Douglas, C.W. and M.A. Strickland. 1987. Fisher. Pages 511-529 in M.J. Novak, A. Baker, M.E. Obbard and B. Malloch, editors. Wild Furbearer Management and Conservation in North America. Ontario Ministry of Natural Resources, Toronto, Ontario.

Fischer, J. and D.B. Lindenmayer. 2000. An assessment of the published results of animal relocations. Biological Conservation 96(1):1-11.

Fontana, A.J., I.E. Teske, K. Pritchard and M. Evans. 1999. East Kootenay fisher reintroduction program, 1996- Ministry of Environment, Lands, and Parks, Cranbrook, British Columbia.

Franceschini, M.D., D.I. Rubenstein, B. Low and L.M. Romero. 2008. Fecal glucocorticoid metabolite analysis as an indicator of stress during translocation and acclimation in an endangered large mammal, the Grevy's zebra. Animal Conservation 11:263-269.

Franklin, J.F. and C.T. Dyrness. 1988. Natural vegetation of Oregon and Washington. Oregon State University Press, Corvallis.

Greenwood, P.J. 1980. Mating systems, philopatry and dispersal in birds and mammals. Animal Behavior 28:1140-1162.

Griffith, B., J.M. Scott, J.W. Carpenter and C. Reed. 1989. Translocation as a species conservation tool: status and strategy. Science 245:477-480.

Grinnell, J., J.S. Dixon and J.M. Linsdale. 1937. Fur-Bearing Mammals of California. University of California Press, Berkeley.

Hartup, B.K., G.H. Olsen and N.M. Czekala. 2005. Fecal corticoid monitoring in whooping cranes (*Grus americana*) undergoing reintroduction. Zoo Biology 24:15-28.

Hayes, G.E. and J.C. Lewis. 2006. Washington State recovery plan for the fisher. Washington Department of Fish and Wildlife, Olympia, Washington. 69 pp. (at: http://wdfw.wa.gov/wlm/diversty/soc/fisher/).

Heinemeyer, K.S. 1993. Temporal dynamics in the movements, habitat use, activity, and spacing of reintroduced fishers in northwestern Montana. MS Thesis, University of Montana, Missoula, Montana.

Henderson, J.A., D.H. Peter, R.D. Lesher and D. C. Shaw. 1989. Forested plant associations of the Olympic National Forest. U.S. Forest Service Ecological Technical Paper 001-88, Portland, Oregon. 502 pp.

Higley, J.M ., J.S. Yaeger, A.B. Colegrove, A.J. Pole and D.A. Whitaker. 1998. Hoopa Valley Indian Reservation fisher study–progress report. USDI Bureau of Reclamation, USDI Bureau of Indian Affairs, and Hoopa Valley Tribe, Hoopa, California.

Higley, J.M. and S. Matthews. 2006. Demographic rates and denning ecology of female Pacific fishers (*Martes pennanti*) in northwestern California: preliminary report October 2004–July 2006. Hoopa Valley Tribe and Wildlife Conservation Society, Hoopa, California.

Houston, D.B. and E.G. Schreiner. 1994. The Olympic Mountains and environs . Pages 14-26 (Chapter 2) in D.B. Houston, E.G. Schreiner and B.B. Morehead, editors. Mountain goats in Olympic National Park: Biology and Management of an Introduced Species. Scientific Monograph NPS/NROLYM/NRSM-94/25. USDI National Park Service. Port Angeles, Washington.

Jordan, M.J. 2007. Fisher ecology in the Sierra National Forest, California. Ph.D. Dissertation, University of California, Berkeley.

Kleiman, D.G., B.B. Beck, J.M. Dietz and L.A. Dietz. 1991. Costs of a re-introduction and criteria for success: accounting and accountability in the Golden Lion Tamarin Conservation Program. Symposia of the Zoological Society of London 62:125–142.

Koen, E.L., J. Bowman and C.S. Findlay. 2007. Fisher survival in eastern Ontario. Journal of Wildlife Management 71(4):1214–1219.

Krohn, W.B., S.M. Arthur and T.F. Paragi. 1994. Mortality and vulnerability of a heavily trapped fisher population. Pages 137–146 in S.W. Buskirk, A.S. Harestad, M.G. Raphael and R.A. Powell, editors. Martens, sables, and fishers: biology and conservation. Cornell University Press, Ithaca, New York, USA.

Larkin, J.L, D.S. Maehr, J.J. Cox, D.C. Bolin and M.W. Wichrowski. 2003. Demographic characteristics of a reintroduced elk population in Kentucky. Journal of Wildlife Management 67:467-476.

Leonard, R.D. 1986. Aspects of reproduction of the fisher, *Martes pennanti*, in Manitoba. Canadian Field-Naturalist100(1):32-44.

Lewis, J.C. 2006. Implementation plan for reintroducing fishers to Olympic National Park. Washington Department of Fish and Wildlife, Olympia. 31 pp. (at: http://wdfw.wa.gov/wlm/diversty/soc/fisher/)

Lewis, J.C. 2013. Implementation plan for reintroducing fishers to the Cascade Mountain Range in Washington. Washington Department of Fish and Wildlife, Olympia. 29 pp. (at: http://wdfw.wa.gov/publications/01556/)

Lewis, J.C. and P.J. Happe. 2008. Olympic Fisher Reintroduction Project: 2008 Progress Report. Washington Department of Fish and Wildlife, Olympia. (at: http://wdfw.wa.gov/wlm/diversty/soc/fisher/)

Lewis, J.C., P.J. Happe, K.J. Jenkins and D.J. Manson. 2010. Olympic Fisher Reintroduction Project: 2009 Progress Report. Washington Department of Fish and Wildlife, Olympia.

Lewis, J.C., P.J. Happe, K.J. Jenkins and D.J. Manson. 2011. Olympic Fisher Reintroduction Project: 2010 Progress Report. Washington Department of Fish and Wildlife, Olympia.

Lewis, J.C. and G.E. Hayes. 2004. Feasibility assessment for reintroducing fishers to Washington. Washington Department of Fish and Wildlife, Olympia. 70 pp. (at: http://wdfw.wa.gov/wlm/diversty/soc/fisher/)

Lewis, J.C., R.A. Powell and W.J. Zielinski. 2012. Carnivore translocations and conservation: insights from population models and field data for fishers (*Martes pennanti*). PLoS ONE 7(3): e32726.

Lofroth, E.C., C.M. Raley, J.M. Higley, R.L. Truex, J.S. Yaeger, J.C. Lewis, P.J. Happe, L.L. Finley, R.H. Naney, L.J. Hale, A.L. Krause, S.A. Livingston, A.M. Myers and R.N. Brown. 2010. Conservation of fishers (*Martes pennanti*) in south-central British Columbia, western Washington, western Oregon, and California–Volume I: conservation assessment. USDI Bureau of Land Management, Denver, Colorado.

McCorquodale, S.M., C.D. Danilson, M.A. Davison, J.S. Bohannon, W.C. Madsen, R.C. Cook and J.G. Cook. 2013. Survival of native and autumn-translocated elk in a recovering western Washington population. Wildlife Society Bulletin 37(2):352-359.

Miller, B., K. Ralls, R.P. Reading, J.M. Scott and J. Estes. 1999. Biological and technical considerations of carnivore translocation: a review. Animal Conservation 2:59-68.

National Park Service. 1998. Strategic plan for Olympic National Park 1998-2002. Olympic National Park, Port Angeles, Washington.

National Park Service. 2005. Fire Management Plan and Environmental Assessment. Olympic National Park, Port Angeles, Washington.

National Park Service. 2005. Fire Management Plan and Environmental Assessment. Olympic National Park, Port Angeles, Washington.

Nielsen, L. 1988. Definitions, considerations, and guidelines for translocation of wild animals. 12–51 in L. Nielson and R.D. Brown, editors. Translocation of Wild Animals. Wisconsin Humane Society and Caesar Kleberg Wildlife Research Institute, Milwaukee, Wisconsin. 333 p.

Paragi, T.F., W.B. Krohn and S.M. Arthur. 1994. Using estimates of fisher recruitment and survival to evaluate population trend. Northeast Wildlife 51:1–11.

Peel, M.C., B.L. Finlayson and T.A. McMahon. 2007. Updated world map of the Köppen-Geiger climate classification. Hydrology and Earth System Sciences 11:1633-1644.

Pollock, K.H., S.R Winterstein, C.M. Bunck and P.D. Curtis. 1989. Survival analysis in telemetry studies: the staggered entry design. Journal of Wildlife Management 53(1):7-15.

Powell, R.A. 1979. Ecological energetics and foraging strategies of the fisher (*Martes pennanti*). Journal of Animal Ecology 48:195-212.

Powell, R.A. 1993. The Fisher: Life History, Ecology, and Behavior. Second edition. University of Minnesota Press, Minneapolis, Minnesota.

Powell, R.A. and R.D. Leonard. 1983. Sexual dimorphism and energy expenditure for reproduction in female fisher *Martes pennanti*. Oikos 40:166–174.

Powell, R.A., J.C. Lewis, B.G. Slough, S.M. Brainerd, N.R. Jordan, A.V. Abramov, V. Monakov, P.A. Zollner and T. Murakami. 2012. Evaluating Translocations of Martens, Sables, and Fishers: Testing Model Predictions with Field Data. Pages 93-137 in K.B. Aubry, W.J. Zielinski, M.G. Raphael, G. Proulx and S.W. Buskirk, editors. Biology and Conservation of Martens, Sables, and Fishers: A New Synthesis. Cornell University Press. Ithaca, NY.

Roy, K.D. 1991. Ecology of reintroduced fishers in the Cabinet Mountains of northwest Montana. MS Thesis, University of Montana, Missoula.

Sarrazin, F. and S. Legendre. 2000. Demographic approach to releasing adults versus young in reintroductions. Conservation Biology 14(2):488-500.

Scott, J.M. and J.W. Carpenter. 1987. Release of captive-reared or translocated endangered birds: what do we need to know? Auk 104:544-545.

Seber, G.A.F. 1982. Estimation of animal abundance and related parameters. Blackburn Press, Caldwell, New Jersey.

Seddon, P.J. 2010. From reintroduction to assisted colonization: moving along the conservation translocation spectrum. Restoration Ecology 18(6):796-802.

Serfass, T.L., R.P. Brooks and W.M. Tzilkowski. 2001. Fisher reintroduction in Pennsylvania: final report of the Pennsylvania fisher reintroduction project. Frostburg State University, Frostburg, Maryland.

Seton, E.T. 1926. Lives of game animals. Vol. 2. Doubleday, Doran & Co., New York.

Sikes, R.S., W.L. Gannon, and the Animal Care and Use Committee of the American Society of Mammalogists. 2011. Guidelines of the American Society of Mammalogists for the use of wild mammals in research. Journal of Mammalogy 92(1):235-253.

Smith, A. and M.M. Peacock. 1990. Conspecific attraction and the determination of metapopulation colonization rates. Conservation Biology 4:320–323.

Stamps, J.A. 1988. Conspecific attraction and aggregation in territorial species. American Naturalist 131:329–347.

Stanley-Price, M.R. 1989. Animal re-introductions: the Arabian oryx in Oman. Cambridge University Press, Cambridge, United Kingdom.

Strickland, M.A., C.W. Douglas, M. Novak and N.P.Hunzinger. 1982. Fisher (*Martes pennanti*). Pages 586-598 in J.A.Chapman and G.A. Feldhamer, editors. Wild Mammals of North America: Biology, Management, and Economics. Johns Hopkins University Press, Baltimore, Maryland.

Teixeira, C.P., C. Schetini De Azevedo, M. Mendl, C.F. Cipreste and R.J. Young. 2007. Revisiting translocation and reintroduction programmes: the importance of considering stress. Animal Behavior 73:1-13.

Truex, R.L., W.J. Zielinski, R.T. Golightly, R.H. Barrett and S.M. Wisely. 1998. A meta-analysis of regional variation in fisher morphology, demography, and habitat ecology in California. Draft report submitted to California Department of Fish and Game, Wildlife Management Division, Nongame Bird and Mammal Section, Sacramento, California.

USDA Forest Service and USDI Bureau of Land Management. 1994. Record of decision for amendments to Forest Service and Bureau of Land Management planning documents within the range of the northern spotted owl. USDA Forest Service and USDI Bureau of Land Management, Portland, Oregon.

U.S. Fish and Wildlife Service. 2004. Endangered and threatened wildlife and plants: 12-month finding for a petition to list the west coast distinct population segment of the fisher (*Martes pennant*i). Federal Register 69(68):18770-18792.

U.S. Fish and Wildlife Service. 2013. Endangered and threatened wildlife and plants; status review of the west coast distinct population segment of the fisher as endangered or threatened. Federal Register 78(53):16828-16829.

Waser, P.M. 1996. Patterns and consequences of dispersal in gregarious carnivores. Pages 267-295 in J.L. Gittleman, editor. Carnivore Behavior, Ecology, and Evolution. Cornell University Press, London.

Washington Department of Natural Resources. 1997. Habitat conservation plan. Washington Department of Natural Resources, Olympia.

Weir, R.D. 1995. Diet, spatial organization, and habitat relationships of fishers in south-central British Columbia. MS Thesis, Simon Fraser University, Burnaby, British Columbia, Canada.

Weir, R.D. and F.B. Corbould. 2007. Factors affecting diurnal activity of fishers in north-central British Columbia. Journal of Mammalogy 88(6):1508–1514.

Weir, R.D. and F.B. Corbould. 2008. Ecology of fishers in sub-boreal forests of north- central British Columbia. Final report PWFWCP Report No. 315. Peace/Williston Fish & Wildlife Compensation Program, Prince George, British Columbia, Canada.

Wengert, G.M. 2013. Ecology of intraguild predation on fishers (*Martes pennanti*) in California. Dissertation. University of California, Davis.

Wengert, G.M., M.G. Gabriel, J.E. Foley, T. Kun and B.N. Sacks. 2013. Field and molecular techniques for identifying intraguild predators of fishers and other North American small carnivores. Wildlife Society Bulletin 37:659-663.

White, G.C. and K.P. Burnham. 1999. Program MARK: survival estimation from populations of marked animals. Bird Study 46 Supplement:120-138.

Wolf, C.M., T. Garland, Jr. and B. Griffith. 1996. Predictors of avian and mammalian translocation success: reanalysis with phylogenetically independent contrasts. Biological Conservation 86: 243-255.

Yalden, D.W. 1993. The problems of reintroducing carnivores. Symposium of the Zoological Society of London 65:289-306.

York, E. 1996. Fisher population dynamics in north-central Massachusetts. MS Thesis, University of Massachusetts, Amherst.

CHAPTER 3

Resource Selection by Fishers Translocated to the Olympic Peninsula of Washington

JEFFREY C. LEWIS, *University of Washington, School of Environmental and Forest Sciences, Seattle, WA 98195 and Washington Department of Fish and Wildlife, 600 Capitol Way N, Olympia, WA 98501*

KURT J. JENKINS, *U.S. Geological Survey, Forest and Rangeland Ecosystem Science Center, 600 E. Park Ave., Port Angeles, WA 98362*

PATRICIA J. HAPPE, *Olympic National Park, 600 E. Park Ave., Port Angeles, WA 98362*

DAVID J. MANSON, *Olympic National Park, 600 E. Park Ave., Port Angeles, WA 98362*

MARC MCCALMON**,** *Washington Department of Fish and Wildlife, 600 Capitol Way N, Olympia, WA 98501*

INTRODUCTION

Carnivore translocations can be complicated, time-consuming, and expensive endeavors, and despite substantial investments to ensure their success, many fail (Reading and Clark 1996, Miller *et al.* 1999, Breitenmoser *et al.* 2001, IUCN 2012). To justify such investments and maintain both public and agency support for translocations, managers should be confident that a targeted recovery area can support a self-sustaining population before initiating a translocation (Yalden 1993). Thus an assessment of habitat suitability prior to initiation is key to planning a successful translocation (IUCN 2012). Although the long-term success of a translocation is a clear indication that habitat conditions are suitable, the failure of a reintroduction does not necessarily indicate that habitat conditions were inadequate for success. Translocations provide the opportunity to investigate resource selection by founder individuals, which can provide valuable information about habitat conditions in the recovery area and resource selection of individuals that establish home ranges within an unfamiliar environment. This investigation would also provide a valuable baseline to determine if descendants are more efficient than founder individuals at exploiting a recovery area.

The fisher (*Pekania pennanti*) is a mid-sized member (2-6 kg) of the weasel family (Mustelidae) that has a long and substantial translocation history (Berg 1982, Powell 1993, Lewis *et al.* 2012). The fisher once occurred throughout boreal and temperate forests in North America, but was extirpated from much of the southern portion of its range (*i.e.*, northern US and southern Canada;

Lewis *et al.* 2012) as a result of over-trapping, incidental capture of fishers in traps set for other furbearers, and loss of habitat (Powell 1993, Powell and Zielinski 1994, Lewis and Zielinski 1996). The fisher is easily trapped and the high value of its pelt (up to $350 per pelt in the early 1900s; Seton 1926, Bailey 1936, Grinnell *et al.* 1937, Dalquest 1948) made it vulnerable to over-exploitation in the US and southern Canada from the mid-1800s to the early 1900s, when there was little or no regulation of furbearer harvests. Extirpation, range contractions, and ongoing threats resulted in many efforts to restore fisher populations throughout much of the southern portion of the species' range through trapping regulations and translocations (Berg 1982, Powell 1993, Lewis *et al.* 2012). In 1998, the fisher was listed as an endangered species in Washington state (Hayes and Lewis 2006), and it is currently a candidate for listing under the federal Endangered Species Act as threatened or endangered throughout its West Coast range (western Washington, western Oregon, and California; USFWS 2004); a 12-month status review of the fisher is currently underway (USFWS 2013). The recovery tasks outlined in the Washington State Fisher Recovery Plan for the fisher stated that reintroductions to the Olympic and Cascade Recovery Areas would be required to recover the fisher in the state (Hayes and Lewis 2006).

Because fishers have been extirpated from their historical range in Washington since the mid-1900s (Aubry and Lewis 2003), there is little information beyond museum and trapping records to indicate the historical distribution or resource selection patterns of fishers in western Washington (Hayes and Lewis 2006). The Olympic Peninsula is unique within the fisher's historical range in that it contains 1) one of the largest remaining temperate rainforests in North America, 2) large areas of federally protected lands within Olympic National Park and Olympic National Forest (*i.e.*, the Olympic Fisher Recovery Area), 3) strong gradients in elevation, topography and climate, and 4) diverse landownerships (*e.g.*, federal, state, tribal, private, municipal) and forest management histories (Henderson *et al.* 1989, Houston *et al.* 1994, Lewis and Hayes 2004, Hayes and Lewis 2006, National Park Service 2008). Consequently, the findings of resource-selection studies from other portions of the fisher's range that were used to evaluate the feasibility of reintroducing fishers to Washington (Lewis and Hayes 2004) may have had limited applicability to the Olympic Fisher Recovery Area.

The reintroduction of fishers to Washington's Olympic Peninsula provided a unique opportunity to examine resource-selection patterns of fishers across a wide range of environmental and forest management conditions that are relatively common in the Pacific Northwest. In western North America, fishers commonly occur in low- and mid-elevation landscapes dominated by forests with moderate-to-high canopy cover containing large forest structures such as snags, logs, cavity trees, and log piles (Buskirk and Powell 1994, Powell and Zielinski 1994, Lofroth *et al.* 2010, Raley *et al.* 2012). Forests at low- to mid-elevations are likely to have higher productivity, support larger trees, and have lesser accumulations of snow than higher elevation forests (Franklin and Dyrness 1988, Meidinger and Pojar 1991). Relatively dense canopy cover is likely to provide: 1) escape cover for fishers (which are good climbers; Powell 1993), 2) cover for fisher prey, 3) suitable microclimates for den and rest sites (Weir *et al.* 2004), and 4) a sufficient canopy to intercept snow that would otherwise accumulate in greater depths on the forest floor and potentially impede efficient travel (Krohn *et al.* 1995, 1997, 2004). The large trees, snags, and logs that fishers use for den and rest sites (Paragi *et al.* 1996, Weir *et al.* 2012, Aubry *et al.* 2013) are commonly found in mid- and late-seral forest stands (Hansen *et al.* 1991, Spies and Franklin 1991).

Numerous studies conducted in western North America have investigated the selection or use of resources by fishers at the home range scale (*i.e.*, the selection or use of a home range within the larger landscape context; Buck *et al.* 1994, Jones and Garton 1994, Zielinski *et al.* 2004, Weir and Corbould 2010, and Sauder and Rachlow 2014). Buck *et al.* (1994) and Zielinski *et al.* (2004) found indications that female and male fishers exhibited different patterns of habitat use at this scale. Two recent studies of resource selection included sample sizes that were too small to investigate sex-specific selection of home ranges by fishers (Weir and Corbould 2010, Sauder and Rachlow 2014). In these cases, selection patterns that were exhibited by one sex could be confounded by the patterns (or lack of a pattern) of the other sex.

Building on hypotheses presented by Buskirk and Powell (1994) and Raley *et al.* (2012), we hypothesized that home range selection would differ between male and female fishers, reflecting gender differences in body size and reproductive requirements (Powell 1993). Females are smaller than males (2-3 kg versus 4-6 kg; Powell 1993, Lewis *et al.* 2011) and are more susceptible to predation by other mid-sized carnivores (e.g., bobcats [*Felis rufus*] and coyotes [*Canis latrans*]; Wengert *et al.* 2013). Consequently females would be likely to avoid highly fragmented forest landscapes (e.g., industrial timberlands), where bobcat and coyote densities are expected to be greater than in continuous forest (Voigt and Berg 1987, Anderson and Lovallo 2003). Moreover, because females must give birth and raise young in large woody structures (e.g., large snags, cavity trees and down logs; Lofroth *et al.* 2010), they would be expected to use forests that have greater abundances of these structures. Hence, we predicted that females would be more closely tied to landscapes dominated by continuous mid- or late-seral forests (Raley *et al.* 2012).

Fishers will cross small forest openings to access forest stands within their home ranges, however large open areas (*e.g.*, wetlands, meadows, agricultural fields, clearcuts >10 ha) are likely to be avoided by fishers (Buskirk and Powell 1994, Powell and Zielinski 1994, Zielinski *et al.* 2004). Consequently, the amount of open habitat (e.g., recently logged, wetlands) was negatively associated with fisher occupancy in British Columbia (Weir and Corbould 2010). Because developed areas, paved roads, and areas with an abundance of human activity (*e.g.*, campgrounds, off-road vehicle recreation areas, timber harvest units, ski areas) may pose threats to fishers (Naney *et al.* 2012), they may be more likely to avoid these areas when establishing a home range.

The IUCN (1987, 1995, 2012) recommends that a feasibility assessment be conducted prior to a reintroduction to determine if it is likely to succeed. Such assessments confirm 1) an adequate amount and distribution of habitat, 2) a suitable and available source population, 3) the removal or cessation of threats that previously caused extirpation in the proposed recovery area, and 4) an ability to effectively implement and monitor the success of the proposed reintroduction. The feasibility assessment conducted for potential fisher reintroductions on the Olympic Peninsula and in the Cascade Range in Washington (Lewis and Hayes 2004) identified the Olympic Peninsula as the most suitable location to conduct the first fisher reintroduction in the state. An investigation of home range selection by translocated fishers would allow us to test assumptions of fisher habitat selection used in the feasibility assessment, refine the assessment process for future translocations, and examine hypothesized differences in large-scale selection patterns between males and females.

To restore a self-sustaining population of fishers in the state, the Washington Department of Fish and Wildlife and National Park Service (Olympic National Park) translocated 90 fishers (50 F, 40 M) from central British Columbia to the Olympic Peninsula from 2008 to 2010. Our objectives in this study were to: 1) identify forest-structure and landscape characteristics associated with home range establishment by translocated male and female fishers, 2) identify differences in home range selection by male and female fishers, 3) evaluate our predictions for home range selection as well as our pre-release predictions of fisher habitat suitability, and 4) assess the implications of our findings for future fisher translocations and resource-selection studies.

STUDY AREA

The study area (14,412 km^2 [5,564 mi^2]) includes most of the Olympic Peninsula in western Washington, which is bordered by the Pacific Ocean to the west, the Strait of Juan de Fuca to the north, and Puget Sound to the east (Figure 3.1). The study area was delineated by the areal extent of all telemetry locations obtained for all fishers released on the Olympic Peninsula (with the exception of an exploratory movement outside the study area by 1 male). The center of the Peninsula is dominated by the Olympic Mountains as well as glaciated headwaters and steep drainages that radiate outward in all directions. Elevations range from sea level to 2,415 m (7,923 ft) at the top of Mt. Olympus near the center of Olympic National Park. The mountainous center of the Peninsula slopes to a pronounced coastal plain to the west and smaller plains to the north and east.

The Olympic Peninsula has a temperate maritime climate characterized by warm summers and cool winters (Peel *et al.* 2007). The one exception to this pattern is the northeastern corner of the Peninsula, which is in the rain shadow of the Olympic Mountains, and has a more Mediterranean climate characterized by warm and dry summers and cool winters (Peel *et al.* 2007). Annual precipitation ranges from 315 to 500 cm on the west slope of the Olympic Mountains; whereas annual precipitation is typically <40 cm in the northeastern corner of the Olympic Peninsula (National Park Service 1998). Eighty percent of annual precipitation on the Olympic Peninsula falls from October through March. Most winter precipitation falls as rain at elevations below 305 m (1,000 ft) and as snow above 760 m (2,500 ft).

The moist climate and broad range of elevations on the Olympia Peninsula profoundly affect vegetation patterns. Conifer forests dominate the Olympic Peninsula and include forests in the Sitka spruce (*Picea sitchensis*) zone (generally <180 m elevation but can be as high as 600 m in western valleys of the Peninsula), western hemlock (*Tsuga heterophylla*) zone (150-600 m on the west side, 0-1,200 m elevation on the east side), Pacific silver fir zone (*Abies amabilis*; mid-elevations), mountain hemlock zone (*Tsuga mertensiana*; generally >1,070 m elevation) and the subalpine fir zone (*Abies lasiocarpa*; >1,200 m elevation; Franklin and Dyrness 1988, Houston and

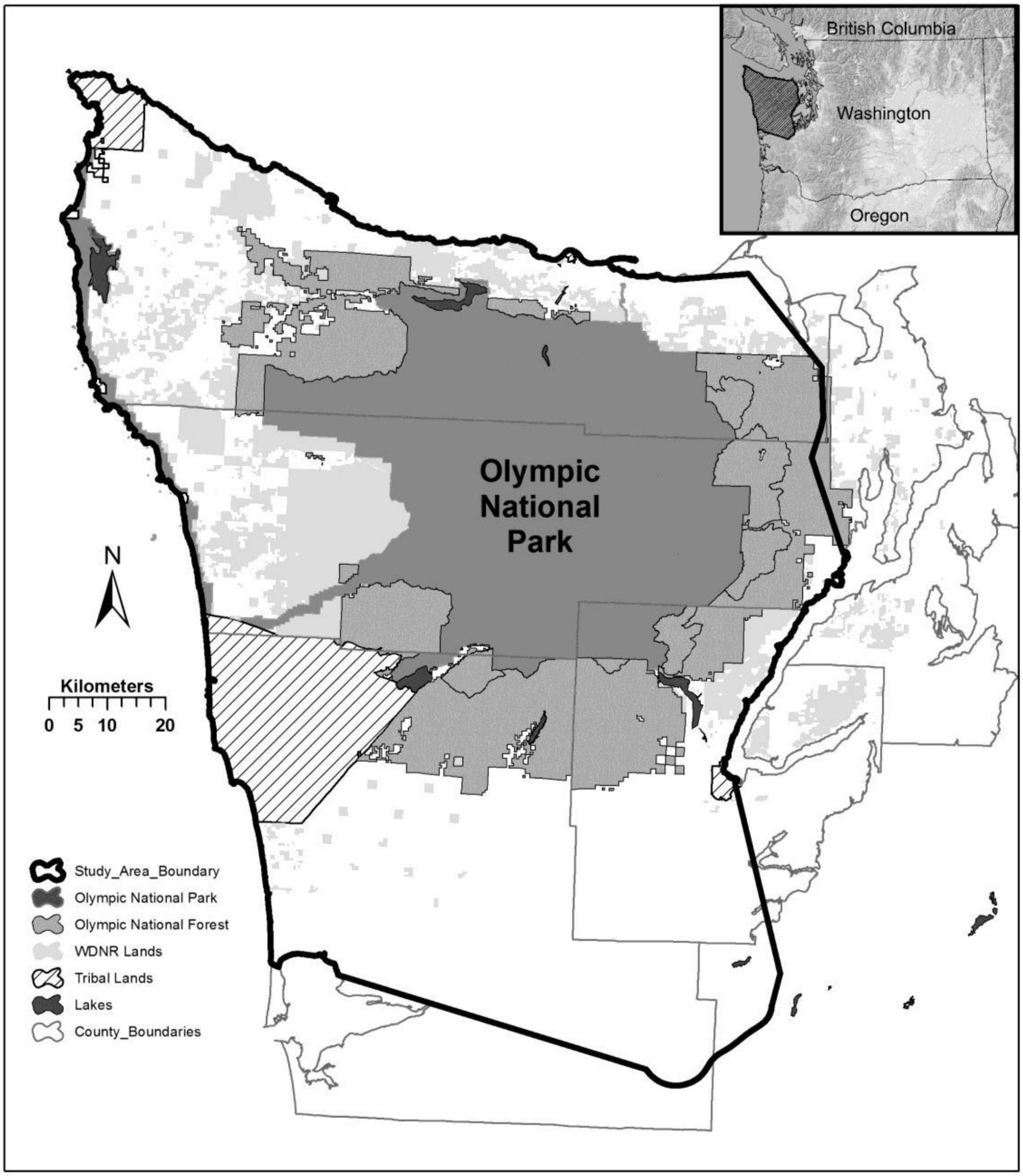

Figure 3.1. The study area of the Olympic fisher reintroduction project on the Olympic Peninsula, Washington, 2008-2011.

Schreiner 1994). Forests in the Sitka spruce zone on the western Olympic Peninsula are often referred to as temperate rainforests, and are characterized by large-diameter trees (>1 m), epiphytic plants, and dense understory vegetation (Franklin and Dyrness 1988). Hardwoods (red alder [*Alnus rubra*], bigleaf maple [*Acer macrophyllum*] and black cottonwood [*Populus trichocarpa*]) are common in riparian forests along major rivers (Houston and Schreiner 1994, National Park Service 2005).

Olympic National Park includes the mountainous center of the Peninsula as well as mid- to low-elevation forested river drainages. The Park interior contains about 284,955 ha (704,139 ac) of forest and almost all (96%) of this area is managed as wilderness. Olympic National Park contains one of the largest areas of contiguous temperate rainforest in North America. Much of the Park's circumference is bounded by Olympic National Forest, which encompasses mountainous terrain in the middle portions of many of the drainages that originate in the Park. Olympic National Park and Olympic National Forest make up the Olympic Recovery Area as outlined in the Washington State Fisher Recovery Plan (Hayes and Lewis 2006). Lower elevation lands outside the Olympic National Forest boundary are owned and managed by the Washington Department of Natural Resources (WDNR), Native American tribes, private timber companies and other private landowners, counties, and local municipalities.

Because of the drier climate in the eastern and northeastern portions of the Olympic Peninsula, wildfires in this area have had a more pronounced role in forest succession than in other areas of the Peninsula (Henderson *et al.* 1989). This history includes numerous fires since the 1850s that did not exceed 4,000 ha (9884 ac) in size and were widely distributed across this portion of the Peninsula. Smaller fires created mid-elevation landscape mosaics that are dominated by unmanaged mid- and late-seral forests in Olympic National Park and Olympic National Forest. A very large fire or series of fires (*i.e.,* 1.2-4.0 million ha [3-10 million ac]) occurred around 1701 within a 15-20 km band of forest along the northern and eastern periphery of the Olympic Peninsula. Remnant stands from the 1701 fire occur throughout the eastern portions and much of the northern portions of Olympic National Forest (Henderson *et al.* 1989). Because remnant stands from this large fire occur across a broad elevational gradient, and because site productivity declines with increasing elevation (Meidinger and Pojar 1991), the size of overstory trees in remnant stands is likely to decrease with increasing elevation. Whereas fires have been prevalent historically in the eastern and northeastern portion of the Olympic Peninsula, large fires have been much less frequent in the southern and western Olympic Mountains. Other disturbances, including wind-throw and landslides, have also altered forest structure and the forest landscape mosaic throughout the study area (Henderson *et al.* 1989).

Forests on Olympic National Forest and WDNR lands have been managed intensively since the 1950s; however, both ownerships also include areas of unmanaged conifer forests. As part of the Northwest Forest Plan (USDA Forest Service and USDI Bureau of Land Management 1994), much of Olympic National Forest is now managed as late-successional forest reserves, where older forests are protected and younger forests are managed to accelerate the development of older forest characteristics. Washington Department of Natural Resources' lands on the Olympic Peninsula are managed under the guidance of a Habitat Conservation Plan (Washington Department of Natural Resources 1997), which targets the retention of older forests for northern spotted owls (*Strix occidentalis caurina*) and marbled murrelets (*Brachyramphus marmoratus*) in portions of the western Olympic Peninsula. Private and tribal lands occur at lower elevations at the periphery of National Forest and WDNR lands; these lands are dominated by second-growth forests as a result of intensive timber management practices.

METHODS

We released 90 fishers (50 F, 40 M) at 21 sites in Olympic National Park from 27 January 2008 to 20 February 2010 with the goal of reestablishing a self-sustaining population of fishers in Washington (Table 3.1; see summaries by Lewis and Happe 2008 and Lewis *et al.* 2011). Release sites were distributed throughout much of the Park to facilitate fisher occupancy in a number of large landscapes dominated by late-seral conifer forests. In years 2 and 3, some fishers were released in the same areas as the previous years to correct imbalances in sex-ratios (Lewis *et al.* 2010, 2011), whereas other fishers were released in new areas to facilitate occupancy throughout the Peninsula. All fishers were immediately released upon arrival at their release site; there was no acclimation period prior to release, as has been the case for some fisher reintroductions (*i.e.,* soft releases; Roy 1991, Heinemeyer 1993, Weir 1995). Of the 90 fishers released, 82 (50 F, 32 M) were equipped with a 40-g VHF radio-collar (model MI-2 with mortality sensor, Holohil Systems Ltd., Carp, Ontario, Canada; http://www.holohil.com). Among the remaining eight fishers, five

Table 3.1. Number of fishers released on the Olympic Peninsula, Washington, from 27 January 2008 to 20 February 2010, by release-year cohort, release dates, sex and age-class.

	Fisher age-class[a]		
Release-year cohort 1 27 Jan 2008 2 Mar 2008	Juveniles	3	5
	Adults	9	1
	Total (18)	12	6
Release-year cohort 2 21 Dec 2008 17 Jan 2009 23 Feb 2009	Juveniles	6	11
	Adults	14	0
	Total (31)	20	11
Release-year cohort 3 24 Dec 2009 21 Jan 2010 20 Feb 2010	Juveniles	7	16
	Adults	11	7
	Total (41)	18	23
	Grand Total (90)	**50**	**40**

[a] For females, juveniles were <1 year old, and adults were ≥1 year old. For males, juveniles were <2 years old, and adults were ≥2 years old.

males (each weighing >4.5 kg) were equipped with a 120-g Argos satellite collar (Kiwisat 202, Sirtrack Ltd, Havelock North, New Zealand; http://www.sirtrack.com) and three males were equipped with a 41-g VHF transmitter surgically implanted in the abdominal cavity (model IMP/310/L with mortality sensor, Telonics Inc., Mesa, AZ, USA; http://www.telonics.com). Animal-handling procedures met or exceeded the guidelines of the American Society of Mammalogists for the use of wild mammals for research (Sikes and Gannon *et al.* 2011) and were approved by Washington Department of Fish and Wildlife's Animal Care and Use Committee.

Radio-Telemetry

Given the remote nature and broad geographic extent of the study area, (Figure 3.1), we used primarily aerial-telemetry methods to locate and track the movements of fishers for up to 2 years following their release. Telemetry flights originated from airports in Port Angeles, Shelton and Olympia, Washington. We attempted to locate each fisher at least once per week, but inclement weather prevented flights during some weeks, and the extensive and rugged nature of the study area, the relatively short transmission distance of VHF radio-collars, and the long-distance movements of many fishers, made it impossible to locate each fisher every week. Fisher relocations were obtained by scanning across the frequencies of all radio-collared fishers until a signal was detected. After a transmitter was detected, the pilot used standard aerial-telemetry methods to home in on radio-collared fishers and locate them as precisely as possible. Once we isolated the fisher to a specific location, we recorded that location with a global positioning system (GPS) and gave it an accuracy rating based on the flight biologist's judgment of signal strength, altitude of the aircraft, and time spent locating the fisher. An accuracy rating of 1.5 was given to locations with an estimated error <500 m, a rating of 2 for locations with an estimated error <1 km, a rating of 3 for locations with an estimated error <5 km, and a rating of 4 for locations with an estimated error >5 km. A location-accuracy rating of 1 was given to ground locations where the location was known (*e.g.*, carcass found, visual observation, fisher occupying a known den) or considered accurate within 100 m. We used standard ground-telemetry techniques to locate fishers at dens and rest sites, and to recover dead fishers.

We evaluated the accuracy of our aerial locations by placing test collars at 31 known locations within the study area that were representative of the range of topographic and vegetation conditions occupied by fishers. Because the locations of test collars were unknown to our pilots, we obtained unbiased estimates of location error by having our pilots estimate test-collar locations with the same intensity of search effort they used when locating a radio-collared fisher. Locations given an accuracy rating of 1.5 had a mean location error of 281 ± 47 m (± SE) and locations given an accuracy rating of 2 had a mean location error of 420 ± 117 m. Thus, the accuracy of most test-collar locations was generally much greater than the least accurate value used for each accuracy rating interval.

For analyses of post-release movements and home range estimation, we only used locations with an accuracy rating ≤3; however, <5% of the locations used in our analyses had an accuracy rating of 3. To avoid temporal autocorrelation, we used locations collected ≥24 hr apart. We also limited the number of ground locations that were included in an individual's data set to 1 during each 7-day period to minimize any bias associated with a greater number of ground locations in the more accessible areas. An exception to this rule was made for male M011 because the area M011 inhabited was entirely accessible by vehicle and his movements were consistently monitored from the ground by the staff of the Makah Tribal Forestry Department.

Home Range Estimation

After release, fishers generally explore their new environment prior to establishing a home range (Heinemeyer 1993, Proulx *et al.* 1994, Weir 1995, Chapter 1). The nature of this exploration may vary considerably among individuals, but once a translocated fisher finds a suitable location, its movements become more localized and shorter in distance as it establishes a home range. Consequently, we determined the timing of home range establishment by examining the movement patterns of individual fishers. To identify a change in movement distances associated with home range establishment, we calculated the mean-squared distance (MSD) between consecutive locations for each fisher, following the methods of Weir (1995) and Weir *et al.* (2009). MSD was calculated by squaring the six individual distances between the first seven locations (*e.g.*, distance from location 1 to 2, 2 to 3,…6 to 7) then averaging those six distances. This calculation was repeated for all locations in each fisher's dataset in increments of seven locations (*i.e.*, distances between locations 1-7, 2-8, 3-9, etc.) until a MSD value was calculated for all groupings of six consecutive movement distances. Because each distance was squared, one large movement (among the six) significantly increased MSD. To estimate the date when home range establishment began, we plotted MSD (y axis) against date (x axis); a minimum of 10 locations collected for $\geq$2 consecutive months were used to indicate home range establishment.

We estimated home range sizes for fishers with $\geq$20 locations collected over 6 or more consecutive months following the estimated date of home range establishment. To allow meaningful comparisons with other studies, we used two methods that have commonly been used to estimate the configuration and size of fisher home ranges in previous studies. We estimated home range boundaries using the fixed-kernel method with smoothing parameters selected by least-squares cross validation (Worton 1989) to create 95% utilization distribution (UD) contours; we delineated home range "core-areas" by generating 50% UD contours. We also estimated home range boundaries using the 100% minimum convex polygon (MCP) method. We used Home Range Tools for Arc GIS (Rodgers *et al.* 2007) to delineate home ranges and program Animal Space Use (Horne and Garton 2009) to calculate smoothing parameters.

Fishers are known to make exploratory forays outside their home range, and it is common for males to move long distances during the breeding season to search for reproductive females (Leonard 1986, Arthur *et al.* 1989, Aubry *et al.* 2004, Weir and Corbould 2007). We defined exploratory forays as locations >2 interquartile ranges from the median x or y location coordinates when they existed as isolated kernels (containing $\leq$2 locations) separated from the primary home range distribution. Because such forays occur outside home range boundaries, we eliminated all locations associated with such exploratory forays from our home range estimates.

Resource Data and Statistical Analysis

We examined the effects of 11 forest structure, landcover, elevation, and road variables on the selection of a home range by radio-equipped fishers; these data were also used to characterize available forest and landscape features within the study area (Table 3.2). Data available from the Landscape Ecology, Modeling, Mapping and Analysis (LEMMA) project of the U.S. Forest Service and Oregon State University (http://www.fsl.orst.edu/lemma/; Ohmann and Gregory 2002) were used to characterize forest structure within the study area at a 30 x 30 m pixel-size. LEMMA data were based on 2006 satellite imagery of the study area, and because of the broad scale of the area assessed, LEMMA project analysts recommended that these data be used for analyses conducted at

Table 3.2. Variables and sources of data used in models for evaluating resource selection by fishers on the Olympic Peninsula, Washington. Data for each variable was inventoried for each used home range (95% utilization contour) and core area (50% utilization contour) and each pseudo home range and core area.

Variable	**Definition**	**Data Source**[a]
Canopy cover	Mean pixel value (%) within a contour for percent canopy cover of all live trees	LEMMA – CANCOV
Mean QMDA	Mean pixel value (cm) within a contour for the Quadratic Mean Diameter of All dominant and co-dominant trees	LEMMA – QMDA_DOM
Small tree class[b]	% of pixels within a contour with CANCOV >40% and QMDA_DOM <25 cm	LEMMA –STRUCCOND (Structural condition class 3)
Intermediate tree class[c]	% of pixels within a contour with CANCOV >40% and QMDA_DOM of 25-50 cm	LEMMA –STRUCCOND (Structural condition class 4)
Intermediate to large tree class	% of pixels within a contour with CANCOV >40% and QMDA_DOM ≥25 cm	LEMMA –STRUCCOND (Structural condition classes 4, 5 or 6)
Large tree class	% of pixels within a contour with CANCOV >40% and QMDA_DOM >50 cm	LEMMA –STRUCCOND (Structural condition classes 5 or 6)
Mean elevation	Mean elevation (m) of the pixels within a contour	USGS 10-meter digital elevation models
Natural open area[d]	% of pixels within a contour that are classified as natural open area	National Landcover Database (2006)
Managed open area[d]	% of pixels within a contour that are classified as human-managed open area	National Landcover Database (2006)
Patch density	Number of patches/km^2 within a contour, where patches were groupings of contiguous pixels of sparse/open forest, small tree class forest, or intermediate to large tree class forest	Derived from LEMMA – STRUCCOND (Structural condition). Structural condition classes 1 and 2 combined = sparse/open forest; class 3 = small tree class; and classes 4, 5, and 6 combined = intermediate to large tree class
Paved roads[e]	% of pixels classified as a paved road	Washington Department of Natural Resources, Transportation data: using the ROAD_SUR_TY variable

[a] LEMMA is the Landscape ecology, modeling, mapping & analysis project of the US Forest Service and Oregon State University (http://www.fsl.orst.edu/lemma/).

[b] The small tree class was composed of early seral regenerating forests at low elevations and older forests at higher elevations.

[c] The intermediate tree class was composed of mid-seral forests at low elevations and older forests at higher elevations.

[d] National Landcover Database (2006) data was available at: http://www.mrlc.gov/nlcd2006.php; where natural open areas included open water, perennial snow/ice, barren land, shrub/scrub, herbaceous, and wetlands, and where managed open areas included developed open space, low to high intensity developed areas, hay/pasture, cultivated crop fields.

[e] Paved road data from Washington Department of Natural Resources was available at: http://fortress.wa.gov/dnr/app1/dataweb/metadata/trans.htm.

large spatial scales (*i.e.,* the landscape scale). We used only LEMMA data with accuracy assessments that indicated >70% agreement between observed (plot) and predicted (model) data (*i.e.,* agreement was 0.75 for canopy cover and 0.72 for quadratic mean diameter of dominant/co-dominant trees). Because resource selection by fishers is strongly influenced by forest structural characteristics (Zielinski *et al.* 2004, Weir *et al.* 2012, Aubry *et al.* 2013), we examined home range characteristics for male and female fishers in relation to forest structural classes based on overstory canopy and tree size (Table 3.2). We used forest-structure and landscape characteristic variables to develop and evaluate 17 models that represented seven *a priori* hypotheses of resource selection by fishers (Table 3.3). We used six sets of these 17 models to evaluate female selection of home ranges (model set 1) and core areas (2), to evaluate male selection of home ranges (3) and core areas (4), and to investigate differential resource use between the sexes based on comparisons between their home ranges (5) and core areas (6).

To evaluate the support for each model in model sets 1-4, we compared the characteristics of occupied home ranges and core areas to a sample of pseudo home ranges and core areas placed randomly throughout the study area (Thomas and Taylor 2006); pseudo home ranges and core areas served as controls for selection analyses. Fifty pseudo home ranges and core areas were created for each fisher for which we had sufficient telemetry locations to estimate a home range. We created pseudo home ranges and core areas with the same shape, size, and configuration as the corresponding occupied home range and core area (Katnik and Weilgus 2005). We randomly placed pseudo home ranges and core areas, and then rotated them in a random direction, within the portion of the study area that was within 66 km (for females) or 108 km (for males) of each fisher's release site. These distances were equal to the greatest distance observed between a release site and the center of an established home range for female and male fishers, respectively, and included the portion of the study area we considered to be most available to that individual. The random placement of pseudo home ranges and core areas for each individual was constrained to prevent overlap; however, overlap was allowed among different individuals. Pseudo home ranges that were located at the periphery of the study area (*i.e.*, the home range centroid was within the boundary) were clipped to exclude areas beyond the study area boundary, including the Pacific Ocean, Strait of Juan de Fuca, and Hood Canal. Spatial overlap of occupied and pseudo home ranges is frequently referred to as contamination, because of the effect of occupied home ranges on the

Table 3.3. Resource selection hypotheses, and corresponding models for evaluating resource selection by fishers on the Olympic Peninsula, Washington. Fisher selection is based on comparisons between occupied home ranges and available pseudo home ranges within the study area.

Hypothesis	Models	Reference(s)
Canopy cover: Fishers will select home ranges that contain forests with moderate to high canopy cover	1) Canopy cover	Carroll *et al.* 1999, Zielinski *et al.* 2004
Large-diameter woody structures: Fishers will select home ranges that contain forests with a greater abundance of large woody structures (*i.e.*, cavity trees, snags, logs); for use as den and rest sites.	2) Quadratic Mean Diameter of All Dominant and Co-dominant trees (QMDA)	Purcell *et al.* 2009, Weir *et al.* 2012
Forest structure class[a]: Fishers will select home ranges with moderate to high canopy cover and forests comprised of mainly intermediate (25-50 cm) or larger (>50 cm) tree sizes.	3) Large tree class (>50 cm) 4) Intermediate to large tree class (>25 cm) 5) Intermediate tree class (25-50 cm) 6) Small tree class (<25 cm)	Jones and Garton 1994, Zielinski *et al.* 2004
Elevation: Fishers will select home ranges at low or mid-elevations	7-12) by adding "elevation" to models 1-6 above	Buskirk and Powell 1994, Raley *et al.* 2012
Forest openings: Fishers will select home ranges with less open area	13) Natural open areas 14) Managed open areas 15) Natural + managed open areas	Weir and Corbould 2010; Sauder and Rachlow 2014
Forest fragmentation: Female fishers will select home ranges in areas with lower patch densities	16) Patch Density	Weir and Harestad 1997, Thompson *et al.* 2011
Human avoidance: Fishers will select home ranges with fewer anthropogenic features (*e.g.*, agricultural areas, developed areas, paved roads)	Model 14 above (Managed open areas) 17) Paved roads	Naney *et al.* 2012

[a] Tree size classes correspond with LEMMA forest structural condition classes 3 (small trees), 4 (intermediate trees), 4-6 (intermediate to large trees) and 5-6 (large trees; http://www.fsl.orst.edu/lemma/).

sample of unoccupied pseudo home ranges (Keating and Cherry 2004). However, due to the large size of the study area and the small number (and spatial extent) of occupied home ranges within the study area, there is a relatively small percentage of the study area that consisted of occupied home ranges. Simulations by Desrochers *et al.* (2010) indicated that sample contamination resulted in unbiased estimates when using a use-availability design with conditional (case-controlled) logistic regression, and Johnson *et al.* (2006) found that bias in probability values was not significant until contamination of available sites (*e.g.*, pseudo home ranges) exceeded 20%.

We used conditional logistic regression to compare occupied and pseudo home ranges and core areas for males and females (model sets 1-4) and used the one to many approach outlined by Hosmer and Lemeshow (2000), where one occupied home range was compared to 50 pseudo home ranges for each individual. We used PROC LOGISTIC with the STRATA command in program SAS (SAS Institute, Cary, NC) to run the conditional logistic regression and make comparisons of occupied and pseudo home ranges and core areas conditional to the individual fisher. This enabled us to control for the influence that the unique shapes and sizes of home ranges and core areas could have on patch density (as defined in Table 3.3). We also used PROC LOGISTIC to compare occupied home ranges and core areas of males with those of females to evaluate sex-specific resource use. We limited the number of variables in candidate models to maintain a >5 to 1 ratio of individual fishers to variables, as recommended by Vittinghoff and McCulloch (2007). To avoid issues of multicollinearity, we limited our use of multivariate models to those with variables that had Pearson correlation coefficients <0.65.

We used the small sample variant of Akaike's Information Criteria (AIC; Burnham and Anderson 2002) to assess the support for candidate models and to rank models in each of the six model sets. To illustrate the effect on selection of the variables in the best model, we calculated the probability of selection (Hosmer and Lemeshow 2000) and plotted it against the predictor variables, based on the range of values for predictor variables within pseudo home ranges and core areas. To demonstrate the support of highly ranked models, we model-averaged parameters from models with ΔAIC_c values <2 (Burnham and Anderson 2002). Lastly, we calculated odds ratios and associated 95% confidence intervals (CI) for the variables in each model to indicate the direction and degree of their effect on the probability of home range use. Model variables were considered influential in predicting fisher selection if the estimated odds ratio and its associated 95% CI did not overlap with 1.

RESULTS

We estimated home ranges and core areas for 19 females and 12 males (Figure 3.2). Mean (±SE) home range size estimated by 95% UDs was 64 ± 39 km^2 (range: 3-137 km^2) for females and 119 ± 19 km^2 (range: 45-239 km^2) for males. Mean age at home range establishment was 29 ± 4 months for females (range: 9-63 months) and 18 ± 2 months for males (range: 10-26 months). Home ranges of 15 (79%) of the 19 females fell primarily within the boundaries of Olympic National Park or Olympic National Forest (*i.e.,* within the Olympic Fisher Recovery Area; Figure 3.2), whereas the remaining four females (21%) occupied home ranges that included primarily non-federal lands (*i.e.,* private, state, or tribal lands; see ownerships in Figure 3.1). By contrast, only five (42%) of the 12 males occupied home ranges that were located primarily within the Recovery Area; the remaining seven males (58%) occupied home ranges that were located primarily on non-federal lands (Figure 3.2). We inventoried data for 11 forest and landscape-characteristic variables within occupied home ranges and core areas; these data were also inventoried for the 950 and 600 pseudo home ranges and core areas we generated for females and males, respectively (Table 3.4).

Home Range Selection by Females

Home range and core area selection by females was most closely associated with forests with a higher percentage of intermediate-sized trees and higher elevations. The model that included the percentage of area in the intermediate tree class and mean elevation was the best among the 17 models for both home ranges (w_i = 0.995) and core areas (w_i = 0.960; Table 3.5). The best models had 331 and 25 times more support than the second-ranking models for home ranges and core areas, respectively; the second ranking model for both model sets included only the percentage of forests in the intermediate tree class. Thus, the inclusion of mean elevation resulted in much greater support from the data.

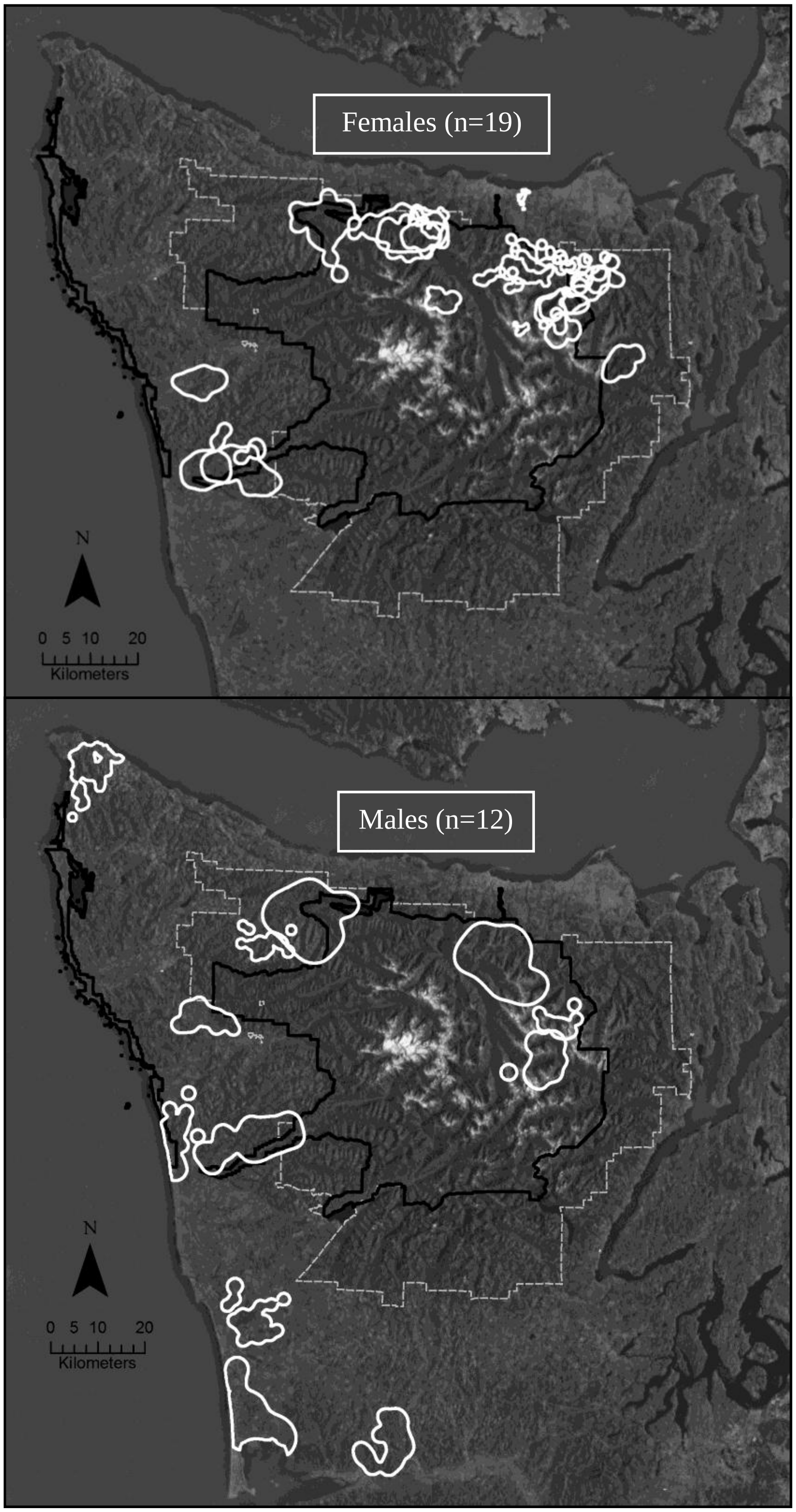

Figure 3.2. Estimated home ranges (95% contours) for female and male fishers on the Olympic Peninsula, Washington, from 2008 to 2011. Olympic National Park is delineated by the black line, whereas Olympic National Forest is adjacent to the Park and is delineated by the dashed white line.

Table 3.4. Summary data for variables used to assess resource selection within occupied and pseudo home ranges and core areas for female and male fishers on the Olympic Peninsula, Washington, from 2008 to 2011.

	Females – home ranges				**Males – home ranges**			
	Occupied (n=19)		**Pseudo (n=950)**		**Occupied (n=12)**		**Pseudo (n=600)**	
Variables	**Mean**	**SE**	**Mean**	**SE**	**Mean**	**SE**	**Mean**	**SE**
% canopy cover	70.61	1.58	69.81	0.18	67.32	1.54	69.15	0.15
Mean QMDA (cm)	34.19	0.99	35.39	0.21	30.54	1.52	33.12	0.21
% small tree class	19.36	2.71	19.56	0.45	32.11	3.70	25.07	0.62
% intermediate tree class	40.55	2.56	30.31	0.43	32.38	2.47	31.06	0.46
% intermediate to large tree class	65.19	3.97	63.45	0.64	49.19	5.00	56.85	0.82
% large tree class	24.64	2.92	33.14	0.78	16.82	3.30	25.78	0.89
Mean elevation (m)	903	102	673	12	585	160	556	16
% natural open area	16.13	3.27	20.38	0.48	26.74	4.23	23.49	0.52
% managed open area	3.84	2.60	3.45	0.28	3.12	0.84	3.52	0.23
Patch density (patches/km^2)[a]	5.82	0.62	5.44	0.10	6.88	0.49	5.81	0.10
% paved roads	0.45	0.21	0.45	0.03	0.50	0.14	0.44	0.03

	Females – core areas				**Males – core areas**			
	Occupied (n=19)		**Pseudo (n=950)**		**Occupied (n=12)**		**Pseudo (n=600)**	
Variables	**Mean**	**SE**	**Mean**	**SE**	**Mean**	**SE**	**Mean**	**SE**
% canopy cover	71.84	1.88	69.94	0.25	66.13	1.87	68.87	0.22
Mean QMDA (cm)	34.74	1.35	36.34	0.29	30.00	1.78	33.37	0.29
% small tree class	20.37	3.67	19.63	0.50	32.49	3.45	24.98	0.66
% intermediate tree class	42.91	3.03	29.97	0.49	31.93	2.59	30.94	0.53
% intermediate to large tree class	68.35	4.29	63.09	0.73	50.20	4.99	56.66	0.90
% large tree class	25.43	3.28	33.12	0.85	18.28	3.76	25.73	0.97
Mean elevation (m)	811	106	642	13	565	168	517	16
% natural open area	12.23	2.26	20.68	0.58	25.90	3.38	23.53	0.63
% managed open area	3.04	1.77	3.47	0.30	3.23	0.88	3.74	0.30
Patch density (patches/km^2)[a]	6.03	0.64	6.39	0.14	7.99	0.55	6.41	0.12
% paved roads	0.32	0.15	0.43	0.04	0.58	0.17	0.47	0.04

[a] Number of patches/km^2 within a core area or home range where patches were groupings of contiguous pixels of sparse/open forest, small tree class forest, or intermediate to large tree class forest $\geq$ 0.0081 km^2.

Table 3.5. Parameters of 17 models used to explain resource selection by female fishers on the Olympic Peninsula, Washington, 2008-2011. Model evaluations were based on data from the occupied home ranges and core areas of 19 female fishers and 950 pseudo home ranges and core areas.

Model	**Odds Ratio Variable 1**	**Odds Ratio Variable 2**	**K**	**AIC_c**	**ΔAIC_c**	**w_i**
Home ranges						
Intermediate tree class + elevation	1.096[a]	1.003[a]	4	127.901	0.000	0.995
Intermediate tree class	1.066[a]		3	139.760	11.860	0.003
Large tree class + elevation	0.967[a]	1.002[a]	4	140.627	12.726	0.002
QMDA + elevation	0.943	1.002[a]	4	144.584	16.683	0.000
Small tree class + elevation	1.031	1.002[a]	4	145.086	17.185	0.000
Canopy cover + elevation	1.024	1.002	4	146.745	18.844	0.000
Intermediate to large tree class + elevation	1.002	1.001	4	147.064	19.163	0.000
Large tree class	0.982		3	148.736	20.836	0.000
Natural open	0.976		3	149.619	21.719	0.000
QMDA	0.968		3	150.716	22.816	0.000
Canopy cover	1.030		3	151.000	23.100	0.000
Patch density	1.048		3	151.084	23.184	0.000
Intermediate to large tree class	1.005		3	151.265	23.365	0.000
Managed open	1.005		3	151.386	23.486	0.000
Small tree class	0.999		3	151.417	23.517	0.000
Paved roads	1.000		3	151.421	23.521	0.000
Natural open + managed open	0.976	1.004	4	151.605	23.704	0.000
Core areas						
Intermediate tree class + elevation	1.079[a]	1.002[a]	4	131.164	0.000	0.960
Intermediate tree class	1.060[a]		3	137.613	6.450	0.038
Natural open	0.959[a]		3	145.785	14.622	0.001
Large tree class + elevation	0.980	1.001	4	147.102	15.938	0.000
Natural open + managed open	0.959	0.997	4	147.782	16.618	0.000
QMDA + elevation	0.965	1.001	4	148.942	17.778	0.000
Canopy cover + elevation	1.041	1.001	4	149.102	17.938	0.000
Small tree class + elevation	1.020	1.001	4	149.245	18.081	0.000
Large tree class	0.987		3	149.613	18.450	0.000
Intermediate to large tree class + elevation	1.010	1.001	4	149.688	18.524	0.000
Canopy cover	1.040		3	150.110	18.947	0.000
Intermediate to large tree class	1.012		3	150.292	19.129	0.000
QMDA	0.977		3	150.745	19.582	0.000
Paved roads	0.880		3	151.197	20.034	0.000
Patch density	0.975		3	151.256	20.093	0.000
Small tree class	1.003		3	151.376	20.213	0.000
Managed open	0.994		3	151.379	20.216	0.000

[a] The 95% confidence interval for this odds ratio did not include 1.

Estimated odds ratios indicated that selection of home ranges by females was influenced positively by the percentage of forest in the intermediate tree class and mean elevation, and negatively by the percentage of forests in the large tree class. Similarly, odds ratios indicated that the selection of a core area by females was positively influenced by the percentage of forests in the intermediate tree class and mean elevation, and negatively by the percentage of natural open areas (Table 3.5). The best models for both home ranges and core areas indicated strong habitat selection by females. A 1% increase in the percentage of forests in the intermediate size class in the landscapes resulted in a 9.6 and 7.9% increase in the odds of selecting home ranges and core areas, respectively. In addition, a 1-m increase in mean elevation resulted in a 0.3 and 0.2% increase in the odds of selecting home ranges and core areas, respectively (Table 3.5). Within core areas, an increase in the probability of selection was greatest as the percentage of forest with intermediate-sized trees increased from 0 to 35% and as elevation increased from 25 to 800 m (Figure 3.3).

Home Range Selection by Males

Within the model set for male home ranges, the model that included mean QMDA and mean elevation had the greatest support ($w_i = 0.322$) for explaining male selection of home ranges (Table 3.6). The remaining 16 models in the data set had ΔAIC_c values >2.0 and <7.7 (Table 3.6), suggesting that males were not strongly selecting among the home range attributes in the models we evaluated. Estimated odds ratios for home ranges indicated selection only for mean QMDA (-) and the percent of forest in the small tree class (+) (Table 3.6). The odds of males establishing a home range decreased by 20.5% for each 1-cm increase in mean QMDA and increased 6.5% for each 1% increase in the percentage of forest in the small tree class.

The highest ranking model for explaining resource selection within male core areas included mean QMDA (-) and mean elevation (+) ($w_i = 0.200$), however there were six models that were highly supported (*i.e.*, ΔAIC_c values <2.0) and all 17 models had some support from the data (*i.e.*, ΔAIC_c values ≤ 6.1; Table 3.6). Shared support among the 17 models suggested that males were not highly selective of the core area attributes included in the models we evaluated. The six highest ranking models indicated that males did select core areas with a greater abundance of smaller trees, higher elevations, a greater percentage of forest in the small tree class, less canopy cover, and greater patch

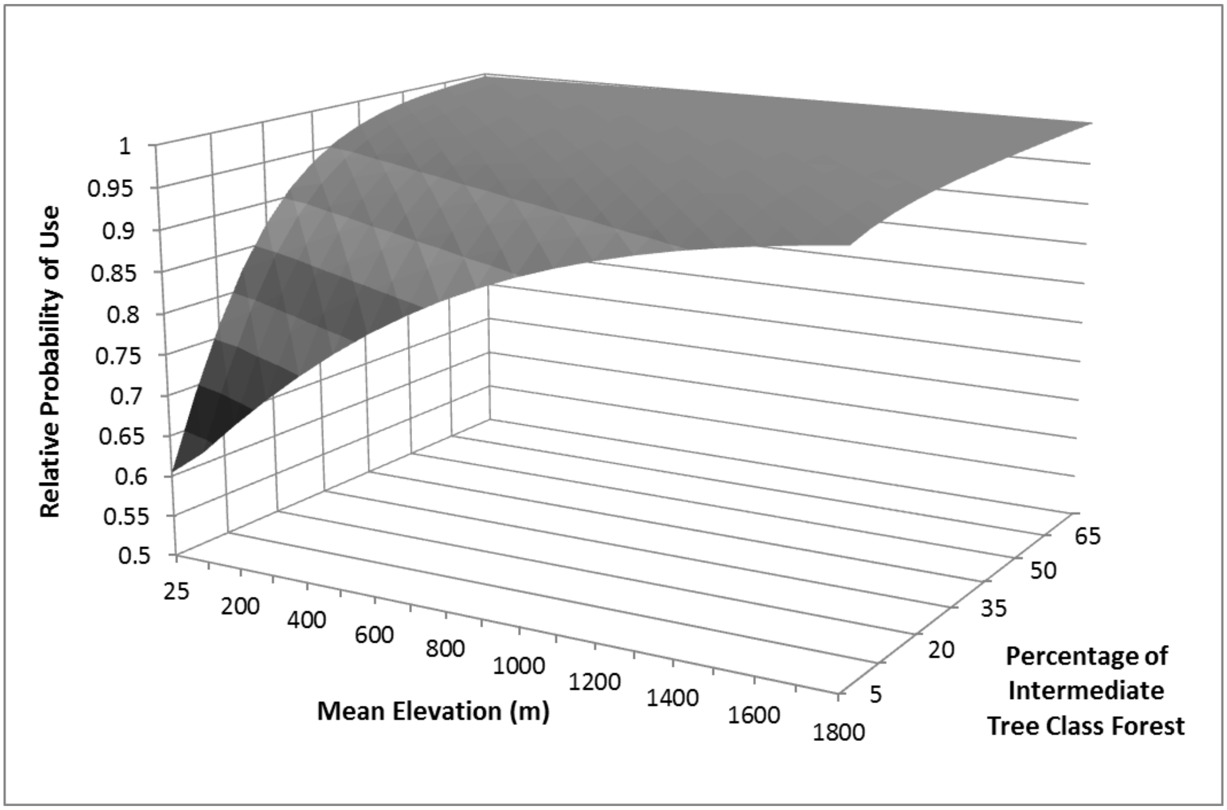

Figure 3.3. Relative probability of female use of core areas based on the percentage of forest in the intermediate tree-class and mean elevation for the range of values found within pseudo core areas. The percentage of forest in the intermediate tree-class and mean elevation comprised the best model for explaining female resource selection in home ranges and core areas. Probability calculations are relative because no intercept values are produced in conditional logistic regression for calculating a true probability.

Table 3.6. Parameters of 17 models used to explain resource selection by male fishers on the Olympic Peninsula, Washington, 2008-2011. Model evaluations were based on data from the occupied home ranges and core areas of 12 male fishers and 600 pseudo home ranges and core areas.

Model	**Odds Ratio Variable 1**	**Odds Ratio Variable 2**	**K**	**AIC$_c$**	**ΔAIC$_c$**	**w$_i$**
Home ranges						
QMDA + elevation	0.795[a]	1.002	4	90.460	0.000	0.322
Small tree class + elevation	1.065[a]	1.002	4	92.516	2.056	0.115
QMDA	0.912		3	93.569	3.109	0.068
Large tree class + elevation	0.953	1.001	4	93.803	3.343	0.061
Canopy cover	0.895		3	93.857	3.397	0.059
Patch density	1.227		3	93.866	3.406	0.059
Large tree class	0.976		3	94.026	3.566	0.054
Small tree class	1.028		3	94.033	3.573	0.054
Canopy cover + elevation	0.849[a]	1.001	4	94.250	3.790	0.048
Intermediate to large tree class	0.981		3	94.689	4.229	0.039
Intermediate to large tree class + elevation	0.970	1.001	4	95.348	4.888	0.028
Natural open	1.019		3	95.651	5.191	0.024
Intermediate tree class	1.011		3	96.222	5.762	0.018
Managed open	0.984		3	96.315	5.855	0.017
Paved roads	1.073		3	96.333	5.873	0.017
Natural open + managed open	1.019	0.980	4	97.571	7.111	0.009
Intermediate tree class + elevation	1.011	1.000	4	98.156	7.696	0.007
Core areas						
QMDA + elevation	0.864[a]	1.001	4	92.052	0.000	0.200
Patch density	1.200		3	93.043	0.991	0.122
Small tree class + elevation	1.052[a]	1.001	4	93.120	1.068	0.117
QMDA	0.926		3	93.466	1.414	0.099
Canopy cover	0.925		3	93.754	1.702	0.086
Small tree class	1.026		3	94.030	1.978	0.075
Canopy cover + elevation	0.911	1.001	4	94.821	2.769	0.050
Large tree class	0.984		3	95.057	3.005	0.045
Intermediate to large tree class	0.987		3	95.385	3.333	0.038
Large tree class + elevation	0.974	1.001	4	95.844	3.792	0.030
Natural open	1.010		3	96.108	4.056	0.026
Paved roads	1.084		3	96.265	4.213	0.024
Intermediate tree class	1.006		3	96.315	4.263	0.024
Managed open	0.988		3	96.316	4.264	0.024
Intermediate to large tree class + elevation	0.982	1.001	4	96.570	4.518	0.021
Natural open + managed open	1.010	0.986	4	98.049	5.997	0.010
Intermediate tree class + elevation	1.007	1.000	4	98.155	6.103	0.009

[a] The 95% confidence interval for this odds ratio did not include 1.

density. The odds ratios, however, indicated support only for male selection of core areas with smaller overstory trees and a greater percentage of forest in the small tree class (Table 3.6). The odds of a male selecting a core area decreased by 13.6% for each 1-cm increase in mean QMDA and increased by 5% for each 1% increase in forests in the small tree class (Table 3.6).

Nine of the 12 males occupied home ranges that were located at relatively low elevations (<644 m; $\bar{x}$ = 305 m), whereas the other three males occupied home ranges at relatively high elevations (1,157, 1,446 and 1,671 m; $\bar{x}$ = 1425 m) in Olympic National Park (Figure 3.2). Home range selection by these three males resulted in the mean elevation of occupied home ranges being slightly higher than the mean elevations of pseudo home ranges (585 [SE=160] m vs 556 [SE=16] m, respectively). Consequently, the addition of elevation in the highest ranking model (QMDA + mean elevation) resulted in an increase in support for the model (decrease of 3.109 in AIC_c), indicating that male selection was influenced by decreasing QMDA values and increasing elevation (Figure 3.4).

Comparing Female and Male Use of Home Ranges

Among the models we constructed to distinguish home ranges and core areas of males and females, three models had substantial support from the data. The highest ranking model included the percentage of forest in the intermediate tree class and mean elevation (w_i = 0.254), whereas the 2nd ranked model included the percentage of forest in the small tree class (w_i = 0.204), and the 3rd included the percentage of forest in the large tree class (w_i = 0.109; Table 3.7). Estimated odds ratios indicated that mean elevation and differences in the percentage of forests within different size classes distinguished female and male home ranges (Table 3.7). The odds of a female establishing a home range (in relation to a male) increased by 13.6% for a 1% increase in the percentage of forest in the intermediate tree class and by 0.2% for a 1-m increase in mean elevation. The best model for distinguishing female and male core areas contained only the percentage of natural open area (w_i = 0.465; Table 3.7); all other models had ΔAIC_c values >2.0. Estimated odds ratios indicate that, compared to males, females occupied core areas with less natural open area and greater percentages of forest in the intermediate to large tree class (Table 3.7). Within the core area, the predicted probability of female versus males use declined substantially (from >0.90 to <0.10) as the percentage of natural open area increased from 0 to 50% (Figure 3.5).

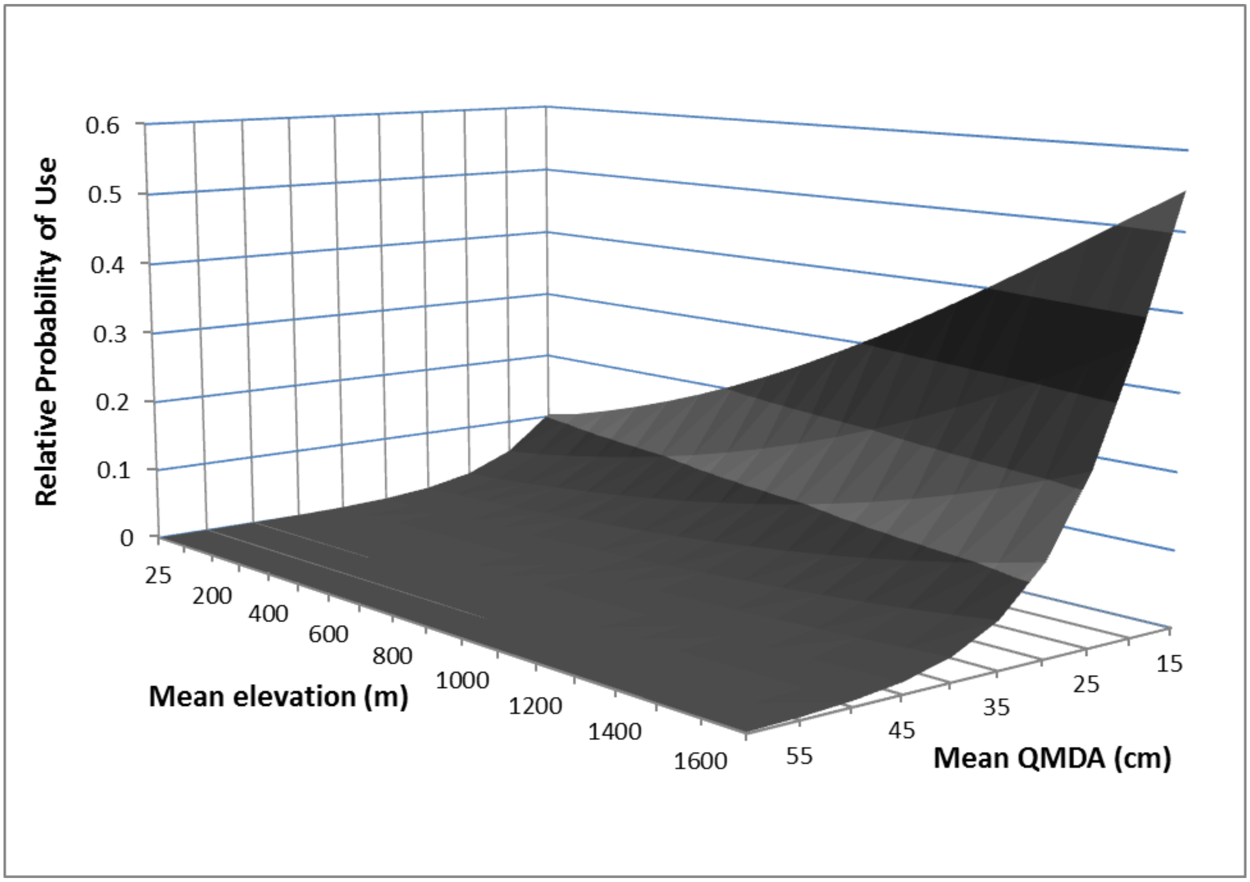

Figure 3.4. Relative probability of male use of core areas based on the mean QMDA and mean elevation for the range of values found within pseudo core areas. Relative probability predictions are based on model-averaged β coefficients from the highest ranking models for core areas that included QMDA and mean elevation. The best model for male home ranges also included only mean QMDA and mean elevation. Probability calculations are relative because no intercept values are produced in conditional logistic regression for calculating a true probability.

Table 3.7. Parameters of 17 models used to distinguish resource use between female and male fishers on the Olympic Peninsula, Washington, 2008-2011. Model evaluations were based on data from occupied home ranges and core areas of 19 female and 12 male fishers.

Model	**Odds Ratio Variable 1**	**Odds Ratio Variable 2**	**K**	**AIC_c**	**ΔAIC_c**	**w_i**
Home ranges						
Intermediate tree class + elevation	1.136[a]	1.002	4	38.395	0.000	0.254
Small tree class	0.923[a]		3	38.830	0.435	0.204
Intermediate to large tree class	1.053[a]		3	40.087	1.692	0.109
Intermediate tree class	1.100[a]		3	40.472	2.077	0.090
Intermediate to large tree class + elevation	1.048	1.001	4	41.209	2.814	0.062
Small tree class + elevation	0.929	1.000	4	41.247	2.852	0.061
QMDA	1.174		3	41.666	3.271	0.049
Natural open	0.951		3	42.041	3.646	0.041
Large tree class	1.055		3	42.829	4.434	0.028
QMDA + elevation	1.142	1.001	4	42.944	4.549	0.026
Canopy cover + elevation	1.082	1.001	4	43.825	5.430	0.017
Canopy cover	1.086		3	43.865	5.470	0.016
Large tree class + elevation	1.037	1.001	4	44.239	5.844	0.014
Patch density	0.823		3	44.325	5.930	0.013
Natural open + managed open	0.951	0.997	4	44.517	6.122	0.006
Managed open	1.010		3	45.775	7.380	0.006
Paved roads	0.913		3	45.790	7.395	0.003
Core areas						
Natural open	0.894[a]		3	35.794	0.000	0.465
Natural open + managed open	0.893[a]	1.016	4	38.231	2.437	0.138
Intermediate to large tree class	1.053[a]		3	39.338	3.544	0.079
Intermediate tree class	1.094[a]		3	39.382	3.588	0.077
Intermediate tree class + elevation	1.093[a]	1.001	4	40.101	4.307	0.054
Small tree class	0.946[a]		3	41.096	5.302	0.033
Patch density	0.721		3	41.378	5.584	0.029
QMDA	1.140		3	41.448	5.654	0.028
Intermediate to large tree class + elevation	1.050[a]	1.000	4	41.637	5.843	0.025
Canopy cover	1.105		3	41.818	6.024	0.023
Canopy cover + elevation	1.099	1.001	4	43.337	7.543	0.011
Small tree class + elevation	0.950	1.000	4	43.476	7.682	0.010
QMDA + elevation	1.126	1.000	4	43.764	7.970	0.009
Large tree classte-seral	1.040		3	43.798	8.004	0.009
Paved roads	0.516		3	44.590	8.796	0.006
Managed open	0.995		3	45.818	10.024	0.003
Large tree class + elevation	1.028	1.001	4	45.919	10.125	0.003

[a] The 95% confidence interval for this odds ratio did not include 1.

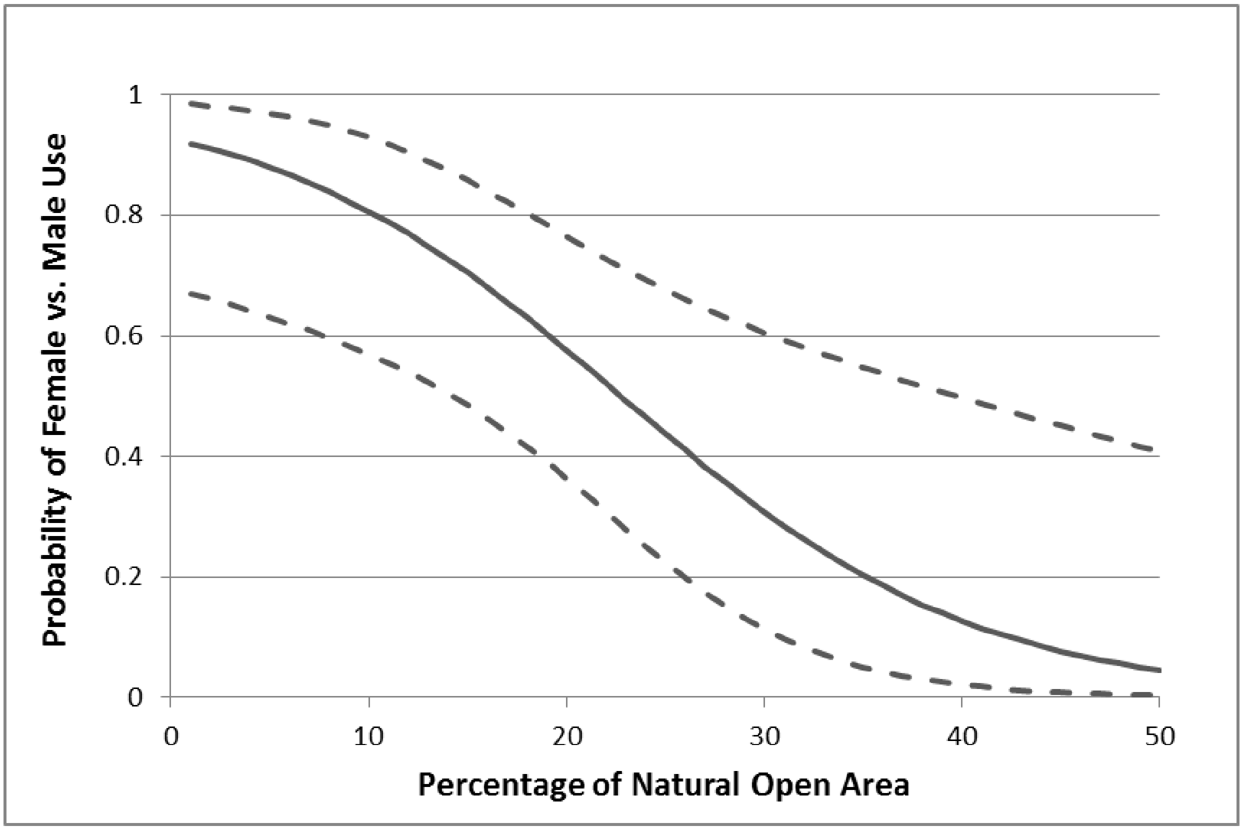

Figure 3.5. Predicted probability of female versus male use of a core area in relation to the percentage of natural open area in the landscape. Probabilities are based on predictions of the best model when incorporating the range of values found within pseudo core areas for the percent of natural open areas. Dashed lines indicate the 95% confidence interval.

DISCUSSION

We found support for two of the seven resource-selection hypotheses we evaluated. This support was limited to two models for females: 1) selection of home ranges and core areas with greater percentages of forest in the intermediate tree class than was found in pseudo home ranges and core areas, and 2) the avoidance of natural open areas within their core areas. Consequently, the data generally supported our predictions that home range selection would differ between the sexes and that females would exhibit greater selectivity for the forest and landscape characteristics we evaluated. We also predicted that fishers would select home ranges at low and mid-elevations; although females did establish home ranges primarily at mid-elevations, the mean elevation of female home ranges was higher than pseudo home ranges. Low- and mid-elevation forests generally have been considered important to fishers because these forests commonly include a greater abundance of the large forest structures that fishers require than forests at higher elevations (Buskirk and Powell 1994, Paragi *et al.* 1996, Raley *et al.* 2012, Weir *et al.* 2012, Aubry *et al.* 2013). Our study area included landscapes dominated by unmanaged forests in the intermediate and large tree classes at mid- and upper-elevations in the Olympic Recovery Area, as well as lower elevation forests that were managed largely as short-rotation commercial timberlands. This arrangement of forests along the elevational gradient in our study area resulted in an abundance of young forests at lower elevations and an abundance of old forests at mid- and upper-elevations. Although this arrangement of forests is not unusual in montane portions of the fisher's West Coast range (*i.e.*, private lands as lower elevations, protected federal forest lands at mid- and upper-elevations), it does explain the greater abundance of large forest structures found in mid- and upper-elevation forests within the study area and may help explain why these forests were selected by females.

Males selected home ranges and core areas with smaller trees and a greater percentage of forest in the small tree class than was generally available, which contradicted two of our predictions (Table 3.3). We did not expect males to select home ranges in managed forest landscapes, nor for their home ranges to have greater percentages of forest in the small tree class and more forest openings than was generally available in our study area. There was some support for all 17 models (all $\Delta AIC_c < 7$, for core area models; Table 3.6), indicating that males exhibited greater flexibility in their selection of home ranges than females. These findings are consistent with our predictions that males would be less selective than females and select different resources than females because of behavioral differences and their larger body size.

The inclusion of elevation in the highest ranking model of home range selection by male fishers initially seemed at odds with the fact that males established only three of 12 home ranges at high elevations, and the majority established home ranges at low elevations outside the Olympic Recovery Area. High elevation habitats, however, were much less extensive within the study area, compared to the large area at low elevations (Figure 3.2). The best home range model for males suggests that male fishers likely occupied a greater proportion of the high elevation habitat than low elevations. However, we suggest that the model support for selection of high elevation home ranges may be misleading. The odds ratio for the effect of elevation was not statistically significant (Table 3.6), mean elevations of male home ranges were essentially indistinguishable from those of the available pseudo home ranges (Table 3.4), females on average used higher home ranges than males (Table 3.7), and the majority of males established home ranges at low elevations, on managed forests outside the Olympic Recovery Area (Figure 3.2). Our results indicate that males exploited both low-elevation habitats in managed forest landscapes and high-elevation habitats in unmanaged landscapes, where many females also located their home ranges.

The reasons for observed differences in resource selection between the sexes are poorly understood. The differences we observed are consistent with the hypothesis that females had greater reproductive and security requirements, including protection from other predators. However, there are many other factors that could explain sex-specific selection patterns including unknown variation in predator and prey densities among available habitats.

Fishers have been shown to be selective for particular forest seral-stages (Jones and Garton 1994, Sauder and Rachlow 2014) and are generally associated with landscapes dominated by mid- and late-seral forests (Raley *et al.* 2012). Females on the Olympic Peninsula selected home ranges and core areas with greater percentages of forest in the intermediate tree class at higher elevations than were generally available, and most (79%) established home ranges in landscapes dominated by unmanaged forests in the intermediate and large tree classes. Although females avoided forests in the large tree class within their home ranges (Table 3.5), these forests made up a large proportion (~ 25%) of female home ranges and core areas (Table 3.4), and tended to be interspersed within a matrix of forests in the intermediate size class of trees. Much of the intermediate tree class forest in these unmanaged landscapes resulted from historical wildfires, including a large fire that occurred in ~1700 (Henderson 1989). Such forests generally contain greater vertical diversity (*i.e.*, multiple vegetation layers) and a greater abundance of large woody structures (large remnant logs and snags) than forest stands within landscapes managed primarily for timber production on short rotations (Franklin and Spies 1991, Spies and Franklin 1991). Under intensive management, the landscape generally comprises large even aged stands rather than the fine-grained mosaics typical of natural disturbance regimes on unmanaged landscapes (Franklin and Dyrness 1988, Henderson *et al.* 1989, Oliver and Larson 1996). Consequently, the difference in context between managed and unmanaged forests is likely to have influenced the use of intermediate tree-class forests by female fishers directly because they provide very different levels of security cover and different densities of predators and prey.

Although fishers prey on a wide variety of small and medium-sized mammalian prey, no previous ecological studies of fishers have been conducted where they co-occur with mountain beavers. Mountain beavers are found in all forest seral stages, but are most abundant in regenerating (seedling, sapling) and young stands, which support a greater abundance of the herbaceous plants, woody shrubs, and seedlings that make up its diet (Hacker and Coblentz 1993, Verts and Carroway 1998, Feldhamer *et al.* 2003, Arjo and Nolte 2006, Arjo *et al.* 2007). Mountain beavers are the dominant prey item in the diet of bobcats in western Washington (Brittell *et al.* 1979, Knick *et al.* 1984, Witmer and DeCalesta 1986), and are known to be among the prey species that fishers use on the Olympic Peninsula (Lewis *et al.* 2010, 2011). Bobcats are habitat generalists that occupy diverse environments throughout their range, and are commonly associated with open areas, brushy habitats and rocky out-crops (Anderson and Lovallo 2003, Riley *et al.* 2010, Wengert 2013). In forested environments, bobcats use all forest seral stages (from regenerating to old-growth stands; Witmer and DeCalesta 1986); however, they appear to be particularly abundant in managed forest landscapes (Yengoyan 1995, Wengert 2013), which may be related to the abundance of mountain beavers and other prey in these environments (Brittell *et al.* 1979, Knick *et al.* 1984).

Bobcats and fishers are likely to compete for prey resources within our study area because of the similarities in their diets (Brittell *et al.* 1979; Knick *et al.* 1984; Martin 1994; Zielinski *et al.* 1999; Anderson and Lovallo 2003; Weir *et al.* 2005; Lewis *et al.* 2010, 2011). The smaller size of female fishers (2-3 kg; Lewis *et al.* 2011) compared to males (4-6 kg), makes them more vulnerable to predation by bobcats (females: 4-16 kg; males: 6-18 kg; Anderson and Lovallo 2003; Chapter 2). In California, Wengert *et al.* (2013) found that predation by bobcats on fishers was limited to females, which they attributed to differences in size between males and females. Our data are consistent with the hypothesis that females (15 of 19) established home ranges in relatively continuous unmanaged forests in the intermediate and large tree classes to reduce predation risks and because such landscapes contain abundant large woody structures that they need for denning and resting. Our data are also consistent with the hypothesis that males selected home ranges in managed forest landscapes because their large size makes them less vulnerable to predation and more capable of competing effectively with other mesocarnivores for mountain beavers and other prey that are abundant in such landscapes. Because of their lower vulnerability and because they do not require den sites, males do not have the same requirements as females for the large down logs, snags, or cavity trees that commonly occur in unmanaged forest landscapes (Franklin and Spies 1991).

Whereas managed forest landscapes have greater densities of mountain beavers, the trade-off in prey availability that female fishers make for the greater amounts of security cover found unmanaged forests may be relatively small. Compared to low-elevation, managed forest landscapes, the continuous unmanaged forests at mid-elevations in the Olympic Recovery Area are likely to support greater abundances of sciurids (Carey 1995) and forest-floor small mammals (Carey and Johnson 1995), which are also important components of the fisher diet (Martin 1994, Zielinski *et al.* 1999, Weir *et al.* 2005). In addition, the juxtaposition of intermediate and large tree-classes in unmanaged forests may create structural diversity at the landscape scale that supports a greater abundance of small and medium-sized mammals than would be present in a landscape dominated by late-seral forests. For example, evidence from spotted owl diets on the Olympic Peninsula (Forsman *et al.* 2001) indicates that snowshoe hare (*Lepus americanus*) densities are greater in the eastern portion of the Olympic Peninsula, where most female fishers settled.

The avoidance of open areas by fishers has been observed throughout their range (Buskirk and Powell 1994, Raley *et al.* 2012). Weir and Corbould (2010) found that the amount of open area (-) was the best predictor of home range selection by fishers in central British Columbia, whereas in Idaho, Sauder and Rachlow (2014) found that the amount of open area (-) and the proximity of mature forest stands were most useful for explaining home range selection by fishers. On the Olympic Peninsula, females selected core areas with less natural open area (odds ratio 0.959, 95% CI = 0.921-0.998); however, the amount of natural open area did not appear to influence home range selection by males. The amount of natural open area was the sole predictor in the best model for distinguishing female and male core areas. Given the small number of males in their sample (8 F, 2 M), it is likely that the strong influence of open areas observed by Weir and Corbould (2010; odds ratio of 0.803, 95% CI = 0.663-0.973) was largely the result of female avoidance. Sauder and Rachlow (2014) also found an avoidance of open areas (odds ratio 0.875, 95% CI = 0.774-0.989) that was only slightly weaker when they investigated resource selection in a population with an even sex ratio (9 F, 9 M).

Fishers commonly occupy landscape mosaics that are dominated by mid or late-seral coniferous forests (Carroll *et al.* 1999, Davis *et al.* 2007); however, the degree of patchiness and the proximity and type of patches that comprise a mosaic are likely to influence its suitability for fishers (Sauder and Rachlow 2014). Patch density on the Olympic Peninsula did not appear to influence selection by females; however, it was the second-ranked model for male core areas (Table 6), which is consistent with males using managed forest landscapes with higher patch densities.

The fisher is often referred to as a secretive carnivore because it avoids humans and areas of human activity (Powell 1993); however, we did not detect avoidance of managed open areas (*e.g.*, agricultural fields, developed areas) or areas with greater densities of paved roads in relation to available areas, by males or females. Models that included these landscape characteristics were among the lowest ranking models in each of the six model sets. In our study area, human influences on fishers took many forms: one male was killed by a domestic dog in a rural developed area, one was treed by domestic dogs in a suburban environment, one female established a home range that included a suburban area of the city of Port Angeles, and one female was captured in a box trap and released unharmed by a landowner that had lost domestic pets to a predator. Although fisher-human interactions were uncommon, the most commonly documented interaction between fishers and humans involved vehicle collisions. Vehicle collisions were an important mortality source (20%) for the 90 fishers we released on the Olympic Peninsula (Chapter 2), which is surprising given the relatively low density of paved roads. However, among the 31 fishers with established home ranges that were included in this study, only one was killed by a vehicle. This finding may indicate that most fishers killed by a vehicle did not have enough locations for us to estimate a home range and include in our sample, or that fishers with established home ranges are less likely to be killed by a vehicle. However, the single mortality within our sample may indicate that most fishers used areas away from the only highway (U.S. Highway 101, where five of the seven collisions occurred among the 90 fishers released), which could decrease their odds of being killed by a vehicle even if their home ranges included other paved roads.

The limited overlap of male and female fisher home ranges and the relatively small portion of the study area that was occupied by fishers, indicates that the study area was below its carrying capacity. Consequently, the habitat use and selection patterns we found may not mirror those of a population at or near carrying capacity. In areas where population densities are below carrying capacity, all individuals could occupy high quality landscapes; thus, our findings could be biased to reflect selection for the highest quality habitats and may not be indicative of the full range of landscape conditions that are capable of supporting fishers.

Conspecific attraction has been reported for territorial species, whereby dispersing or newly released individuals tend to establish home ranges (or territories) in close proximity to those of resident individuals (Stamps 1988, Smith and Peacock 1989). Although we found little overlap among the home ranges of male and female fishers or among males, there was a significant clustering of female home ranges in the northern and northeastern portions of our study area (Figure 3.2). These portions of the study area may have been the highest quality habitats within the recovery area, however the concentration of 15 of the 19 female home ranges in these two relatively small areas, the large size of the study area, and the wide distribution of potentially suitable habitat (Lewis and Hayes 2004), suggest that home range establishment by females within the study area may have been influenced by the presence of other females.

This study is the first to investigate resource selection by a founding population of translocated fishers. Based on the success of other fisher translocations (Lewis *et al.* 2012) and the abundance of suitable habitat (Lewis and Hayes 2004), we expected released fishers to establish a self-sustaining population in the Olympic Recovery Area. Our study documented the resource selection by fishers that had originated in managed, sub-boreal conifer-forests, and were translocated to an area dominated by unmanaged, temperate conifer-forests. These fishers were adapted to resource-selection cues available in the sub-boreal forests of central British Columbia, and many of these cues may be missing in the temperate conifer forests of the Olympic Peninsula or may not represent resources that enable released fishers to effectively exploit available habitats (Stamps and Swaisgood 2007). Consequently, the resource selection behaviors we observed may be less effective for exploiting available habitats on the Olympic Peninsula than the behaviors native Washington fishers exhibited prior to their extirpation. However, we expect that the descendants of founder individuals will have modified their resource selection behaviors to respond more efficiently to resource cues within the study area, which could result in different resource-selection patterns.

RESEARCH AND MANAGEMENT IMPLICATIONS

When conducting the feasibility assessment for the reintroduction of fishers to the Olympic Peninsula and Cascade Range, Lewis and Hayes (2004) identified optimal habitat as landscapes dominated by late-seral conifer forests at low and mid-elevations, because such forests were thought to provide the highest quality habitat for both male and female fishers. In this study, we found that females selected home ranges in landscapes dominated by unmanaged forests in the intermediate and large tree classes at mid-elevations, which are similar to the forests and landscapes identified by Lewis and Hayes (2004) as being important for fishers, especially females. In contrast, males in our study area selected forests in the small-tree class and areas with lower QMDA values, and most established home ranges in low-elevation managed forests. Consequently, assessing the suitability of landscapes for supporting viable fisher populations should include habitat models that identify unmanaged forests in the intermediate and large tree classes at low and mid-elevations, which appear to be particularly important to females. Our findings also indicate that managed forest landscapes can contribute to population recovery by providing habitats that are suitable for male fishers.

Translocations provide opportunities for managers to investigate resource selection of a large number of individuals of both sexes. Sufficient samples of both sexes are valuable because of the important differences in selection that may exist between the sexes. Studies that combine data of males and females may miss or dilute important differences in selection between the sexes. These differences will be important for managing landscapes for female fishers and successful reproduction.

Our research suggests that many important interactions exist among fishers, their predators (*i.e.*, bobcats), their prey (*i.e.*, mountain beavers, snowshoe hares, and squirrels), and forest landscapes that influence how fishers select home ranges. Reliable data on the potential prey and predators are needed to elucidate potential interactions, especially information on how the densities of these species vary in response to the composition and configuration of forest landscapes and how predation risks and foraging tradeoffs influence resource selection strategies of male and female fishers. Future studies would also help to clarify how resource selection strategies of translocated fishers released into a vacant and new environment may change over time as resource selection of this translocated population adapts to local variations in food resources and predation risks, and as the growing population reaches carrying capacity.

The LEMMA data available for the Olympic Peninsula included large-snag densities and volumes of large down wood; however, because these data had poor validation scores (<0.50; http://www.fsl.orst.edu/lemma/), we did not use them in our analyses. It is challenging for land management agencies to obtain high-quality data on these structures at regional scales. However, such data would be particularly useful for determining the suitability of managed landscapes that contain low amounts of late-seral forest, but have sufficient large woody structures for denning and resting. Because the LEMMA project was not designed to support fine-grained evaluations of specific habitat features, higher resolution data would enable us to conduct analyses that included measures of both landscape configuration at finer scales and the role of specific forest structures, in addition to measures of landscape composition (Sauder and Rachlow 2014).

Most reintroductions involve the release of individuals at locations where they are surrounded by broad expanses of high-quality habitat. This approach is used to prompt founders to settle within these high-quality habitats and to increase the likelihood of survival and reproductive success. Because females were more selective when establishing home ranges and core areas, and are the more critical sex for achieving adequate recruitment in a founder population, managers may achieve greater success by releasing fishers within large landscapes dominated by unmanaged forests in the intermediate or large tree classes. Where unmanaged forest landscapes are limited, managers could consider releasing fishers in managed landscapes with large expanses of forests in the intermediate or large tree classes, especially those that are adjacent to protected or unmanaged forest landscapes (*e.g.*, WDNR's Hoh-Clearwater block; WDNR 1997). Males are more likely than females to leave these targeted landscapes, but their wide movements during the breeding season suggests there may be minimal effects of spatial segregation of the sexes on reproductive success (Chapter 1).

The affinity of female fishers for large, unmanaged forested landscapes may indicate that fisher recovery in western Washington will be limited to the Olympic Peninsula and Cascade Range, where such landscapes exist. Female selection of home ranges dominated by unmanaged forests of mid-sized and large overstory trees and few natural openings can provide planning targets for landscape management plans. The area required to support an individual fisher is very large ($\bar{x}$ = 64 km^2 for females; Chapter 1), and the area needed to support a self-sustaining population is much larger and likely to encompass multiple land ownerships (*e.g.*, National Forests, National Parks, State Forests). Planning across the boundaries of ownerships may be complicated but is essential for species that require management at regional scales. The Northwest Forest Plan (USDA Forest Service and USDI Bureau of Land Management 1994) is a plan of this magnitude and given its provisions to maintain and develop older forests across large landscapes, we expect it to support the reestablishment of self-sustaining fisher populations in Washington, once they have been reintroduced.

LITERATURE CITED

Anderson, E. and M. Lovallo. 2003. Bobcat and lynx. Pages 758–786 in G. A. Feldhamer, B. C. Thompson, and J. A. Chapman (Eds.), Wild Mammals of North America: Biology, Management, and Economics. Johns Hopkins Press, Baltimore, Maryland.

Arjo, W.M., and D.L. Nolte. 2006. Boomer or bust: managing a Pacific Northwest pest species. Proceedings of the 22nd Vertebrate Pest Conference 22:181-186.

Arjo, W.M., R.R. Huenefeld and D.L. Nolte. 2007. Mountain beaver home ranges, habitat use, and population dynamics in Washington. Canadian Journal of Zoology 85:328-337.

Arthur, S.M., W.B. Krohn and J.R. Gilbert. 1989. Home range characteristics of adult fishers. Journal of Wildlife Management 53:680-688.

Aubry, K.B and J.C. Lewis. 2003. Extirpation and reintroduction of fishers (*Martes pennanti*) in Oregon: implications for their conservation in the Pacific states. Biological Conservation 114:79-90.

Aubry, K.B., C.M. Raley, S.W. Buskirk, W.J. Zielinski, M.K. Schwartz, R.T. Golightly, K.L. Purcell, R.D. Weir, J.S. Yaeger. 2013. Meta-analyses of habitat selection by fishers at resting sites in the Pacific coastal region. Journal of Wildlife Management 77(5):965–974.

Aubry, K.B., S.M. Wisely, C.M. Raley and S.W. Buskirk. 2004. Zoogeography, spacing patterns, and dispersal in fishers: insights gained from combining field and genetic data. Pages 187-206 in D.J. Harrison, A.K. Fuller and G. Proulx, editors. Marten and fishers (Martes) in human-altered environments: an international perspective. Springer, New York, New York.

Bailey, V.O. 1936. The mammals and life zones of Oregon. North American Fauna 55:1-416.

Berg, W.E. 1982. Reintroduction of fisher, pine marten and river otter. Pages 159-175 in G. C. Sanderson, editor. Midwest furbearer management. North Central Section of The Wildlife Society, Bloomington, Illinois.

Breitenmoser, U., C. Breitenmoser-Wursten, L.W. Carbyn and S.M. Funk. 2001. Assessment of carnivore reintroductions. Pages 240-281 in J.L. Gittleman, S.M. Funk, D. W. Macdonald and R.K. Wayne, editors. Carnivore Conservation, Cambridge University Press, New York, New York, USA.

Brittell, J.D., S.J. Sweeney and S.T. Knick. 1979. Washington bobcats: diets, population dynamics and movement. Pages 107-110 in P. C. Escherich and L. Blum, editors. Proceedings of the 1979 bobcat research conference. National Wildlife Federation, Wildlife Technical Series 6.

Buck, S.G., C. Mullis, A.S. Mossman, I. Show and C. Coolahan. 1994. Habitat use by fishers in adjoining heavily and lightly harvested forest. Pages 368-376 in S.W. Buskirk, A. Harestad, M. Raphael and R.A. Powell, editors. Martens, Sables, and Fishers: Biology and Conservation. Cornell University Press. Ithaca, New York, USA.

Burnham, K.P. and D.R. Anderson. 2002. Model selection and inference: a practical information-theoretic approach, 2nd edition. Springer-Verlag, New York.

Buskirk, S.W. and R.A. Powell. 1994. Habitat ecology of fishers and American martens. Pages 283-296 in S.W. Buskirk, A.S. Harestad, M.G. Raphael and R.A. Powell, editors. Martens, Sables, and Fishers: Biology and Conservation. Cornell University Press, Ithaca, New York.

Carroll, C., W.J. Zielinski and R.F. Noss. 1999. Using presence-absence data to build and test spatial habitat models for the fisher in the Klamath Region, U.S.A. Conservation Biology 13:1344-1359.

Carey, A.B. 1995. Sciurids in Pacific Northwest managed and old-growth forests. Ecological Applications 5(3):648-661.

Carey, A.B. and M.L. Johnson. 1995. Small mammals in managed, naturally young, and old-growth forests. Ecological Applications 5:336-352.

Dalquest, W.W. 1948. Mammals of Washington. University of Kansas, Lawrence, Kansas.

Davis, F.W., C. Seo and W.J. Zielinski. 2007. Regional variation in home-range scale habitat models for fisher (*Martes pennanti*) in California. Ecological Applications 17:2195–2213.

Desrochers, A., E.J.B. McIntire, S.G. Cumming, J. Nowak and S. Sharma. 2010. False negatives—A false problem in studies of habitat selection? Ideas in Ecology and Evolution 3: 20-25.

Feldhamer, G.A., J.A. Rochelle and C.D. Rushton. 2003. Mountain beaver. Pages 179-187 in G.A. Feldhamer, B.C. Thompson and J.A. Chapman, editors. Wild Mammals of North America: Biology, Management and Conservation, 2nd edition. Johns Hopkins University Press, Baltimore, MD.

Forsman, E.D., I.A. Otto, S.G. Sovern, M. Taylor, D.W. Hays, H. Allen, S.L. Roberts and D.E. Seaman. 2001. Spatial and temporal variation in diets of spotted owls in Washington. Journal of Raptor Research 35:141-150.

Franklin, J.F. and C.T. Dyrness. 1988. Natural vegetation of Oregon and Washington. Oregon State University Press, Corvallis, Oregon.

Franklin, J.F. and T.A. Spies. 1991. Composition, function, and structure of old-growth Douglas-fir forests. Pages 71–90 in L.F. Ruggiero, K.B. Aubry, A.B. Carey and M. Huff, technical coordinators. Wildlife and vegetation of unmanaged Douglas-fir forests. General Technical Report PNW-GTR-85. U.S. Department of Agriculture Forest Service, Pacific Northwest Research Station, Portland, Oregon.

Grinnell, J., J.S. Dixon, J.M. Linsdale. 1937. Fur-Bearing Mammals of California: Their Natural History, Systematic Status and Relations to Man. Volume 1. University of California, Berkeley.

Hacker, A.L. and B.E. Coblentz. 1993. Habitat selection by mountain beavers recolonizing Oregon Coast Range clearcuts. Journal of Wildlife Management 57: 847-853.

Hansen, A.J., T.A. Spies, F.J. Swanson and J.L. Ohmann. 1991. Conserving biodiversity in managed forests. BioScience 41:382-392.

Hayes. G.E. and J.C. Lewis. 2006. Washington State recovery plan for the fisher. Washington Department of Fish and Wildlife, Olympia, Washington. 69 pp. (at: http://wdfw.wa.gov/wlm/diversty/soc/fisher/).

Heinemeyer, K.S. 1993. Temporal dynamics in the movements, habitat use, activity, and spacing of reintroduced fishers in northwestern Montana. MS Thesis, University of Montana, Missoula, Montana.

Henderson, J.A., D.H. Peter, R.D. Lesher and D.C. Shaw. 1989. Forested plant associations of the Olympic National Forest. U.S. Forest Service Ecological Technical Paper 001-88, Portland, Oregon. 502 pp.

Horne, J.S. and EO. Garton. 2009. Animal Space Use 1.3 <http://www.cnr.uidaho.edu/population_ecology/animal_space_use>.

Hosmer, D.W. and S. Lemeshow. 2000. Applied Logistic Regression, 2nd edition. John Wiley and Sons, Inc., New York, NY. 375 pp.

Houston. D.B. and E.G. Schreiner. 1994. The Olympic Mountains and Environs . Pages 14-26 (Chapter 2) in D.B. Houston, E.G. Schreiner, and B. B. Morehead, editors. Mountain goats in Olympic National Park: biology and management of an introduced species. Scientific Monograph NPS/NROLYM/NRSM-94/25. USDI National Park Service. Port Angeles, Washington.

Houston, D.B., E.G. Schreiner and B.B. Morehead, editors. 1994. Mountain goats in Olympic National Park: biology and management of an introduced species. Scientific Monograph NPS/NROLYM/NRSM-94/25. USDI National Park Service. Port Angeles, Washington.

IUCN. 1987. IUCN position statement on translocation of living organisms: introductions, reintroductions, and re-stocking. Available at: www.iucn.org/themes/ssc/pubs/policy/transe.

IUCN. 1995. IUCN/SSC Guidelines for re-introductions. Forty-first meeting of the IUCN Council, Gland, Switzerland. May 1995, 6 pp. Available at: www.iucn.org/themes/ssc/pubs/policy/reinte.

IUCN. 2012. IUCN guidelines for reintroductions and other conservation translocations. Available at: http://www.issg.org/pdf/publications/Translocation-Guidelines-2012.pdf

Johnson, C.J., S.E. Nielsen, E.H. Merrill, T.L. McDonald and M.S. Boyce. 2006. Resource selection functions based on use-availability data: theoretical motivation and evaluation methods. Journal of Wildlife Management 70:347–357.

Jones, J.L. and E.O. Garton. 1994. Selection of successional stages by fishers in North- Central Idaho. Pages 377-387 in S.W. Buskirk, A.S Harestad, M.G. Raphael and R.A. Powell, editors. Martens, Sables, and Fishers: Biology and Conservation. Cornell University Press, Ithaca, New York.

Katnik, D.D., R.B. Wielgus. 2005. Landscape proportions versus Monte Carlo simulated home ranges for estimating habitat availability. Journal of Wildlife Management 69(1):20-32.

Keating, K.A. and S. Cherry. 2004. Use and interpretation of logistic regression in habitat-selection studies. Journal of Wildlife Management 68(4):774-789.

Knick, S.T., S.J. Sweeney, J.R. Alldredge and J.D. Brittell. 1984. Autumn and winter food habits of bobcats in Washington State. Great Basin Naturalist 44:70-74.

Krohn, W.B., K.D. Elowe and R.B. Boone. 1995. Relations among fishers, snow, and martens: development and evaluation of two hypotheses. Forestry Chronicle 71:97-105.

Krohn, W.B., W.J. Zielinski and R.B. Boone. 1997. Relations among fishers, snow, and martens in California: results from small-scale spatial comparisons. Pages 211-232, in G.H. Proulx, H.N. Bryant and P.M. Woodard, editors. *Martes*: Taxonomy, Ecology, Techniques and Management. Provincial Museum of Alberta, Edmonton, Canada. 474 pp.

Krohn, W.B., C. Hoving, D. Harrison, D. Phillips and H. Frost. 2004 *Martes* foot-loading and snowfall patterns in eastern North America: implications to broad scale distributions and interactions of mesocarnivores. Pages 115-131 in D.J. Harrison, A.K. Fuller and G. Proulx, editors. Martens and Fisher (*Martes*) in Human-Altered Environments: An International Perspective. Kluwer Academic Publishers, Norwell, Massachusetts, USA.

Leonard, R.D. 1986. Aspects of reproduction of the fisher, *Martes pennanti*, in Manitoba. Canadian Field-Naturalist 100(1):32-44.

Lewis, J.C. 2013. Implementation plan for reintroducing fishers to the Cascade Mountain Range in Washington. Washington Department of Fish and Wildlife, Olympia. 29 pp. (at: http://wdfw.wa.gov/publications/01556/)

Lewis, J.C. and P.J. Happe. 2008. Olympic fisher reintroduction project: 2008 progress report. Washington Department of Fish and Wildlife, Olympia. (at: http://wdfw.wa.gov/wlm/diversty/soc/fisher/)

Lewis, J.C., P.J. Happe, K.J. Jenkins, and D.J. Manson. 2010. Olympic fisher reintroduction project: 2009 progress report. Washington Department of Fish and Wildlife, Olympia. (at: http://wdfw.wa.gov/wlm/diversty/soc/fisher/)

Lewis, J.C., P.J. Happe, K.J. Jenkins and D.J. Manson. 2011. Olympic fisher Reintroduction Project: 2010 progress report. Washington Department of Fish and Wildlife, Olympia. (at: http://wdfw.wa.gov/wlm/diversty/soc/fisher/)

Lewis, J.C. and G.E. Hayes. 2004. Feasibility assessment for reintroducing fishers to Washington. Washington Department of Fish and Wildlife, Olympia. 70 pp. (at: http://wdfw.wa.gov/wlm/diversty/soc/fisher/)

Lewis, J.C., R.A. Powell and W.J. Zielinski. 2012. Carnivore translocations and conservation: insights from population models and field data for fishers (*Martes pennanti*). PLoS ONE 7(3): e32726.

Lewis, J.C. and W.J. Zielinski. 1996. Historical harvest and incidental capture of fishers in California. Northwest Science 70:291-297.

Lofroth, E.C., C.M. Raley, J.M. Higley, R.L. Truex, J.S. Yaeger, J.C. Lewis, P.J. Happe, L.L. Finley, R.H. Naney, L.J. Hale, A.L. Krause, S.A. Livingston, A.M. Myers and R.N. Brown. 2010. Volume I - Conservation Assessment for Fisher (*Martes pennanti*) in South-central British Columbia, Western Washington, Western Oregon, and California. USDI Bureau of Land Management, Portland, OR.

Martin, S.K. 1994. Feeding ecology of American martens and fishers. Pages 297–315 in S.W. Buskirk, A.S Harestad, M.G. Raphael and R.A. Powell, editors. Martens, Sables, and Fishers: Biology and Conservation. Cornell University Press, Ithaca, New York.

Meidinger, D. and J. Pojar. 1991. Ecosystems of British Columbia. British Columbia Ministry of Forests, Victoria, Canada. http://www.for.gov.bc.ca/hfd/pubs/Docs/Srs/Srs06.htm.

Miller, B., K. Ralls, R.P. Reading, J.M.Scott and J. Estes. 1999. Biological and technical considerations of carnivore translocation: a review. Animal Conservation 2:59-68.

Naney, R.H., L.L Finley, E.C. Lofroth, P.J. Happe, A.L. Krause, C.M. Raley, R.L. Truex, L.J. Hale, J.M. Higley, A.D. Kosic, J.C. Lewis, S.A. Livingston, D.C. Macfarlane, A.M. Myers and J.S. Yaeger. 2012. Conservation of fishers (Martes pennanti) in south-central British Columbia, western Washington, western Oregon, and California–Volume III: Threat Assessment. USDI Bureau of Land Management, Denver, Colorado, USA.

National Park Service. 1998. Strategic Plan for Olympic National Park 1998-2002. Olympic National Park, Port Angeles, Washington.

National Park Service. 2005. Fire Management Plan and Environmental Assessment. Olympic National Park, Port Angeles, Washington.

National Park Service. 2008. Final General Management Plan /Environmental Impact Statement: Olympic National Park, Port Angeles, Washington. (Available at: http://parkplanning.nps.gov/projectHome.cfm?projectID=10233)

Ohmann, J.L. and M.J. Gregory. 2002. Predictive mapping of forest composition and structure with direct gradient analysis and nearest-neighbor imputation in coastal Oregon, U.S.A. Canadian Journal of Forest Research 32:725-741.

Oliver, C.D. and B.C. Larson. 1996. Forest Stand Dynamics. John Wiley & Sons, Inc. New York, NY.

Paragi, T.F., S.M. Arthur and W.B. Krohn. 1996. Importance of tree cavities as natal dens for fishers. Northern Journal of Applied Foresty 13:79-83.

Peel, M.C., B.L. Finlayson and T.A. McMahon. 2007. Updated world map of the Köppen-Geiger climate classification. Hydrology and Earth Systems Science 11:1633-1644.

Powell, R.A. 1993. The fisher: Life history, ecology, and behavior, 2nd edition. University of Minnesota Press, Minneapolis, Minnesota.

Powell, R.A. and W.J. Zielinski. 1994. Fisher. Pages 28-73 in L.F. Ruggiero, K.B. Aubry, S.W. Buskirk, L.J. Lyon and W.J. Zielinski, editors. The scientific basis for conserving forest carnivores: American marten, fisher, lynx, and wolverine in the western United States. U.S. Forest Service, General Technical Report RM-254.

Proulx, G.A., J. Kolenosky, M. Badry, M.K. Drescher, K. Seidel and P.J. Cole. 1994. Post-release movements of translocated fishers. Pages 197-203 in S.W. Buskirk, A. Harestad, M. Raphael and R.A. Powell, editors. Martens, Sables, and Fishers: Biology and Conservation. Cornell University Press, Ithaca, New York.

Purcell, K.L., A.K. Mazzoni, S.R. Mori and B. Boroski. 2009. Resting structures and resting habitat of fishers in the southern Sierra Nevada, California. Forest Ecology and Management 258: 2696–2706.

Raley, C.M., E.C. Lofroth, R.L. Truex, J.S. Yaeger and J.M. Higley. 2012. Habitat ecology of fishers in western North America: a new synthesis. Pages 231-254 in K.B. Aubry, W.J. Zielinski, M.G. Raphael, G. Proulx and S.W. Buskirk, editors. Biology and Conservation of Martens, Sables, and Fishers: A New Synthesis. Cornell University Press, Ithaca, New York.

Reading, R.P. and T.W. Clark. 1996. Carnivore reintroductions: an interdisciplinary examination. Pages 296-336 in J.L. Gittleman, editor. Carnivore Behavior, Ecology, and Evolution. Volume 2. Cornell University Press, Ithaca, New York, USA.

Riley, S.P.D., E.E. Boydston, K.R. Crooks and L.M. Lyren. 2010. Bobcats (*Lynx rufus*). Pages 121-140 in S.D. Gehrt, S.P.D. Riley and B.L. Cypher, editors. Urban Carnivores: Ecology, Conflict, and Conservation. Johns Hopkins University Press, Baltimore, MD. 285 pp.

Rodgers, A.R., A.P. Carr, H.L. Beyer, L. Smith and J.G. Kie. 2007. HRT: Home Range Tools for ArcGIS. Version 1.1. Ontario Ministry of Natural Resources, Centre for Northern Forest Ecosystem Research, Thunder Bay, Ontario, Canada.

Sauder, J. and J.L. Rachlow. 2014. Both forest composition and configuration influence landscape-scale habitat selection by fishers (*Pekania pennanti*) in the Rocky Mountains of Idaho. Forest Ecology and Management 314:75-84.

Seton, E.T. 1926. Lives of Game Animals. Vol. 2. Doubleday, Doran & Co., New York.

Sikes, R.S., W.L. Gannon, and the Animal Care and Use Committee of the American Society of Mammalogists. 2011. Guidelines of the American Society of Mammalogists for the use of wild mammals in research. Journal of Mammalogy 92(1):235-253.

Spies, T.A. and J.F. Franklin. 1991. The structure of natural young, mature,and old-growth Douglas-fir forests in Oregon and Washington. Pages 91–109 in L.F. Ruggiero, K.B. Aubry, A.B. Carey, and M. Huff, technical coordinators. Wildlife and vegetation of unmanaged Douglas-fir forests. General Technical Report PNW-GTR-85. U.S. Department of Agriculture Forest Service, Pacific Northwest Research Station, Portland, Oregon.

Smith, A. and M.M. Peacock. 1990. Conspecific attraction and the determination of metapopulation colonization rates. Conservation Biology 4:320–323.

Stamps, J.A. 1988. Conspecific attraction and aggregation in territorial species. American Naturalist 131:329–347.

Stamps, J.A. and R.R. Swaisgood. 2007. Someplace like home: experience, selection, and conservation biology. Applied Animal Behaviour Science 102:392-409.

Thomas, D.L. and E.J. Taylor. 2006. Study designs and tests for comparing resource use and availability II. Journal of Wildlife Management 70(2):324-336.

Thompson, C.M., W.J. Zielinski and K.L. Purcell. 2011. Evaluating management risks using landscape trajectory analysis: a case study of California fisher. Journal of Wildlife Management 75(5):1164-1176.

USDA Forest Service and USDI Bureau of Land Management. 1994. Record of decision for amendments to Forest Service and Bureau of Land Management planning documents within the range of the northern spotted owl. USDA Forest Service and USDI Bureau of Land Management, Portland, Oregon.

USFWS (US Fish & Wildlife Service). 2004. Endangered and threatened wildlife and plants: 12-month finding for a petition to list the west coast distinct population segment of the fisher (*Martes pennanti*). Federal Register 69(68):18770-18792.

USFWS (US Fish & Wildlife Service). 2013. Endangered and Threatened Wildlife and Plants; Status Review of the West Coast Distinct Population Segment of the Fisher as Endangered or Threatened. Federal Register 78:16828-16830.

Verts, B.J. and L.N. Carroway. 1997. Land Mammals of Oregon. University of California Press, Berkeley, CA. 668 pp.

Vittinghoff, E. and C.E. McCulloch. 2007. Relaxing the rule of ten events per variable in logistic and Cox regression. American Journal of Epidemiology 165(6):710–718.

Voigt, D.R. and W.E. Berg. 1987. Coyote. Pages 345-357 in M. Novak, J.A. Baker, M.E. Obbard and B. Malloch, editors. Wild Furbearer Management and Conservation in North America. Ministry of Natural Resources, Ontario, Canada.

Washington Department of Natural Resources. 1997. Habitat Conservation Plan. Washington Department of Natural Resources, Olympia.

Weir, R.D. 1995. Diet, spatial organization, and habitat relationships of fishers in south-central British Columbia. MS Thesis, Simon Fraser University, Burnaby, British Columbia, Canada.

Weir, R.D. and F.B. Corbould. 2007. Factors affecting diurnal activity of fishers in north-central British Columbia. Journal of Mammalogy, 88(6):1508–1514.

Weir, R.D. and F.B. Corbould. 2010. Factors affecting landscape occupancy by fishers in north-central British Columbia. Journal of Wildlife Management 74(3):405–410.

Weir, R.D., F.B. Corbould and A.S. Harestad. 2004. Effect of ambient temperature on the selection of rest structures by fishers. Pages 196-210 in D.J. Harrison, A.K. Fuller and G. Proulx, editors. Martens and Fisher (*Martes*) in Human-Altered Environments: An International Perspective. Kluwer Academic Publishers, Norwell, Massachusetts, USA.

Weir, R.D. and A.S. Harestad. 1997. Landscape-level selectivity by fishers in south-central British Columbia. Pages 252-264 in G. Proulx, H. N. Bryant and P. M. Woodward, editors. Martes: Taxonomy, ecology, techniques, and management. Provincial Museum of Alberta. Edmonton, Alberta, Canada.

Weir, R.D., A.S. Harestad and F.B. Corbould. 2009. Home ranges and spatial organization of fishers in central British Columbia. Canadian Field-Naturalist 123(1):126-132.

Weir, R.D., A.S. Harestad, and R.C. Wright. 2005. Winter diet of fishers in British Columbia. Northwestern Naturalist 86:12-19.

Weir, R.D., M. Phinney and E.C. Lofroth. 2012. Big, sick, and rotting: Why tree size, damage, and decay are important to fisher reproductive habitat. Forest Ecology and Management 265: 230-240.

Wengert, G.M. 2013. Ecology of Intraguild Predation on Fishers (Martes pennanti) in California. PhD dissertation. University of California, Davis. 119 pp.

Wengert, G.M., M.G. Gabriel, J.E. Foley, T. Kun and B.N. Sacks. 2013. Field and molecular techniques for identifying intraguild predators of fishers and other North American small carnivores. Wildlife Society Bulletin 37:659-663.

Witmer, G.W. and D.S. DeCalesta. 1986. Resource use by unexploited sympatric bobcats and coyotes in Oregon. Canadian Journal of Zoology 64:2333-2338.

Worton, B.J. 1989. Optimal smoothing parameters for multivariate fixed and adaptive kernel methods. Journal of Statistical Computation and Simulation 32:45-57.

Yalden, D.W. 1993. The problems of reintroducing carnivores. Symposium of the Zoological Society of London 65:289-306.

Yengoyan, L.P. 1995. Habitat utilization and home range size of the bobcat in managed forests of western Washington. MS thesis. University of Washington, Seattle. 86 pp.

Zielinski, W.J., R.L. Truex, G.A. Schmidt, F.V. Schlexer, K.N. Schmidt and R.H. Barrett. 2004. Home range characteristics of fishers in California. Journal of Mammalogy 85(4):649-657.

Zielinski, W.J., N.P. Duncan, E.C. Farmer, R.L.Truex, A.P. Clevenger and R.H. Barrett. 1999. Diet of fishers (*Martes pennanti*) at the southernmost extent of their range. Journal of Mammalogy 80:961–971.