Fisher Recolonization and Recovery in Oregon:

Implementation Plan for the Reintroduction of Fishers



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Executive Summary

Fishers (*Pekania pennanti*) are a forest-dependent, medium-sized carnivore classified as a species of conservation concern in much of their distribution in western North America. Fishers were once widely distributed throughout the forests of western Oregon. Today, fishers occupy approximately 15% of their former distribution in Oregon between the west slope of the Cascade Mountains and the Pacific Ocean, with contemporary locations only occurring in the southern-most portion of the state. There is a need to reduce the extinction risk and restore resiliency, redundancy, and representation of fishers in Oregon. Under a no-action alternative, partners involved in fisher conservation and research Oregon concluded there is a moderately-high risk of decline or extirpation of extant fisher populations in Oregon. We intend to reduce these risks by implementing a conservation action of reintroducing fishers to an unoccupied portion of their historical distribution to enhance the probability of persistence for fisher populations in Oregon. By implementing this action, our goal is to reduce the potential for decline and/or population loss to a less than 20% risk of extinction over the next 50 years, and contribute towards broader connectivity and recovery of fisher populations in Oregon and the Pacific Northwest.

Herein, we provide a hierarchical decision-making framework (*sensu* Armstrong and Seddon 2008, Pérez et al. 2012) to evaluate reintroduction as a conservation action for fishers in Oregon. The decision-making framework poses a series of questions centered on necessity, risk evaluation, and logistical capacity intended to guide adaptive decision-making throughout a potential reintroduction. We address each decision level and associated questions in turn, providing a blueprint for a reintroduction of fishers to a portion of the southern Oregon Cascades.

Fisher reintroduction and recovery provides an opportunity to engage a diverse group of partners and the public in wildlife conservation and management. A fisher reintroduction in Oregon will require a sustained commitment by partners to achieve objectives and monitor short- and long-term trends of source and translocated populations. This implementation plan, in conjunction with the companion reintroduction feasibility assessment by Matthews et al. (2021), serve as a blueprint to reintroduce fishers to a portion of their former distribution in Oregon, offer opportunities to fill existing information gaps to inform recovery and management decisions, achieve conservation objectives of partners, and contribute to the long-term recovery of the species.

Introduction

The recovery and conservation of mammalian carnivores are important for the restoration of ecological processes, ecological function, and resilience of global ecosystems (Estes et al. 2011, Ripple et al. 2014). Recovery for some carnivore species require active and adaptive management, including the reintroduction of species to parts of their historical distributions where they have been extirpated (Armstrong and Seddon 2008). Reintroductions of species of conservation concern to formerly occupied portions of their distributions provide a practical touchstone for a diverse group of partners with measurable and tangible steps toward species recovery (Seddon et al. 2007).

The fisher is a monospecific member of the genus *Pekania* in the weasel family, Mustelidae (Koepfli et al. 2008, Sato et al. 2012, Bradley et al. 2014). Fishers are native to the boreal and mixed northern hardwood–conifer forests of North America (Hagmeier 1956, Hall 1981, Powell 1981, Douglas and Strickland 1987, Gibilisco 1994). Historically, fishers occurred throughout large portions of the boreal forests in southern Canada and temperate, mixed-

deciduous-evergreen and conifer-dominated forests throughout North America (Lofroth et al. 2010, U.S. Fish and Wildlife Service 2016). The distribution of fishers contracted significantly following European colonization in North America (Powell 1993, Lewis et al. 2012) and at the end of the Little Ice Age around 1800 as the climate warmed (Krohn 2012). Unregulated trapping for fur, predator- and pest-control efforts, and habitat loss and fragmentation contributed to population declines and contractions in their distribution in the late 1800s and early 1900s (Lofroth et al. 2010, U.S. Fish and Wildlife Service 2016).

Fishers have since recovered in much of the eastern and central portions of their historical distribution in Canada and the United States (Powell 1993, Buskirk and Powell 1994, Gibilisco 1994, Lewis and Stinson 1998, Proulx et al. 2004, Lewis et al. 2012). However, recovery of fishers in the Pacific states has been limited despite similar conservation efforts (e.g., trapping bans, translocation/reintroduction, forest restoration; Lofroth et al. 2010, U.S. Fish and Wildlife Service 2016) and an abundance of habitat predicted to be suitable for fishers in the Pacific states (Aubry et al. 2017, Barry et al. 2021). Thus, fishers in the Pacific states are a species of conservation concern for federal and state agencies, Tribes and First Nations, private-land managers, and the public (U.S. Fish and Wildlife Service 2016, 2020*a*, 2025*a*).

The U.S. Forest Service and Bureau of Land Management classified fishers as a sensitive species of management concern (Interagency Special Status/Sensitive Species Program 2015). The Oregon Department of Fish and Wildlife listed fishers as a Species of Greatest Conservation Need addressed in the State Wildlife Action Plan (Oregon Department of Fish and Wildlife 2025). The Oregon Biodiversity Information Center identified fishers in Oregon as imperiled, defined as “taxa that are threatened with extirpation or presumed to be extirpated from the state of Oregon” (Oregon Biodiversity Information Center 2019). In 2020, the U.S. Fish and Wildlife

Service published a final rule that listed the Southern Sierra Nevada Distinct Population Segment of fishers as endangered under the federal Endangered Species Act and concluded the remaining subpopulations in California and Oregon were not warranted for listing (U.S. Fish and Wildlife Service 2020*a*). Following a complaint challenging the 2020 Final Rule (U.S. Fish and Wildlife Service 2020*a*), the U.S. Fish and Wildlife Service reached a not-warranted finding for the Northern California Southern Oregon Distinct Population Segment of fishers (U.S. Fish and Wildlife Service 2025*a*). The U.S. Fish and Wildlife Service, however, concluded that changing drought patterns, degrading forest health, and increasing wildfire severity present a moderate risk of extinction for fishers in northern California and southern Oregon over the next 70 years and acknowledged the conservation benefits of fisher reintroductions conducted in Washington and California (U.S. Fish and Wildlife Service 2025*b*).

The contemporary distribution of fishers in Oregon includes two populations in the southernmost portion of the state (Aubry and Lewis 2003, Lofroth et al. 2010, U.S. Fish and Wildlife Service 2016, Barry 2018, U.S. Fish and Wildlife Service 2020*a*). A native population, referred to as the Northern California Southern Oregon population, is limited to the Siskiyou mountains west of Ashland and south of Grants Pass, extending south into northern California (Aubry and Lewis 2003, U.S. Fish and Wildlife Service 2016, Barry 2018). A reintroduced population, referred to as the Southern Oregon Cascades population, is descended from fishers that were reintroduced to the southern Oregon Cascades and were sourced from south-central British Columbia and northern Minnesota between 1977 and 1981 (Aubry and Lewis 2003, U.S. Fish and Wildlife Service 2016, Barry 2018).

In 2020, the Oregon Fish and Wildlife Office of the U.S. Fish and Wildlife Service developed the Oregon Fish and Wildlife Interim Fisher Conservation Strategy to guide priorities,

actions, and collaboration for fisher conservation in western Oregon (U.S. Fish and Wildlife Service 2020*b*). Therein, the U.S. Fish and Wildlife Service defined successful fisher conservation in Oregon as achieving "self-sustaining, interacting populations of fisher well-distributed throughout available habitat" (U.S. Fish and Wildlife Service 2020*b*). The objectives designed to meet the conservation goal of this strategy were:

- Conserve and increase the size and distribution of existing fisher populations
- Improve connectivity of fisher populations
- Restore and maintain landscapes that are resilient to disturbances and support fisher population viability
- Develop and support collaboration among partners and stakeholders for fisher conservation

The U.S. Fish and Wildlife Service's goal of achieving self-sustaining populations may include expansion as a result of natural colonization of fishers into currently unoccupied habitat. It may also include carefully considered reintroductions to historically occupied areas to increase distribution and redundancy, or to provide demographic or genetic support to existing populations. The distribution of fishers in Oregon, however, is not currently expanding (Barry 2018). Some evidence suggests the spatial distribution of fishers in Oregon may be contracting despite the availability of forest conditions often associated with fisher occurrence (Barry et al. 2021). Thus, reintroduction is being considered as a conservation action to increase population resiliency and redundancy (U.S. Fish and Wildlife Service 2017, 2020*b*).

The Oregon Department of Fish and Wildlife identifies fishers as a Species of Greatest Conservation Need in the State Wildlife Action Plan (Oregon Department of Fish and Wildlife 2025). Beyond maintaining complex forest structure with large trees and improving connectivity

for dispersal, genetic exchange, and population expansion, the Department recommended exploring the feasibility of reintroductions, including potential ecological and genetic constraints. The Department further recommended implementing feasibility studies to guide specific conservation actions and management decisions for potential reintroductions.

Our objective in this Implementation Plan is to develop a blueprint for the reintroduction of fishers to an unoccupied portion of their historical distribution in western Oregon and address conservation objectives described by the U.S. Fish and Wildlife Service in the Oregon Fish and Wildlife Office Interim Fisher Conservation Strategy (U.S. Fish and Wildlife Service 2020*b*) and by the Oregon Department of Fish and Wildlife in the State Wildlife Action Plan (Oregon Department of Fish and Wildlife 2025). This Implementation Plan is an extension of the *Fisher Recolonization and Recovery in Oregon: Feasibility Assessment for the Reintroduction of Fishers* authored by Matthews et al. (2021). Matthews et al. (2021) concluded that unoccupied portions of the historical distribution of fishers in the Oregon Cascades could support the reintroduction of fishers based on an evaluation of forest conditions, potential for connectivity to extant fisher populations, and potential risk factors.

In July 2022, a group of partners representing state and federal agencies, land managers, regulatory entities, non-profit organizations, wildlife researchers, and the forestland owners began regularly meeting to discuss the contemporary status of fishers in Oregon and potential conservation measures. Partners agreed there is a need to reduce extinction risk and restore resiliency, redundancy, and representation of fishers in Oregon. The partners also agreed that under a no-action alternative there is a moderately-high risk of decline or extirpation of extant populations of fishers (native and introduced populations in Oregon). Given this consensus, we are seeking to reduce risk by implementing a conservation-focused reintroduction of fishers to an

unoccupied portion of their historical distribution to enhance the probability of persistence for fisher populations in Oregon. By implementing a reintroduction, our goal is to reduce the potential for decline and/or population loss to less than a 20% risk of extinction over the next 50 years, and contribute towards broader connectivity for fisher populations in the Pacific Northwest. Reintroductions require rigorous justifications prior to implementation, and must balance conservation benefits against the costs and risks to ecological, social, and economic interests (Armstrong and Seddon 2008, Pérez et al. 2012, IUCN Species Survival Commission 2013).

In order to objectively evaluate the necessity, potential risks, and logistical suitability in justifying and planning a reintroduction, partners adopted a hierarchical decision-making framework (*sensu* Armstrong and Seddon 2008, Pérez et al. 2012) to evaluate reintroduction as a

conservation action for fishers in Oregon (

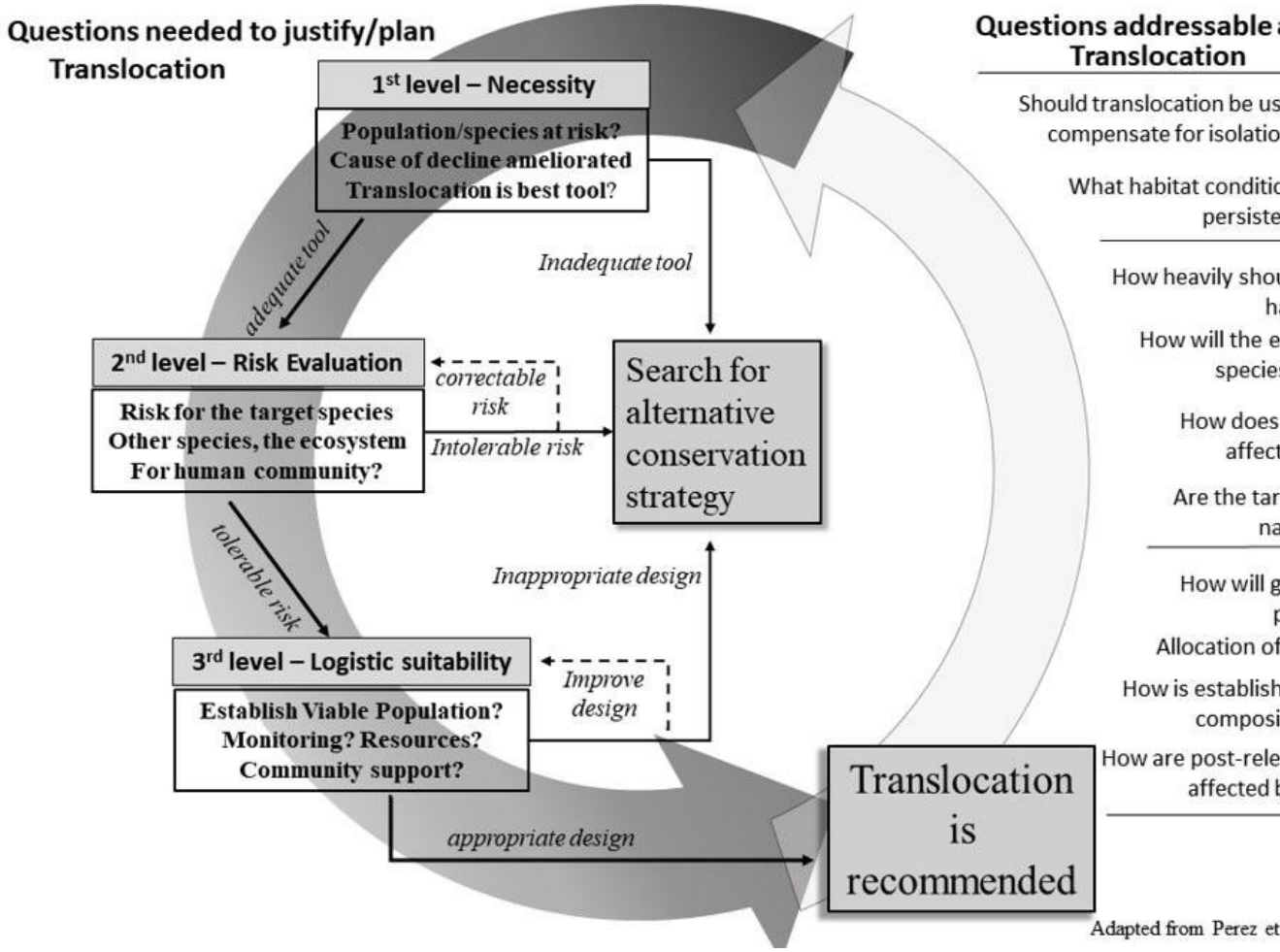

Figure 1). The system poses a series of questions to guide adaptive decision making for a reintroduction. We address each decision level and associated questions in turn. Our implementation plan follows the IUCN Species Survival Commission (2013) Guidelines for Reintroductions and Other Conservation Translocations, the Policy and Procedures for Conservation Translocations of Animals and Plants (California Department of Fish and Wildlife 2017), and best-practice guidance from the primary and secondary literature on conservation reintroduction science.

Hierarchical Decision-making System for Reintroduction

We address 10 criteria for reintroductions nested into three successive decision levels recommended by Pérez et al. (2012). The successive decision levels and criteria are as follows and visually depicted in Figure 1:

Level 1 – Necessity of the Reintroduction
1. Is the species or population under threat?
2. Have the threatening factors been removed or controlled, or were they absent in the release area?
3. Is reintroduction the best tool to mitigate conservation conflicts?

Level 2 – Risk Evaluation
4. Are risks for the target species acceptable?
5. Are risks for the other species or the ecosystem acceptable?
6. Are the possible effects of the translocation acceptable to local people?

Level 3 – Technical and Logistical Suitability
7. Does the reintroduction maximize the likelihood of establishing a viable population?
8. Does the reintroduction include clear goals and monitoring?
9. Do enough economic and human resources exist?
10. Do scientific, governmental, and partner groups support the reintroduction?

Level 1 – Necessity of the Reintroduction

The recovery of species of conservation concern often requires a multitude of activities and engaged partners. Conservation actions may include ameliorating threats, improving habitat conditions, genetic management, or removing invasive species. In circumstances where natural recolonization is unlikely, reintroductions may be necessary to re-establish populations following extirpation (IUCN Species Survival Commission 2013). Reintroductions take many forms to achieve a diverse array of objectives (Armstrong and Seddon 2008, Seddon 2010, IUCN Species Survival Commission 2013, Taylor et al. 2017). A conservation reintroduction is defined as "the intentional movement and release of a living organism where the primary objective is a

conservation benefit: this will usually comprise improving the conservation status of the focal species locally or globally, and/or restoring natural ecosystem functions or processes" (IUCN Species Survival Commission 2013). Herein, we provide an evaluation of historical and contemporary threats to fishers and an evaluation of reintroduction as a conservation tool to recover fishers in Oregon.

Population or Species at Risk

Fishers were once widely distributed throughout forested portions of western Oregon (Aubry and Lewis 2003, Barry et al. 2021). Today, fishers occupy approximately 15% of their former distribution between the west slope of the Oregon Cascades and the Pacific Ocean, with contemporary locations occurring in the southern-most portion of the state (Aubry and Lewis 2003, Barry et al. 2021). Fisher population declines and range contractions were attributed to unregulated trapping for fur, predator- and pest-control campaigns, and habitat loss of structurally-complex forests (Powell 1993, Aubry and Lewis 2003, Krohn 2012, Powell et al. 2017). Despite the reduction or cessation of activities that were suspected to have caused extirpations of fishers in Oregon, fishers have failed to repatriate a majority of their historical distribution in the Coastal and Cascade Mountains of Oregon (Barry et al. 2021). Small, isolated populations, a reliance on late-seral forest components, limited reproductive and dispersal capacity, and vulnerability to wildfire and rodenticides have catalyzed conservation concern for fishers in the western United States (U.S. Fish and Wildlife Service 2020*a*). Thus, fishers remain a species of conservation concern in Oregon, California, and Washington (California Department of Fish and Wildlife 2025, Oregon Department of Fish and Wildlife 2025, Washington Department of Fish and Wildlife 2025).

Amelioration of Cause of Decline

The causes of previous species extirpations require critical consideration and strong evidence that they have been removed or significantly reduced prior to justifying a species reintroduction (IUCN Species Survival Commission 2013). The extirpation of fishers from a majority of their historical distribution in Oregon in the late 1800s and early 1900s was attributed to unregulated trapping for furs, predator- and pest-control efforts, and habitat loss and fragmentation (Aubry and Lewis 2003, Lofroth et al. 2010, U.S. Fish and Wildlife Service 2016). These threats have been either removed or reduced since the extirpation of fishers from large portions of their distribution in Oregon (U.S. Fish and Wildlife Service 2016, Matthews et al. 2021). The closure of trapping seasons for fishers, elimination and regulation of widespread predator- and pest-control campaigns, and changes in forest management have alleviated the mechanisms leading to fisher extirpations across their range (Aubry and Lewis 2003, Spies et al. 2018).

The unregulated trapping of fisher for fur was recognized as a threat to fisher persistence by the early 1900s (e.g., Bailey 1936, Grinnell et al. 1937). Fisher trapping was closed in Oregon in 1937 and across the Pacific states by 1946 (Aubry and Lewis 2003). However, fishers are susceptible to incidental capture in traps set for other species (Earle 1978, Luque 1983, Lewis and Zielinski 1996). In Oregon, it is legal to harvest many mammals that are found in the historical and contemporary distribution of fishers, including bobcat (*Lynx rufus*), gray fox (*Urocyon cinereoargenteus*), red fox (*Vulpes vulpes*), coyote (*Canis latrans*), mink (*Neovison vison*), and marten (*Martes caurina*). Only five incidental fisher captures have been reported since 1975, two resulting in mortality (U.S. Fish and Wildlife Service 2016). Although it is unknown how many fishers are illegally harvested annually, the U.S. Fish and Wildlife Service

(2016) expected incidental fisher captures to remain infrequent into the foreseeable future assuming current trends continue. Further, ODFW Furbearer Regulations emphasize fisher conservation and require that any incidental capture or other forms of take must be reported within 48 hours. No incidental capture or other forms of take have been reported (Mathew Vargas, Oregon Department of Fish and Wildlife, personal communication).

The widespread use of strychnine and other pesticides as predator- and pest-control agents was likely a direct and indirect source of fisher mortality in the early and mid-1900s, despite fishers not being an intended target of systematic control efforts (Douglas and Strickland 1987, Powell 1993). The feeding of forest-dwelling herbivores and granivores (e.g., mice, porcupine) on aerial-dispersed seeds and regenerating seedlings in managed forests can decrease projected yields and increase the length of stand rotations (Black and Lawrence 1992). Strychnine, zinc phosphide, sodium fluoroacetate (1080), and many other rodenticides were frequently dispersed in managed forests to control small mammals and to protect seeds and seedlings (Black and Lawrence 1992), but resulted in secondary poisoning of a diverse array of wildlife species (Evans and Ward 1967).

Predator- and pest-control programs underwent extensive reform following the 1964 Leopold Report (Leopold et al. 1964), culminating in an executive order in 1972 banning the use of pesticides for predator control on federal lands (Feldman 2007). Today, the legal application of strychnine and other pesticides in the management of forests is regulated by federal and state legislation and forest practice rules (e.g., Oregon Department of Forestry 2020). The illegal application of anticoagulant rodenticides and other toxicants, particularly related to trespass marijuana cultivation, however, remain a significant and persistent threat to fishers (Gabriel et al. 2012, 2015, Thompson et al. 2014, U.S. Fish and Wildlife Service 2016, 2020*a*).

The magnitude and intensity of timber harvest through the 1990s was among the suspected primary causes for fisher declines across their range in North America (Douglas and Strickland 1987, Powell 1993, Powell and Zielinski 1994). Since 1990, the volume of timber extracted throughout the historical distribution of fishers has declined sharply in Oregon. Federal timber harvest volume in Oregon has dropped by over 90% between 1989 and 1995 (Gale et al. 2012, Oregon Forest Resources Institute 2020). Although harvest from private forestlands in Oregon declined following the recession of 2007-2009 and the global pandemic of 2020-2023, annual harvest volume has remained relatively stable, between 3.8 billion board feet, since 2013 (Gale et al. 2012, Oregon Forest Resources Institute 2020).

Although the volume of timber harvest has declined since the late 1980s, particularly on federal land, net percent changes in the amount of older forests (OGSI-80; Franklin and Spies 1991) in the area designated under Northwest Forest Plan declined 2.8% on federal lands and 11.7% on non-federal lands between 1993 and 2012 (Davis et al. 2015). These declines occurred despite gross losses of 6.0% on federal lands and 23.2% on non-federal lands from wildfire, timber harvest, and insects or other causes, suggesting that forest succession compensated for losses of older forests following disturbance (Davis et al. 2015). Indices of older forest are often an indicator of fisher habitat suitability (e.g., Fitzgerald et al. 2013, Aubry et al. 2017). Thus, the rate of forest succession will need to exceed the rate of gross losses over several decades to achieve increases in the availability of older forest (Davis et al. 2015) and predicted habitat suitability for fishers (U.S. Fish and Wildlife Service 2016).

The availability of younger, structurally-complex forests that are occupied by fisher and found on private forestlands (e.g., Green et al. 2018, 2022*b*, Powell et al. 2025), may facilitate access to additional resources (e.g., Parsons et al. 2020). In addition, private forestland owners in

Oregon that have enrolled in a Candidate Conservation Agreement with Assurances (CCAA) with the U.S. Fish and Wildlife Service, have management plans that retain complex structures important to fisher (U.S. Fish and Wildlife Service 2017). A Candidate Conservation Agreement with Assurances (CCAA) for fisher in Oregon is a voluntary agreement where public or private landowners agree to manage their lands to remove or reduce threats to fisher. The U.S. Fish and Wildlife Service developed a template Candidate Conservation Agreement with Assurances for the West Coast distinct population segment of the fisher in Oregon. In general, these agreements are intended to promote conservation of candidate species. For partners who are enrolled landowners that fully implement conservation measures for fishers, those partners receive assurances that additional restrictions will not be required if fishers become listed as threatened or endangered under the Endangered Species Act (50 CFR 17.22(d) and 17.32(d). The agreement establish general forest management guidelines and identify minimum conservation measures for project partners including, but not limited to 1) promoting the development of habitat structures that would support fisher, 2) allowing access to covered lands to conduct fisher surveys, 3) allowing the potential future translocation of fishers onto enrolled lands, 4) protecting fisher dens and their young by limiting disturbance and impacts to denning structures and, 5) limiting control of other animals by trapping/nuisance that could pose a risk to the fisher (U.S. Fish and Wildlife Service 2017).

Reintroduction as a Management Tool

The removal and alleviation of threats offer opportunities for habitat recovery and species to reoccupy portions of their historical distribution from which they were extirpated (Soulé and Terborgh 1999). The alleviation of threats allowed for fishers to reoccupy large portions of their historical distribution through natural recolonization and successful reintroductions in the eastern

United States and Canada (Lewis et al. 2012). Fishers in the western United States, however, have failed to reoccupy equivalent portions of their historical distribution, despite similar conservation measures (Lewis et al. 2012, Barry et al. 2021).

Limited expansion and recovery in the Southern Oregon Cascades population raised questions about the presence of fisher habitat in the Oregon Cascades (Barry 2018). Recent models suggest the presence of mature forest structure are often associated with fisher occurrence both within the historical distribution and within the dispersal capability of fishers in unoccupied portions of the Oregon Cascades (Fitzgerald et al. 2013, Aubry et al. 2017, Spencer et al. 2019, Barry et al. 2021). To further evaluate the availability of environmental conditions to support a fisher reintroduction in Oregon, Matthews et al. (2021) developed an ensemble model of four habitat suitability and occupancy models developed by Allen (1983), Fitzgerald et al. (2013), Aubry et al. (2017), and Barry et al. (2021). Matthews et al. (2021) concluded that portions of the Oregon Cascades could feasibly support the reintroduction of fishers based on forest conditions and proximity to extant fisher populations.

Fishers failing to reoccupy predicted habitat in the historical range is not unique to Oregon. The northern limit of the Southern Sierra Nevada fisher population is an area along the northwestern portions of Yosemite National Park and the Stanislaus National Forest (U.S. Fish and Wildlife Service 2016). Spencer et al. (2016) identified habitat north of this area that fishers have failed to reoccupy due to physical barriers, indicating human intervention may be required to facilitate repatriation. It is important to note that drought and wide-spread tree mortality have also dramatically altered forest composition and structure within the southern Sierra Nevada since 2016 (Fettig et al. 2019). Similarly, in northern California, Davis et al. (2007) and Carroll et al. (2005) identified predicted habitat to the east of the contemporary distribution limit of

fishers in northern California that fishers had failed to reoccupy. These findings were a contributing catalyst for the translocation of fisher to the northern Sierra Nevada of California between 2009 and 2011 (Callas and Figura 2008, Green et al. 2022*a*, Powell et al. 2025).

Predicted habitat, quantified by environmental conditions and climate variables, is not the only possible explanation for the failure of fishers to recolonize a larger extent of the Oregon Cascades. Alternative hypotheses include 1) the source individuals from British Columbia and Minnesota were maladapted to the reintroduction area in the southern Oregon Cascades (Lewis et al. 2012), 2) patterns of male-biased dispersal and female philopatry may limit population expansion (Aubry et al. 2004, Matthews et al. 2013*b*, Tucker et al. 2017), 3) prey diversity and abundance may not be sufficient to support the energetic requirements of a persisting population (Powell 1977), 4) predation risk (Wengert et al. 2014, Gabriel et al. 2015), and 5) competition pressure exceed thresholds (Green et al. 2018), and 6) rodenticide exposure may affect survival and reproduction (Gabriel et al. 2012, 2015, 2018). These alternative hypotheses are considered further in Level 2 – Risk Evaluation.

Recovery efforts for fisher have relied heavily on translocations, with fisher being one of the most frequently and successfully reintroduced carnivores (Lewis et al. 2012, Powell et al. 2012). Translocations, however, are not always successful (Breitenmoser et al. 2001). Lewis et al. (2012) reported that 23% of fisher reintroductions between 1900 and 2011 failed to establish a sustaining population. Nonetheless, recent reintroductions of fishers in Washington and California offer insights and promise for the reintroduction of fishers in Oregon.

Fishers reintroduced to the Olympic Peninsula of Washington between 2008 and 2010 exhibited stable trends in occupancy between 2013 and 2016, expansion beyond the primary study area, and evidence of successful reproduction suggested the successful establishment of a

population (Happe et al. 2019). The movement patterns of 30 of the 69 (43%) fishers released in the first two cohorts to the southern Cascades of Washington between 2015 and 2020 indicated home range establishment. Mean annual survival rates (and 95% confidence intervals) the first year following reintroduction were 0.77 (0.59–0.95) and 0.67 (0.48–0.86) for the first two cohorts and 0.67 (0.38–0.96) for the second year following reintroduction for the first cohort (Lewis et al. 2019). Female reproduction, however, was only documented on three occasions between 2016 and 2018 due to poor weather conditions, aircraft availability, transmitter failure, and limited ground-based telemetry capacity that hindered relocating released fishers (Lewis et al. 2019).

The reintroduction of 40 fishers to the northern Sierra Nevada of California from the Klamath-Siskiyou and Coastal mountains of northern California between 2009 and 2011 offered the opportunity to expand the spatial distribution of fishers in California and evaluate the capacity of commercial timberlands to support a reintroduced fisher population (Callas and Figura 2008, Powell et al. 2025). Released fishers exhibited high survival and reproduction during the initial post-release period and were consistent with a growing population through 2017 (Green et al. 2022*a*, Powell et al. 2025). The population appeared unlikely to go extinct prior to the year 2040, barring dramatic decreases in survival and reproduction caused by unpredictable events, such as wildfire or an epizootic disease outbreak (Powell et al. 2025).

Concurrent with the translocation of fishers to the northern Sierra Nevada, partners monitored a portion of the source population in the Klamath-Siskiyou region of northern California (Green et al. 2018). Fishers were monitored using hair snares and genetic samples to identify individual fishers. Green et al. (2018) reported their spatial capture-recapture model failed to detect any significant annual changes in the fisher density for the three years following

the removal of approximately 20% of the estimated population between 2009 and 2010 for reintroduction to the northern Sierra Nevada. Green et al. (2018), however, reported a significant decrease in fisher occupancy following the first year of removals in 2009 and occupancy rates remained low through 2013. These results highlight discrepancies between patterns of density and occupancy over time and at fine spatial scales (Matthews et al. 2011, Green et al. 2018). These discrepancies between density and occupancy emphasize the importance of measuring changes in density over time to accurately monitor the effects of translocation on source and reintroduced populations.

Taken in sum, these contemporary reintroductions of fishers in Washington and California were successful in establishing a population in the target reintroduction areas, expediting the recovery of fishers to unoccupied portions of their historical distribution. Further, there is evidence to suggest source populations are robust to the removal of translocated individuals. Thus, reintroductions offer an opportunity to achieve the respective conservation goals of these projects. The contrasting outcomes in post-release monitoring between these three translocations, however, demonstrate the variation in available insights and emphasize the importance of post-release monitoring for fisher reintroduction in Oregon.

Level 2 – Risk Evaluation

Critically evaluating the feasibility, implementation, and outcomes of a reintroduction program requires an empirical understanding of the ecology at source and release areas (Armstrong and Seddon 2008, IUCN Species Survival Commission 2013, Halsey et al. 2015, Green et al. 2018). Reintroduction programs frequently fail to monitor and evaluate the impacts of a translocation on source populations (IUCN Species Survival Commission 2013). In order for a reintroduction to provide a net conservation benefit, the removal of founding individuals

should limit any negative effects on the source population and avoid reducing long-term viability of source populations (IUCN Species Survival Commission 2013). For a reintroduction of fishers to Oregon, monitoring source populations prior to and following the removal of founding individuals can be done cost-effectively using robust, non-invasive survey methods (Green et al. 2018) and should be a key component of the reintroduction of fishers to the southern Oregon Cascades.

Priorities and resources of an agency, however, heavily influence the feasibility of monitoring source populations post removals. An agency may rely on simulations and previous empirical results to support post-translocation decisions. Simulation modeling based on contemporary population vital rates indicated the resiliency of the Northern California Southern Oregon population to provide founding individuals for a reintroduction (Padilla 2025). Further, Green et al. (2018) reported no change in fisher density in a portion of the Northern California Southern Oregon population up to three years following the removal of approximately 20% of the population for translocation to the northern Sierra Nevada. An important caveat to the results of Padilla (2025) and Green et al. (2018) is that the forests occupied by fisher in northern California and southern Oregon are experiencing rapid change attributed to wildfire, forest pathogens, and climate change (Serra-Diaz et al. 2018, Aparício et al. 2026), with negative, population-level effects on fishers (Green et al. 2022*b*). Where population abundance estimates are lower (Appendix B), monitoring certain source populations may be warranted. While the recommendation is to monitor the source population post-translocation, that may not be reasonable across a large spatial scale, and if the source population exists across state lines, the decision on intensity of monitoring is up to the corresponding state agency. See subsection *Consequence for source population* for more details.

Reintroduction programs often focus on components of the physical environment (e.g., tree species composition, forest age) at candidate reintroduction areas to evaluate the suitability and capacity of a landscape to support a reintroduced population (Steury and Murray 2004, Jachowski et al. 2011). However, these components of the physical environment are often surrogates for other ecological processes that can limit reintroduction feasibility, such as predation risk, competition, and prey availability (Hetherington and Gorman 2007, Cook et al. 2010, Carlson et al. 2014). Further, specific to fishers in Oregon, physical components of the environment, including forest structure and climate, do not appear to be the main factor limiting fisher recovery in extirpated portions of its range (Barry et al. 2021). The diversity, abundance, and distribution of predator, competitor, and prey communities may also influence the success of reintroduction efforts. Empirical evaluations and comparisons of these factors at source and reintroduction areas are an important component to evaluating risk of fisher removal and translocation.

Understanding how predation influences reintroduction success is an important and understudied aspect of reintroduction science. Predation is a naturally occurring mechanism of population regulation between predators and prey (Palomares et al. 1999, Coonan et al. 2005, Moehrenschlager et al. 2007, Hunter and Caro 2008). Predation combined with all other sources of mortality, however, could negatively affect the persistence of a reintroduced fisher population if annual predation rates exceed annual juvenile recruitment rates (Roy 1991, Lewis 2014). Bobcats, cougars, and coyotes are frequent predators of fishers throughout their western distribution (Wengert et al. 2014) and are the leading source of fisher mortality (Gabriel et al. 2015). In addition to top-down pressures from larger, dominant carnivores, fishers also compete with carnivores in the same guild (e.g., grey fox, Green et al. 2018). Overlap in distribution, diet,

and habitat requirements drive competitive interactions for resources among many small carnivores (Lesmeister et al. 2015). These competitive interactions can influence carnivore community dynamics, spatial patterns, and, potentially, recolonization and reintroduction outcomes (Green et al. 2018). Fishers and grey foxes often co-occur throughout the Pacific Northwest and are similar in body size and foraging patterns (Campbell 2004, Green et al. 2018). Green et al. (2018) found fisher occurrence negatively affected the persistence of grey fox but had a positive effect on grey fox colonization. They posited the additive effects of interference competition with predators and exploitative competition with grey fox could influence natural recolonization and reintroduction outcomes for fisher. Thus, a spatially-explicit assessment of predation risk and potential competition in the proposed reintroduction area compared to portions of the Northern California Southern Oregon population area would inform discussions about risk and provide a baseline to evaluate reintroduction outcomes (Heezik et al. 1999, Seddon et al. 2007, Poessel et al. 2011, Halsey et al. 2015).

Mustelids have some of the highest relative energetic expenditures of all carnivores due to their elongated shape and limited capacity to store fat (Brown and Lasiewski 1972, Buskirk and Harlow 1989). These adaptations provide efficiencies for the pursuit of prey (Andruskiw et al. 2008), but place stringent demands on foraging efficiency to avoid metabolizing limited fat reserves and structural protein (e.g., Buskirk and Harlow 1989). Fishers, therefore, are under relatively high pressure to maximize the energy gained from prey items and minimize acquisition costs (Charnov 1976, Powell et al. 2017). Thus, spatial and temporal variations in prey abundance could have direct implications for fisher reintroduction outcomes (e.g., Cook et al. 2010).

Conducting surveys in the proposed release area and in portions of the extant Northern California Southern Oregon population area prior to the reintroduction of fishers to the South Cascades are important steps in evaluating how the presence and abundance of predators, competitors, and prey may affect the success of a reintroduced population. Comparison of density, occupancy, and community composition where a native fisher population persists can help determine thresholds that guide where reintroduction and release occur to maximize establishment and retention. Although important for planning current and future fisher reintroductions, this information is rarely evaluated (however, see Halsey et al. 2015) and would address key information gaps. See Appendix B for more case-specific details.

Risk for the Target Species

An important criteria for evaluating a reintroduction is considering the potential risks that could threaten the persistence of the source and reintroduced populations (Armstrong and Seddon 2008, Pérez et al. 2012, IUCN Species Survival Commission 2013). In addition to the potentials for risk to the source population, risks to those fishers in the reintroduced population in Oregon include consequence for genetic mixing, disease spread, energetic requirements, competition, predation, wildfire, and direct mortality (e.g., incidental trapping, road-strike, or rodenticide exposure) (Pérez et al. 2012, U.S. Fish and Wildlife Service 2016, 2020*a*). Many of these risks can be mitigated by conducting pre-reintroduction population-modeling simulations and surveys combined with thoughtful planning and implementation of the reintroduction process. The empirical evaluation of some risk factors will require well-designed post-reintroduction monitoring, adaptive strategies, and ongoing discussions between partners to re-evaluate risk (e.g., Pérez et al. 2012).

Consequence for source population

Theoretical and empirical evidence suggest the proximity of source populations and reintroduction areas play an important role in reintroduction outcomes (e.g., Miller et al. 1999, Lewis et al. 2012). The IUCN Species Survival Commission (2013) recommended wild source populations and founding individuals should be genetically, morphologically, physiologically, and behaviorally similar to the original population that occupied the proposed reintroduction area. This typically results in source populations being geographically close to reintroduction areas. In their empirical evaluation of factors contributing to the success of 38 fisher translocations in North America, Lewis et al. (2012) found the proximity of source populations to release areas was an important predictor of translocation success. These considerations lead Matthews et al. (2021) to conclude the Northern California Southern Oregon population of fishers should serve as the source population for a translocation to the southern Oregon Cascades.

The viability of candidate source populations must be evaluated prior to and following the removal of founding individuals (Dimond and Armstrong 2007, Armstrong and Seddon 2008). In preparation for the translocation of fishers to the northern Sierra Nevada of California, Powell and Zielinski (2005) concluded that the removal of up to 20 fishers per year for as many as eight years from northwestern California would result in <5% increase in the probability of extinction of the northwestern California population. Powell and Zielinski (2005) parameterized their model with a carrying capacity of the northwestern California source population set at 2,000 $\pm$ 250 fishers. More recently, Furnas et al. (2017) estimated the expected abundance of fishers in northern California and southern Oregon as 3,196 fishers (95% confidence interval:

2,507 – 4,184), though we acknowledge their estimates resulted from a Royle-Nichols expected abundance model and did not account for closure between primary sampling periods.

Subsequent, empirical evidence support the conclusions of Powell and Zielinski (2005). A 465 km^2 area of mixed U.S. Forest Service and private ownership in the Klamath-Siskiyou Ecoregion of northern California and southern Oregon served as one of several sources of fishers for translocation to the northern Sierra Nevada (Powell et al. 2019). Population surveys using non-invasive genetic methods began in 2006 and nine fishers (7F, 2M) were removed for translocation between 2009 and 2010 (Green et al. 2018). The removal of the nine fishers, approximately 20% of the estimated population, had no detectable effect on the average abundance of fishers up to three years following the removal (Green et al. 2018).

A caveat to the predictive and empirical model results from Powell and Zielinski (2005) and Green et al. (2018) for the removal of fishers from this population for future translocations was the uncertainty associated with landscape-scale change. Powell and Zielinski (2005) acknowledged a shortcoming of their modeling was a failure to account for the potential influences of timber management on fisher survival and reproduction and consequent trends in post-removal demographics. Additional research on the impacts of varying levels of timber management intensities on fisher survival and reproduction is a pending research need (although see Weir and Corbould 2006, Green et al. 2022*a*). However, recent research on the potential impacts of private forest management on fisher survival and reproduction found encouraging results, as fishers translocated to private forestlands in California exhibited high survival and reproduction during an initial post-release period (Green et al 2022a, Powel et al 2025). Recent analyses also suggest fuels management and wildfire have negative impacts on fisher occupancy and density (Scheller et al. 2011, Sweitzer et al. 2016, Green et al. 2022*b*). These uncertainties

and the stochastic nature of wildfire reinforce the need for a reintroduction to Oregon to increase resilience and redundancy and the need to monitor source populations prior to and following the removal of founding individuals (Armstrong and Seddon 2008, IUCN Species Survival Commission 2013).

To further address the potential risk to the source population, Padilla (2025) conducted a population viability analysis to model the effect of removing animals from extant fisher populations in northern California and southwest Oregon. Padilla (2025) used a modified Euler-Lotka female-based model to estimate the maximum intrinsic growth rate of the reintroduced population. Three alternate removal scenarios were modeled involving the removal for 40 individual female fishers, including: 1) all removals in year one; 2) 14 removals in year one, 13 removals in year two, and 13 removals in year three; and 3) 20 removals in year 1, 10 removals in year five, and 10 removals in year 10. Padilla (2025) also modeled a 0.025 annual fire probability, equivalent to a 40-year fire return interval. For each fire, the proportion of the fisher population that died due to fire was drawn from a uniform distribution between 0.2 and 0.75. Baseline vital rates (Table 1) were set based on unpublished estimates from extant fisher populations in southwest Oregon (Benjamin Padilla and Keifer Titus, unpublished data, Oregon Department of Fish and Wildlife) and estimates from California, Oregon and Washington reported in peer-reviewed literature and unpublished reports (Padilla 2025). Padilla (2025) evaluated and defined the probability of failure as a population with fewer than 20 female fishers 15-years post removal. The modeled probability of failure for each scenario was <0.05, indicating that the simulated source populations recovered and reached carrying capacity in all three removal scenarios with a 0.025 annual fire probability. Thus, these results suggest that extant fisher populations could sustain the removal of founding individuals for a reintroduction.

We suggest monitoring the proposed source population in northern California and southern Oregon prior to and following the removal of founding individuals. Monitoring will offer opportunities to estimate annual vital rates, validate the baseline demographic rates used by Padilla (2025), and support future modeling and decision making about potential source removals. Pre- and post-translocation monitoring will provide opportunities to confirm the viability of portions of the Northern California Southern Oregon fisher to serve as a source population for additional translocations and empirically evaluate the effect of removing individuals for translocation (IUCN Species Survival Commission 2013, Green et al. 2018). Use of standardized survey protocols and analytical methods, such as those implemented by Green et al. (2018, 2022*b*) for non-invasive monitoring of fishers and other mesocarnivores in northern California and southern Oregon (Appendix B), would ensure consistent survey effort and the ability to compare demographic rates and variation among study areas.

Genetic mixing

The Southern Oregon Cascades population, is descended from fishers that were reintroduced to the southern Oregon Cascades and were sourced from south-central British Columbia and northern Minnesota between 1977 and 1981 (Aubry and Lewis 2003, U.S. Fish and Wildlife Service 2016, Barry 2018). Recent genetic evidence suggests, however, that the genetic haplotype unique to fishers from Minnesota (Drew-hap10; Drew et al. 2003) no longer appears to be present in the contemporary Southern Oregon Cascades population (K. Pilgrim, U.S. Forest Service National Genomics Center for Wildlife and Fish Conservation, personal communication). There is a possibility of reintroduced fishers hybridizing with fishers in the Southern Oregon Cascades population and their alternate genetic haplotype (Drew-hap9; Aubry and Lewis 2003, Drew et al. 2003).

Given an observed level of hybridization between fishers from the Northern California Southern Oregon population in the Southern Oregon Cascades population (Kristy Pilgrim, U.S. Forest Service National Genomics Center for Wildlife and Fish Conservation, personal communication), there is no evidence hybridization between translocated fishers from the Northern California Southern Oregon population with extant fishers in the Southern Oregon Cascades population would negatively impact the success of a fisher reintroduction to the central Oregon Cascades and could provide a net benefit through the introduction of new genetic material and an increase in genetic diversity (T. Levi, Oregon State University, personal communication, Matthews et al. 2021). Nonetheless, strategic pre- and post-translocation monitoring using a combination of live-capture and non-invasive methods would facilitate genetic monitoring to track potential hybridization and introgression between extant and reintroduced individuals.

Disease spread

A thorough health assessment of each individual fisher considered a candidate for translocation should be conducted by a wildlife veterinarian (Lewis and Hayes 2004, Callas and Figura 2008, Facka and Powell 2021, Lewis et al. 2022). A health assessment should include 1) a physical examination to evaluate individuals for obvious signs of illness or trauma, 2) the collection of samples for disease screening, and 3) the administration of vaccinations such as Imrab-3 rabies vaccine, Purevax ferret distemper vaccine and Ivermectin and Droncit (praziquantel) to minimize the risks of translocating nematodes, tapeworms, and flukes to reintroduction sites and bolster reintroduced individuals against potential zoonotic diseases. Fishers that fail to meet the criteria for translocation should be released at their original location of capture. Once deemed suitable for translocation, each fisher can be transported from the

captive facility and released at a pre-designated release site in the reintroduction area. All staff involved in the capture and handling of fishers will follow COVID protocols to prevent transmission including wearing gloves and KN95 respirators when handling or working near captive animals. Matthews et al. (2021) provided additional details, modified from Callas and Figura (2008).

Energetic requirements

Relative to body size, mustelids have some of the highest energetic needs of all carnivores due to their elongated shape and limited capacity to store fat (Brown and Lasiewski 1972, Buskirk and Harlow 1989). Thus, spatial and temporal variations in prey abundance or availability can have direct implications for maximizing seasonal energy demands and fisher reintroduction outcomes (e.g., Cook et al. 2010). Fishers consume a diverse array of prey, including small and medium-sized mammals, birds, reptiles, insects, and carrion, with a majority of their diet composed of small and medium-sized mammals (reviewed in Matthews et al. 2021). Mammals tend to dominate fisher diet (Table 3), with mammal remains found in >78% of fisher scats collected across five study areas in California, Oregon, and Washington (Zielinski et al. 1999, Golightly et al. 2006, Slauson et al. 2011, Parsons et al. 2020, Raley and Aubry 2020). The frequency of occurrence of other taxa, including reptiles, insects, and plant-based food items, were reported to be larger in southern portions of the distribution in California (reviewed in Raley and Aubry 2020).

Despite being a dietary generalist, fishers have evolved unique behaviors to kill porcupines (*Erethizon dorsatum*) (Powell 1993). Porcupines are among the highest ranked prey item consumed by fishers, second only to carrion in the ratio of energy available to handling time (Powell 1979, 1993, Powell et al. 2017). The catalyst for the reintroduction of fishers to the

southern Oregon Cascade Mountains in 1961 and between 1977 and 1981 was to restore a predator of porcupine (Kebbe 1961, Aubry and Lewis 2003). Raley and Aubry (2020), however, found evidence that only male fishers preyed on porcupine in the Upper Rogue River basin in southern Oregon Cascades between 1995 and 2002. Porcupine occurred in 8% (7 of 84) of male fisher scats and 6 captured male fishers had porcupine quills embedded in their skin. They found no evidence for female consumption of porcupine in 303 scats or evidence of quills in captured females. Raley and Aubry (2020) hypothesized the larger body size of males may have made them more effective at killing larger prey. Alternatively, they hypothesized that a majority of female scats were collected at den sites and females may have preferentially selected prey items that could more easily be provisioned to kits.

Slauson and Zielinski (2019) found evidence for the reproductive provisioning hypothesis in Humboldt marten (*Martes caurina humboltensis*) in the interior mountains of coastal California, where female martens appeared to consume more prey items that could be provisioned to kits during the denning season than in the non-denning season. These patterns could also be a reflection of seasonal prey availability and sampling bias (Slauson and Zielinski 2017). More recently, Parsons et al. (2020) and Happe et al. (2021) evaluated the diet composition of fishers within the fisher reintroduction areas in the Olympic Peninsula between 2008 and 2010 and the southern Cascade Mountains of Washington between 2016 and 2017, respectively. These studies reported that porcupines were uncommon on the Olympic Peninsula and in the southern Washington Cascades, and did not detect porcupine (Happe et al. 2021) or did not evaluate the presence of porcupine (Parsons et al. 2020) in the diet of fishers.

There is recent evidence to suggest that porcupine populations may be declining range-wide with steeper declines in the Pacific states (Mally 2008, Allen and Casady 2012, Appel et al.

2017). In the Pacific Northwest, the extant distribution of porcupines has declined in forested areas and is more stable in desert scrub and grassland vegetation communities (Appel et al. 2021). These results align with anecdotal evidence of recent declines in porcupine numbers in forested areas (Joe Doerr and Josh Chapman, U.S. Forest Service. personal communication). Appel et al. (2021) hypothesized the potential decline of porcupines could be attributed to a history of lethal control of porcupines and changes in forest management practices. The debarking foraging behavior of porcupines reduces the timber quality of commercially-valuable conifers (however, see Curtis 1941) and previously was the impetus for the wide-spread lethal-control of porcupines (Hoffer 1967, Hooven 1971, Anthony 1986, Borrecco and Black 1990). Additionally, the extensive logging of late-seral forests and subsequent forest succession beginning in the 1920s may have resulted in increases in porcupine abundance in western forests by rapidly increasing the abundance of early-seral forest conditions suitable to porcupine foraging and persistence (Hooven 1971, Yocom 1971). Thus, an alternative hypothesis is that porcupine declines may be a function of rapid reductions in harvest volume in recent decades and resultant declines in available early-seral forest conditions (Appel et al. 2021). Despite porcupine declines, fishers in the Pacific states consume a diverse array of prey items (Golightly et al. 2006, Parsons et al. 2020, Raley and Aubry 2020, Smith et al. 2022, Pilgrim et al. 2023).

Snowshoe hares (*Lepus americanus*) are another common prey item for fisher (Powell 1979, 1993, Powell et al. 2017) and make up a dominant component of their diet across much of North America (e.g., Powell and Zielinski 1994, Smith et al. 2023). Raley and Aubry (2020) reported Leporids (i.e., hares and rabbits) comprised 27 and 8% of the female and male fisher diets, respectively, in the southern Oregon Cascades. Similarly, Parsons et al. (2020) found hares and mountain beavers (*Aplodontia rufa*), which were indistinguishable using stable isotopes

analyses, comprised 68% (95% credible interval: 36-89%) of the diet of reintroduced fishers in the southern Washington Cascades. Parsons et al. (2020) also found snowshoe hares in the southern Washington Cascades were most abundant in early- and mid-seral stands with high stem density of small-diameter trees and dense shrub cover. Similarly, Happe et al. (2021) reported snowshoe hares were the dominant mammal species found in fisher scats collected on the Olympic Peninsula in Washington, occurring in 40% of scats and comprising an average of 40% of sample contents by volume. Mountain beavers were the second-most prevalent prey item on the Olympic Peninsula, occurring in 30% of scats and averaging 14% of sample contents by volume. Importantly, the combined frequencies of occurrence of snowshoe hares and mountain beavers on the Olympic Peninsula (70%, Happe et al. 2021) were similar to the estimated contribution of snowshoe hares and/or mountain beavers in the diet of fishers in the southern Washington Cascades (68%, Parsons et al. 2020). Recent research using DNA metabarcoding of scats collected from reintroduced fishers in the North and South Cascades also found snowshoe hares were a predominant prey item, though they occurred almost 7x more often in scats collected in the South Cascades (68%) than in the North Cascades (~10%) (Shively et al. 2025).

Squirrels and smaller mammals are also consumed by fishers throughout their western distribution (Aubry and Jagger 2006, Golightly et al. 2006, Slauson et al. 2011, Parsons et al. 2020, Shively et al. 2025). Raley and Aubry (2020) reported squirrels were approximately 25% of the diet of female fishers, particularly California ground squirrel (*Otospermophilus beecheyi*) and Douglas' squirrel (*Tamiasciurus douglasii*) that occurred in 14% and 3% of female fisher scat, respectively. Other small mammals (e.g., Geomyidae, Cricetidae) occurred in <10% of scats collected by Raley and Aubry (2020). Parsons (2020) concluded squirrels and small mammals contributed 4 and 8% of fisher diet in the southern Washington Cascades, respectively.

Birds comprised measurable portions of fisher diet in previous studies in the Oregon and Washington Cascade Mountains (Parsons et al. 2020, Raley and Aubry 2020), although birds are considered a lower quality food item based on digestible energy and handling time (Powell et al. 2017). Raley and Aubry (2020) reported evidence of Aves species in 28% and 25% of female and male fisher scats, respectively. They identified Steller's jay (*Cyanocitta stelleri*) and northern flicker (*Colaptes auratus*) in some scats, but were unable to identify most Aves remains to species. Similarly, Parsons et al. (2020) estimated 18% (95% credible interval: 1-53%) of fisher diet was comprised of ruffed grouse (*Bonasa umbellus*).

Powell (1979) estimated the daily energy expenditure of four wild male fishers ranged between 1,059 and 1,865 kJ and one female fisher expended 862 kJ per day during the winter in the Ottawa National Forest of Michigan. Based on energy expenditure and depending on the weight of individual prey items and fishers, Powell (1979) concluded a free-living fisher during the winter would need to consume a daily average of 380-665 g of a porcupine, 280-528 g of a snowshoe hare, 125-333 g of deer carrion, 295-590 g of a squirrel, 211- 376 g of mice, or an equivalent combination of prey to avoid catabolization of tissue. Further, Powell and Leonard (1983) estimated daily energy expenditure of a reproductive female fisher at the time of weaning (approximately 10 weeks post-parturition) could approach 2,500-2,900 kJ, approximately three times the average daily energy expenditure during the non-reproductive season. These results emphasize the importance of abundant and accessible prey in the proposed reintroduction area for the survival of translocated animals and recruitment of offspring (e.g., Mech and Barber-Meyer 2015).

Recent research highlights the importance of fine-scale forest conditions that mediate the food-safety tradeoff for fishers, as for other mesocarnivores (Bischof et al. 2014). Parsons et al.

(2019) reported reintroduced fishers and their progeny in the southern Washington Cascades selected sites with intermediate activity levels of hares and avoidance of sites with the highest activity levels of hares. Hare activity levels were positively correlated with bobcat activity levels, suggesting sites with more hare activity also presented an increased predation risk to fisher (Parsons et al. 2019). Parsons et al. (2019, 2020) concluded that fishers in southern Washington selected habitat mosaics of late-seral forests that provide necessary structural components and recently disturbed stands that support frequently-consumed prey species.

The success of reintroductions is influenced by the availability of prey in reintroduction areas (Steury and Murray 2004, Hetherington and Gorman 2007, Jachowski et al. 2011, Parsons et al. 2020). Diverse prey communities can reduce the probability of failure of a reintroduction by providing energetic stability through time and buffer against environmental stochasticity (Doak et al. 1998, Tilman et al. 1998, Petchey 2000). For example, survival rates of reintroduced fishers were significantly greater in areas of greater relative abundance of important fisher prey in Washington state, although there were other differences between these areas that likely also influenced fisher survival (Humphries 2021).

An evaluation of prey diversity and availability will offer insights into the prey resources available for a reintroduced fisher population and how those resources compare to areas with extant fisher populations. Prey diversity in the source population area and the proposed reintroduction area could be evaluated using remotely-triggered cameras or through the quantification of the diet of fisher predators and competitors (e.g., cougar, bobcat, coyote, gray fox) using detection-dog teams to collect scat samples and DNA metabarcoding to identify scat contents. A more complete understanding of diet composition of co-occurring carnivores will offer insights to niche partitioning and species coexistence pre- and post-fisher reintroduction.

Pre- and post-reintroduction surveys should be conducted in the source population area and the proposed reintroduction area to further evaluate prey diversity and availability for a reintroduced fisher population. Results from pre-translocation surveys will offer opportunities for partners to evaluate potential risks and make empirically-informed, adaptive decisions.

Competition and predation

Competitive species interactions are an aspect of carnivore habitat rarely considered prior to reintroductions and can directly influence reintroduction outcomes (Cook et al. 2010, Fisher et al. 2014). Interspecific killing, the most extreme form of interference competition, imposes top-down pressures on mesocarnivores and forces sacrificing other activities for predator avoidance (Bischof et al. 2014). Mortality due to interspecific killing is common for fishers (Wengert et al. 2013, 2014) and is the leading cause of mortality (70% of mortalities) in western portions of their range (Gabriel et al. 2015). The most frequent predators of fishers in western North America in declining frequency are bobcat, mountain lion, and coyote (Wengert et al. 2014). Subdominant species, like fishers to the aforementioned carnivores, may reduce competitive interactions through spatial and temporal segregation (Hunter and Caro 2008, Bischof et al. 2014). Fishers may also face exploitative competition, where one species reduces resources available to another by consuming them, from both larger-bodied and smaller-bodied species. For example, in the Great Lakes region, Smith et al. (2023) found that fishers ate small-bodied prey (i.e., mice, chipmunks) more often at sites where martens were absent and ate more larger-bodied prey items (i.e., snowshoe hares, porcupines) as martens became more locally abundant. Fishers overlap spatially with many small-carnivore species that have similar distributions, diet, and habitat requirements which may increase competitive interactions for resources (Prugh et al. 2009, Roemer et al. 2009, Lesmeister et al. 2015, Smith et al. 2023, 2024).

Estimating occupancy and abundance patterns of carnivores can be challenging because these species often occur at low densities, have broad geographical distributions, and may be wary of traditional survey devices (e.g., live traps, hair snares; Thompson 2004). Thus, surveys for these species frequently result in low detection probabilities and high uncertainty around occupancy or abundance estimates (Williams et al. 2002). Recent advances in data integration using hierarchical frameworks can provide opportunities to incorporate data from multiple sources to inform detection probabilities and improve estimates of population parameters or occurrence (Blanc et al. 2014, Chandler and Clark 2014, Ruprecht et al. 2021). Non-invasive survey methods, such as detection dog surveys or remote cameras, are cost-effective and can be combined in modeling efforts, making them an appealing option to survey for elusive carnivores (Long et al. 2008, Kéry and Schaub 2012). Non-invasive techniques are an important alternative to capturing and collaring animals that can avoid the stresses of capturing and handling, are less costly to deploy over large spatial scales, and can achieve the same survey and monitoring objectives.

Remotely-triggered cameras offer opportunities to detect target species and estimate occupancy patterns (Long et al. 2008, O'Connell et al. 2011). We recommend a minimum pre-translocation survey effort using remotely-triggered cameras to estimate the occupancy patterns of potential fisher predators (cougars, bobcats, and coyotes) and competitors (grey foxes) in the source population area and the reintroduction area (e.g., Halsey et al. 2015, Parsons et al. 2019). Alternatively, or in addition to occupancy surveys using remotely-triggered cameras, detection dog teams can efficiently locate carnivore scats across large areas (Beckmann 1997, Thompson et al. 2012, Moriarty et al. 2018, Ruprecht et al. 2021, Martin et al. 2025*a*). Genetic analyses of scats allow researchers to estimate population abundance and demography (Davidson et al. 2014,

Wultsch et al. 2014, Eriksson et al. 2020) and diet composition and partitioning (e.g., Kartzinel et al. 2015, Martin et al. 2025*a*). Pre-translocation surveys using non-invasive methods will also enable a further confirmation of the absence of fishers within the proposed reintroduction area, acknowledging that Barry et al. (2021) found no evidence of fisher presence in the proposed reintroduction area using remotely-triggered cameras and detection-dog teams between 2015 and 2017.

A contemporary and empirical evaluation of the occupancy patterns of potential fisher predators and competitors will offer opportunities to quantify potential predation and competition risk between the source population area and the reintroduction area. A recent multi-species survey and modeling effort offered an empirical evaluation of landscape-scale occupancy patterns of bobcats, cougars, and coyotes in the Oregon Cascades between 2015 and 2017 (Brent Barry and Katie Moriarty, USDA Pacific Northwest Research Station, unpublished data). These patterns offer initial insights into potential predation risk, but require more recent and finer spatial evaluations of predator occupancy, abundance, and habitat associations to understand potential impacts to a fisher reintroduction in Oregon. Pre-translocation results will also support the selection of fisher release sites within the proposed release area to reduce acute predation risk of founding individuals and their offspring. Results from these pre-translocation surveys will provide opportunities for partners to evaluate potential risks and make empirically-informed, adaptive decisions.

We recommend conducting surveys using a combination of remotely-triggered cameras and/or detection-dogs at least once prior to the translocation of fishers and at annual intervals for at least four years following translocation. We recommend surveys are designed to maximize the detection of cougars, bobcats, coyotes, grey foxes, and fishers. Surveys can be distributed using a

grid across the proposed reintroduction area (e.g., Ruprecht et al. 2021). Survey effort should be distributed across randomly selected grid cells and include a representative range of elevations, vegetation communities, and terrain conditions in the proposed reintroduction area. We suggest survey design be based on the designs described by Thompson et al. (2012) and Ruprecht et al. (2021).

Surveys by detection-dog teams could be complemented with remotely-triggered camera surveys conducted in alternate years to collect detection/non-detection data of fisher predators and competitors. Scat collection using detection-dog teams and genetic identification of species and individuals can be used to estimate species density while correcting for imperfect detection (Thompson et al. 2012, Royle et al. 2014). These methods, however, are costly to implement annually and recent research has demonstrated the combination of individually-identified genetic data collected by detection-dog teams and detection/non-detection (e.g., camera data) can be analyzed jointly to estimate species abundance (Blanc et al. 2014). Density estimates of cougars, bobcats, coyotes, and grey foxes are limited and variable throughout the historical distribution of fishers in the western states (Table 2). Available estimates, however, provide an opportunity for comparison between the proposed reintroduction area and areas occupied by extant fisher populations. Additional survey details can be found in ***Error! Reference source not found.***.

Efforts to evaluate competition and predation in areas where native populations persist and reintroduction is planned can be costly, both in funds and personnel capacity. Practitioners need to candidly understand the resources available to dedicate to such an expensive effort and communicate with team members or upper-level management on prioritization from a holistic scale, as well as comfortability of executed efforts and limitations of their results when making reintroduction decisions. Funding opportunities are limited by scope and amount, and are

difficult to obtain. While density estimates via non-invasive survey methods and a genetic-based detection/non-detection survey design using hair snares (e.g., Green et al. 2018), scat-detection dog teams (e.g., Thompson et al. 2012), or a combination of techniques in an integrated approach (e.g., Ruprecht et al. 2021) provide more specific intel on the mesocarnivore community to guide reintroduction decisions, oftentimes occupancy surveys using cameras are more feasible to conduct despite low detectability and achievability of robust analyses from photo-data. Additionally, photo-processing rate is based on directives within the team versus the time to process genetic samples is based on the queue of the entity contracted. Depending on the results of a more cost-effective survey (i.e., low detections of primary predators; Appendix B), enough comfortability may be achieved to move forward with reintroduction planning. A cost-benefit analysis should be conducted when weighing the pros and cons of different methodologies, the limitations of the dataset and the reasonable inferences of analyses, and the associated temporal constraints in providing contemporary information to guide decision-making. See additional details in *Appendix D – Costs, Benefits, and Efficacy of Different Sampling Methods*.

Wildfire

Forests occupied by fishers, particularly in northwestern California and the Sierra Nevada, are expected to become more vulnerable to wildfires over the next century (Safford and Van de Water 2014, Davis et al. 2017). Empirical evidence indicates wildfire leads to declines in fisher occupancy (Scheller et al. 2011, Thompson et al. 2011, Garner 2013, Sweitzer et al. 2016, Barry 2018), declines in fisher density regardless of burn severity (Green et al. 2022*b*), and alters interspecific interactions between fishers and other carnivores (Green et al. 2022*b*). Given the threat of wildfire to fishers, particularly to small populations, accounting for the current and future fire environment is an important consideration for reintroduction. The susceptibility of the

Oregon forests to wildfire presents a potential risk to a reintroduced fisher population, though fuels management programs, particularly through those that restore historical fuel loads and fire return intervals (e.g., broadcast burning), can help to minimize risks to fisher populations when these management actions are considerate of the need for habitat connectivity. Monitoring of the response of the reintroduced population to any future fire events will be essential to evaluating the impacts of wildfire on reintroduction success.

Rodenticide exposure

Rodenticide exposure and poisoning is an emerging conservation concern for non-target wildlife (Erickson and Urban 2004, Fournier-Chambrillon et al. 2004, Coeurdassier et al. 2014). Recent research has demonstrated that anticoagulant rodenticides are a direct and indirect source of fisher mortality in California (Gabriel et al. 2012, 2015), prompting conservation concerns for fishers in the Pacific states (U.S. Fish and Wildlife Service 2016, 2020*a*). Empirical data on the use of rodenticides and exposure rates to wildlife in Oregon is limited and attributed to a lack of systematic survey effort (but see Wiens et al. 2019). Exposure risk is expected to be more prevalent in southwestern Oregon than in the central and northern Oregon Cascades, likely attributed to favorable growing conditions for cannabis where rodenticides are frequently used (e.g., Wengert et al. 2021). Model results for central and northern portions of the Oregon Cascades are forthcoming (Greta Wengert, Integral Ecology Research Center, personal communication) and will require further consideration in evaluating and monitoring a fisher reintroduction.

Risk for Other Species and the Ecosystem

Planning the reintroduction and recovery of imperiled mammalian carnivores and their functional role in an ecosystem requires an understanding of the destination community and the

preservation of co-occurring species, particularly those of conservation concern (Seddon et al. 2007, IUCN Species Survival Commission 2013). Forty-three terrestrial vertebrates of conservation concern identified by the U.S. or Oregon Endangered Species Act (Oregon Legislature 1995), the federal Interagency Special Status/Sensitive Species Program (U.S. Forest Service and Bureau of Land Management 2025), or the Oregon State Wildlife Action Plan (Oregon Department of Fish and Wildlife 2025) occupy at least a portion of the historical range of fisher in Oregon. The reintroduction of fishers could potentially impose interference and exploitative competitive pressure on co-occurring species. Recent research into exploitative competition between fishers and intra-guild, co-occurring species also offer some insights into the implications of a reintroduction of fishers in Oregon (e.g., Wengert et al. 2013, Gabriel et al. 2015, Halsey et al. 2015, Green et al. 2018, Parsons et al. 2019). Previous fisher translocations in Washington and California evaluated the potential risk to other species and concluded the potential costs of fisher reintroduction were outweighed by the potential conservation benefits (Lewis and Hayes 2004, Callas and Figura 2008). Matthews et al. (2021) reviewed the available literature for species of conservation concern in Oregon and arrived at similar conclusions. A more complete understanding of the effects of fisher reintroduction will require monitoring of co-occurring species of conservation concern prior to and during the reintroduction and establishment of a reintroduced fisher population.

Fishers and martens (American marten [*Martes americana*] and Pacific marten [*M. caurina*]) are sympatric over much of their North American distributions (Gibilisco 1994). The Humboldt marten (*M. caurina humboldtensis*), a low-elevation coastal subspecies, is listed a federally threatened (U.S. Fish and Wildlife Service 2020*c*). Humboldt martens in Oregon occur in the central coast (generally in Douglas, Lane, and Lincoln counties) and in the southern coast

(generally in Coos and Curry counties) (Moriarty et al. 2016, Linnell et al. 2018, U.S. Fish and Wildlife Service 2020*c*). In areas of overlap, fishers and Humboldt martens may partition along dimensions of diet and microsite habitat selection, but these axes of co-occurrence require further investigation. A reintroduction of fishers to the central Oregon Cascades will not spatially overlap with extant Humboldt marten.

Pacific martens occupy portions of the Oregon Cascades between 1,252 and 2,237 m (Mcfadden-Hiller and Hiller 2015). Aubry and Raley (2006) and Moriarty et al. (2019) reported fishers rarely occupied areas >1,550 m, suggesting limited overlap in elevation in the Cascades. It is suspected that deep, soft snow increases energetic demands on fishers, whereas sympatric marten populations may be limited by the presence of fishers (Krohn et al. 1995, 1997). Surveys are currently underway in the Gifford Pinchot National Forest to evaluate the co-occurrence of reintroduced fishers and Pacific martens in the Southern Washington Cascades Fisher Recovery Area (E. Anderson, Oregon State University, unpublished data).

Anecdotal accounts and more recent empirical surveys of Sierra Nevada red foxes (*Vulpes vulpes necator*) indicate Sierra Nevada red foxes and fishers spatially partition along an elevation gradient, resulting in limited interspecies competition (Bailey 1936, Verts and Carraway 1998, Mcfadden-Hiller and Hiller 2015, Quinn et al. 2018). Sierra Nevada red foxes are associated with subalpine woodlands and meadows at elevations >1,300 m along the Cascade crest (Quinn et al. 2018). Alternatively, fishers in the southern Oregon Cascades occupy late-seral forests at elevations between 610 and 1,525 m (Aubry and Raley 2006, Moriarty et al. 2019) and are expected to occupy similar elevations in the proposed reintroduction area.

Risk for the Human Community

The reintroduction of a species can have direct and indirect negative effects on human interests (IUCN Species Survival Commission 2013). Reintroductions can have direct effects on people and livelihoods and indirect ecological effects that could threaten food supplies or ecosystem services (IUCN Species Survival Commission 2013). Fishers are a late-seral-forest obligate species with an uncertain regulatory future (U.S. Fish and Wildlife Service 2023), and thus raise concerns for the management of Oregon's nearly 9,600 ha of timberlands (Oregon Forest Resources Institute 2020). Approximately 34% of Oregon's forestlands are in private ownership and the remainder are under federal, state, or Tribal ownership (Oregon Forest Resources Institute 2020). Although a majority of forestlands are in public or tribal ownership, nearly 78% of the 3.8 billion board feet of timber harvested in Oregon annually comes from private timberlands (Oregon Forest Resources Institute 2020).

Recognizing the concerns of landowners and the conservation opportunity, the U.S. Fish and Wildlife Service completed a Template Candidate Conservation Agreement with Assurances (CCAA) for the fisher in western Oregon (U.S. Fish and Wildlife Service 2017). The CCAA, and recently developed Conservation Benefit Agreements, are voluntary agreements whereby landowners agree to conservation measures that remove or reduce threats to a species that may become listed under the Endangered Species Act before the species is listed as threatened or endangered. In return, enrolled landowners receive regulatory assurances against future obligations for a listed species. One goal of the program is to encourage the public to voluntarily develop and implement conservation actions for at-risk species. Species conservation under a CCAA may contribute to not needing to list the species, listing it as threatened instead of

endangered, or accelerating species recovery if it is listed. The programmatic fisher CCAA was developed to achieve the following goals:

- Expand the understanding of fisher distribution, densities, and responses to forest management activities;
- Promote conservation measures that reduce or remove threats to fisher in Oregon;
- Provide a program of voluntary proactive recovery efforts that deliver conservation benefits intended to meet the U.S. Fish and Wildlife Service CCAA standard;
- Provide enrolled landowners assurances that they will not be held responsible for additional conservation measures or incur additional future regulatory obligations if the fisher becomes listed under the Endangered Species Act, provided the CCAA is being fully and completely implemented.

Seven private landowners and the Oregon Department of Forestry, representing over 700,000 ha of timberlands, have enrolled in the western Oregon fisher template CCAA, ensuring regulatory certainty for these landowners should the fisher become listed. The CCAA enrollees and other forestland partners are key stakeholders in the reintroduction of fishers to Oregon.

Level 3 – Technical and Logistical Suitability

Establishing a reintroduced population of fishers in the southern Oregon Cascades and improving the connectivity of fisher populations at regional scales will involve a series of technical and logistical considerations (Pérez et al. 2012, IUCN Species Survival Commission 2013). These considerations include:

1. Maximizing the likelihood of establishing a viable population
2. Establishing clear reintroduction project goals
3. Identifying economic and human resources
4. Engaging partner support

Establish Viable Population

Environmental conditions at release areas contribute significantly to the likelihood of establishing self-sustaining, reintroduced populations (Griffith et al. 1989, Seddon et al. 2014).

Release areas need to contain resources for all seasonal habitat requirements at all life stages for released individuals and their progeny, be large enough to provide conservation benefit, and have adequate connectivity to allow for population expansion (IUCN Species Survival Commission 2013). Matthews et al. (2021) utilized habitat ensemble modeling methods to identify portions of the historical distribution of fishers in Oregon that could support a reintroduction. Matthews et al. (2021) identified a 1,683 km^2 area in the central Oregon Cascades as a potential reintroduction area that could support a reintroduced fisher population based on an evaluation of current forest conditions, connectivity potential based on proximity to extant fisher populations, and potential risk factors. Following a series of partner meetings, a site visit to portions of the southern Willamette National Forest in 2022, and wildfire activity in 2022 and 2025, partners revisited the methods of Matthews et al. (2021) to select an alternative reintroduction area in the southern Oregon Cascades east of Interstate 5 between Roseburg and Grants Pass and west of the area currently occupied by the Southern Oregon Cascades fisher population (Barry et al. 2021).

Release Area Selection

We began our process of release area selection with the ensemble model of fisher habitat suitability developed by Matthews et al. (2021) that included habitat models developed by Allen (1983), Fitzgerald et al. (2013), Aubry et al. (2017), and Barry et al. (2021). We rescaled each individual model to a 90 m resolution and calculated the mean habitat suitability value for each 90 m grid cell. We then calculated the mean habitat suitability value within polygons estimated to be occupied by fishers in Oregon from Barry (2021; Figure 2). We then masked all ensemble model grid cells with mean habitat suitability values less than the mean habitat suitability value of 0.40 estimated within occupied portions of Oregon. We masked all ensemble model grid cells that were subject to >25% loss in canopy cover due to wildfire between 2012 and 2023 based on

USDA Rapid Assessment of Vegetation Condition after Wildfire (RAVG) data. We also masked all ensemble model grid cells where the suitability of a large wildfire by the year 2100 under climate scenario RCP8.5 was high (Davis et al. 2017).

We generated potential release area polygons by dissolving neighboring individual ensemble model grid cells to polygons and dropping any polygons less than 480 km^2, an area estimated to accommodate between 20 and 50 female fishers based on average female home range sizes of 24.6 km^2 (Aubry and Raley 2006, Moriarty et al. 2019) and 9.6 km^2 (Lofroth et al. 2010), respectively. We also filled any polygon holes less than 20 km^2 and smoothed the final polygon using Gaussian kernel regression (Figure 2).

We further reduced the size of the potential release area by only including polygons within the potential reintroduction area identified by Oregon Department of Fish and Wildlife and U.S. Fish and Wildlife Service partners, an area bounded on the west and south by Interstate 5, on the east by Oregon Highways 62 and 230, and on the north by Oregon Highway 138 (Figure 2). We then buffered these roadways and the population area polygons of Barry (2018) by 5.62 km based on the diameter of the mean female fisher home ranges sizes reported by Aubry and Raley (2006) and Moriarty et al. (2019) and clipped any areas within those buffers (Figure 3). We further reduced the potential release area by buffering each paved road by 2.81 and 5.62 km, estimated as one and two female home range diameters based on mean female fisher home ranges sizes reported by Aubry and Raley (2006) and Moriarty et al. (2019). We clipped any potential release areas within the 2.81 and 5.62 km road buffers (Figure 4 and Figure 5). Land managers represented in the potential release areas include the U.S. Forest Service, the U.S. Bureau of Land Management, the Cow Creek Band of Umpqua Tribe of Indians, and

private forestland partners enrolled in a Candidate Conservation Agreement with Assurances with the U.S. Fish and Wildlife Service (Figure 6).

The monitoring of reintroduced fishers and their progeny will be required to evaluate population establishment, population persistence, and meta-population connectivity (Armstrong and Seddon 2008, Happe et al. 2019, Green et al. 2022*a*, Lewis et al. 2022). Releases and monitoring will be most efficiently conducted in areas with sufficient road access. Based on the Oregon Bureau of Land Management Ground Transportation Road Network data (Figure 7), we concluded the identified release areas and surrounding landscapes included sufficient road access for releases and monitoring.

For visualization purposes, we overlayed a 4x4 km (16 km^2) grid over the potential release area polygons (Figure 8). We assumed a 16 km^2 grid cell would approximate a mean female fisher home range size based on mean female home range sizes presented by Loforth et al (2010), Aubry and Raley (2006), and Moriarty et al. (2019). We anticipate the mean female home range size of female fishers translocated from the native portion of the Northern California Southern Oregon population to a release area in the southern Oregon Cascades will be larger than the mean home range size in northern California and southern Oregon and smaller than those reported in the southern Oregon Cascades (e.g., Martin et al. 2025*b*).

The establishment and persistence of a reintroduced population of fishers will require areas larger than the proposed release areas. The proposed release areas are centrally located within larger patches of forested habitat that should provide connectivity among release areas, previously reintroduced fishers in the extant Southern Oregon Cascades population, and, potentially within several generations post-release, occupied and unoccupied habitat within the Northern California Southern Oregon population west of Interstate 5 (Figure 9). Partners selected

this area because it was determined to be the best placement to achieve a self-sustaining, interacting populations of fisher well-distributed throughout available habitat due to historical occupancy, proximity to contemporary source populations, and potential to support future connectivity of fisher populations within the Cascade Range.

Release Site Selection

Release site selection will be adaptive as pre-translocation survey data are analyzed and results are summarized. From noninvasive methods (see Appendix B), quantification of the carnivore and small mammal communities in areas where native fisher populations persist and where the translocated fisher population is proposed (see Release Area Selection) is ongoing. Acknowledging that translocated individuals are most vulnerable when initially released, these analyses will guide recommendations on where specifically within the release area fishers should be released. Release site selection will be based on comparatively lower estimates of relative carnivore occupancy (i.e., felid species prioritized over canid species), and higher estimates of relative small mammal occupancy and diversity. Estimates of carnivore occupancy and small mammal occupancy and diversity from areas where fishers will be sourced will guide thresholds of release sites within the release area that have tolerable levels of predators/competitors and sufficient prey. Based on other reintroduction efforts, it is recommended that fewer release sites are selected to encourage home range establishment that overlaps and or is in close proximity to one another.

Population Viability Simulations

The population viability analysis conducted by Padilla (2025) and described in the Consequence for source population section above was further leveraged to model the establishment and persistence of a reintroduced population of fishers to the southern Oregon

Cascades. Padilla (2025) used a modified Euler-Lotka female-based model to estimate the maximum intrinsic growth rate of the reintroduced population. The probability of failure was defined as a population of fewer than 20 female fishers 15 years post release and was evaluated for a variety of scenarios with varying population vital rates, release number and timing, and wildfire risk. Population vital rates and estimates of variance were selected from the literature focused on fisher populations in California, Oregon, and Washington (Table 1).

In the first scenario, Padilla (2025) varied the number of individual female fishers released in year one, including: 10, 30, 50, 70, and 90 fishers. Assuming high quality habitat where rates of competition and predation are moderate to low and prey resources are abundant, they concluded a fisher reintroduction was likely to be successful when more than 10 individuals were released. Based on these analyses, other reintroduction efforts, and on the available resources, we recommend reintroducing a minimum of 40 female fishers and 26 male fishers.

In the second scenario, Padilla (2025) varied adult female survival rates with values of mean survival of 0.5, 0.6, 0.7, 0.8, and 0.9 with a fixed standard deviation of 0.1 for all mean values. Unsurprisingly, the effects of adult survival on reintroduction success were significant with probability of failure above 90% when mean adult survival was 0.6 or below. With the median tested survival of 0.7, 13% of simulations were considered a failure after year 15. As expected, mean adult survival at 0.8 and above yielded effectively 0% probability of failure. Adult female survival is generally the most influential factor for population growth in most long-lived mammal species (e.g., Gaillard et al. 2000). Estimates of true and apparent survival based on GPS telemetry and capture-recapture models in the extant Northern California Southern Oregon populations range from approximately 0.68 to 0.85 (B. Padilla and K. Titus, Oregon Department of Fish and Wildlife, and E. Anderson, Oregon State University, unpublished data,

Green et al. 2022*b*). This suggests that a mean adult female survival of 0.7, while low, is within the range of values experienced by nearby extant populations. Adult female survival should be closely monitored following reintroduction to ensure survival does not drop to or below 0.7.

In the third scenario, Padilla (2025) varied the fecundity and juvenile survival rates of the reintroduced population. Fecundity was defined as the mean number of female kits per female of breeding age per year assuming equal sex ratios. Juvenile survival was defined as the survival probability from birth through the first winter, thus, the product of juvenile survival and fecundity is recruitment into the population. Padilla (2025) tested three values of fecundity: low = 0.75, baseline = 1.25, high = 1.5, and three levels of juvenile survival: low = 0.2, mean = 0.3, high = 0.45. Per capita recruitment to adulthood in fishers is a function of litter size and juvenile survival. Model simulations varying these values showed that low litter size was mitigated when juvenile survival was high or vice versa. For example, with mean fecundity of 0.75 and high juvenile survival of 0.45 there was only a 5.3% probability of failure. By contrast, a mean fecundity of 0.75 with the baseline juvenile survival of 0.3 results in a 77.3% probability of failure in our model simulations. Litter size and juvenile survival are difficult to monitor but should be prioritized in post release monitoring because of the significant impact variation in recruitment can have on population trajectories. Research and monitoring efforts should also identify environmental, ecological, and biological factors that influence fecundity and juvenile survival to enable more proactive decision making. These vital rates, though likely correlated in practice, may be responding to different proximate stressors. For example, fecundity could be influenced by prey availability through female body condition (e.g., Sand 1996) while juvenile survival is often driven by winter severity (e.g., Norton et al. 2021).

In the fourth scenario, Padilla (2025) explored the combined effects of varying wildfire probability and the number of female fishers released on reintroduction success. Annual wildfire probabilities ranged were: low = 0.05, med = 0.10, and high = 0.15, corresponding to 20-, 10-, and 7-year fire return intervals, respectively. For each fire, the percentage of the fisher population that died due to fire was drawn from a uniform distribution between 0.2 and 0.75. The stochastic nature of wildfire occurrence makes it difficult to draw definitive conclusions, however, clear patterns emerged from this simulation. First, when fewer individuals were released, the risk that fire resulted in driving the population toward failure was higher across all fire probabilities. Second, fires that occurred earlier in the reintroduction timeline were more likely to result in reintroduction failure, particularly if multiple events occurred in a relatively short time scale.

In the fifth scenario, Padilla (2025) staggered the timing of the release of fishers with baseline population vital rates and variable adult survival shortly after release. During the 'Establishment Phase' a reintroduced population can experience significant variability in vital rates as individuals disperse, home ranges are established, and interspecific dynamics established. Padilla (2025) conducted an analysis where there was a stochastic reduction in adult female survival probability during the first 15 years post release using the baseline mean vital rates and a 0.02 annual fire probability. Padilla (2025) also tested three different release scenarios: 40 individuals released all in year 1, 40 individuals released in years 1 (n=14), 2 (n=13), and 3 (n=13), and 40 individuals released in years 1 (n=20), 5 (n=10), and 10 (n=10). As with other wildfire scenarios, stochastic fire events had a significant influence on reintroduction success or failure, with the timing of fire relative to release as an important factor. With baseline mean vital rates, all release scenarios had failure rates below 2%.

In the sixth scenario, Padilla (2025) staggered the timing of the release of fishers and included lower vital rates and more variable adult survival shortly after release. Here, values were 0.75 for adult survival and 0.25 for juvenile survival while fecundity remained at a baseline of 1.25 but with a slightly higher standard deviation of 0.3 (vs baseline of 0.2). The probability of failure was substantially higher with only slightly lower mean vital rates in all release timing scenarios. In all three release scenarios probability of failure was nearly 20% or greater. Low mean vital rates coupled with reduced survival during the establishment phase, possibly due to predation, roadkill, or other stochastic variation, had a clear effect on simulated populations. To maximize potential for interactions of translocated individuals during breeding seasons and minimize impacts of potential inbreeding, we recommend release of cohorts in consecutive years.

The population viability analysis of a reintroduced population of fishers to the souther Oregon Cascades conducted by Padilla (2025) incorporated unpublished vital rates from extant fisher populations in southwest Oregon. Thus, Padilla (2025) necessarily assumed that vital rates of a reintroduced population would be within the range of variability observed in extant populations in southwest Oregon. As fishers are translocated, released, and monitored, we will rerun the population viability analysis to update modeled vital rates from the reintroduced population. Key vital rates of adult survival and recruitment should be closely monitored through the establishment phase to ensure that reintroduced populations are on a trajectory toward persistence. With incorporation of vital rates into the population viability analysis, we will be able to evaluate our goal of reducing the potential for decline and/or population loss to a less than 20% risk of extinction over the next 50 years.

Initial Capture

The capture and handling of fishers will require the approval and permitting of several agencies. We recommend the translocation of fishers follow guidelines described in the California Department of Fish and Wildlife (2017) Policy and Procedures for Conservation Translocations of Animals and Plants. An Oregon Department of Fish and Wildlife Scientific Take Permit and California Department of Fish and Wildlife Scientific Collecting Permit may be required if those conducting the capture efforts are not agency-personnel. The importation and release of fishers into Oregon would be administered under the authority of the Oregon Department of Fish and Wildlife and the Oregon Department of Agriculture (Oregon Administrative Rules 603-011-0255 and 603-011-0382). Capture and handling protocols will follow recommendations of the Animal Care and Use Committee of the American Society of Mammalogists (Sikes and The Animal Care and Use Committee of the American Society of Mammalogists 2016).

Simulation modeling by Padilla (2025) presented above and previous research indicate that partners should plan on reintroducing a minimum of 40 female fishers from the Northern California Southern Oregon fisher population to the central Oregon Cascades (e.g., Lewis et al. 2012, Facka and Powell 2021). Matthews et al. (2021) recommended a translocation of fishers to the Oregon Cascades source fishers from the Northern California Southern Oregon population. This conclusion was based on theoretical and empirical evidence suggesting the proximity of source populations and reintroduction areas have a strong influence on reintroduction outcomes (e.g., Miller et al. 1999, Lewis et al. 2012, Manlick et al. 2017, 2019). Generally, the proximity of source and release areas can serve as a proxy for genetic relatedness between the source population and the population that historically occupied the release area (Lewis et al. 2012). The

decision to reintroduce a minimum of 40 female fishers is also intended to minimize genetic-related concerns (e.g., inbreeding depression; Weeks et al. 2011) and lack of evidence of inbreeding from the reintroduced population of 40 fishers to the northern Sierra Nevada (Facka et al. 2016, Powell et al. 2019).

The translocation of each fisher will involve three stages, 1) initial capture, field health assessment, and processing, 2) veterinary health assessment and disease screening, and 3) release. We recommend capturing fishers from across northern California and southern Oregon, sourcing approximately five individuals from each capture site and capture sites being a minimum of 8.8 km from other capture sites. We selected 8.8 km based on the mean diameter of male home range estimates (Aubry and Raley 2006, Lofroth et al. 2010, Moriarty et al. 2019) and the documented movement patterns of reintroduced fishers (Facka and Powell 2021), assuming male fisher home ranges overlap several female fisher home ranges. By implementing a limit of individuals sourced from each capture site and a minimum distance between capture sites, we intend to reduce likelihood of translocating related individuals and thereby increase genetic diversity of the reintroduced population.

We recommend the initial capture and processing be used to identify candidate fishers for translocation based on sex, age estimate, and general physical condition. Priority should be given to translocating sexually-mature, young-adult female fishers and large, young-adult male fishers in good body condition. Younger adult fishers have the greatest reproductive potential and large males tend to be more successful breeders (Matthews et al. 2013*b*, Powell et al. 2019). Candidate fishers will be transported to a captive facility and a health assessment will be conducted by a wildlife veterinarian following recommendations in Callas and Figura (2008). Already-permitted options in Oregon can be found here (with additional options permitted on a case-by-case basis

by agency veterinarians), but will need to be consulted on their capacity and interest, and an agreement will likely need to be established. Importantly, prior to release, biological samples should be collected for all individuals intended for translocation for unique identification via later-genotyping and for long-term post-release monitoring purposes.

We suggest fishers be opportunistically captured throughout the Northern California Southern Oregon fisher population and released to the southern Oregon Cascades for two consecutive years between September and December. Success in year one and support for year two will be measured through the post-release monitoring of the founding individuals, quantifying their survival and reproduction rates, comparing those rates to other populations, and using those rates as a baseline to re-run the population viability analyses presented in Padilla (2025). Based on Padilla (2025), we recommend that 40 individuals are translocated over the two consecutive years, with 50-75% being female. Methods for live-capture should follow those described in Appendix C, in coordination with agency veterinary guidance.

Release

We recommend release sites be selected within the reintroduction area based on 1) areas identified to be suitable for fishers (Matthews et al. 2021), 2) the results of pre-release surveys for distribution, occupancy, and densities of fisher predators, competitors, and prey and 3) on-site evaluations by species experts and partners. We suggest fishers be released from three to five release sites as the number of release sites has been positively correlated to simulated and empirical reintroduction outcomes (Lewis et al. 2012, Powell et al. 2012). Release sites should be spaced within 8.8 km from other release sites and from known and established home ranges of previously reintroduced fishers. These distances are based on the mean diameters of female and male home range estimates, respectively, (Aubry and Raley 2006, Lofroth et al. 2010, Moriarty

et al. 2019) and the documented movement patterns of reintroduced fishers (Facka and Powell 2021). There is no empirical or modeled evidence to suggest the type of release (hard versus soft) influences reintroduction outcomes (Lewis et al. 2012, Powell et al. 2012), but results from an American marten reintroduction in Wisconsin indicated soft-released individuals were more likely to experience non-life-threatening injuries (e.g., tooth, nail, or digit fractures) than hard-released individuals (Woodford et al. 2013). Thus, we suggest fishers be released as soon as they are deemed suitable for translocation based on demographic and health criteria.

We recommend fishers be released to the reintroduction area between September and December. Facka et al. (2016) investigated the reproductive outcomes of female fishers reintroduced to the Olympic Peninsula and the northern Sierra Nevada. The authors reported female fishers released in November and December had a 66% higher rate of denning their first den season post-release than those fishers released between January and March. The authors also found releasing fishers by December had a long-term positive influence on mean population size and on quasi-extinction thresholds.

Post-Release Monitoring

Sufficient monitoring of a reintroduced population is required to evaluate population establishment, population persistence, and meta-population connectivity (Armstrong and Seddon 2008). We recommend the monitoring of the reintroduced fisher population in central Oregon address a sequence of objectives: 1) quantifying the survival of the release generation; 2) quantifying reproduction by the release generation; 3) evaluating the establishment of the reintroduced population; 4) evaluating the persistence of the reintroduced population; 5) evaluating any threats to these population parameters; and 6) evaluating measures of population connectivity (Seddon 1999). The survival and reproduction of the release generation can be

monitored using telemetry. The establishment and persistence of the reintroduced population can be further evaluated using non-invasive survey methods. These data will enable agencies and partners to conduct population viability analyses (*sensu* Padilla 2025) with empirical population vial rates, evaluate the trajectory of the reintroduced population, and make adaptive management decisions.

Additionally, reintroductions are increasingly being used to answer *a priori* questions that address uncertainty and inform recovery and management decisions (Taylor et al. 2017). Significant information gaps remain in our knowledge of fisher demographics, habitat selection, diet, and movement patterns in the forests of the Oregon Cascades (Lofroth et al. 2010, U.S. Fish and Wildlife Service 2016). Landscape connectivity for species of conservation concern is also a central topic for species recovery and conservation planning (Hilty et al. 2020, Perzanowski et al. 2020). A reintroduction of fishers to the southern Oregon Cascades would provide opportunities to further investigate and refine connectivity modeling and conservation outcomes (Oregon Department of Fish and Wildlife 2016, Spencer et al. 2019, U.S. Fish and Wildlife Service 2020*a*).

Translocated Individuals

Accordingly, we recommend monitoring translocated fishers using GPS and VHF-radio telemetry to document post-release movement, home-range establishment, habitat selection, survival, cause-specific mortality, and reproduction (*sensu* Woodford et al. 2013). A reintroduction should not be executed if funding for post-release monitoring through telemetry is not secured. A mixture of VHF and GPS collars could be deployed to balance funding constraints, and prioritization based on sex (to quantify reproduction) could be considered. At minimum, a subset of the founding cohort should be radio-collared. Decisions related to the use

GPS or VHF technology should be critically evaluated based on objectives and acknowledging the benefits and costs of each technology. For example, VHF technology enables longer deployments between recapturing individual fishers, however, the technology requires additional personnel time and costs per location. Alternatively, GPS technology provides a larger number of relocations at a lower per-location cost but require more frequent recapture and recollaring (e.g., Moriarty et al. 2019). Additional questions (e.g., fine scale movement patterns and habitat selection) can be addressed using short-term deployment of GPS technology, but at added costs for purchasing transmitters, transmitter refurbishment, and more frequent recapture efforts to re-collar individuals. Radio-collar battery-life is dependent on the scheduled fix interval, and with the priority of maximizing monitoring longevity of founding individuals, we would recommend a fix rate of 1-4x/day with a VHF-beacon active during typical workday hours. While radio-collars imply staff resources allocated to radio-telemetry, these tools provide worthwhile datasets to evaluate reintroduction population establishment.

Telemetry tracking can be done primarily from vehicles and on foot using a handheld receiver and antenna to relocate active and inactive fishers. Aerial telemetry could be conducted on an as-needed basis to relocate translocated fishers that go missing for more than two weeks. Locations of fishers can be obtained using the loudest-signal method and by hiking to the source of the signal (Springer 1979). We suggest translocated fishers be monitored using telemetry for three years post-release. Three years is the minimum data required to estimate trends in population vital rates. As such, we recommend fewer fix rates per day to extend the battery life as much as possible in anticipation of the potential that only the founding individuals will be radio-marked. If funding allows, we recommend an additional radio-marking effort once the

translocation phase is completed. We suggest additional post-release monitoring of translocated fishers and their progeny be completed using the non-invasive survey methods.

Population Monitoring

Understanding the ecological mechanisms influencing population persistence and outcomes of reintroduction efforts, is imperative to achieve reintroduction objectives. Post-release population monitoring via non-invasive methods should be conducted annually after the radio-marking effort has concluded, and then at regular intervals to evaluate population establishment and persistence (Armstrong and Seddon 2008, David et al. 2013, Facka 2017). Reintroduced populations are established when population processes (e.g., reproduction and recruitment) support a stable or growing population independent of augmentation from additional founding individuals (Bright and Smithson 2001, Armstrong and Seddon 2008, Facka 2017). Reintroduced populations are persistent when they are of sufficient size to be resilient to environmental and demographic stochasticity (Bright and Smithson 2001, Armstrong and Seddon 2008, Facka 2017).

Green et al. (2022*a*) documented the establishment and continued annual growth of a reintroduced population of fishers in the northern Sierra Nevada for six years following the final release of founding individuals, despite a population decline during the two years following the final release of fishers. Population monitoring can be conducted using non-invasive techniques developed and refined for long-term population monitoring of fishers and co-occurring species (e.g., Truex et al. 2012, Green et al. 2018) as well as incorporating telemetry data on fishers for the period of telemetry monitoring (e.g., Ruprecht et al. 2021).

We recommend conducting population monitoring using non-invasive survey methods and a genetic-based detection/non-detection survey design using hair snares (e.g., Green et al.

2018), scat-detection dog teams (e.g., Thompson et al. 2012), or a combination of techniques in an integrated approach (e.g., Ruprecht et al. 2021). Non-invasive hair snare surveys require comparatively-reduced staff resources and reasonable funding requirements to genotype samples to identify individuals while contracting scat-detection dog teams is more effective but also more costly. We suggest a non-invasive survey area be determined by the post-release movements of the founding individuals based on telemetry data and include as many established home ranges as possible (*sensu* Facka and Powell 2021). Estimates of annual population size, density, apparent survival, apparent recruitment, and population growth rate could be calculated using spatial capture-recapture analytical methods (Royle and Young 2008, Royle et al. 2014). Detection histories of individual fishers can be supplemented with telemetry data to refine movement parameter estimates and population demographic variables (Ruprecht et al. 2021, Green et al. 2022*a*, Padilla 2025). Non-invasive survey methods using a genetic-based detection/non-detection survey design should be implemented after most of the radio-collars exceed their estimated battery life for a minimum of 3 consecutive years.

As the reintroduced population expands and time progresses where radio-marked individuals from the founding population are no longer on the landscape and a baseline trend in population density estimation from annual non-invasive survey methods using a genetic-based detection/non-detection survey design has been established, alternative strategies to continue monitoring the reintroduced population while balancing resource limitations could be considered. Non-invasive survey methods using a genetic-based detection/non-detection survey design could be employed at regular intervals that are reasonable given resource capacities. A reliable low-cost method to monitor the occupancy and distribution of the reintroduced population is through implementation of a camera trapping grid. A camera trapping grid could

supplement non-invasive survey methods using a genetic-based detection/non-detection survey in off-years, or could be exclusively utilized if managers are comfortable and confident in the establishment of the reintroduced population. Considerations should be weighed on the trade-off between a low-maintenance and cost strategy with sacrificing density or abundance estimate conclusions, and after how many generations after reintroduction each method is appropriate. Further, recognition of dataset limitations (i.e., low detection rates of mesocarnivores) and photo-processing capacities should be realized. See Appendix B and D for more details.

Reintroduction Goals

Fishers currently occupy approximately 15% of their former distribution in Oregon between the west slope of the southern Oregon Cascades and the Pacific Ocean, with contemporary locations only occurring in the southern-most portion of the state (Aubry and Lewis 2003, Barry et al. 2021). There is a need to reduce the extinction risk and restore resiliency, redundancy, and representation of fishers in Oregon (U.S. Fish and Wildlife Service 2022, Oregon Department of Fish and Wildlife 2025). Under a no-action alternative, there is a moderately-high risk of decline or extirpation of one or several populations of fisher over the near-term. Our goals in conducting a reintroduction of fishers to the southern Oregon Cascades are 1) to reduce the potential for decline and/or population loss to less than a 20% risk of extinction over the next 50 years and 2) improve broader connectivity for fisher populations in the Pacific Northwest. Accordingly, to achieve these goals our objectives are to establish a persistent population of reintroduced fishers in the southern Oregon Cascades. We will evaluate establishment and persistence with empirical estimates of survival and reproduction of translocated fishers and their progeny and the expansion of the population beyond the reintroduction area.

Population establishment, persistence, and extinction risk for the reintroduced population in the southern Oregon Cascades will be evaluated using demographic parameters collected while monitoring the reintroduced population and updating those parameters in the population viability analysis methods described in Padilla (2025). Survival and reproduction parameters can be from radio-telemetry (known-fate models and or apparent estimates) or from non-invasive genetic-based surveys (SCR and Cormack-Jolly-Seber models). Meta-population connectivity will be evaluated while monitoring the reintroduced population and in close coordination with the Oregon Department of Fish and Wildlife (Oregon Department of Fish and Wildlife 2023). Through non-invasive genetic sampling after establishment and several generations, proportion of haplotype presence and admixed individuals, as well as percent of the sampled-population that are new individuals (i.e., assumed a portion of which immigrated in), could be used to infer integration between established and reintroduced populations.

Agencies and partners collaborating on the reintroduction logistics have identified a range of demographic parameter values from which discussions will be warranted to identify limiting factors to success of the reintroduction and how to attempt to remedy them. This includes thresholds, when, if reached, the reintroduction effort will temporarily cease until such actions to alleviate those attributing pressures have been taken. These thresholds were determined based on contemporary research (Table 1 and Appendix B) and with consideration that reintroduced individuals experience elevated stress levels and increased vulnerability that negatively influence demographic parameters. Because survival is feasible to quantify even with varying amounts of resources (e.g., VHF or GPS collars, size of crew), we used the population viability analysis, which integrated survival rates documented in reintroduced populations (Lewis et al. 2022, Facka 2017), to guide our thresholds for these parameters. If annual adult

female survival or juvenile (young of year) female survival is estimated between 0.80-0.65 and 0.30-0.15 respectively, agencies and partners would initiate conversations aimed at mitigating causes of mortality and potential actions to ameliorate them. If annual adult female survival or juvenile female survival is estimated below 0.65 and 0.15 respectively, subsequent releases (within same year or following years) would not occur until the reintroduction team felt identified limiting factors have been sufficiently mitigated.

Quantifying reproduction is highly dependent on available resources. If resources allow, estimation of female fecundity would be the preferred metric. To minimize disturbance and thereby potential abandonment, we recommend using trail cameras to document the number of kits per reproducing adult female and estimating fecundity by dividing those data by 2 (assuming 0.5 sex ratio). A range (95% confidence interval) of fecundity estimates, along with an aforementioned range of survival estimates, could be integrated into the developed population viability analysis to evaluate and forecast reintroduction success. Using documented fecundity rates from reintroduced populations (Lewis et al. 2022, Facka 2017), if estimated fecundity of translocated females is estimated between 1.25-0.85, agencies and partners would initiate conversations aimed at mitigating limiting factors to denning and recruitment success and potential actions to ameliorate them. If estimated fecundity is estimated below 0.85, subsequent releases (within same year or following years) would not occur until the reintroduction team felt identified limiting factors have been sufficiently mitigated. Preferably, fecundity would be monitored using noninvasive methods to minimize disturbance to, and thereby potentially success of, reproducing females.

Alternatively, if resources do not allow for estimation of fecundity, using clusters from GPS collars, proportion of females initiating denning and proportion of females confirmed to

have at least one surviving kit by the end of weaning will both be used to qualify female reproduction. These estimates are not a proxy for fecundity and would not be integrated into the population viability analysis. With proportion of radio-marked females reproducing being within 52-87% and proportion of which confirmed to have at least 1 surviving kit by end of weaning between 40-86% (Matthews 2012, Facka et al. 2016, Green 2017; Appendix B), if <50% of released females reproduce and, from which, <40% are confirmed to have at least 1 surviving kit, the team will engage in conversations aimed at addressing conservation threats to denning habitat in the release area. If no released females reproduce in the first spring after release, subsequent releases (within same year or following years) would not occur until the reintroduction team felt limiting factors have been sufficiently mitigated. Lastly, if a wildfire greater than moderate intensity burns >50% of the cumulative release area, the reintroduction effort will terminate and new locations will be reconsidered.

Once all cohorts have been translocated, success of the reintroduction effort will be measured through several means. First, with incorporation of the demographic parameters of the reintroduced population into the population viability analysis where appropriate, we will be able to determine if we reduced the potential for decline and/or population loss to a less than 20% risk of extinction over the next 50 years for the fisher population in Oregon. Initial establishment will be evaluated with survival and post-release movement metrics and secondarily by reproduction. Initial persistence will be evaluated using reproduction parameters (Armstrong and Seddon 2008). The thresholds in the previous paragraphs will be used to determine success of initial establishment and persistence.

We will continue to monitor population persistence, density, occupancy, and distribution for at least three years using non-invasive sampling methods once radio-collars are no longer

active or being deployed. We will delineate a non-invasive study area based on the merged home ranges of released individuals. We will continue to evaluate population viability using non-invasively collected data and annual estimates of apparent survival and apparent recruitment. We will further evaluate successful population persistence as a stable or increasing trend in annual estimates of population size and occupancy or a trend not in excess of a 30% annual decline based on annual estimates of population abundance of a reintroduced population in California (Green et al. 2022*a*).

We will also monitor long-term population expansion and connectivity using non-invasive sampling deployed using a 4-km (1 female home range diameter; Aubry and Raley 2006, Moriarty et al. 2019) grid across the non-invasive study area and to the estimated boundaries of the NCSO and SOC populations. Initially, monitoring expansion and connectivity may be implemented annually and then at less-frequent intervals, depending on resources and documented trends of the reintroduced population. Similarly, non-invasive sampling may include a combination of genetic and camera methods. If fisher detections occurs beyond the boundaries of the merged home ranges of the founding individuals, the long-term expansion component of the reintroduction effort will be considered a success. If a genetic component is integrated into the non-invasive sampling effort, through proportion of haplotypes and new individuals (specifically females), we will be able to determine connectivity between the reintroduced population and the previously existing fisher populations in Oregon. If haplotypes 1, 2, and 9 (Drew et al. 2003, Knaus et al. 2011) are present, the connectivity component of the reintroduction effort will be considered a success. As trends of the aforementioned metrics consistently increase and then plateau once occupancy-boundaries of the three populations begin

to overlap, we will feel confident that the fisher population in Oregon is more connected and have achieved our long-term goal.

In the event that some of the metrics of reintroduction success are observed but others are not, partners will discuss ways to encourage retention and expansion of the reintroduced fisher population. Methods could involve habitat management or modification actions to increase abundance of preferred forest stand features, incentivized predator harvest or contracted predator removal, accessibility restrictions to mitigate vehicles or recreation, or other efforts to reduce negative pressures (determined by cause of mortality of released individuals). If none of the qualifiers of reintroduction success are realized, a detailed report by the reintroduction team will discuss potential causes and retrospective recommendations of how to increase the success of such a management decision.

Economic and Human Resources

Sufficient economic resources and experienced personnel are necessary throughout a reintroduction project. A draft timeline for a reintroduction of fishers to Oregon is available in the *Timeline* section below. A detailed estimate of project expenses for the duration of the project, including post-release monitoring, is being developed. Potential funding sources include Competitive State Wildlife Grant funds, Pitman-Robertson funds administered by the Oregon Department of Fish and Wildlife, new funding opportunities through the passing of HB4134, and partner support. While submitting grant proposals and receiving responses, practitioners should consider if they are comfortable with the data already collected within their allocated resources and when the relevancy of those results expire, ultimately driving the decision of how to proceed with the reintroduction effort.

Partner Support

Successful reintroductions require sustained engagement from a diverse suite of partners and the public (Hayes and Lewis 2006, Callas and Figura 2008, IUCN Species Survival Commission 2013). Engagement from all partners can help strengthen the reintroduction study design and monitoring objectives, implementation of field research, community-science contributions, data analyses, publication of results, and public outreach. Establishing clear expectations, roles, and responsibilities for all partners is a key element in the success of a translocation. During a series of meetings, partners collectively identified roles needed to implement a fisher reintroduction and organized those roles into four categories: 1) planning and coordination, 2) source population surveys, 3) reintroduced population monitoring, and 4) wildlife health (Table 4).

Support from upper-level management, division administrators, and the director's office of the Oregon Department of Fish and Wildlife and approval from the Oregon Fish and Wildlife Commission is essential for a reintroduction. Transparent conversations with all tiers in the agency need to be frequent and fluid. Further, conversations with other state agencies may be warranted depending on the location of the desired source population. These discussions should occur early and outline expectations of who will be conducting the work (any pre- or post-translocation surveys of the source population, capture of sourced individuals) and the allocated resources to support such work (or plan for options to obtain funding). Collaboration may involve Memorandums of Agreements/Understanding, Data Sharing and Non-Disclosure Agreements, or other agreements that are explicit in accountability and deliverables. Lastly, a reintroduction implementation plan needs to be approved by the Oregon Fish and Wildlife Commission.

A diverse group of partners is meeting on a monthly basis to advance a program of work around reintroducing fishers to an unoccupied portion of their historical distribution in Oregon. Partners include approximately 40 individuals representing approximately 10 entities and organizations. Importantly, partners represent state and federal land and wildlife management agencies, land owners and managers enrolled in Candidate Conservation Agreements with Assurances (now Conservation Benefit Agreements) with the U.S. Fish and Wildlife Service, and academic research. The development and revisions of this implementation plan have been a central focus of partners.

The partners have discussed all the elements of a proposed reintroduction, come to consensus on shared concerns, and present an agreed upon program of work in this document. Shared concerns centered on the potential of the U.S. Fish and Wildlife Service listing fishers in northern California and southern Oregon under the federal Endangered Species Act. Partners shared concerns about the implications of a federal listing and the additional hurdles and steps for land owners and management agencies when conducting management actions (Figure 1). Partners discussed uncertainties around 1) an experimental designation for a reintroduced population, referred to as a 10(j) designation, if fishers were federally listed; 2) scarce datasets and limitations in conclusions leading to a range of uncertainty and comfort in our decision-making process for release sites and potential reintroduction areas; 3) timelines moving from planning to implementation and stochastic environmental events that could happen within that time; 4) other options to increase population resiliency and redundancy (such as a captive breeding program and large-scale habitat projects); and 5) a no-action alternative. Discussions have also focused on resolving reintroduction goals and objectives, determining where reintroduction is most feasible, and using modeling approaches to better understand how many

animals may be needed to ensure success under different threat scenarios. Despite differences in the goals, mandates, and missions of the many partners involved in these conversations, all partners have continued to express support with a common agreement that a reintroduction would benefit the conservation of fisher within northern California and southern Oregon.

Timeline

Task	Year 1				Year 2				Year 3				Year 4				Year 5				Year 6				Year 7				Year 8				Year 9			
	Spring	Summer	Fall	Winter	Spring	Summer	Fall	Winter	Spring	Summer	Fall	Winter	Spring	Summer	Fall	Winter	Spring	Summer	Fall	Winter	Spring	Summer	Fall	Winter	Spring	Summer	Fall	Winter	Spring	Summer	Fall	Winter	Spring	Summer	Fall	Winter
Obtain permits																																				
Pre-release predator and competitor surveys – Cascades and Source population																																				
Fisher noninvasive surveys – Source population																																				
Translocate fishers – capture, health assessments, release																																				
Fisher radio telemetry – Cascades																																				
Fisher noninvasive surveys – Cascades																																				
Data analyses/ reporting																																				

Conclusions

The implementation of a reintroduction of fishers to a portion of their historical distribution in western Oregon would address conservation objectives described by the U.S. Fish and Wildlife Service in the Oregon Fish and Wildlife Office Interim Fisher Conservation Strategy (U.S. Fish and Wildlife Service 2020*b*) and the Oregon Department of Fish and Wildlife in Oregon's State Wildlife Action Plan (Oregon Department of Fish and Wildlife 2025) and engage a diverse group of public and private partners in fisher conservation and management. This Implementation Plan offers a blueprint of goals, methods, and timeline for a fisher reintroduction to Oregon, leveraging best practices from previous reintroductions involving fishers and other species of conservation concern. Our recommendations are an adaptive process of implementation, monitoring, analyses, and adjusting to environmental and non-environmental conditions. Our recommendations include monitoring of fisher source and translocated populations and known fisher predators, competitors, and prey in the reintroduction area to evaluate the reintroduction effort and inform adaptive, post-release intervention. A reintroduction of fishers in Oregon will require sustained commitment by partners and stakeholders to achieve measurable species conservation objectives and the persistence of fishers in Oregon.

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Tables

Table 1. Baseline vital rates used in our population viability analyses based on unpublished estimates from extant fisher populations in southwest Oregon (Benjamin Padilla and Keifer Titus, unpublished data, Oregon Department of Fish and Wildlife) and estimates from California, Oregon and Washington reported in peer-reviewed literature and unpublished reports.

Vital rate	Mean	Standard deviation
Adult survival	0.8	0.1
Juvenile survival	0.3	0.1
Fecundity	1.25	0.2
Carrying capacity	K = 150, $\theta = 2$	
Age at first reproduction	2	

Table 2. Density estimates of carnivores known to predate on and compete with fishers (*Pekania pennanti*) within western portions of the historical range of fishers (Wengert et al. 2014, Gabriel et al. 2015). The fisher population column indicates whether a co-occurring population of fishers currently exists on each study area. Density estimates of fisher predators from occupied and unoccupied portions of the historical range of fishers in the western states could be used to further evaluate conditions in the proposed reintroduction area.

Species	Study Area	Density per 100 km^2	Fisher Population	Reference
Cougars	Blue Mountains in northeastern Oregon	1.7 (0.9 – 3.6, 95% BCI)	No	Ruprecht et al (2021)
	North Cascades in north-central Washington	1.55 $\pm$ 0.44	Reintroduction underway	Beausoleil et al. (2021)
	Blue Mountains in southeastern Washington	2.79 $\pm$ 0.35	No	Beausoleil et al. (2021)
	West slope of the Central Cascades in western Washington	2.34 $\pm$ 0.08	No	Beausoleil et al. (2021)
	East slope of the Central North Cascades in central Washington	2.37 $\pm$ 0.56	No	Beausoleil et al. (2021)
	Between the Kettle Range and Selkirk Mountains in northeastern Washington	1.96 $\pm$ 0.20	No	Beausoleil et al. (2021)
	Northern Rocky Mountains in west-central Montana	3.7 (2.3 – 5.7, 95% CI) – 6.7 (3.1 – 11.0, 95% CI)	Yes	Russell et al. (2012)
Bobcats	Blue Mountains in northeastern Oregon	12.1 (8.4 – 16.2, 95% BCI)	No	Ruprecht et al (2021)
	Central Oregon Cascades in western Oregon	3.0 – 4.0	No	Toweill (1986)
	Central Oregon Cascades in central Oregon	9.38 (7.61 – 11.7, 95% BCI)	No	Martin and Matthews (unpublished)

Coyotes	Blue Mountains in northeastern Oregon	25.0 (21.0 – 30.9, 95% BCI)	No	Ruprecht et al (2021)
Grey foxes	Klamath-Siskiyou Ecoregion in northern California	46 (31 – 65, 95% BCI) – 103 (68 – 138, 95% BCI)	Yes	Green et al. (2022)

Table 3. Prey items consumed by fishers on four study areas in Oregon and California. The number of scats collected and analyzed are presented for each study area. Columns indicated the number and percentage of scats containing a particular prey item. Taxonomy was updated from presented research using Integrated Taxonomic Information System (U.S. Geological Survey and Smithsonian Institution 2020).

	Klamath and California North Coast Golightly et al. 2006 n=388		Hoopa Valley Indian Reservation Slauson et al. 2011 n=140		Northern Sierra Nevada Facka et al. unpublished data n=382		Southern Oregon Cascades Raley and Aubry 2020 n=380	
Taxa	n	%	n	%	n	%	n	%
Mammalia	361	93.0	86	61				
Rodentia	193	49.7	64	46			16	4.2
Sciuridae	104	26.8	23	16	66	17.3		
Sciurinae	85	21.9						
Glaucomys oregonensis	16	4.1	3	2	6	1.6	7	1.8
Neotamias sp.	44	11.3						
Sciurus griseus	19	4.9	4	3	321	84.0	1	0.3
Spermophilus sp.	4	1.0						
Tamiasciurus douglasii	13	3.4	6	4	11	2.9	10	2.6
Xerinae								
Callospermophilus lateralis							10	2.6
Otospermophilus beechyii			2	1	14	3.7	43	11.3
Tamias sp.			2	1			11	2.9
Tamias minimus					1	0.3		
Cricetidae	78	20.1	38	27				
Neotominae	31	8.0						
Peromyscus sp.	11	2.8	7	5	5	1.3	2	0.5
Peromyscus maniculatus					2	0.5		
Neotoma sp.	20	5.2			15	3.9	1	0.3

Neotoma fuscipes			27	19	5	1.3		
Arvicolinae	36	9.3						
Arborimus sp.	15	3.9						
Arborimus pomo			4	3				
Microtus sp.	13	3.4					2	0.5
Myodes sp.	2	0.5					1	0.3
Ondatra zibethicus							9	2.4
Geomyidae								
Thomomys sp.	5	1.3					2	0.5
Dipodidae								
Zapus princeps							1	0.3
Erethizontidae								
Erethizon dorsatus							7	1.8
Soricomorpha	81	20.9	4	3			4	1.1
Soricidae								
Sorex sp.	6	1.5			23	6.0	3	0.8
Talpidae	77	19.8						
Neurotrichus gibbsii	22	5.7	1	1			8	2.1
Scapanus sp.	57	14.7	2	1			5	1.3
Scapanus latimanus					38	9.9		
Lagomorpha			1	1				
Leporidae	16	4.1					87	22.9
Sylivlagus bachmani			1	1				
Ochotonidae								
Ochotona princeps							1	0.3
Artiodactyla			1	1				
Cervidae	81	20.9	1	1			33	8.7
Odocoileus hemionus			1	1	119	31.2		
Didelphimorphia								
Didelphidae								
Didelphis viginiana					2	0.5		
Carnivora	87	20.9			6	1.6		
Canidae			2	1				

Urocyon cinereoargenteus	8	2.1	2	1				
Mustelidae								
Mustela vision					2	0.5		
Mephitidae	30	7.7	3	2			10	2.6
Mephitis mephitis	4	1.0	1	1				
Spilogale gracilis			1	1	1	0.3		
Procyonidae								
Procyon lotor	4	1.0						
Ursidae			1	1				
Ursus americanus			1	1	2	0.5		
Chiroptera	3	0.8						
Vespertillionidae								
Myotis sp.	2	0.8						
Unknown Mammalia							55	14.5
Aves	101	26.0			24	6.3		
Galliformes			1	1				
Phasionidae			1	1				
Bonasa umbellus			1	1				
Piciformes			3	2				
Picidae			3	2				
Picoides sp. or *Melanerpes* sp.			1	1				
Colaptes auratus	4	1.0	2	1			1	0.3
Passeriformes	1	0.3	5	4				
Corvidae			1	1				
Cyanocitta stelleri			1	1			4	1.0
Turdidae			4	3				
Ixoreus naevius or *Turdus migratorius*			4	3				
Sturnidae								
Sturnus vulgaris					15	3.9		
Accipitriformes								
Accipitridae								
Accipiter gentilis					2	0.5		

Unknown Aves							98	25.8
Reptilia	95	24.5	20	14	7	1.8		
Squamata	94	24.2					3	0.8
Iguania								
Iguanidae								
Sceloporus sp.			1	1				
Sauria	56	14.4						
Serpentes	14	3.6						
Colubridae			3	2				
Colubridae or Viperidae							8	2.1
Autarchoglossa								
Anguidae			13	9				
Elgaria sp.			13	9			3	0.8
Unknown Reptilia							11	2.9
Amphibia								
Caudata	8	2.1						
Unknown Chordata	40	10.3						
Insecta	214	55.2						
Hymenoptera	128	33.0			3	0.8	38	10.0
Vespidae								
Vespula sp.	64	16.5			4	1.0		
Formicidae	74	19.1			2	0.5	8	2.1
Camponotus sp.							18	4.7
Lepidoptera								
Torricidae							22	5.8
Orthoptera	19	4.9						
Stenopelmatidae								
Stenopelmatus fuscus	12	3.1						
Acrididae	6	1.5						
Coleoptera	81	20.9			5	1.3	17	4.5
Silphidae							2	0.5
Mallophaga	4	1.0						
Diptera	1	0.3						

Unknown Insecta			10	2.6
Other Invertebrates	17	4.4		
Acarina	8	2.1		
Araneida	2	0.5		
Gastropoda	4	1.0		

Table 4. Roles of partners in the implementation of a fisher reintroduction to an unoccupied portion of their historical distribution in the southern Oregon Cascades. Roles are indicated as leadership roles (L) and participant roles (P).

	Partner								
Roles	ODFW Research	ODFW Other	OSU	USFWS	Donor and Recipient Landowners	Non-Recipient Landowners/ Partners	Contracted Entities	ODFW Wildlife Health Lab	Private Partner Wildlife Care
Planning and Coordination									
Implementation plan revision	P	P	L	P					
Implementation plan approval	P	L	L	L	P	P	P	P	P
Identification of release area	L	L	L	L	P	P			
Partner coordination meetings	P	L	P	P	P	P	P	P	P
Regulatory compliance and permitting	P	L	L	P	P	P			
Grant solicitation	L	L	L	L	P	P			
Grant administration and reporting	L	L	L	L					
Source Population Surveys									
Capture and processing			L	P					
Movement including interstate transfer coordination		L		P					
Monitoring post-translocation			L	P					
Reintroduced Population Monitoring									
Pre-translocation surveys									
Identify data needs including existing data sources	L	P	L	P					
Design, planning, coordination	L	P	L	P	P				
Collect field data	L	P	L		P		P		
Analysis and reporting	L	P	L		P		P		
Information sharing	L	P	L	P	P	P	P		
Release	L	P	L	P	P			L	P
Post-translocation surveys									
Design, planning, coordination	L	P	L	P	P	P			
Collect field data	L	P	L		L	P	P		
Analysis and reporting	L	P	L	P	L		P		
Information sharing	L	P	L	P	L	P	P		

	Partner								
Roles	ODFW Research	ODFW Other	OSU	USFWS	Donor and Recipient Landowners	Non-Recipient Landowners/ Partners	Contracted Entities	ODFW Wildlife Health Lab	Private Partner Wildlife Care
Wildlife Health									
Assess animal health	P	P	P					L	P
Animal holding								P	L

Figures

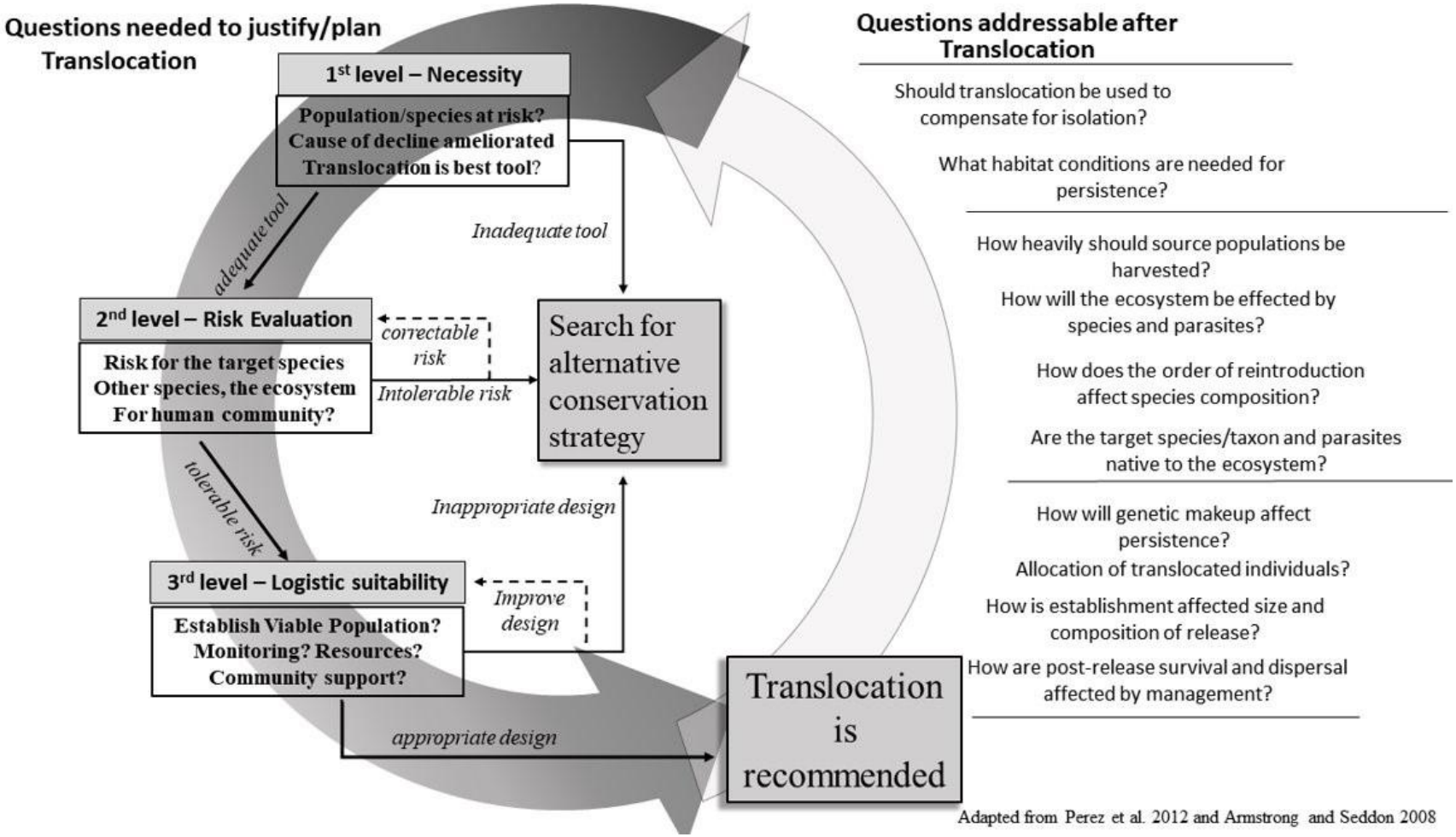

Figure 1. Hierarchical decision-making framework used by partners to evaluate reintroduction as a conservation action for fishers in Oregon.

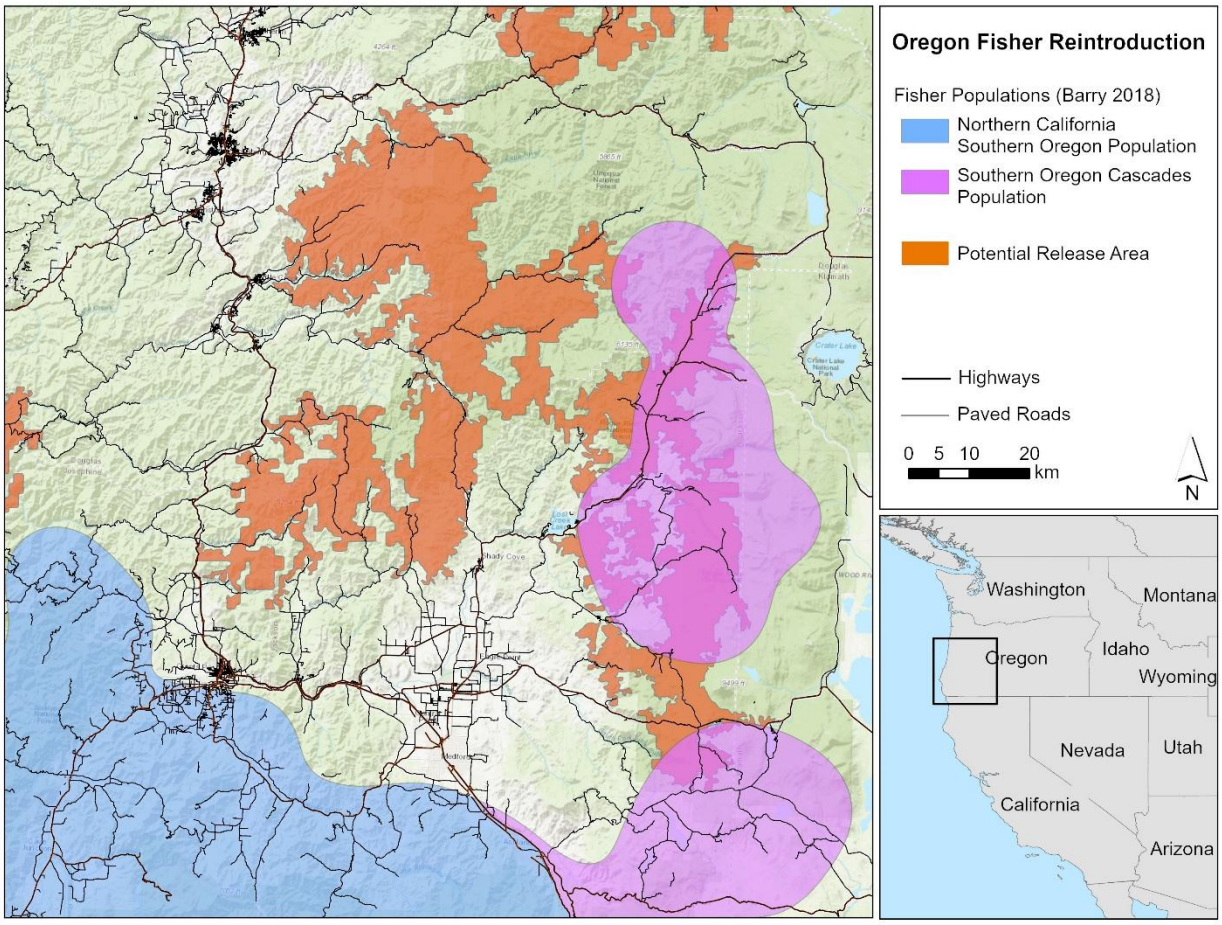

Figure 2. Polygons representing potential fisher release areas were developed from an ensemble model of habitat suitability models developed by Allen (1983), Fitzgerald et al. (2013), Aubry et al. (2017), and Barry et al. (2021). We rescaled each individual habitat suitability model to a 90 m resolution and calculated the mean habitat suitability value for each 90 m grid cell. We then calculated the mean habitat suitability value within polygons estimated to be occupied by fishers in Oregon from Barry (2018). We then masked all ensemble model grid cells with mean habitat suitability values less than the mean habitat suitability value estimated within occupied portions of Oregon. We masked all ensemble model grid cells that experienced >25% loss in canopy cover due to wildfire between 2012 and 2023 based on USDA Rapid Assessment of Vegetation Condition after

Wildfire (RAVG) data. We also masked all ensemble model grid cells were the suitability of a large wildfire by 2100 under climate scenario RCP8.5 was high (Davis et al. 2017). We generated the potential release area polygons by 1) converting the remaining ensemble model grid cells to polygons, 2) dropping any polygons less than 480 km^2 based on an average female home range size of 9.6 km^2 (Lofroth et al. 2010) and an baseline 50 female fishers, 3) filling any polygon holes less than 20 km^2, and 4) using kernel smoothing.

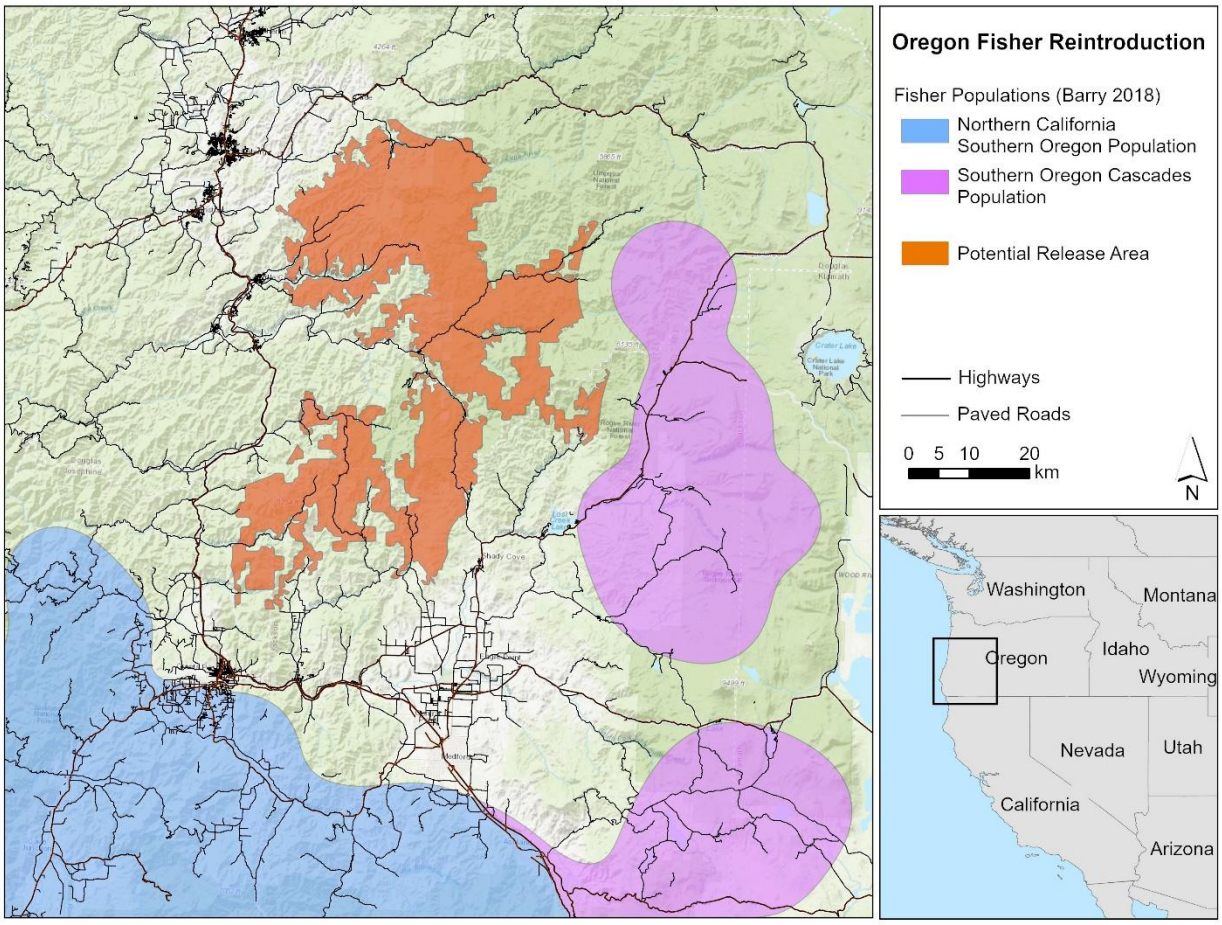

Figure 3. We further reduced the potential release area by clipping any areas west and south of Interstate 5, north of Oregon highway 138, and east of Oregon highway 62. We also buffered these roadways and the population area polygons of Barry (2018) by 5.62 km based on mean female fisher home ranges sizes reported by Aubry and Raley (2006) and Moriarty et al. (2019) and clipped any areas within those buffers resulting in the above potential release areas.

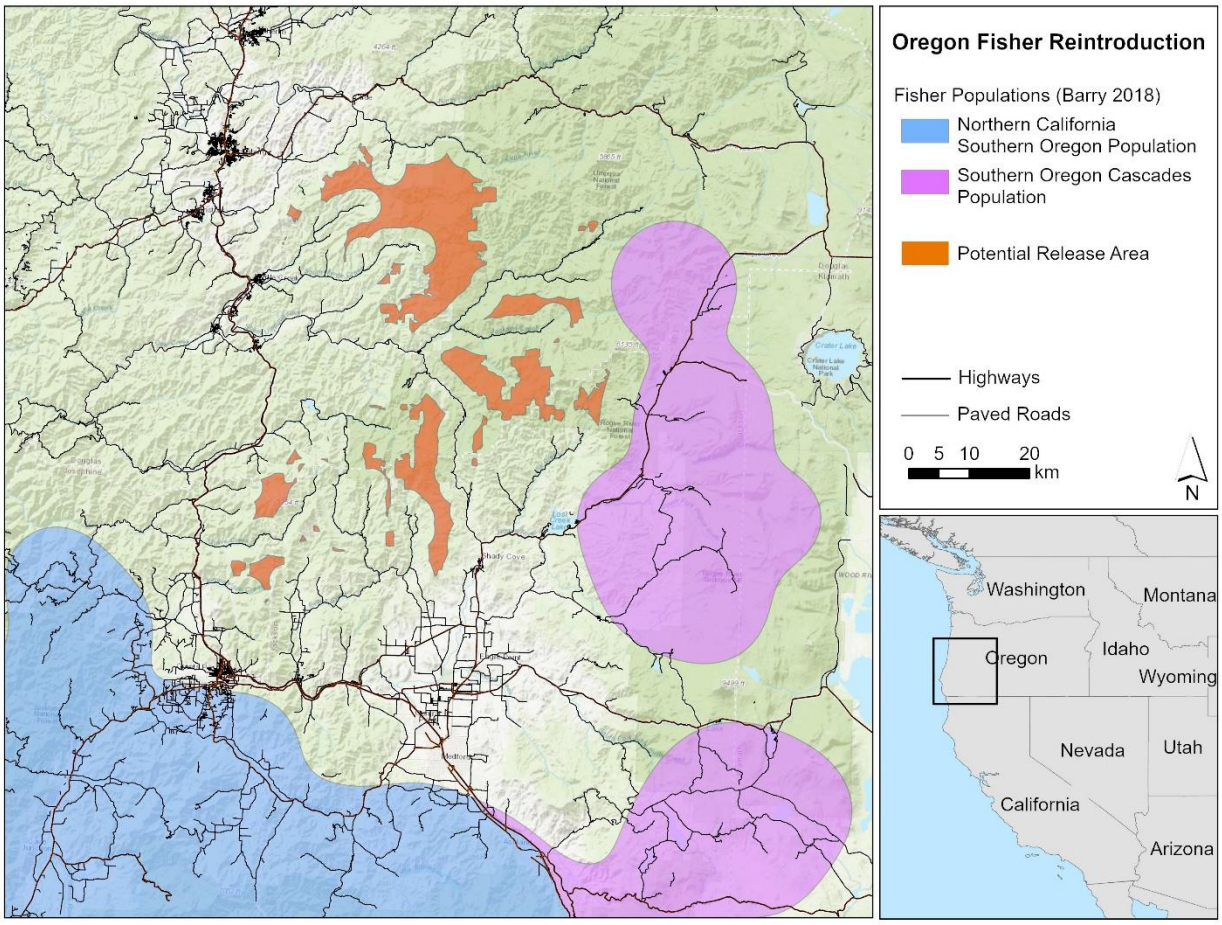

Figure 4. We further reduced the potential release area by buffering each paved road by 2.81 km, estimated as one female home range diameter based on mean female fisher home ranges sizes reported by Aubry and Raley (2006) and Moriarty et al. (2019), and clipped any areas within those buffers resulting in the above potential release areas.

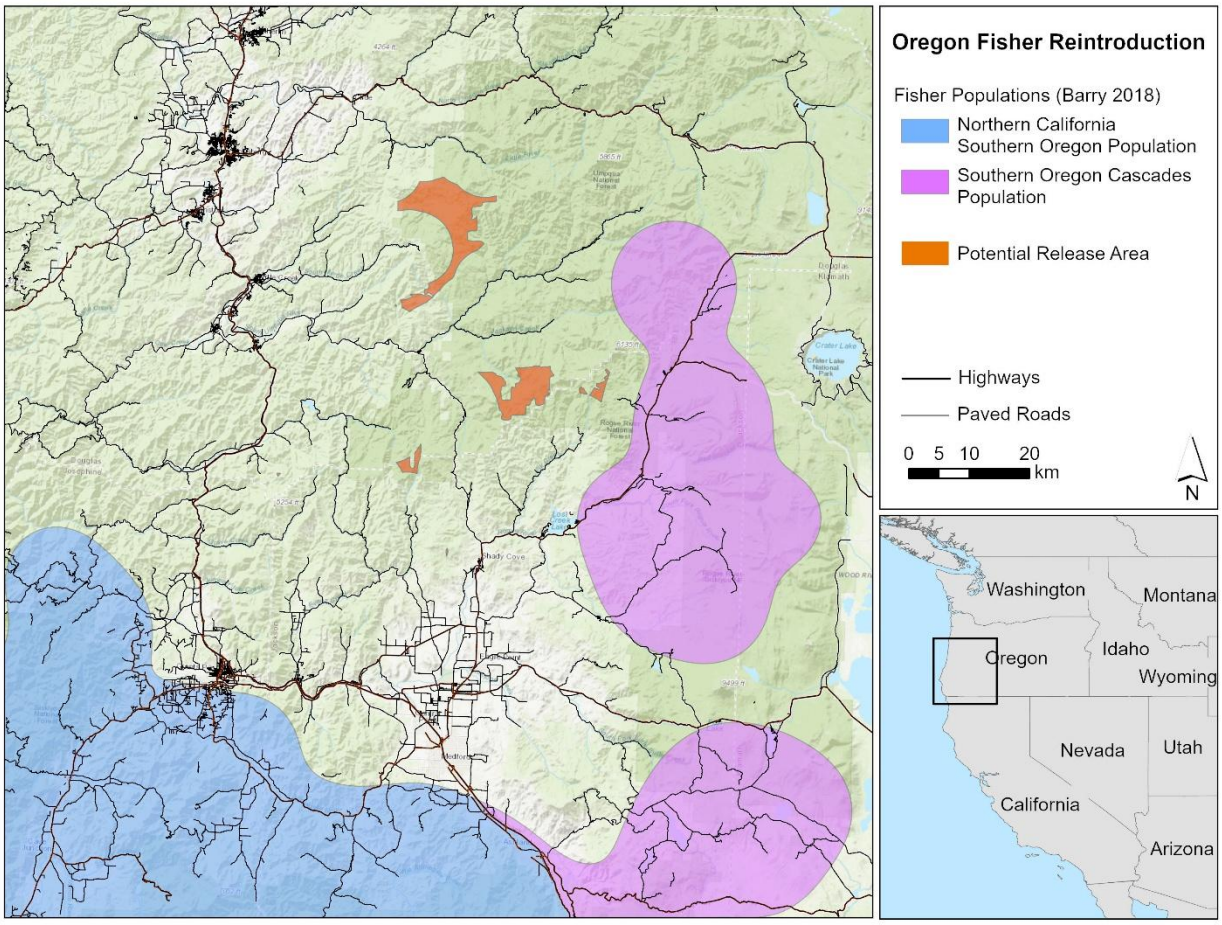

Figure 5. We further reduced the potential release area by buffering each paved road by 5.62 km, estimated as two female home range diameters based on mean female fisher home ranges sizes reported by Aubry and Raley (2006) and Moriarty et al. (2019) and clipped any areas within those buffers resulting in the above potential release areas.

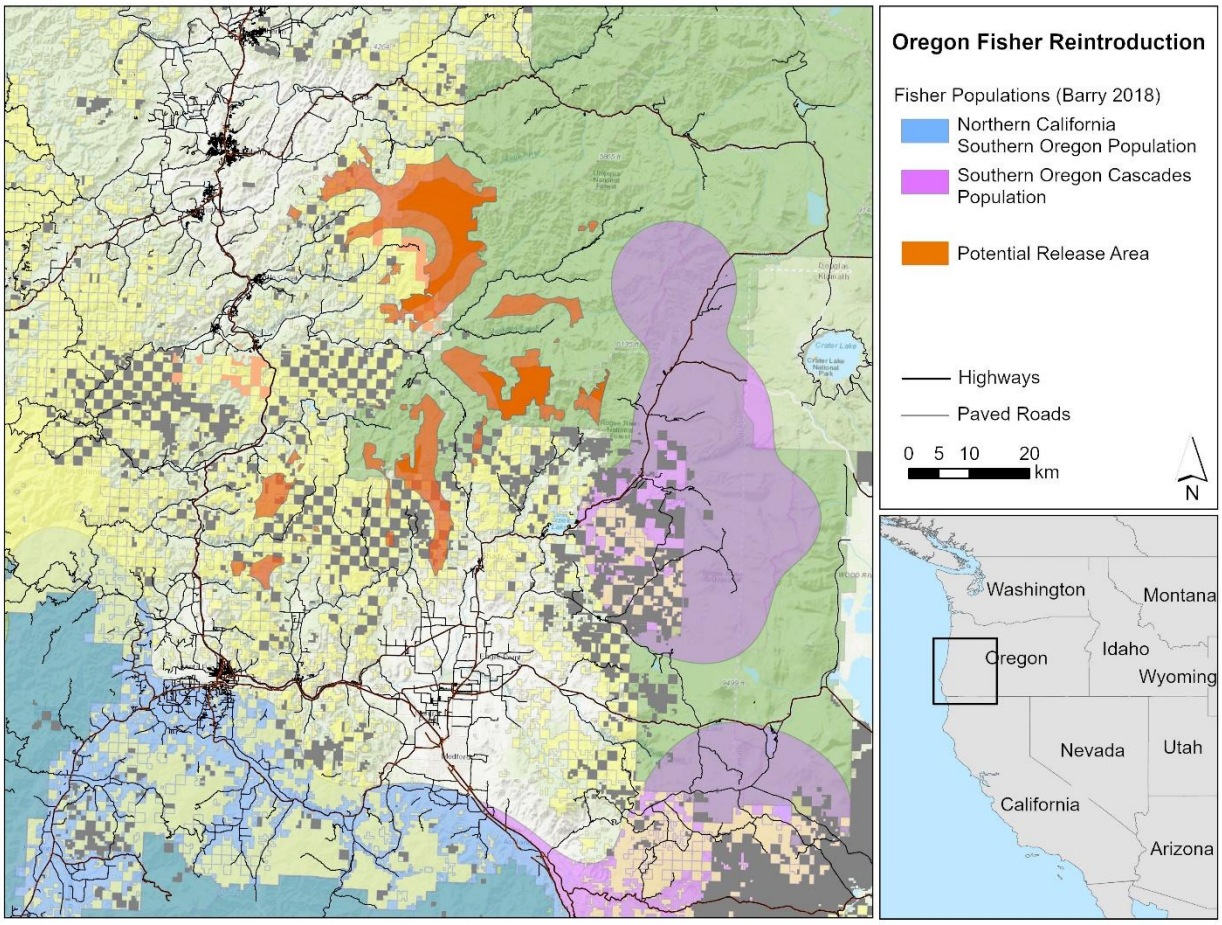

Figure 6. We overlayed a layer of landowners over the polygons presented in Figures 3 and 4. Light green represents U.S. Forest Service, light purple represents Bureau of Land Management, peach represents Cow Creek Band of Umpqua Tribe of Indians, gray represents private and public lands enrolled in a Candidate Conservation Agreement with Assurances with the U.S. Fish and Wildlife Service, and no color represents other private lands.

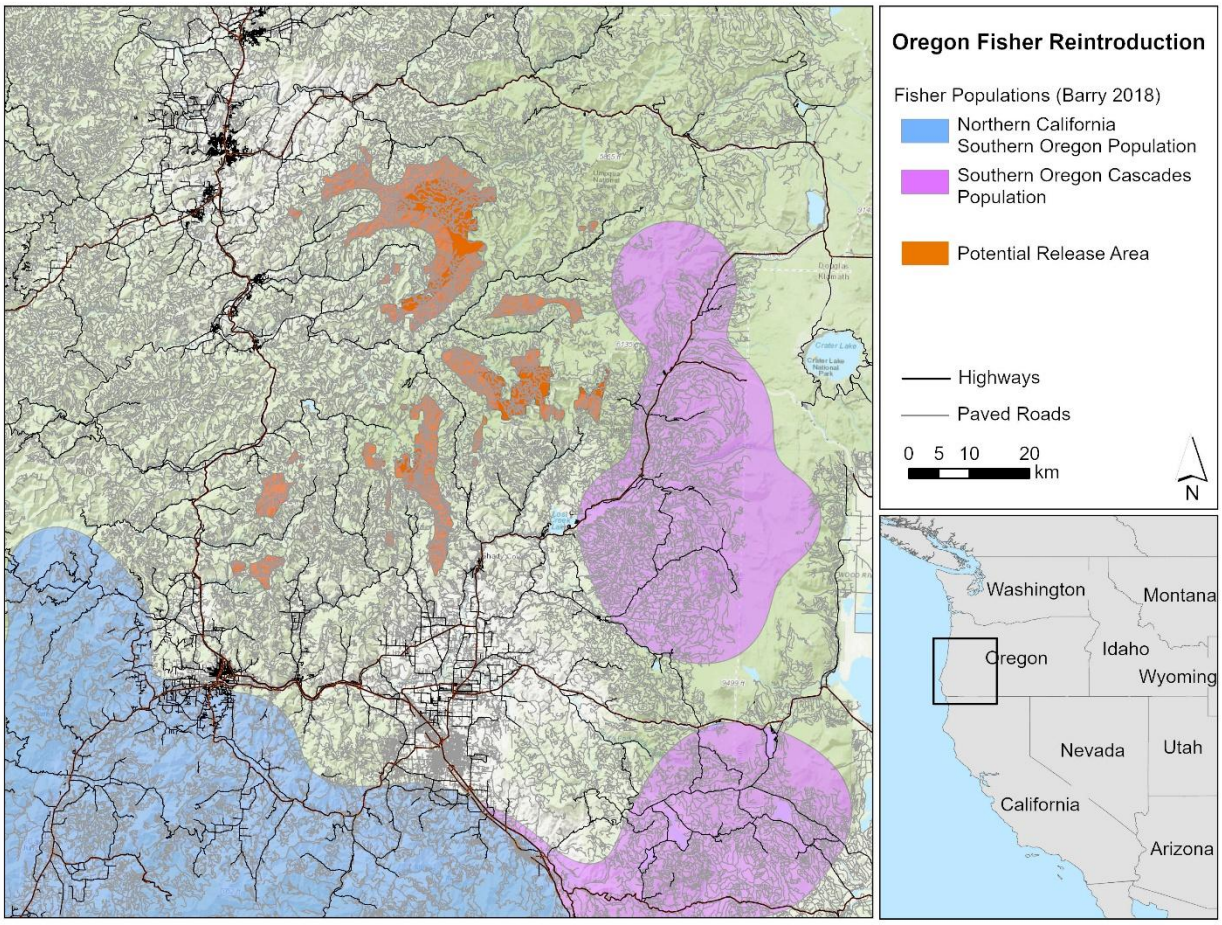

Figure 7. We determined the availability of roads in and adjacent to the potential release areas to be sufficient for the release and monitoring of reintroduced fishers and their progeny based on Oregon Bureau of Land Management Ground Transportation Road Network data and the unpaved roads depicted in light gray.

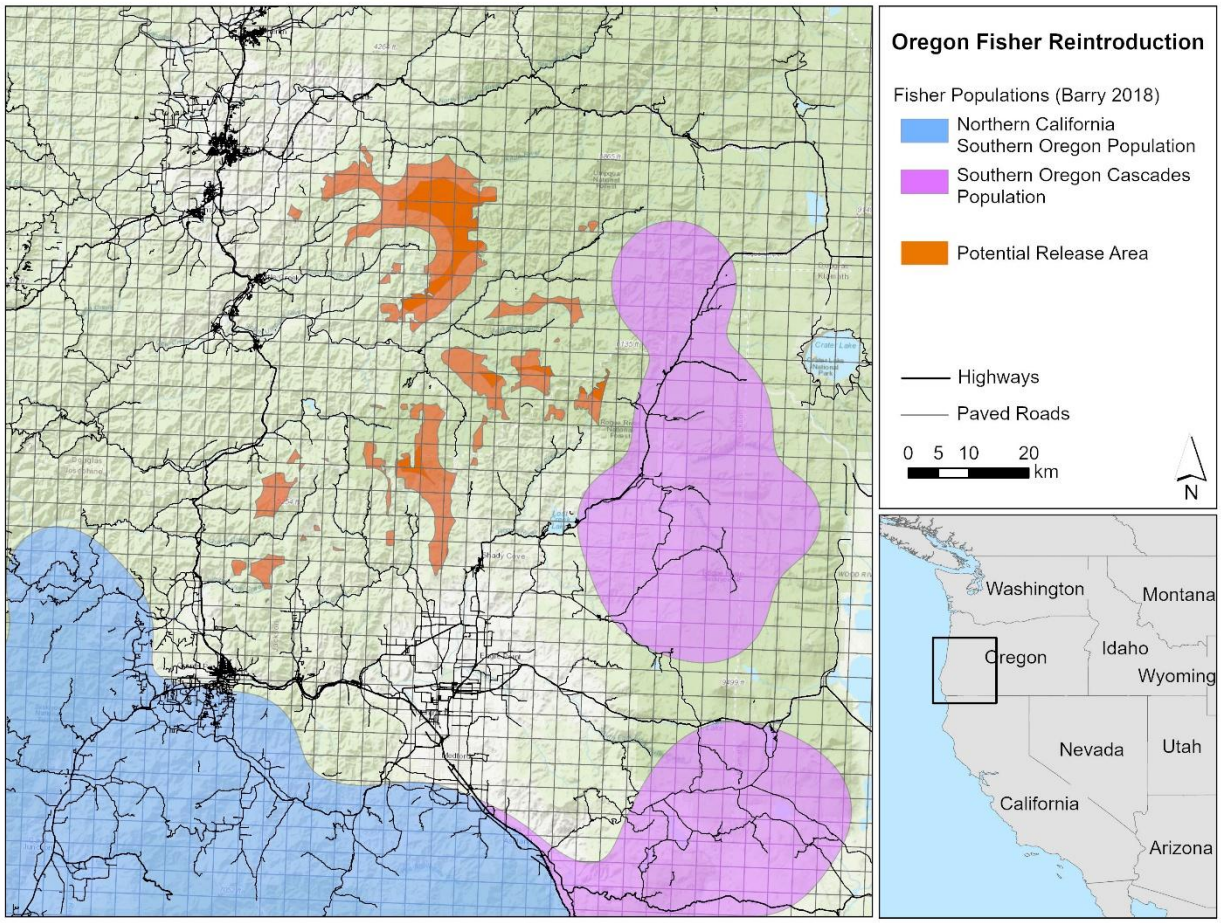

Figure 8. We overlayed a 4x4 km (16 km^2) grid over the potential release area polygons. We assumed a 16 km^2 grid cell approximates a mean female fisher home range size based on mean female home range sizes presented by Loforth et al (2010), Aubry and Raley (2006), and Moriarty et al. (2019). We anticipate the mean female home range size of female fisher translocated from the native portion of the Northern California Southern Oregon (NCSO) population to a release area in the southern Oregon Cascades (SOC) will be larger than the mean home range size in the NCSO and smaller than those reported in SOC.

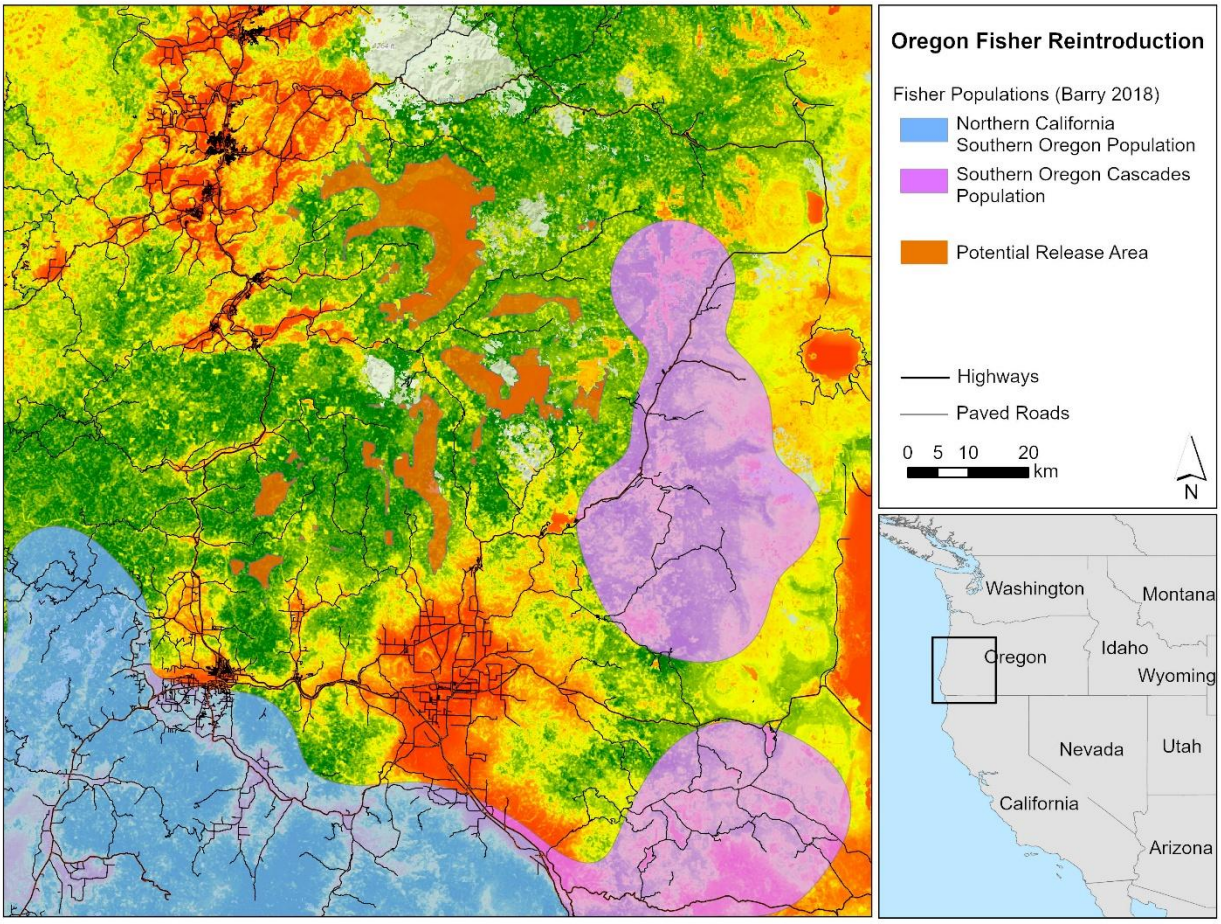

Figure 9. Proposed release areas identified for the reintroduction of fishers and ensemble habitat suitability values (Matthews et al. 2021) in the southern Oregon Cascades. Green pixels represent high habitat suitability, yellow pixels represent moderate habitat suitability, and red pixels represent low habitat suitability.

Appendix A – Partner MOU and CDFW Conservation Translocation Policy Attachment C

Memorandum of Understanding Between Partners Involved in the Reintroduction of Fishers to the Umpqua National Forest

Note: A pdf of the MOU and CDFW Conservation Translocation Policy Attachment C will be added to the final pdf version of this document.

Appendix B – Methods and Results for Pre-Translocation Surveys

Abundance estimation of the source population

The persistence of source populations from which translocated animals are removed is a key consideration for reintroductions intended to improve species conservation outcomes (IUCN Species Survival Commission 2013). Population persistence is evaluated via monitoring of source populations and is an important metric of reintroduction success. The monitoring of portions of the Northern California Southern Oregon fisher population selected for the removal of fishers for translocation to central Oregon could leverage survey protocols and analysis methods following the non-invasive methods described by Green et al. (2018).

In brief, survey sites are distributed within a grid of 10 km^2 cells and two survey sites are placed at least 1 km apart within each surveyed grid cell. Sites are selected to maximize the detection of fishers (e.g., riparian areas, ridgetops, dense overhead canopy cover, multi-story canopy, large down wood) and are within 100 m of drivable roads to optimize survey effort. Detection devices consist of a rectangular box (26 x 26 x 81 cm) made of corrugated plastic baited with a raw chicken leg and a can of wet cat-food. The back of the box is closed with 1.3 cm galvanized hardware cloth and the front is partially obstructed with three wooden slats. A strip of non-poisonous glue-board is attached to the underside of the bottom wooden slat so that mammals coming into the box are likely to leave hair with follicles attached to the glue strip to be used for genetic analyses. DNA is extracted from hair samples and genotyped at the U.S. Forest Service National Genomics Center for Wildlife and Fish Conservation (Tucker et al. 2014). Fisher population vital rates (i.e., apparent survival and apparent recruitment) and density can be estimated with a spatial Jolly-Seber open population model. Jolly-Seber models fit with

spatial capture-recapture integrate the movements of individuals directly into their formulation to provide spatially-explicit estimates of population density.

For the last decade, Oregon State University and the Oregon Department of Fish and Wildlife have been collaborating on Northern California Southern Oregon fisher population monitoring using non-invasive survey methods and a genetic-based detection/non-detection survey design with hair snares (Green et al. 2018). From 2017 to 2024, OSU deployed 100 sampling stations at locations that have been surveyed annually since 2006. The Oregon Department of Fish and Wildlife deployed an additional 42 stations in 2019, 63 stations in 2020 and 95 stations in 2021-2024 to the north and adjacent to the 100 historical stations. Collectively, this study area was representative of the extant fisher population in northern California and southern Oregon. Survey stations remained open for a maximum of six consecutive weeks each year and were checked weekly. Each hair sample was collected, stored in a vial with desiccant, and sent to the National Genomics Center for Wildlife and Fish Conservation within the USDA Forest Service Rocky Mountain Research Station to determine species for all samples, and if identified as fisher samples, were subsequently genotyped to determine individual identity and sex. We used single-season, hierarchical spatial capture-recapture (SCR) models for each year of surveys to estimate the abundance, density, and distribution of fishers within the study area (Anderson et al. 2023). To build upon these results, we incorporated data into a Cormack-Jolly-Seber model following Green et al. (2022) to estimate apparent survival, recruitment, and emigration of the population.

From 2017 to 2022, our sampling units were open each year for a cumulative average of 6758.5 ± 606.6 sampling nights. During this time, we collected an average of 694 ± 173.4 samples annually that were submitted for genetic analyses. We identified a total of 153 unique

individuals from 2017 to 2022, with an average of 40 ± 10 individual fishers detected each year. Generally, our single season spatial capture recapture models revealed a declining trend in fisher abundance over a 6-year period across the overall study area (Figure 1).

From 2022 to 2024, our sampling units were open for 7618.33 ± 725.11 sampling nights. We collected a mean of 739.33 ± 94.09 samples that were submitted to The National Genomics Center for Wildlife and Fish Conservation for genetic analyses. Of the samples collected, we identified the species for 716.33 ± 119.24 samples (96.56% ± 0.04%). Fisher was identified for 163.33 ± 50.33 samples, and of these, 122.33 ± 46.69 (75.12% ± 14.12%) successfully amplified to individual and sex identification. We identified 31, 36, and 41 unique fishers in 2022, 2023, and 2024, respectively. Individual fishers were detected at an average of 1.9 ± 0.3 hair snares each year.

More recently, we've identified a change in trends in both the OSU and ODFW portions of the study area. Our model estimated a median fisher abundance of 40 (CRI 30-52) fishers in 2022, 45 (CRI 35-56) fishers in 2023, and 49 (CRI 40-58) fishers in 2024 for all combined study areas (Figure 2). Most importantly, fisher abundance increased in ODFW's portion of the study area from 8 (CRI 5-12) fishers in 2022 to 16 (CRI 12-19) fishers in 2024 (Figure 2; 30 unique individuals identified that year when also considering radio-collaring efforts). We also documented a shift in sex-ratio from male-biased to female-biased and a stabilization in estimates fisher abundance in the OSU portion of the study area (Figure 2) that extends from Mt. Ashland in Oregon to the southernmost portion of the study area. There, annual fisher abundance ranged from 24 (CRI 16-31) fishers in 2022 to 25 (CRI 18-30) fishers in 2024. This is encouraging given the extensive, high-severity fires that occurred within the study area in 2016, 2021, and 2022. The increasing trend in estimated population abundance in Oregon suggest that

the ODFW portion of the study area may serve as a leading edge for fisher population expansion and recolonization into historically occupied portions of the NCSO. Due to high annual survival estimates of radio-marked fishers in southwestern Oregon (73%, seasonal survival ranges from 88-96%) and their increasing population trend, ODFW can comfortably source some individuals from southwest Oregon for a reintroduction effort, with the majority of individuals being sourced from California. See the cumulative studies and results described in *Consequence for source population* for more details.

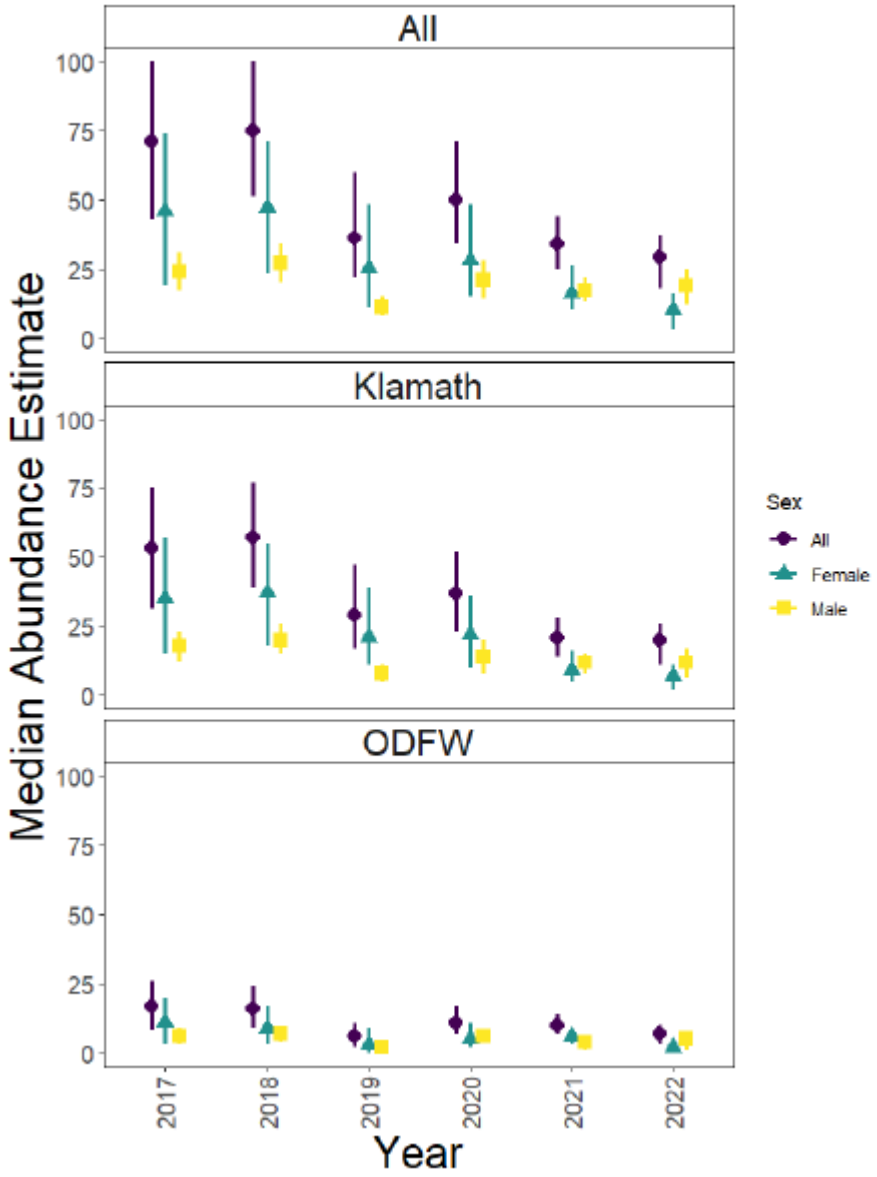

Figure 1. Annual abundance estimates from 2017 to 2022 for all fishers (purple circles), female fishers (green triangles), and male fishers (yellow squares) in the 1. Combined Klamath and ODFW study area, 2. Klamath study area, and 3. ODFW study area. Estimates for 2017-2021 were presented in Anderson et al. (2023)(2023).

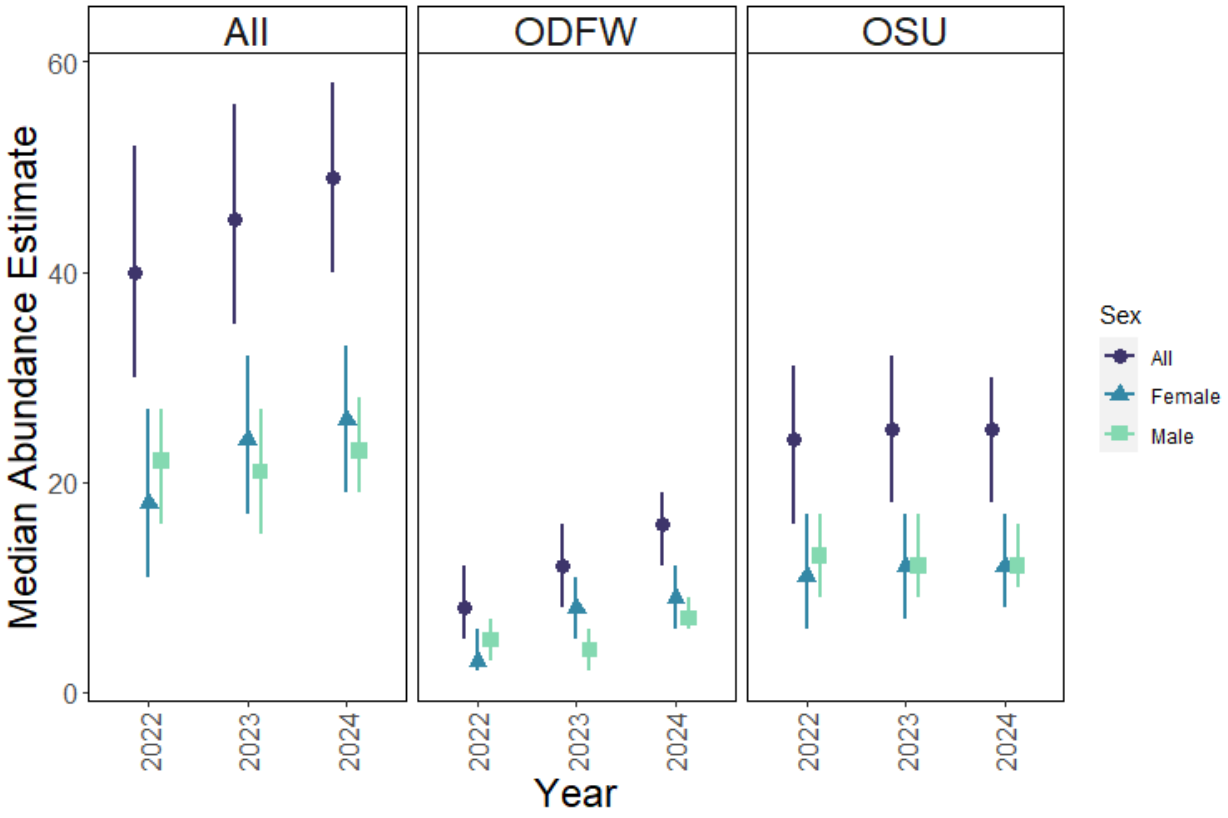

Figure 2. Annual abundance estimates from 2022 to 2024 for all fishers (purple circles), female fishers (blue triangles), and male fishers (green squares) in the 1. Combined Klamath and ODFW study area, 2. ODFW study area, and 3. Klamath study area.

Radio-marking individuals of the source population

Understanding how fisher populations use their landscape and what landscape characteristics they select for is a vital step to guide decision-making when selecting release areas for a reintroduction. Relative to other techniques for monitoring, documenting fisher locations using radio-tracking can provide more information about landscape or habitat features that are important to fishers, providing data at relevant scales that can be used to evaluate suitability of potential release areas. Results from a radio-marking effort will highlight specific forest features and at specific scales or metrics that could be compared to unoccupied areas of fisher historical range identified as a site for a potential release area. Through this quantitatively-based process, our goal is to ensure that the reintroduction area is similar to areas already occupied by fishers. By selecting appropriate habitat in the relocation area, we hope to reduce risks to translocated fisher..

In an effort to better understand the spatiotemporal movements of fisher and quantify niche partitioning with gray fox, a hypothesized limiting factor to fisher population expansion and recolonization, ODFW radio-marked fisher and gray foxes from 2023-2025. Radio-collars were programmed to record a GPS point every 3-hours with an offset of 1-hour so that all 24 hours of the day were represented with location data. From 20 fishers (11F:9M) and 23 gray foxes (14F:9M), we collected spatiotemporal data resulting in, on average, 239 days and 1429 locations for fisher and 251 days and 1805 locations for gray fox. With these data, we summarized general space use metrics (e.g., average home range size), quantified activity patterns temporally, and evaluated 3rd-order selection of landscape and habitat covariates.

Mean home range sizes for fishers are large relative to their body size, estimated to be 19.30 km^2 (13.87-27.11 km^2) for females and 89.12 km^2 (62.80-126.47 km^2) for males (similar to

that of other fisher populations and studies). This was not surprising given their food, refuge, and denning needs. What was surprising was that we found fishers to be more active during the day (opposite observation from the fisher population in the Sierra Nevada, Jody Tucker pers. comm.) and gray foxes to be more active at night. In addition to temporal niche partitioning, we also observed spatial niche partitioning with fishers selecting for increased and high-mid canopy cover (canopy cover: $\beta = 0.349 \pm 0.049$ and canopy cover^2: $\beta = -0.052 \pm 0.048$) and gray foxes selecting for decreased and mid-low canopy cover (canopy cover: $\beta = -0.690 \pm 0.133$ and canopy cover^2: $\beta = -0.311 \pm 0.134$). Due to spatiotemporal niche partitioning, we do not believe gray fox to be a limiting factor to fisher population persistence and growth, nor to expansion or recolonization. As such, gray fox densities or abundance are less important than other considerations in evaluating release sites and reintroduction areas. Instead, when identifying and justifying the reintroduction area and release sites, we are putting a greater focus on habitat characteristics such as increased canopy cover that would encourage effective niche partitioning, or features such as coarse woody debris or snags that have positive influences on foraging and denning.

Occupancy estimation of the source population

To better understand population thresholds of potential fisher predators (cougar, bobcat, coyote) and competitors (bobcat, coyote, grey fox), as well as prey resource availability, we sought to quantify the mesocarnivore and small mammal communities in the areas where fisher populations persist. To do this, we capitalized on data collected in our OSU-ODFW collaborative hair snare survey and applied those data (cameras and track plates paired at hair snare detectors) to an occupancy framework. To date, the scales and models are not directly comparable to the surveys and analyses conducted in the release area, but still provide some preliminary insight

into what occupancy estimates of other mesocarnivores are tolerable as well as of prey species are necessary to support a sustainable fisher population.

Using a stacked single-season occupancy model and data from 95 sites at least 1km apart for 6 weeks in 2019-2024, we estimated mean occupancy probability for several small mammal groups. As expected, occupancy fluctuated for several species and often demonstrated contrasting occupancy probabilities of other small mammals in the community suggesting a consistent prey base. Only western gray squirrels had a mean occupancy probability of consistently below 0.25. Douglas squirrels, ground squirrels, flying squirrels, and chipmunks all had mean occupancy probabilities between 0.25-0.75. Woodrats and small rodents as a stable mean occupancy probability at about 0.50 and 0.75, respectively.

Occupancy surveys in the release area

Pre-translocation surveys to estimate the occupancy patterns of potential fisher predators (cougar, bobcat, coyote) and competitors (bobcat, coyote, grey fox) using remotely-triggered cameras can be conducted using a variety of methods (Long et al. 2008). Initial survey efforts in the northern portion of the proposed reintroduction area were conducted by the Oregon Department of Fish and Wildlife in fall 2025. The objectives of this effort was to characterize the composition and distribution of potential competitors and predators to translocated fishers and assess the small mammal prey community.

Remote cameras were deployed in patches of mature forest in pre-selected release areas. Using a 4km x 4km grid based on average female fisher home range size in the literature, we established two camera sites per cell (to increase probability of detection of mesocarnivores) and cameras were deployed with a scent lure for 8 weeks. Two additional camera sites were selected

≥100 m from the mesocarnivore camera site and each surveyed for 4 weeks for small mammals (Littlewood et al. 2021).

Given clustering of sampling sites within ownership polygons, we used polygon-level random intercepts when supported by the data. Model structure was selected on a species-specific basis according to detection frequency, dispersion diagnostics, and random effect singularity checks. We used negative binomial generalized linear mixed models (NB GLMMs) when overdispersion and spatial clustering were supported and Poisson GLMMs when dispersion parameters indicated minimal overdispersion. For species with sparse detections where random effects were not supported, we used negative binomial generalized linear models (NB GLMs; intercept-only). This modeling approach retains zero-detection sites, accounts for unequal sampling effort, accommodates overdispersion common in camera trap data, and incorporates spatial clustering where warranted. Accordingly, results are interpreted as indices of relative activity rather than unbiased estimates of abundance or occupancy.

We analyzed small mammal detections using a parallel negative binomial modeling framework. We initially explored a single-season occupancy model (4 × 7-day sampling occasions); however, we observed significant lack of fit across all modeled species (MacKenzie–Bailey goodness-of-fit $p < 0.01$; $\hat{c}$ range 2.14–5.59), indicating substantial detection heterogeneity. Given these results, we deemed occupancy modeling inappropriate for this dataset and retained relative detection rate models. We assigned species to modeling tiers based on detection frequency and spatial coverage. Species with sufficient detections were analyzed using negative binomial GLMMs with polygon-level random effects where supported. Species with moderate detections were modeled using intercept-only negative binomial GLMs. Rare species

were summarized descriptively without model-based inference. As with carnivore analyses, we interpreted small mammal results as indices of relative activity at baited stations.

Among focal carnivores, striped skunk (Mephitis mephitis), gray fox (Urocyon cinereoargenteus), and American black bear (Ursus americanus) were the most frequently detected species in terms of both total detections and site-level occurrence (0.59, 0.54, and 0.37 respectively), and gray fox exhibited the highest estimated relative activity rate. Small mammal surveys detected 13 species or species groups across sampling stations. Deer mouse (Peromyscus maniculatus) was the most frequently detected taxon and was widely distributed across the study area, occurring at 84% of survey sites. Townsend's chipmunk (Tamias townsendii) and shrew species (Soricidae spp.) were also broadly distributed and commonly detected, occurring at 68% and 48% of sites, respectively. Vole species (Cricetidae spp.) and Douglas' squirrel (Tamiasciurus douglasii) were detected at moderate frequencies, occurring at 42% and 38% of survey sites. Site-level detection counts indicated substantial spatial heterogeneity for several taxa, particularly shrew species, which showed high detection counts at a small number of sites but were absent from many others. Paying particular attention to these locations as potential release sites may be beneficial for females who may den post-translation. Collectively, small mammal detections indicated a broadly distributed prey community across surveyed parcels, with assemblages dominated by small granivorous and insectivorous taxa. If detection frequencies are highly colinear to and or representative of occupancy probabilities (see aforementioned section on the small mammal community in the NCSO), then our results suggest that prey availability in the release area is sufficient enough to support a reintroduced fisher population.

Low detection rates of mesocarnivores and inability to model occupancy raises some levels of concern. Although we are hopeful photo collection during the winter months produced increased detection rates and more photo data, we are considering another year of pre-translocation surveys to increase our understanding of the proposed release areas to inform decisions regarding the suitability of the proposed reintroduction area. However, preliminary results suggest low levels of predation risk, indicating that translated fishers may not face significantly elevated risk in the proposed reintroduction area. Preliminary results from the small mammal perspective also suggests a robust prey community and sufficient food resources, although these metrics cannot be compared to other areas with persistent fisher populations because those types of efforts were not executed. These data help to further refine where translocated fisher may be released and provide scientifically-based and provide evidence to support moving forward with a fisher reintroduction in the Umpqua National Forest.

Abundance surveys and diet composition in the release area

Pre-translocation surveys to estimate population densities and habitat associations of potential fisher predators (cougar, bobcat, coyote) and competitors (grey fox) using detection-dog teams would offer additional insights into pre-reintroduction decision making. A survey design could be based on the designs described by Thompson et al. (2012) and Ruprecht et al. (2021). Briefly, survey effort by detection-dog teams can be stratified across 4-km^2 grid cells within which detection-dog teams can hike for 6-8 hours covering 5-15 km during each survey day. Survey effort can be measured using a global positioning system (GPS) data logger placed inside the dog's harness. Handlers can record date, time, GPS coordinates, species identification, and scat condition at the location of each scat. Handlers can collect entire scats in a paper bag and label samples with the date of collection, handler initials, and a unique sample identifier.

Handlers should take precautions while collecting scats to avoid unnecessary exposure to diseases by wearing gloves and sanitizing their hands.

Scat samples can be analyzed for genetic species and individual identification and diet composition. Genetic subsamples can include 2-3 mL of scat subsampled from the tapered ends and widest portions of the scat to collect the largest number of epithelial cells and stored in 15 mL tube of 95-100% EtOH. Diet subsamples can include three lentil-sized pellets of scat from the interior of the scat stored in 1 mL of 95-100% EtOH in a 2-mL Eppendorf tube. Species and individual identification will require the development of regionally-specific molecular tools based on single-nucleotide polymorphisms (SNPs) that allow for more affordable and accurate identification of individual carnivores.

Detection-dog surveys can be complemented with remotely-triggered camera surveys conducted in alternate years to collect detection/non-detection data of fisher predators and competitors. Remotely-triggered cameras can be used to collect detection/non-detection data for a wide range of carnivores (Kays and Slauson 2008, O'Connell et al. 2011), including cougars, bobcats, coyotes, grey foxes and fishers (e.g., Kelly et al. 2008, Green et al. 2018, Ruprecht et al. 2021). We recommend deploying cameras in alternate years from the detection-dog surveys (see *Timeline*) and using a survey grid of 4-km^2 cells with two cameras deployed at least 1 km apart in each grid cell. Cameras should be placed along ridges, saddles, drainages, or other terrain features that could be likely travel routes for target carnivores. Fine-scale site selection should involve placing cameras along wildlife trails or locations of target carnivore sign (e.g., tracks, scats, scrapes). Cameras should be deployed for at least six continuous weeks. Species identification can be completed using Wildlife Insights (https://www.wildlifeinsights.org/home). Species detection/non-detection data could be included in a hierarchical modeling of cougar,

bobcat, coyote, and grey fox densities and could further elucidate the suspected absence of fisher in the reintroduction area.

Appendix C – Methods for Fisher Live-Capture and Field Health Assessments

The live-capture and handling of fishers for translocation to southern Oregon will require coordination and approval of agency partners, veterinarians, and an Institutional Animal Care and Use Committee. We propose the following series of best practices for the live-capture and handling of fishers. These recommendations are a culmination of a series of projects involving the live-capture and handling of fishers (e.g., Aubry and Raley 1996, Matthews et al. 2011, Lewis 2014, Sweitzer et al. 2015, Green 2017). Sourcing of founding individuals for translocation should occur over a wide landscape across southern Oregon and northern California at multiple sites to minimize impacts to source populations (i.e., Powell et al. 2025). Priority for sourcing founding individuals should be from long-term study areas with monitoring data available to inform the live-capture of founding individuals (e.g., Green et al. 2018, Anderson et al. 2024). It is advised that fewer sedations and handling events are prioritized, so data and sample collection should be preemptively planned to align with capture events and health assessments.

We recommend the selection of sites to place live-traps should follow methods developed for the translocation of fishers from northern California to the northern Sierra Nevada (Stuart Farber, Manulife Investment Management, personal communication). Genetic data from long-term study areas in the Klamath and Ashland regions are available for assessments of the number of individuals, sex, and minimum age detected at sites, Genetic relatedness tests for female fisher should be conducted to inform founder population selection. In an effort to achieve a genetically diverse founder population, live-capture locations should be selected based on the following criteria, 1) repeated detections of the same individual male and female fishers with minimum age estimates of at least three year of age and 2) live-capture and translocation of only one female

fisher at sites with two or more closely-related individuals (indicating a mother-daughter or sister-sister relationship).

Site selection should also involve using a Geographic Information System (GIS) in a stratified design to account for access via drivable roads, the presence of ground and overhead vegetation, and the presence downed logs and drainages (Green et al. 2018, 2022*b*, Anderson et al. 2025, Smith et al. 2025). Overhead cover will be particularly important to increase trapping success and to minimize temperature fluctuations at the trap. A 1 m long and 0.3 m wide area should be cleared from debris and flattened such that a trap can be placed and stabilized. Traps should ideally be placed against a log or large tree for additional support.

Fishers should be captured using cage-type live traps (model 207, Tomahawk Live Trap Company, Tomahawk, Wisconsin, USA), modified with a plywood cubby (Wilbert 1992, Seglund 1995). The floor of the cubby should be covered with leaf litter or a piece of fleece material. The cubby should serve as a refuge to reduce stress and self-inflicted injuries (e.g., broken canines). Traps and cubbies should be covered with bark and other natural materials to conceal traps, to keep the interior dry, and to provide insulation. Traps should be baited with chicken tied to the top of the trap behind the treadle or tied directly to the treadle with monofilament line or string or placed inside a sock and tied to the top of the trap behind the treadle. Bait lure should be applied to the chicken (Hawbaker's Fisher Lure, Hawbaker and Sons, Fort Loudon, PA or Fisher Red Lure, Proline Lures, Indianapolis, IN) and a call lure can be applied on a nearby tree (Caven's Gusto, Minnesota Trapline Products, Pennock, MN or Outreach Call Lure, Proline Lures, Indianapolis, IN).

Traps should be checked every morning and any captured fishers should be processed at the site of capture after all traps are checked. Captured fishers should be anesthetized with

ketamine hydrochloride (40 mg/kg) and midazolam (0.25 mg/kg). Ketamine is a dissociative anesthesia drug, is rapidly absorbed, and the animal maintains cardiovascular and respiratory function. Midazolam is a sedative, muscle relaxant, and has slight analgesic, or pain-killing, properties. Midazolam also counteracts the likelihood of convulsions from ketamine (Mortenson and Moriarty 2015). The sex and age class of each captured animal should be estimated by visually assessing the size and physical attributes of the animal in the trap. Sex of the animal can be confirmed by palpating the groin for presence of a baculum when the animal enters the handling cone. Dosages should be drawn with 1 mL syringes and 1 inch, 25 gauge needles. Drug volumes should be combined in a 3 mL syringe and delivered via intramuscular injection (IM) using a ¾ inch, 20-gauge needle.

A canvas bag attached to a cylindrical steel-handling cone should be secured onto the end of the cubby box. The bag and handling cone should be spread out such that the animal can see light through the cone once the door on the cubby box is opened. Large dowels (8-10 cm) can be used to assist in directing the animal into the cone. Once in position, the rear sliding door of the cubby box is opened. As the fisher advances to the cone, the handler can coax the animal into the cone area and dowels can be placed such that the animal cannot turn around. Handlers should keep their hands away from the cone and only use dowels to direct the fisher prior to immobilization to minimize risk of bites or scratches. Fishers should not be physically handled while conscious. The ketamine/midazolam should be injected in one of the hind leg muscles. Once injected, the handling cone and the fisher should be covered with a cloth, ensuring the face and eyes of the fisher are covered to minimize visual stimuli and stress. Ketamine/midazolam take approximately 2-3 minutes to have effect regardless of dosage. Once it is clear that the

animal is no longer physically responsive to a slight tap of its foot, it is then removed from the cone and placed on a mat for processing.

Opthalmic ointment should be applied to both eyes of the animal because the immobilization drug inhibits blinking. The eyes should then be covered with a cloth (tubular piece of a sock) to limit the number of visual stimuli. The animal should then be weighed by gently placing it in a pillow slip and measured using a Pesola scale (PESOLA Präzisionswaagen AG, Schindellegi, Switzerland). The sex of the animal can then be determined by inspection of external genitalia. Females should be examined for signs of reproductive activity (e.g. teats examined and measured) (Frost et al. 1999, Matthews et al. 2013*a*). A systematic physical exam of the eyes, ears, nose, bones, joints, and pelage will be conducted to examine for any obvious injuries. Each fisher should be marked with a 12.5 mm passive integrated transponder (PIT) tag (Biomark FDX-B HPT12, Biomark, Boise, Idaho). The tag should be inserted beneath the skin by injection with a 10 g trocar between the scapulae so recaptured individuals can be identified without immobilization and processing.

Age is an important demographic parameter for modeling the persistence of populations of conservation concern (Wisdom et al. 2000). Photographs of the wear of the dentition will be taken and age class of the animal will be determined (i.e., juvenile, sub-adult, adult). Age estimation based on tooth wear, however, is highly variable as wear can be related to an individual's diet independent of age. A more accurate estimate of age is gained through the use of cementum annuli analysis, requiring the extraction of a vestigial, first upper premolar from each recovered fisher carcass and analyses of cementum layers (Strickland et al. 1982, Arthur et al. 1992, Poole et al. 1994). The premolar will be extracted using a dental extractor (Jorgenson Dental Extractor Small J42T). The tooth will be placed in a manila envelope, labeled, stored in a

cool, dry place, and shipped to a commercial laboratory for analysis and age estimation (Matson's Laboratory, Milltown, Montana, http://www.matsonslab.com/preparing-specimens-teeth-and-calcified-tissue.html).

Genetic variability is an important consideration in the conservation of recovering populations (e.g., Pimm et al. 2006, Sacks et al. 2010, Whiteley et al. 2015) and especially important for conservation translocations (IUCN Species Survival Commission 2013). We recommend tissue and hair samples be collected from captured fishers for genetic analysis. Tissue samples should be collected using a sterile 4-mm punch biopsy along the outer margin of the fisher's ear. Sterilized forceps should be used to remove the tissue sample. Tissue samples should be stored at room temperature in a cryogenic tube filled with 95% ethanol (labeled with the date, individual, and coordinates of the trap location). Hemostasis should be achieved using direct pressure with a 2.5 cm^2 gauze pad. Two hair samples with follicles (i.e., white bulb at the end of the hair) should be collected between the handler's thumb and index finger from 3-4 locations approximately 1.5 cm^2 each along the shoulders and back of the captured fisher. Each hair sample should contain a minimum of thirty hairs. Samples should be placed in two, labeled No. 1 coin envelopes (labeled with the date, individual, and coordinates of the trap location). The paper envelopes should be placed in a plastic bag with desiccant in the lab.

Disease is an emerging concern for wildlife populations and is an important consideration for translocations (Sainsbury and Vaughan-Higgins 2012). The collection of blood and membrane swabs provide opportunities to investigate disease presence and prevalence in wildlife populations. Whole blood samples (<6 mL) should be collected from animals for serology testing (canine distemper virus, canine parvovirus, *Toxoplasmosis gondii*) following protocols outlined by the Integral Ecology Research Center (IERC, Blue Lake, California). A 6 mL whole

blood sample represents <10% of a fisher's estimated circulating blood volume, as recommended by policy for regulating the volume of experimental blood samples. Samples should be collected via venipuncture of the jugular or femoral vein, using 23-25 gauge needles attached to 3 or 6 mL syringes. Blood samples should be separated in the field. Half of each sample (2-3 mL) should be injected and gently mixed into plastic EDTA (lavender top) blood tubes with anticoagulant. The other half (2-3 mL) should be injected into a red/grey top blood tube, allowed to clot, and centrifuged to remove the serum. Blood and serum tubes should be labeled with permanent marker and frozen.

Fecal samples should be collected and are usually available at during live-captures. If not available or obtained during initial processing, a fecal sample could be collected using a sterile Dacron swab, ensuring the end of the swab is covered with fecal material following protocols outlined by the Integral Ecology Research Center (IERC, Blue Lake, California). The sample should be placed in an airtight vial or whirl-pack, labeled, and frozen. Ocular and nasal swabs should be done using Dacron swabs. Fluid from both eye rims/lids should be sampled, being cautious not to touch eyes with swab to minimize infection or retinal scratches. Another swab should be used to wipe the inside of both nasal passages. Each swab will be placed in labeled vials and frozen.

Ectoparasite samples should be collected for disease screening following protocols outlined by the Integral Ecology Research Center (IERC, Blue Lake, California). Ticks should be removed with a tick key or fine-tipped forceps to minimize damage and preserve mouthparts for accurate identification of specimens. Fleas should be collected using a flea comb and fine-tipped forceps. Ecotoparasites should be immediately placed and stored at room temperature in

cryogenic tubes half-filled with 70% ethanol. Each tube should be labeled with a permanent marker for sample identification.

Signs of trematode infection should be examined. A previously undescribed trematode was found in fishers captured in Humboldt and western Trinity counties of California for translocation to the northern Sierra Nevada (Powell et al. 2019). No fishers infected with the trematode were translocated in the northern Sierra Nevada. Any fisher exhibiting signs of trematode infection should not be considered for translocation and released at their location of capture.

The temperature and respiration rate of anesthetized fishers should be monitored at least every five minutes. Dopram (Doxapram HCL, concentration 20 mg/mL) can be administered intravenously in the event of a respiratory emergency to stimulate respiration during anesthesia and/or to speed awakening and reflexes. Heat packs can be applied to the groin and chest and the animal is covered with blankets in the event of hypothermia (body temperature <37.8 degrees C). Isopropyl alcohol can be applied to the foot pads to promote evaporative cooling in the event of hyperthermia (body temperature >39.5 degrees C).

At the completion of the handling process, immobilized fishers should be placed back in the nest box and given at least one hour from the time immobilization drugs were administered to recover. The animal should be checked frequently by capture personnel during the recovery period to detect any complications. Fishers should be fully mobile prior to transport or release. Fishers that show signs of illness or trauma should not be considered fit for translocation and released at their site of capture. Signs of illness or trauma include, but are not limited to, abscesses, significant oculonasal discharge, diarrhea, dehydration, emaciation, grossly swollen lymph nodes, oily or abnormally rough hair, high numbers of ticks or fleas, serious traumas

including broken bones, exhibiting blindness, multiple broken or missing canines, missing limbs or toes, and serious lacerations.

Traps should be thoroughly disinfected with Roccal-D veterinary grade disinfectant following every capture or any animal sign at a trap. The leaf litter or fleece should also be replaced. Chicken should be replaced every day in ambient high temperatures in excess of 25 degrees C and once every three days in cooler conditions.

Appendix D – Costs, Benefits, and Efficacy of Different Sampling Methods

Summary

Tracking the status of rare or imperiled populations and species of conservation interest is essential to inform conservation decisions and management actions. Surveys and monitoring are essential to track the status of populations of interest, particularly in reintroduction areas to inform potential needs for augmentation, rescue, or cessation of activities. Nonetheless, monitoring animals, and particularly low-density, rare animals is both financially and temporally costly. Previous work by Moriarty et al. (2018) compared the efficacy of different non-invasive survey methods and showed that detection dog surveys provided the most cost-effective method to detect their species of interest (coastal marten, *Martes caurina humboldtensis*), but that hair snares at remote camera stations provided the most cost-effective method to collect individually-identified samples. At that time, mustelid scats were genotyped with microsatellites which generally result in lower genotyping success rates than hair samples (Moriarty et al. 2018, Green et al. 2022). SNP-panel development for fishers at the Levi lab at Oregon State University and at the U.S. Forest Service National Genomics Center for Wildlife and Fish Conservation (NGCWFC) will likely facilitate better genotyping success rates of mustelid scats (Hallerud et al. In revision).

Below, we briefly outline the expected annual costs to survey an area (465 km^2) similarly to that monitored by the Oregon State University portion of the Klamath-Siskiyou Carnivore Project via three non-invasive approaches: 1) remote camera only surveys, 2) surveys combining remote cameras, track plates, and hair snares (the current sampling strategy for the Klamath-Siskiyou Carnivore Project lead by the Institute for Natural Resources at Oregon State University and partners), and 3) grid-based detection dog surveys with Rogue Detection Teams, Inc. In

general, remote camera only surveys are the lowest-cost survey approach overall given decreased human time needed to set and maintain remote camera stations and decreased post-season data processing costs (Table 1). Costs for human-based non-invasive surveys (cameras, hair snares, track plates) and detection dog-based surveys were similar (Table 1) with the latter being about 8 cheaper than the former depending on the genetics lab used to process samples. Despite differences in costs for remote camera only vs. genetics-based monitoring, practitioners should consider the question of interest, and the potential benefits and limitations of each method (Table 2) prior to selecting their survey method.

Table 1. A non-invasive monitoring cost-comparison using the EKSA fisher project as an example. Note that "total cost" only reflects total cost to collect and process data but does not include senior staff time for study design, training, or analysis and reporting.

Method	**Season length**	**Number stations**	**Data collected**	**Collection cost**	**Sample processing cost**	**Total cost**
Remote cameras only	4 months	100 stations	250,000 images	$112,311	$10,405 for two weeks photo ID	$122,716
Cameras, hair snares + trackplates	5 months	100 stations	200-600 hair samples + tracksheets, 250,000 images	$177,180	$30,405 1. $20,000 for genetics 2. $10,405 for two weeks photo ID	$207,585
Detection dog surveys	60 sampling days	30 4 x 4 km^2 grid cells	1,000 scats	$67,532 for contract to Rogue	$98,000 to $109,000 1. NGCWFC: $100/scat for species ID and vert, invert, plant metabarcoding, $30/ scat for genotyping 2. Levi lab: $80/scat for species ID and vert metabarcoding, $60/scat for genotyping	$165,532 to $176,532

Table 2. Summary of primary sampling methods that could be used to track the status of a fisher population post-release. Data type broadly reflects the data collected via the method which dictates the models the data can subsequently be incorporated into. Parameters reflect model parameters the data collected via the respective method can be used to estimate. Data delay reflects the expected delay between the end of a sampling season and the data being finalized and ready to incorporate into analysis. Benefits briefly describe the strengths of the method, while limitations briefly describe its drawbacks (additional details in text below table).

Method	**Data type**	**Parameters**	**Data delay**	**Benefits**	**Limitations**
Remote cameras	Detection-non-detection, counts	Occupancy, abundance indices [RN, N-mixture], distribution	1-3 months	Co-occurring species, lower cost than hair snares, devices can be re-used, devices can be left out for long periods of time	Most effective when re-baited and lured, no individual ID, abundance indices less precise than true abundance, parameters not sensitive to small changes
Trackplates	Detection-non-detection	Occupancy, abundance indices [RN, N-mixture], distribution	<1 week	Co-occurring species, very low cost, potential for sex and/or individual IDs in development	Need to be rebaited, co-occurring species detections limited to mesocarnivores and small mammals, no individual IDs, no temporal component to data
Detection dog surveys	Detection-non-detection, counts, individual identities	Occupancy, abundance, density, distribution, apparent survival, apparent recruitment, fine-scale diet	3-5 months for species ID, 6-8 months for individual ID	Co-occurring species, high efficacy, less expensive than fixed monitoring by human crews, detectability less affected by individual behavior/personality, ability to estimate demographics over time	If species is very rare may end up too few samples to formally model, genotyping for fisher scat on SNPs panels still in works, grid cells typically sampled once
Hair snares	Detection-non-detection, counts, individual identities	Occupancy, abundance, density, distribution, apparent survival, apparent recruitment, coarse diet/foraging	3-5 months for species ID, 6-8 months for individual ID	Co-occurring species, individual identities, ability to estimate demographics over time	Need to be rebaited, co-occurring species detections limited to mesocarnivores and small mammals, more expensive than detection dog surveys
Live-capture	Detection-non-detection, counts, individual identities	Abundance, behavior, density, distribution, health, space-use, known-fates survival	Variable	Individual identities and characteristics, ability to collect bio samples for complementary questions, ability to track individuals over space and time	Most expensive, dispersing or released animals may be difficult to track post-release, female fishers still too small for iridium collars

Remote cameras

Data type

Remote cameras can provide detection-non-detection data and counts data.

Parameters

Data collected from remote cameras can be incorporated into a variety of models to elucidate fisher occurrence and distribution. Most commonly, detection-non-detection data can be incorporated into occupancy and species distribution models to understand occurrence-habitat relationships. Detection-non-detection data and detections formatted as counts data can further be incorporated into models that facilitate estimates of abundance indices or expected abundance (e.g., N-mixture, Royle-Nichols; Royle and Nichols 2003, Royle 2004).

Data delay

Data delays to process remote camera data depend on crew capacity and the volume of data collected. Since the EKSA project started collected remote camera data in concert with trackplate and hair snare surveys, the project has collected ~200,000 to 400,000 images per year. Typically, the crew waits until the end of the season to process photographs and identify species and typically takes approximately 1 to 2 months working part-time (0.25 - 0.50 FTE) to process all photographs. Recent work (Tucker et al. 2026) shows that modifications can be made to remote camera settings that reduce the volume of photographs while not affecting occupancy estimates which would reduce the person-time needed to process photographs. Wildlife Insights, which is currently used by EKSA for data storage and processing, also provides automated identification of photo contents (Ahumada et al. 2020). While species-level identifications for rarer species are often less reliable, the model is effective at classifying blank images (Ahumada et al. 2020).

Modifying camera settings or using a blank image filter via Wildlife Insights could reduce processing time and delays for data to finalized.

Benefits

One of the major benefits of remote cameras is their ability to detect the target species and any co-occurring animals that enter the camera's viewshed without a need for an animal to enter or interact with a sampling device. This "bycatch" can facilitate complementary analyses allowing us to understand the ecology of co-occurring prey, predators, and competitors (Burton et al. 2022) and clarifying potential biotic interactions with fishers (e.g., Rich et al. 2019, Dymit et al. 2025). Cameras can operate continuously without much maintenance over long periods of time, though their efficacy in detecting animals may decline as bait/lure dissipate. In general, camera-only sampling is less expensive given decreased time spent servicing sites and no costs associated with genetics analysis. We estimate (Table 1) that crew time would be reduced by ~20% during remote camera only surveys given crews would not have to construct, scout, deploy, and maintain trackplate boxes and hair snares. Cameras also provided a temporal signature for detections, allowing more flexible formatting and indexing for occupancy models and facilitating estimates of temporal partitioning or behavior (Kautz et al. 2021, Devarajan et al. 2025).

Limitations

One of the primary limitations of remote camera surveys for fishers is the inability to collect individual identities from camera detections. Though statistical advances allow for estimates of unmarked abundance, expected abundance, or abundance indices with detection-non-detection data, these are often less precise than "marked" methods and are sensitive to the spacing and design of sampling arrays (Linden et al. 2017, Burgar et al. 2018, Twining et al. 2022).

Similarly, using occupancy as a parameter to track the status of populations over time can be less effective when effective sampling area is low, detection probability is low, or declines are small (Tucker et al. 2021).

Trackplates

Data type

Trackplates can provide detection-non-detection data and sex ID for fisher tracks depending on timing of surveys.

Parameters

Data collected from trackplates can primarily be incorporated into models that estimate fisher occurrence and distribution. Detection-non-detection data can be incorporated into occupancy and species distribution models to understand occurrence-habitat relationships. Depending on the timing of data collection, fisher tracks can be identified to sex (Tucker et al. 2023) which can facilitate sex-specific estimates of occupancy and distribution.

Data delay

Trackplate data can be processed very quickly and there is very little delay between collection and data being finalized so long as staff time is available. For the OSU portion of the EKSA carnivore project, crews will save trackplate ID until the end of the season and identify all species on tracksheets at once. This process typically takes a crew of three people approximately two workdays (i.e., ~48 work-hours) to identify tracks from 250 to 600 tracksheets.

Benefits

The greatest benefits of trackplates are their low cost and the small amount of time needed to process data. Trackplates are very inexpensive given they are comprised of a corrugated plastic sheet (~$20), aluminum track plate (~$8), and a sheet of contact paper ($0.33) dusted with non-

toxic printer toner ($0.06). The aluminum trackplates last indefinitely, so long as they are not irreparably bent or lost due to black bears. Recent developments in modeling techniques allow for reliable identification of sex from fisher tracks (Tucker et al. 2024) though this requires sampling at a time of year when juvenile males are large enough to be discriminated from adult females. Trackplates are typically constructed to have a hair snare included as part of its structure, so another benefit is simultaneously collecting a hair sample from the animal that enters while also collecting tracks from any animals that subsequently enter.

Limitations

The biggest limitation of trackplates as a survey method for fishers is that they cannot currently be used to identify unique individuals, and as such are limited to use for estimating occupancy, distribution, or models that use detection-non-detection data to estimate indices of abundance (e.g., N-mixture, Royle-Nichols). Unlike remote cameras, detections at trackplates do not have a temporal signature (i.e., not able to discern when tracks were applied between check 1 and check 2) and are therefore not able to formatted as counts data. Tracks at trackplates can be harder to identify or distinguish when applied in very wet conditions and are most effective when visited to collect tracksheets and replace bait weekly.

Detection dog surveys

Data type

Scat samples collected via detection dog surveys can provide detection-non-detection data, counts data, and individual identities.

Parameters

Data collected via detection dog surveys can be incorporated into a variety of models to elucidate fisher demographics, ecology, and distributions. Scat samples identified to species can be used to

estimate occupancy or distribution of fishers and co-occurring species, though this is typically limited by the study design and target species identified before surveys begin. Scat samples that are successfully genotyped can be incorporated into spatial capture-recapture models (Davidson et al. 2014, Martin et al. 2023) to estimate abundance, density, apparent survival, and apparent recruitment, though the latter two parameters require at least three sampling periods to estimate. Scat samples can also be used to estimate diet patterns via metabarcoding, though these estimates are temporally fine and limited to items consumed in the couple of days before the animal defecated (Hoenig et al. 2022).

Data delay

The delay in time between when scat samples are collected and finalized data are received varies depending on the genetics lab used to process samples. Most recently, the EKSA sent carnivore scats collected in the central Oregon Cascades to the Levi lab at Oregon State University to be processed for species ID, scat contents, and genotyping. Timelines for species ID at this lab were approximately 12 months from sample submission to species ID and metabarcoding results being received and an additional 12 months from receiving species IDs to receive bobcat and cougar genotypes from the Levi lab. Genotyping timelines were likely longer than is typical as the lab was developing a new GTSeq SNP panel for genotyping these species rather than using existing RADSeq panels (Buchalski et al. 2022, Ahrens et al. 2024). The Levi lab is currently refining a GTSeq SNP panel for genotyping fishers which should be ready for use by the end of 2026. The NGCWFC is another option for submitting scats for species ID and genotyping scat samples from fishers and other co-occurring carnivores. The timeline at this lab for sample processing is anticipated to be approximately three months from sample submission to receive species IDs and

metabarcoding results and and approximately 6 to 7 months from sample submission to receive individual identifications.

Benefits

A primary benefit of detection dog surveys is their efficacy in challenging conditions and more agnostic approach to sampling individuals. Previous research shows that detection dog teams are also often more effective at successfully detection species of interest when compared to other non-invasive methods like remote cameras and hair snares (Long et al. 2007, Moriarty et al. 2018). Individual-based sampling via devices that require an animal to interact with them (e.g., hair snares, trackplates) can be affected by individual behavior, whereas detection dog teams are able to more effectively sample all individuals in a population (Long et al. 2007, Davidson et al. 2014, Thomas et al. 2020).

Limitations

When the target species is very rare, the effort per target species detection for detection dog teams can be very high and sometimes results in too few detections to formally model abundance, distribution, or occupancy. Currently, genotyping of mustelid scats, including fishers, is still challenged by the efficacy of existing techniques. Several SNP panels are in development by the Levi lab at OSU and the NGCWFC and should facilitate more effective genotyping of fisher scats by the end of 2026. In general, genotyping success rates of mustelid hair with microsatellites are still higher than genotyping success rates of scats with SNP panels.

Hair snares

Data type

Hair samples collected at hair snare devices can provide detection-non-detection data, counts data, and individual identities.

Parameters

Data collected from hair snares can be incorporated into a variety of models to elucidate fisher demographics, ecology, and distributions. Hair samples identified to species can be used to estimate occupancy or distribution of fishers and co-occurring species, though this is typically limited to similar-sized species (e.g., ringtail, gray fox) that can fit into trackplate boxes with hair snares (e.g., Green et al. 2021, 2022). Hair samples that are successfully genotyped can be incorporated into spatial capture-recapture models to estimate abundance, density, apparent survival, and apparent recruitment of fishers (Green et al. 2022), though the latter two parameters require at least three sampling periods to estimate. Hair samples can also be used to estimate diet and foraging patterns using stable isotope analysis, though these estimates are coarser and focused on functional prey groups (Manlick et al. 2017, 2019, Smith et al. 2023).

Data delay

The delay in time between when samples are collected and finalized data are received varies depending on the genetics lab used. Since its inception, the EKSA fisher project has worked with the NGCWFC. The EKSA sends samples to NGCWFC weekly, but NGCWFC waits until the final week of samples is received, typically early November of each sampling season, to begin genetic analysis. The timeline at this lab for sample processing is approximately three months from the last week of sampling to receive species IDs for all hair samples (i.e., February), and approximately 6 to 7 months from the last week of sampling (i.e., May or June) to receive individual identifications.

Benefits

Hair snares provide the opportunity to collect samples that can be used to abundance, density and distribution of fishers. Spatial capture-recapture with known individual identities is considered

the "gold standard" for estimating abundance via non-invasive methods (Royle and Chandler 2014). This approach is typically more precise than abundance indices and unmarked approaches (Linden et al. 2017, Burgar et al. 2018, Twining et al. 2022) and recent work has emphasized that individual-based monitoring is particularly important in for rare species and small populations (Gerber and Gilbert 2026). If data is collected over at least three sampling periods, ideally years, the data can be incorporated into an open population model which additionally provides the opportunity to estimate apparent survival and apparent recruitment (Green et al. 2022). Given genotyping only requries use of the hair follicle, the shafts of hairs collected via hair snares can be used to estimate diet and foraging patterns using isotope analysis (e.g., Manlick et al. 2017, 2019, Smith et al. 2023).

Limitations

One of the primary limitations of sampling for fishers via hair snares is the inherent financial and temporal cost of processing genetic samples. Financial costs and time needed for sample processing vary depending on the genetics lab that samples are sent to. Currently, the EKSA fisher project sends samples to the NGCWFC. In most years, the OSU portion of the EKSA fisher project collects 250 to 600 hair samples, with anywhere from 10 to 40% subsequently identified as fisher. The most recent budgeted cost to identify all hair samples to species and subsequently genotype samples identified as fisher to individual was approximately $20,000 per year. This cost can be higher when sent to genetics labs at academic institutions which typically have higher overhead/indirect costs. The delay in time from sample collection to data finalization, typically 3 months for species ID and 6 months for individual identities, can affect timelines for estimating abundance and density which can have downstream effects for decision making. Hair snares also rely on an animal interacting with the device, meaning that detection

probability can be affected by individual behavior (e.g., trap-shy individuals less likely to be detected, trap-happy individuals more likely to be detected). While this can be modeled on the detection probability process in a spatial-capture-recapture model, it still requires enough recaptures of the respective groups of interest (e.g., females vs. males) to explicitly account for it.

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Appendix E – Veterinary Health Assessment and Disease Screening

Fishers meeting the demographic and health criteria for translocation in the field should undergo a veterinary health assessment in a captive setting. Assessments should include 1) a physical examination to evaluate obvious signs of illness or trauma, 2) the collection of additional samples for disease screening, and 3) the administration of vaccinations. Our recommendations are modified from Callas and Figura (2008).

In the captive facility, fishers should be housed in transport crates. Crates should be constructed of 1.9 cm (0.75 inch) or thicker plywood. The floor of the cates should allow urine to drain away from captive animals. Crates should be slightly elevated and an absorbent material (cloth or hay) set under the crates to minimize transfer of urine, fecal, or other body fluids to adjacent crates. The environment at the housing facility should be very quiet, crates should be out of direct sunlight, and ambient temperatures should be controlled. The crates should allow for adequate ventilation and for the introduction of food and water during housing without risk of fishers escaping or incurring injury to personnel. Fishers should be allowed access to water (preferably *ad libitum* but at least every four hours) and fed several dead mice daily. Each fisher should be transported and housed in an individual crate out of sight of other fishers to minimize stress.

At the captive facility, a wildlife veterinarian should supervise an additional chemical immobilization of captive fishers, conduct a thorough physical examination, and collect samples for disease screening. Chemical immobilizations should follow the procedures of the captive facility and attending veterinarian. Fishers that show signs of illness or trauma should not be considered fit for translocation and released at their site of capture. Signs of illness or trauma include, but are not limited to, abscesses, significant oculonasal discharge, diarrhea, dehydration,

emaciation, grossly swollen lymph nodes, oily or abnormally rough hair, high numbers of ticks or fleas, serious traumas including broken bones, exhibiting blindness, multiple broken or missing canines, missing limbs or toes, and serious lacerations.

Biological samples taken for disease screening should include blood samples, fecal samples, ectoparasite samples, and swabs of mucus membranes. Samples should be collected during the health assessment if additional samples beyond those collected during the initial capture and immobilization are required. Sample collection should be conducted using techniques recommended by the attending veterinarian.

Fisher feces and swabs from the back of the mouth will be evaluated using PCR for current infection of canine distemper virus, canine parvovirus, and canine adenovirus. Blood may be evaluated using PCR for current infection of *Anaplasma phagocytophilum* and using indirect fluorescent antibody tests for antibodies specific for canine distemper virus, canine parvovirus, canine adenovirus, and canine herpesvirus. Fishers shedding canine distemper virus or canine parvovirus DNA or fishers found to be previously exposed to canine distemper virus should be returned to their capture site and released. Fishers determined to have been exposed to canine herpesvirus or canine adenovirus, without exposure to canine distemper virus or canine parvovirus, should be evaluated for clinical signs of disease. If other signs indicate poor health, these fishers should be returned to their capture site and released. Fishers that appear otherwise healthy should be considered appropriate for translocation, but they should be monitored during translocation and post-release for signs of disease.

Once a fisher is deemed to have met the demographic and health criteria for translocation, it should be chemically immobilized to administer vaccinations and fitted with a GPS/VHF radio transmitter (e.g., Holohil model MI-2, Telonics model MOD80, ATS model W-

500, Lotek models R40 and R60). Each translocated fisher should receive 1 ml Imrab-3 rabies vaccine injection intramuscularly and 1 ml Purevax ferret distemper vaccine injected subcutaneously. Each translocated fisher should also receive 0.2 mg/kg Ivermectin and 5 mg/kg Droncit (praziquantel) injected subcutaneously to minimize the risks of translocating nematodes, tapeworms, and flukes to reintroduction sites. Signs of trematode infection should be monitored for all captured fishers. No other vaccines should be administered unless they have a history of safe use in the *Martes* complex. Minor wounds should also be treated as necessary.