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Responses of Pacific Fishers to Habitat Changes as a Result of Forestry Practices in Southwestern Oregon

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RESPONSES OF PACIFIC FISHERS TO HABITAT CHANGES AS A RESULT OF FORESTRY PRACTICES IN SOUTHWESTERN OREGON

by

Tessa R. Smith

A thesis submitted in partial fulfillment
of the requirements for the degree

of

MASTER OF SCIENCE

in

Wildlife Biology

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ABSTRACT

Responses of Pacific Fishers to Habitat Changes as a Result of Forestry Practices in Southwestern Oregon

by

Tessa R. Smith, Master of Science

Utah State University, 2021

Major Professor: Dr. Eric M. Gese
Department: Wildland Resources

Timber practices are increasingly being implemented to improve stand resiliency to high-severity wildfires. Treatments such as thinning and selective harvests mitigate fire impacts by removing hazardous fuels and separating high-density stands. However, sudden changes in habitat quality may adversely affect wildlife specialists like the Pacific fisher (*Pekania pennanti*). As a habitat-obligate species, fisher require late-seral forests with large trees, dense canopy, and three-dimensional stand complexity. Fishers frequently use tree cavities, standing snags, and large trees with interlocking crowns, all of which form over long periods of time. Yet, it is unknown whether fishers can adapt to rapidly changing forest conditions through anthropogenic means. We explored this concept by tracking the behavior of a small population of fishers in an Ashland, Oregon, watershed during 2010-2017. Fuel reduction treatments were applied, decreasing vegetation density and canopy cover through tree removal and light understory thinning. We tracked 8 GPS-collared fishers before and after thinning treatments to elucidate fisher spatial use relative to treatment placement in each home range. Our results indicated there was individual variation in fisher tolerance levels to treatments, and that season, treatment size, and vegetation type were driving factors. Although 6 of 8 fishers had a weak negative correlation to treatment effects, we observed them utilizing untreated portions of their home ranges at a threshold limit of ~2000 m from the nearest treatment area. Additionally, we investigated habitat characteristics selected by fishers at the home range scale using resource selection functions (RSF). We performed 3 habitat RSFs on 9 individuals and separated RSFs into 2-year increments (i.e., 2011-2012, 2013-2014, and 2015-2016). Model variables included topographic and vegetation characteristics. Habitat selection differed among biennial RSFs, but elevation and ruggedness were found to be significant features in at least two of three models. Fishers were also positively associated with dense canopy cover in the first 2 RSFs; however, vegetation type replaced canopy cover as an important variable in the third RSF. Our research results concluded that fishers would use recently thinned habitat as long as sufficient overhead cover (≥50%) and high-valued resources (e.g., den sites) were retained within their home range.

(164 pages)

PUBLIC ABSTRACT

Responses of Pacific Fishers to Habitat Changes as a Result of Forestry Practices in Southwestern Oregon

Tessa R. Smith

The fisher (*Pekania pennanti*) is a medium-sized carnivore found in mature forest stands across much of the northern United States. Although historically abundant in the west, fisher populations declined rapidly after fur trapping, extensive logging, and urban development reduced their numbers. Currently, biologists are concerned about the effects timber harvest practices have on fisher tolerance and adaptability when faced with changes to high-quality habitat stands. Tree removal and thinning of understory vegetation are frequently used to alleviate the spread of wildfires in previously dense forest stands with a potential for large-scale habitat loss; yet, a deficit of large trees and important vegetation attributes could be detrimental to fisher survival. We explored the impacts of timber treatments on fisher behavior and habitat preferences in a watershed system near Ashland, Oregon between 2010 and 2017. In our study, we assessed where fishers were found in their home ranges before and after treatments occurred (i.e., measuring fisher distance to treatment units), as well as the habitat features they selected pre- and post-treatment. Our results indicated that although most fishers moved away from treated areas, they still used untreated portions of their home ranges. For habitat selection, fishers chose sites at lower elevations, with low to moderate rugged topography, and they selected moderately steep slopes. They also preferred canopy cover 60% or higher and vegetation types consisting of conifers and hardwoods. We concluded that fishers were able to tolerate ongoing treatments in their home ranges as long as adequate canopy cover and large structures remained for their use on the landscape.

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Even before attending college as an undergraduate student, I knew attaining a graduate degree was my ultimate educational goal; though at the time, I didn't understand how long that would take. As it so happens, it took 20 years of hard field work, moving 15 times to different field stations, and meeting truly amazing people that helped make that wish come true. And though experience, intelligence, and motivation are handy attributes to get one through a graduate program, I never could have achieved this without the friends, mentors, and colleagues that encouraged me throughout my time at Utah State University. First and foremost, I humbly thank my advisor, Eric Gese, who is one of the finest research biologists and life teachers I have had the privilege of knowing. Eric took a chance on me despite my doubtful GPA, my data set that I deemed "a challenging mess", and my frustratingly long timeline to complete a thesis that should have been finished, at best, 2 years ago. I cannot thank him enough for being rock solid in his support of my research and his confidence in me to finally attain this goal. He has my sincerest gratitude.

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CHAPTER 1

INTRODUCTION

Though more than a century has passed since widespread logging began in the Pacific Northwest, the harvesting of trees for commercial profit continues to be a billion-dollar industry due to a rising demand in paper production and construction lumber (Oregon Forest Resources Institute 2019). In the conterminous United States, Oregon remains the largest producer of commercial timber, with approximately 80% of wood harvested from private forests spanning across almost half the state (Simmons et al. 2016). Although Oregon also holds the distinction of being the first state to initiate land conservation efforts for sustainability purposes (Conservation Act of 1941), its landscape has nevertheless been changed by decades of extensive logging and fire suppression in the early 20th century (Oregon Department of Forestry 2009). In contrast, by volume forests have increased exponentially due in part to the uninhibited growth of stands without fire disturbance (Wells 2009). However, many of these same stands harbor fire-intolerant tree species and dense understories with heavy fuel loads, a dangerous combination for extreme wildfire conditions (Odion et al. 2004, Halofsky et al. 2020).

To combat the risk of frequent, high-intensity forest fires, managers across the Pacific Northwest have adopted a plan of strategically placed silviculture treatments. The objective is to comply with the Healthy Forests Restoration Act of 2003, which targets the return of our forests to a robust and resilient ecological landscape (Healthy Forests Restoration Act of 2003, H. R. 1904, 108th Congress, 2003). Under this law, silviculture treatments are not intended to replace the natural processes responsible for forest succession, but rather assist in maintaining the quality and character of healthy stand establishments. Silviculture practices include variable-density management, which create more open, fire-resistant stands reminiscent of historic conditions. An example of this method includes the thinning or removal of small-stemmed, fire-intolerant trees combined with a reduction in surface and ladder fuels (Agee and Skinner 2005). Decreasing the density of sapling and pole trees, along with a reduction in understory vegetation, will substantially alleviate the chances of a ground to crown wildfire, which cause the most damage to ecosystem properties (Raymond and Peterson 2005, Kalies and Yokom Kent 2016). In combination with density management, foresters also employ prescribed burns to moderate hazardous fuel loads and reinvigorate soil nutrient cycling, thus emulating the natural, low-intensity surface fires that dry forest-types experience under normal circumstances (Spies and Franklin 1991, Agee and Skinner 2005). These strategies seek to mitigate wildfire activity in high-risk forests, which have encountered hotter, drier summers and increasingly limited snowpack levels in the last several decades (Halofsky et al. 2020). Though these silviculture practices will improve forest conditions in the long term, it is probable the changes in surrounding habitat occur more rapidly than natural succession. With key structures suddenly removed and the reconfiguration of three-dimensional stand complexity, biologists are concerned wildlife may have little time to adapt to their new living arrangements (Bull et al. 2001, Zielinski et al. 2013). These habitat transformations could lead to species experiencing impaired fitness, occupancy decline, lost foraging opportunities, or reproductive failure if the environment contains limited structures and resources necessary to support healthy wildlife populations.

Further research is needed to evaluate the impact silviculture treatments would impose on wildlife survival, distribution, and behavior. The Pacific fisher (*Pekania pennanti*), a habitat-obligate species associated with mixed-aged and late-seral forests, has been selected for observation as fuel-reduction treatments are implemented in a portion of southwestern Oregon. Like the Northern spotted owl (*Strix occidentalis*), the fisher is considered a key species in forest health due to its close ties with mature stands and its diverse resource needs (Buskirk and Powell 1994). A medium-sized carnivore, the fisher requires habitat characterized by large-diameter trees, multi-layered canopies, and riparian drainages filled with fallen logs, heavy underbrush, and rock piles for searching out small prey (Powell and Zielinski 1994). Cavity chambers found in live and dead trees are also incredibly important physical features in terms of reproductive and resting habitat for fishers (Lofroth 2010, Green 2017). These unique structures may take decades, or even centuries to form, and are often sparsely distributed throughout a fisher home range (Purcell et al. 2009, Weir et al. 2012, Delheimer et al. 2019). Although mature forest conditions are crucial to fisher survival and fecundity, they have nevertheless been observed in alternative habitats for foraging or exploratory purposes (Sauder and Rachlow 2015). For instance, timber-managed stands, open shrub areas, and low-elevation pine forests are a few examples of disturbed, or natural, plant communities set within mature forest fisher habitat. Though limited in range and extent, these alternative fisher environments provide additional food opportunities that may not be found in denser forests (Swanson et al. 2011). A study by Golightly et al. (2006) discovered that the Klamath fisher population had a more variable diet than any other region, with the exception of the southern Sierra Nevada group which was equally diverse (Zielinski et al. 1999). Fishers in the Klamath bioregion foraged in multiple ecological zones, including oak woodlands and harvested stands where tree squirrels and woodrats resided, respectively. Moreover, fishers were found to hunt snowshoe hares and mountain beavers, two species closely associated with younger, managed timber stands (Parsons et al. 2020). Despite the wide diversity of prey found within unconventional habitats, these areas may offer few established structural resources large or old enough to support the long-term needs of fisher breeding (Zhao et al. 2012, but see Niblett et al. 2017). Thus, fisher conservation is complicated as it necessitates protecting a variety of habitat types, many of which may be vulnerable to fragmentation from urban, economic, and recreational development as well as fuel reduction efforts.

Considering the fisher's comprehensive needs, an abrupt change in habitat conditions through anthropogenic activities suggests they could be negatively impacted by silviculture treatments, specifically where mature resting and denning structures are suddenly removed. Consequently, fishers might abandon or shift their home range to find suitable food and shelter, or be forced to remain in areas of reduced habitat quality if conspecific competition limits movement to higher-quality areas. In related studies, a decline in habitat conditions has been proposed as a source of decreased fitness potential in other habitat-obligate species, such as the Pacific marten (*Martes americana*) and the Northern spotted owl (Tempel et al. 2014, Moriarty et al. 2016). A reduction in reproductive output, site fidelity, and foraging rates of owls and martens occurred when their habitat was modified by heavy extraction and treatment operations applied in formerly dense forests (Moriarty et al. 2016, Ganey et al. 2017, Gallagher et al. 2019). Similarly, the scope and intensity of treatments may reveal negative response behaviors of fishers (e.g., erratic movement patterns or home range shifts), especially if a large proportion of trees or canopy cover are removed in the area. Yet, lightly applied silviculture applications might be tolerated by fishers, particularly because of their observed presence in early seral and mixed-aged stands (Lewis et al. 2016). Thus, for fisher, the impacts of habitat transformations may be more nuanced due to their extensive use of integrated ecotones and opportunistic foraging behavior (Powell 1994).

In southwestern Oregon, fisher research had previously been conducted using camera surveys and track plate studies (Barry 2018, Green et al. 2018). To our knowledge, no studies had explored the effects of silviculture treatments in this area on radio-collared fishers. Therefore, we investigated fisher habitat preferences and behavior given that managers were interested in fisher tolerance to treatment activities planned in the Ashland, Oregon watershed region. Our main objectives included evaluating the behavior response and habitat selection of fisher to silviculture treatments, with the following research questions:

1. Using spatial distance as a proxy for behavioral responses, are fishers tolerant of silviculture treatments within their home range?
2. What are the habitat preferences of fisher in the Ashland, Oregon watershed area?
3. Do fisher habitat preferences vary as vegetation changes through time?

Answering the proposed questions will contribute new information in fisher ecology that can help improve forest restoration plans by conserving essential habitat components for regionally sustainable fisher populations while reducing the risk of large-scale habitat loss. In addition, we will gain knowledge regarding the threshold at which fishers can tolerate anthropogenic changes affecting habitat quality. Thus, land management strategies can be modified at spatial and temporal scales in order to balance the protection of vital fisher habitat with the safety of communities and other resource values.

LITERATURE CITED

Agee J. K., and C. N. Skinner. 2005. Basic principles of forest fuel reduction treatments. Forest Ecology and Management 211:83–96.

Barry, B. R. 2018. Distribution, habitat associations, and conservation status of Pacific fisher (*Pekania pennanti*) in Oregon. Thesis, Oregon State University, Corvallis, Oregon.

Bull, E. L., K. B. Aubry, and B. C. Wales. 2001. Effects of disturbance on forest carnivores of conservation concern in Eastern Oregon and Washington. Northwest Science 75:180–184.

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

Delheimer, M. S., K. M. Moriarty, M. A. Linnell, and B. V. Woodruff. 2019. If a tree falls in a forest: implications of forest structure persistence for the Pacific marten (*Martes caurina*). Ecosphere 10(8): e02819.

Gallagher, C. V., J. J. Keane, P. A. Shaklee, H. A. Kramer, and R. Gerrard. 2019. Spotted owl foraging patterns following fuels treatments, Sierra Nevada, California.

Journal of Wildlife Management 83:487–501.

Ganey, J. L., H. Y. Wan, S. A. Cushman, and C. D. Vojta. 2017. Conflicting perspectives on spotted owls, wildfire, and forest restoration. Fire Ecology 13:146–165.

Golightly, R. T., T. F. Penland, W. J. Zielinski, and J. M. Higley. 2006. Fisher diet in the Klamath/North Coast bioregion. Unpublished report, Department of Wildlife, Humboldt State University, Arcata, California.

Green, D. S., S. M. Matthews, R. C. Swiers, R. L. Callas, J. S. Yaeger, S. L. Farber, M. K. Schwartz, and R. A. Powell. 2018. Dynamic occupancy modelling reveals a hierarchy of competition among fishers, grey foxes, and ringtails. Journal of Animal Ecology 87:813–824.

Green, R. E. 2017. Reproductive ecology of the fisher (*Pekania pennanti*) in the southern Sierra Nevada: an assessment of reproductive parameters and forest habitat used by denning females. Ph.D. dissertation, University of California, Davis.

Halofsky, J. E., D. L. Peterson, and B. J. Harvey. 2020. Changing wildfire, changing forests: the effects of climate change on fire regimes and vegetation in the Pacific Northwest, USA. Fire Ecology 16(4). https://doi.org/10.1186/s42408-019-0062-8.

Kalies, E. L., and L. L. Yocom Kent. 2016. Tamm Review: are fuel treatments effective at achieving ecological and social objectives? A systematic review. Forest Ecology and Management 375:84–95.

Lewis, J. C., K. J. Jenkins, P. J. Happe, D. J. Manson, and M. McCalmon. 2016. Landscape-scale habitat selection by fishers translocated to the Olympic Peninsula of Washington. Forest Ecology and Management 369:170–183.

Lofroth, E. C., C. M. Raley, J. M. Higley, R. L.Truex, J. S. Yaeger, J. C. Lewis, and P. J. Happe. 2010. Conservation of fishers (*Martes pennanti*) in South-Central British Columbia, Western Washington, Western Oregon, and California-Volume I: conservation assessment. U.S. Department of the Interior Bureau of Land Management, Denver, Colorado, USA

Moriarty, K. M., C. W. Epps, and W. J. Zielinski. 2016. Forest thinning changes movement patterns and habitat use by Pacific marten. Journal of Wildlife Management 80:621–633.

Niblett, M. R., R. L. Church, S. H. Sweeney, and K. H. Barber. 2017. Characterizing habitat elements and their distribution over several spatial scales: The case of the fisher. Forests 8:186. doi:10.3390/f8060186.

Odion, D. C., E. J. Frost, J. R. Strittholt, H. Jiang, D. A. DellaSala, and M. A. Moritz. 2004. Patterns of fire severity and forest conditions in the Western Klamath Mountains, California. Conservation Biology 18:927–936.

Oregon Department of Forestry. 2009. Forest facts: Forest practices [Fact sheet]. https://www.oregon.gov/odf/Documents/aboutodf/ForestPracticesFactsheet.pdf.

Oregon Forest Resources Institute. 2019. Oregon’s Forest Economy. 2019 Forest Report: An economic snapshot of Oregon’s Forest Sector. http://theforestreport.org/wp-content/uploads/2019/07/OFRI-2019-EconSummary-Web.pdf.

Parsons, M. A., J. C. Lewis, J. N. Pauli, T. Chestnut, J. I. Ransom, D. O. Werntz, and L. R. Prugh. 2020. Prey of reintroduced fishers and their habitat relationships in the Cascades Range, Washington. Forest Ecology and Management 460:117888. https://doi.org/10.1016/j.foreco.2020.117888.

Powell, R. A. 1994. Effects of scale on habitat selection and foraging behavior of fishers in winter. Journal of Mammalogy 75:349–356.

Powell, R. A., and W. J. Zielinski. 1994. The fisher. Pages 38-73 *in* L. F. Ruggiero, K. B. Aubry, S. W. Buskirk, L. J. Lyon, and W. J. Zielinski, editors. The scientific basis for conserving forest carnivores: American marten, fisher, lynx, and wolverine in the western United States. General Technical Report. GTR RM-254. U.S. Department of Agriculture, Forest Service, Rocky Mountain Forest and Range Experiment Station. Fort Collins, Colorado.

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

Raymond, C. L., and D. L. Peterson. 2005. Fuel treatments alter the effects of wildfire in a mixed-evergreen forest, Oregon, USA. Canadian Journal of Forest Research 35:2981–2995.

Sauder, J. D., and J. L. Rachlow. 2015. Forest heterogeneity influences habitat selection by fishers (*Pekania pennanti*) within home ranges. Forest Ecology and Management 347:49–56.

Simmons, E. A., M. G. Scudder, T. A. Morgan, E. C. Berg, and G. A. Christensen. 2016. Oregon's forest products industry and timber harvest 2013 with trends through 2014. General Technical Report. PNW-GTR-942. U.S. Department of Agriculture, Forest Service, Pacific Northwest Research Station, Portland, OR. 58 p.

Spies, T. A., and J. F. Franklin. 1991. The structure of natural young, mature, and old-growth Douglas-fir forests in Oregon and Washington. General Technical Report PNW-GTR-285. U.S. Department of Agriculture, Forest Service, Pacific Northwest Research Station, Portland, Oregon.

Swanson, M. E., J. F. Franklin, R. L. Beschta, C. M. Crisafulli, D. A. DellaSala, R. L. Hutto, D. B. Lindenmayer, and F. J. Swanson. 2011. The forgotten stage of forest succession: Early-successional ecosystems on forest sites. Frontiers in Ecology and the Environment 9:117–125.

Tempel, D. J., R. J. Gutierrez, S. A. Whitmore, M. J. Reetz, R. E. Stoelting, W. J. Berigan, M. E. Seamans, and M. Z. Peery. 2014. Effects of forest management on California spotted owls: implications for reducing wildfire risk in fire-prone forests. Ecological Applications 24:2089–2106.

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

Wells, G. 2009. What do Western Oregon’s forests look like after a century of management? [Infographic]. Oregon Forest Resources Institute. Portland, Oregon. Retrieved from https://oregonforests.org/sites/default/files/2017-05/What_do_western.pdf.

Zhao, F., R. A. Sweitzer, Q. Guo, and M. Kelly. 2012. Characterizing habitats associated with fisher den structures in the Southern Sierra Nevada, California using discrete return lidar. Forest Ecology and Management 280:112–119.

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

Zielinski, W. J., C. M. Thompson, K. L. Purcell, and J. D. Garner. 2013. An assessment of fisher (*Pekania pennanti*) tolerance to forest management intensity on the landscape. Forest Ecology and Management 310:821–826.

CHAPTER 2

DETERMINING SPATIAL RESPONSES OF PACIFIC FISHER (*PEKANIA PENNANTI*) TO SILVICULTURE TREATMENTS IN SOUTHWESTERN OREGON

ABSTRACT

Historical forestry practices such as fire suppression and heavy timber logging have contributed to a discernable change in stand composition and distribution of western forests in the United States. Current forest conditions are now comprised of a tinderbox mixture of increased surface and ladder fuels, dense stands, and fire-intolerant species, all of which can ignite into a full conflagration from one errant spark. Forest managers are mitigating this concern by implementing silviculture practices such as selective logging, thinning, prescribed burning, and canopy closure reduction to remove high fuel loads and open the understory to improve stand resiliency. However, interest in the impacts to habitat specialists, such as the Pacific fisher (*Pekania pennanti*), have arisen because they may be negatively influenced by subtle modifications to their environment. To address this issue, we initiated a study from 2010-2017 in the municipal watershed of Ashland, Oregon, to determine the behavioral response of fishers to a diverse array of fuel reduction polygon treatments applied in forested stands. We measured the distance of each location from 8 GPS-collared fishers to all treatment polygons before and after they were treated within each home range, and performed three statistical tests for robustness, including multi-response permutation procedure (MRPP), chi-squared test of independence, and a Kolmogorov-Smirnov assessment. We found there was high

variation between individuals in regards to tolerance of habitat manipulation. Using effect size to interpret the magnitude of fisher response to pre- and post-treatment effects, we found that one fisher showed a moderate negative relationship to fuel reduction treatments, five exhibited a weak negative response, and two had a weak positive association with treatments. We performed analysis of variance (ANOVA) regression tests on the three fishers that exhibited the largest effect sizes to treatment disturbance, and used treatment, temporal, and habitat covariates to explore if these factors may have influenced behavioral differences. We found that season and vegetation class were two of the mutually-shared factors in the pre-treatment period influencing response distance. Post-treatment variables that elicited a negative treatment response were season and treatment size, and results were slightly different when parsing out individual effects than from a pooled sample set. Our findings suggest that the seasonal timing and location of management activities could influence fisher movement throughout their home range, but it is largely context-dependent based on the perceived risks or benefits to individuals. Finally, because of individual variation observed between fisher behavioral patterns, our research investigating population-level responses was difficult to interpret. We suggest that future research should review how environmental stressors impact dynamic interactions of fisher behavior, such as changes in predator dynamics, reallocation of prey resources, and removal of crucial denning structures for females. If additional long-term monitoring efforts and preservation of important habitat elements are maintained, forest managers and researchers can gain substantial insight on threshold limits of fisher tolerance to changing anthropogenic environments.

INTRODUCTION

The Pacific fisher (*Pekania pennanti*), a mesopredator in the family Mustelidae, is primarily associated with late-successional forests in the western United States (Powell and Zielinski 1994, Zielinski et al. 2004, Thompson et al. 2011). A few common attributes in these mature forest communities include a mixture of conifer and deciduous trees, multiple canopy layers, high woody biomass, and a complex arrangement of vertical structures (Buskirk and Powell 1994). Previous research indicates that in late-seral forests, fishers use an assortment of physical features at variable scales for different activities. For instance, at the forest-level scale, successful home range establishment occurs where large diameter trees, standing snags, riparian drainages, and continuous overhead cover are spread throughout a heterogeneous landscape of contrasting ecotones (Buskirk and Powell 1994, Powell and Zielinski 1994, Davis et al. 2007, Lofroth et al. 2010). On a finer scale, fishers use homogenous stands within their home range, which offer foraging opportunities, rest areas, and travel corridors to habitat patches with similar resources. Finally, on a microsite level, fishers choose specific structures within habitat patches that provide security during resting and denning, or while evading predators. Often fishers are located in distinctive structures that take decades to form, such as hollow tree cavities, broad tree limb platforms, or mistletoe broom clusters in large trees (Purcell et al. 2009, Green 2017). All of the features selected at coarse and fine scales are vital to the sustainability of fisher populations in the West; however, management objectives that remove or modify critical habitat properties may conflict with fisher conservation efforts (Thompson and Purcell 2015). As such, recommendations are needed to assess the habitat structures essential for fisher survival and their behavioral response to forest restoration practices as environmental conditions change.

According to Lofroth et al. (2010), the historical distribution of fishers once spanned throughout most of North America's coniferous and mixed forests. However, their range has contracted considerably in the United States and currently consists of only a small portion of its former extent (Powell 1993, Gibilisco 1994). The reduction in the western population of fishers has been the most troubling, as reproductive recruitment and genetic flow between subpopulations remains low (Wisely et al. 2004, Zielinski et al. 2005, Tucker et al. 2012). Several factors have been posited for limiting population growth and dispersal. In the early 1900's, overharvesting of coveted fisher pelts extirpated many western populations (Aubry and Lewis 2003, Zielinski et al. 2013). Concurrently, widespread logging operations also fragmented fisher habitat, resulting in fewer dispersal events and a loss in genetic diversity (Davis et al. 2007). More recently, the planting of illegal marijuana gardens in the Pacific states has created an alarming dilemma, where fishers and other wildlife are exposed to numerous toxic substances at these gardens. The direct and indirect consumption of these toxins impairs fisher survival and can negatively affect fecundity rates in isolated populations (Gabriel et al. 2015).

Although fur trapping is currently regulated or banned in the western states (Lewis and Zielinski 1996), genetic diversity and population growth remain lower than expected between populations, leading biologists to speculate if suboptimal habitat and human disturbance are the principal factors now limiting fisher demographics (Lacy 1997, Wisely et al. 2004). In particular, forest composition across fisher ranges has changed considerably in the last century due to the combination of extensive timber extraction and fire suppression (Zielinski et al. 2013). Late successional forests are now comprised of fewer large diameter trees, reduced amounts of coarse woody debris, and limited understory flora (Hessburg et al. 2005). Long fire-return intervals coupled with intensive logging practices have resulted in high fuel loads and dense, homogeneous stands with small-stemmed trees and fire-intolerant species (Agee and Skinner 2005). These current landscape conditions are the remnants of historical practices and management policy that now produce an elevated risk of frequent, severe wildfires in the West. Thus, mitigation to resolve this problem involves forest managers advocating for a more fire-resilient ecosystem using fuel reduction methods. Nevertheless, these anthropogenic changes may remove critical habitat requirements that fishers rely on to hunt, travel, rest, and reproduce.

To investigate whether restrictions in fisher space use are influenced by habitat manipulation, we initiated a study within a protected watershed of southwestern Oregon. The Ashland Forest Resiliency (AFR) project, in part with the nationwide Healthy Forest Initiative, proposed strategically placed applications of fuel reduction treatments in select forest stands. The AFR's main objectives included reducing surface and ladder fuel loads along with the creation of fire-adapted stands, ultimately producing forest stands with natural fire suppression conditions. Additionally, the project aimed to protect unique biological resources found in the Ashland watershed such as rare botanical sites and late-successional reserve habitat where refugia existed for native species. (U.S. Forest Service 2005). Indeed, local endemism is pronounced in the area, with several biotic species restricted to site-specific habitat patches, including the fisher (Olson et al. 2012). High-quality habitats, such as those found in the AFR watershed, were at greatest risk of loss if severe wildfires spread throughout the area. Therefore, management priority was to safeguard these refugia sites, along with the forest community at large, and managers were urged to employ well-established policies of stand manipulation methods (i.e., silviculture practices). Principally, thinning in conjunction with prescribed burns were used to decrease the risk of crown fire and reduce fuel loads in densely occupied tree stands (Agee and Skinner 2005). By removing a predetermined amount of trees to reduce bulk density and eliminate surface/ladder fuels, silviculture methods can slow fire spread and repress crown scorch in severe fire situations (Raymond and Peterson 2005, McIver et al. 2013, Kalies and Yokom Kent 2016). These prescription treatments may be necessary to modify fire behavior, but we wanted to ascertain whether simplifying the stand structure had potential consequences on space use by fishers.

Although research describing the implications of management activity on habitat-obligate species remain deficient in the historical literature, recent studies have focused on how silviculture methods and wildfire affect the behavioral response of sympatric species occupying late seral forests (Fisher and Wilkinson 2005). For example, researchers reported that old-growth specialists, such as the spotted owl (*Strix occidentalis*), exhibited negative response behavior to restoration activities in areas where overhead cover was substantially reduced (Tempel et al. 2014, Ganey et al. 2017). In contrast, owl pairs displayed mixed reactions to variable wildfire intensities, suggesting positive use of heterogeneous habitat created through natural mixed-severity disturbances (Ganey et al. 2017). Research on the American marten (*Martes americanus*), another habitat specialist, consistently reported negative associations with timber harvests. Martens were found to avoid high-contrast edges adjacent to logged stands (Hargis et al.

1999, Sirén et al. 2016), and circumvented openings in recently cut stands consisting of lower basal area and canopy cover (Soutiere 1979, Fuller and Harrison 2005, Moriarty et al. 2016). Similarly, researchers examined potential management effects on fisher occupancy, habitat use, and survivorship. A long-term monitoring project in the Sierra National Forest, California, concluded that fisher occupancy and annual survival were lower in areas with fuel reduction treatments (Sweitzer et al. 2016). However, research assessing the direct impacts of restoration activities (e.g., thinning, burning) on important habitat components concluded fishers may tolerate ground-disturbing events temporarily, depending on the extent or intensity of the treatment (Truex and Zielinski 2013, Zielinski et al. 2013). Notably, several researchers used theoretical models to assess how habitat quality changed after varying levels of stochastic events, both natural and man-made. The results of these models inferred that temporary trade-offs in local habitat loss from restoration activities warranted consideration as it would reduce the possibility of more widespread ecological damage. In other words, habitat quality and heterogeneity would potentially recover faster during the revegetation process after thinning than what would occur after a stand-replacing wildfire, the repercussions of which included reduced habitat quality over larger landscapes and longer timeframes (Scheller et al. 2011, Zielinski et al. 2012).

A few empirical studies have recently concentrated their focus on the specific impact fuel reduction practices have on fisher habitat preferences (Scheller et al. 2011, Thompson et al. 2011, Zielinski et al. 2013), with additional insight on long-term cumulative effects to local persistence of fisher populations (Sweitzer et al. 2016). Nevertheless, little work has been completed in determining whether fisher will significantly change their behavior in a before/after context due to disturbance events, or if vegetation removal as a result of silviculture activity will affect which structural features fishers use within their home range. Intending to narrow this knowledge gap concerning fisher behavior, our first objective was to examine whether the effects of mechanical fuel treatments discouraged fisher use near those management areas. We hypothesized that before treatments occurred (pre-treatment), fishers would use their entire home range, including the treatment areas. After treatments were applied (post-treatment), we expected them to avoid commercially treated areas up to a particular threshold distance; however, they would continue to move through the noncommercial units within their home range. Agee and Skinner (2005) described commercially treated stands as sites where mechanical methods (i.e., logging, thinning) removed large, healthy trees to reduce canopy density while providing economic profit for the agency or company. In contrast, noncommercial treatments were less intensive; the process left high crown canopy untouched but reshaped the understory by extracting shrubs, downed wood, and small saplings of no economic value. Both methods affected overhead cover at different levels and intensities, and a decline in canopy shelter may affect wildlife distributions in a spatial or temporal context. For fisher in particular, Truex and Zielinski (2013), and Olson et al. (2014), stated that dense canopy cover was the principal habitat element required when fishers dispersed or established home ranges. Therefore, in addition to exploring the relationship between the positional distances of a fisher to a treated unit, we further examined what biological and anthropogenic variables (e.g., canopy cover or treatment type) might drive fisher movement towards or away from a treatment disturbance.

STUDY AREA

The study was conducted in a municipal watershed near Ashland in southwestern Oregon. As part of the Siskiyou mountain range, the 6,300-ha protected site sits within two national forests: Rogue River-Siskiyou and Klamath National Forests. Land is jointly owned and managed by the U.S. Forest Service, Bureau of Land Management, City of Ashland, and private individuals for the purpose of maintaining recreational, commercial, and municipal resources (U.S. Department of Agriculture 2008). Topographical relief varies extensively across the study site; moderate inclines near the urban interface quickly transition to steep slopes (up to 70%), with riparian corridors emerging from deep drainages. Elevations in the watershed range from approximately 600 to 2000 m above sea level, with Mount Ashland as the highest peak in the area. Climate is typically Mediterranean with warm, dry summers, and cool, wet winters. The majority of precipitation occurs in the winter (November-March) as heavy rain or snow, with the summer often having extended periods of drought (Franklin 1972). Recognized by the World Wildlife Fund as a biologically unique region for vegetation (DellaSala 2006), the research area contains a floristic diversity closely tied to the surrounding elevation, soil, and moisture gradients. Habitat classifications are characterized by plant association groups, in which vegetation communities are organized by dominant tree or understory species (Agee 1993). At higher elevations, mixed-conifer zones support stands of Douglas-fir (*Pseudotsuga menziesii*), white fir (*Abies concolor*), ponderosa pine (*Pinus ponderosa*), sugar pine (*Pinus lambertiana*), and incense cedar (*Calocedrus decurrens*). As elevation decreases, hardwood components such as Pacific madrone (*Arbutus menziesii*), California black oak (*Quercus kelloggii*), and Oregon white oak

(*Quercus garryana*) are interspersed with conifers. Below tree canopy level, understory flora usually consists of a variety of shrubs, grasses and forbs, with ocean spray (*Holodiscus discolor*), *Ceanothus* spp., manzanita (*Arctostaphylos* spp), hazel (*Corylus cornuta*), dogwood (*Cornus nuttallii*), *Ribes* spp., Oregon grape (*Mahonia aquifolium*), poison oak (*Toxicodendron diversilobun*), serviceberry (*Amelanchier alnifolia*), and blackberry (*Rubus ursinus*) as predominant species.

Because the Ashland community strives to protect the unique biotic and abiotic environments existing in the Siskiyou range, timber management has been limited in the area over the last century. Remnants of past clear cuts are barely visible on the outer edges of the watershed boundary, whereas small-scale timber harvests scattered throughout the interior have resulted in stand successional stages of varying degrees (U.S. Forest Service 2005). While historical logging practices have contributed to shaping forest structure and complexity, the primary mechanistic force for landscape change is mainly from wildfires. Fire return intervals for the region are generally 5 to 75 years; however, fire regimes are difficult to assess in the Siskiyou mountain range because of the distinctive vegetation and topographic variation. Interestingly, the region experiences more lightning strikes during storms than any other forest in the Pacific Northwest (Agee 1993). Past wildfires in the watershed occurred in 1959, 1992, 1994, 2001, 2002, and 2018 with lightning strikes cited as the main source of ignition (Schilling 2009, Cox 2018).

METHODS

Capture, Handling, and Monitoring

Fishers were captured, radio-tagged, and tracked within the Ashland watershed boundary between 2010 and 2017. Trapping occurred in late fall and mid-winter of each year, with limited trapping in late summer months to replace failing transmitters on targeted individuals. Spring and early summer trapping were not feasible due to critical breeding and reproductive periods for female fishers (Green 2017). During the first year, traps were placed across the entire watershed in all representative habitats to assess fisher distribution and document "hot spots" for future trapping. Duration of trapping lasted between 1-3 weeks depending on personnel availability and extenuating circumstances (e.g., prescribed fire closures or inaccessible roads). We set traps within 100 m of roads to increase trap check efficiency and reduce hazards for personnel safety when working on steep slopes. Tomahawk traps (Model 108, Tomahawk Live Trap, Hazelhurst, WI) were used with a modified cubby box attached to the cage for added stability, animal safety, and weather protection (Wilbert 1992, Seglund 1995). During winter, an additional layer of corrugated plastic (Coroplast, Vanceburg, KY) covered the traps to prevent rain or snow from accumulating inside the cage. We baited traps using raw chicken hung behind the treadle and an assortment of scent lures (Hawbaker's Fisher Lure, Hawbaker and Sons, Fort Loudon, PA; Fisher Red Lure, Proline Lures, Indianapolis, IN) smeared on the bait. Long-distance call lures, such as Cavin's Gusto (Minnesota Trapline Products, Pennock, MN) and Outreach (Proline Lures, Indianapolis, IN), were liberally applied to several trees to serve as broadcast scent signals. Traps were checked every morning and occasionally moved to different locations if bears or insects were problematic to trap success or animal welfare. Trapping and capture protocols were followed by U.S. Forest Service technicians and approved through the Kings River Fisher Project Institutional Animal Care and Use Committee (No. 2018-01), California Department of Fish and Wildlife permit (No. SC-5479, SC-2730), and Oregon Department of Fish and Wildlife permit (2017 029-17). All field methods for animal handling procedures were in accordance with guidelines set by the American Society of Mammologists (Sikes et al. 2011).

Once captured, fishers were coaxed into a handling cone and anesthetized with a mixture of ketamine (22.5 mg/kg) and diazepam or midazolam (0.125 mg/kg) administered via hand syringe. Collection of biological data included morphometric measurements, swab samples for pathogen detection, blood draws for epidemiological analysis, and pulled hair follicles or ear tissue biopsies for genetic sampling. We inserted a passive integrated transponder (PIT) tag (Biomark, Boise, ID) subdermally to uniquely mark individuals. Fishers were sexed, then aged based on sagittal crest development, tooth wear, and weight condition into one of three age classes: juvenile, subadult, or adult (Sauder and Rachlow 2013, Green 2017). Adult fishers were fitted with either micro-GPS radio collars (various Quantum models, Telemetry Solutions, Concord, CA) or VHF (very high frequency) radio collars (Holohil System Ltd., Carp, Ontario, Canada). Once processing was completed, we released fishers at their original capture location and tracked them 24 hours later to confirm successful recovery.

Experimentation with GPS-collar scheduling occurred sporadically throughout the study period because of malfunctioning transmitters or terrain interference with satellite fixes. However, the majority of transmitters were programmed to attempt satellite fix locations every 10 hours on average in order to extend battery life. We estimated GPS-collar accuracy by placing two test collars in contrasting habitats for 48 hours and calculating the mean distance between the collar locations and known GPS location from a handheld Garmin. Monitoring of animals occurred once a week using ground telemetry to determine home range extent and assess survival status. Every 3-4 weeks, we tracked GPS-collared individuals within 100 m of the animal's position and remotely downloaded stored locations onto a laptop computer.

Data Preparation

We prepared data for analysis by filtering GPS locations to remove outliers, autocorrelated observations, and inaccurate positions. We reduced temporal autocorrelation for each animal by retaining locations collected ≥10 hours apart. We retained all three-dimensional locations and those two-dimensional locations with HDOP values ≤7, which preserved sufficient sample size and precision quality for our analysis tests (Dussault et al. 2001, Lewis et al. 2007, Recio et al. 2011).

Home Range Estimates and ArcGIS Analysis

Home range delineations for each fisher were derived using the Minimum Convex Polygon (MCP) method calculated by the program Geospatial Modeling Environment (http://www.spatialecology.com/gme). We calculated the 100% MCP for each individual fisher territory. To determine the minimum number of locations required for home range estimates, we found that approximately 25 locations per fisher were needed to reach the asymptote of the area-observation curve for home-range size (Haines et al. 2009).

Treatment polygons were defined for management activity by AFR forestry personnel as part of their environmental impact statement, and we obtained this data from them after treatments were completed. We imported the polygons into ArcGIS 10.3.1 (Environmental Systems Research Institute [ESRI], Redlands, CA) and executed the Clip tool to intersect treatment polygons within each fisher territory. In order to determine the behavioral tolerance of a fisher to a treated unit before and after mechanical operations, we generated distance measurements using the GPS locations of each fisher to all respective treatment units inside the home range. Distance to the nearest edge of a treatment polygon was used instead of the centroid since the entire unit was manipulated. Because multiple treatments occurred simultaneously or within short periods of one another, our approach of calculating the distance to each treatment unit instead of the nearest unit decreased our chance of violating assumptions of independence between sampling observations. Our next step involved separating locations into pre- or post-treated points with respect to the date of the treatment. Dates acquired from operation managers were limited to a month/year format; therefore, data were filtered to exclude animal locations <30 days of a unit being treated. Finally, the tables for distance measurements and treatment attributes were spatially joined in ArcGIS to connect all corresponding information.

Statistical Analyses

We focused our data inquiry on mechanical treatments, where tree density management and canopy closure reduction were the primary objectives. Subsequent pile burns to clean up ground debris and reduce surface fuels occurred in the same units; however, we did not examine fisher response to this method due to the difficulty in interpreting spatially related multiple effects. Histogram distributions for pre- and post-treatment observations for each fisher revealed highly skewed data, so we assessed whether unequal variances existed in the data between the periods using Levene's test (Levene, 1960). Because variances were unequal in a majority of the sampling data, we performed nonparametric tests for statistical analyses. However, we realized that large amounts of data using nonparametric tests often lead to small p-values, indicating significance when it may not exist (i.e., false positive). Therefore, we addressed this predicament by performing three different analyses to compare statistically significant results, but did not make conclusions based on p-value outcomes as they can be confounded by their dependence on sample size (Sullivan and Feinn 2012). Instead, we represented the substantive strength of the relationship between the treatment periods and fisher distances by measuring the effect size, which essentially quantifies the magnitude of the difference between pre- and post-treatment distances.

Three different statistical tests (multi-response permutation procedure, Kolmogorov-Smirnov test, and chi-squared test of independence) were used to compare the distributional response of each fisher to pre- and post-treatment applications, using Euclidean distance from a treated unit as a proxy for behavioral responses. The null and alternative hypotheses were the same for all tests:

H_0: The data follow the same distribution.

H_A: The data do not follow the same distribution.

The Kolmogorov-Smirnov (K-S) test evaluated whether the empirical cumulative distribution functions (ECDF) of sampled distances pre- and post-treatment period differed from one another (Smirnov 1939). A K-S statistic was calculated based on the maximum vertical distance between the pre- and post- treatment ECDF curves with significance levels (α = alpha) generated using the test statistic, D.

We used the multi-response permutation procedure (i.e., MRPP, McCune and Grace 2002) to analyze the differences in the means between pre- and post-treatment distances for each fisher. The MRPP test computed pairwise distances between all observations within each group (i.e., pre- or post-treatment distance measurements) and calculated a weighted average for each group (Cai 2006). The test generated a test statistic, delta, to determine dissimilarities between the groups. We compared an observed delta to an expected delta to produce an approximate Pearson type III *P* value. Expected deltas were obtained by permuting all possible combinations of each group and then recalculating the weighted average. Significance was determined by evaluating whether dissimilarities existed between the permuted datasets and observed dataset, indicated by an appropriate *P* value statistic (α <0.05). Although the *P* value determined significant differences in the sampling groups, we also evaluated the effect size of the differences, which would be independent of sample size. A chance-corrected agreement coefficient (A) signified within-group homogeneity using the expected and observed deltas. The A-coefficient was calculated on a scale between 0 and 1. When A = 1, all within-group observations were identical, whereas when A approached zero, dissimilarities existed between groups (McCune and Grace 2002).

The third approach to examine differences in distances within each fisher among treatments was the chi-squared test of independence, which was performed by binning the distances into pre-defined intervals, with short-range distances split into numerous classes and long-range distances lumped into fewer bins. We examined the dataset distributions to determine what distance increments were biologically relevant when related to behavioral responses, using previous literature of fisher treatment response as an additional guide in partitioning our bin categories (U.S. Department of Agriculture 2008, Garner 2013). We were interested in determining the threshold distance at which fishers could tolerate treatment effects, so we focused on splitting our data into smaller bins below the mean in order to capture a defined distance measurement. Because chi-squared tests are sensitive to sample size requirements, we aimed to retain appropriate sample sizes (i.e., > 5) within each bin category (McHugh 2013). Per McHugh (2013), we subsequently followed our statistical tests with strength tests to determine the direction and magnitude of significance, using effect size as our strength test. Thus, the Cramer's V test from chi-squared results and the A-coefficients from the MRPP test were used to calculate appropriate effect sizes (Cohen 1992, McCune et al. 2002). All statistical analyses were implemented using program R (R Software, version 3.4.3; R Development Core Team 2017). The K-S test, chi-squared test, and effect size computations were run using base R packages, while the 'mrpp' procedure was executed using the 'vegan' package (R software, version 2.6.2; R Development Core Team 2008).

To determine whether certain environmental or treatment factors influenced fisher distance movements in the pre- and post-treatment periods, we extracted site-specific variables from the animal locations and the treatment areas for a post-hoc regression analysis. We ran ANOVA models with categorical covariates that included treatment variables (treatment size, treatment type), temporal variables (season, time of day), and floristic variables (canopy cover, habitat classification). Treatment and temporal data were derived from information collected by U.S. Forest Service personnel performing density management practices, while floristic data were obtained using LANDFIRE datasets (www.landfire.gov/vegetation.php). Canopy cover was binned into 5 categorical classes: open-low (0-19%), low-moderate (20-39%), moderate (40-59%), moderate-high (60-79%), and high (≥80%). For habitat classifications, we reclassified similar habitat types into fewer categories for ease of model interpretation. Vegetation categories from the LANDFIRE data set were reduced to 4 classes, with each representative proportion signifying its prevalence in the study area: low elevation shrubland (24%), Sierra mixed conifer (19%), conifer-hardwood (17%), and conifer (40%). No collinearity tests were performed to search for variable correlation since all of the covariates were categorical. The ANOVA models were run with the assumption that some individual variation would be evident in response behavior (i.e., fisher distance) to treatment disturbance. To address this expectation, we began by testing a pooled subset of fisher data which exhibited significant findings from our first analyses (i.e., distance response and effect size tests). We then performed ANOVA tests on individual fishers from the pooled subset above to compare within-population response variability of fisher distance to treatment effects.

RESULTS

Capture and Monitoring

From 2010 through 2017, we captured 40 (23 F, 17 M) individual fishers in the Ashland watershed. All adult females were fitted with GPS collars, with the exception of two adults and a subadult receiving VHF radio-collars. A mix of male age classes also received GPS collars if the canister assembly weighed less than 4% of their body mass;

otherwise, VHF or no collars were affixed. In total, 23 GPS collars and 4 VHF collars were deployed (Table 2.1). Although a majority of fishers radio-collared had GPS technology, only 10 individuals over the course of the study had reliable GPS units that remained on-air with successful fix rates. We acquired a total of 1,352 GPS locations for analysis, ranging from 24 to 263 points per fisher ($\bar{x} = 135.2 \pm 108.36$ standard deviation (SD) Table 2.2). Over the period of the study, we tracked fishers a total of 925 days ($\bar{x} = 92.5 \pm 75.24$ SD), though the females accounted for a larger proportion of time followed due to the ease in covering their home ranges and re-capture success to replace failing transmitters (Table 2.2). Acquisition rates for our collars were among the lowest of any GPS brand on the market, averaging 32% among all individuals.

Home Ranges and Treatment Units

Home ranges differed between the sexes, with a mean female size of 16.27 $\text{km}^2 \pm 8.87$ and males averaging 69.29 $\text{km}^2 \pm 31.19$, more than four times larger (Wilcoxon test, $V = 55$, $P = 0.002$, Table 2.3, Fig. 2.1). The number of treatments within each fisher home range varied widely, and our analysis only considered mechanically altered units (i.e., vegetation removal by machine) which consisted of 60% of all treatments. Sections that received burn-only treatments were not evaluated. The number and proportion of mechanical treatments per individual home range area varied between 8 and 83 units (8% to 34%; Table 2.3). Notably, some of the same treatment units coincided with multiple female territories due to the proximity and overlap of home ranges (Fig. 2.1). The percentage of treatments within male home ranges was lower, mainly due to large territories and the propensity of males to explore areas outside of the prescribed treatment blocks. Most activities across the entire study area comprised mechanical methods in treatment units categorized as either commercial or noncommercial (Table 2.4). Both commercial and noncommercial treatment units were similar in size (hectares) and number of units treated. When combined, the mechanically-treated units totaled 72.7% of the entire area managed for vegetation removal or understory improvement.

Statistical Analyses

We did not perform any statistical analyses on two fishers, F04 and M04, as both individuals had insufficient location data for the post-treatment phase of the study. For the remaining fishers, we found variability in the distributional frequencies of distance to treatments between the pre- and post-treatment periods. Two analyses tests, Kolmogorov-Smirnov (K-S) and MRPP, resulted in P-values <0.05 for all 8 fishers of interest (Table 2.5), indicating significant shifts towards or away from treated units. The chi-squared test of independence also produced significant outcomes ($\alpha < 0.05$) for all fishers except M02 (Table 2.5). Overall, 6 fishers (F01, F02, F06, F08, F09, M02) revealed intolerant behaviors to mechanical treatments, meaning a majority of fishers were located further from treated units in the post-operational period. However, measurements further from treatment areas were negligible for 2 fishers (F06, M02) during the post-treatment period. In contrast, 2 fishers (F03, M10) displayed movement closer to treatments after mechanical activity was completed (Fig. 2.2, 2.3).

The K-S curves (Fig. 2.2) illustrate the relationship between treatment periods and the cumulative frequency of the response variable of fisher distance to treatment. Two important considerations with this test are the maximum distance between curves and the shift in curves relative to distance. The first observation of separation between curves denotes whether there is a significant difference between treatment data distributions.

The second observation explains which treatment period the fisher tolerated best according to the frequency of distances closer to treated areas. Several fishers (F01, F08, F09, M02, M10) exhibited the greatest distance difference between curves, although the K-S analysis resulted in a significant outcome for all fishers. As noted previously, all but two fishers (F03, M10) exhibited intolerant behavior to areas after treatments occurred, indicated by a shift in the K-S post-treatment distance curve to the right (Fig. 2.2).

The chi-squared test of independence mirrored the K-S test patterns, excluding the insignificant outcome of M02's behavior (Fig. 2.3). Again, F03 and M10 were the only individuals who displayed tolerance to treated units, moving closer to those areas after operations took place. However, their response to treatments were weak in comparison to other individuals.

The relative frequencies (i.e., proportion) of post-treatment distances were higher for 5 fishers (F01, F02, F06, F08, F09) when they were located approximately ≥2000 m from restoration areas (Fig. 2.3), regardless of whether the units were commercially altered or not. At this threshold limit, fisher distances were either equivalent between the two periods or showed higher post-treatment proportions. This observation supports our original hypothesis that fishers would avoid treated units up to a particular threshold distance.

When evaluating effect size, only the MRPP and chi-squared test statistics resulted in measureable effects, using the A-coefficient and Cramer's V test, respectively. We were unable to compute effect sizes with the KS statistics. We elucidated the Cramer's V results using range values classified in "negligible", "small", "medium", or "large" effect size categories. The MRPP method, however, did not have an associated range interpretation for effect size. Thus, graphical results were primarily used to validate similar patterns in the Cramer's V effects. We found that treatments in the home ranges of 3 fishers (F01, F08, F09) rendered stronger effect size responses compared to the other fishers (Fig. 2.4 A, B). According to the Cramer's V descriptive index, no fishers exhibited "large" effects and only one animal (F01) had a measured "medium" effect to treatment activities. Three fishers (F02, F08, F09) revealed "small" effect treatment response, while the remaining fishers fell into the "negligible" effect category. Although the MRPP and chi-squared analyses resulted in different effect size scales, similar patterns emerged with seven out of eight fishers tested. The lone exception was F02's larger discrepancy in the Cramer's V test (Fig. 2.4 A).

Regression Analyses

Three fishers (F01, F08, F09) demonstrated the largest effect sizes in our first analysis (Fig. 2.4), therefore we used these females as our pooled sample data set for the first ANOVA model. In the pre-treatment period, the significant variables found to influence fisher response to treatment disturbance were season, vegetation class, treatment type, treatment size, and canopy cover (Table A.1, Fig. A.1). The only variable without a significant effect was time of day. The post-treatment period revealed significant variables of season, vegetation class, and treatment size. Shared factors that lent a substantial effect on response behavior during both time periods were season, treatment size, and vegetation class (Table A.1, Fig. A.1).

The 3 females were also tested individually to ascertain if specific factors influenced their behavior to treatment effects and if any variables were common on a subpopulation scale. One fisher (F01), despite data deficiency in one variable category (treatment size), did not show significant results in the pre- or post-periods for treatment type (commercial versus noncommercial), indicating it was not an influential factor (Table 2.6 A, Fig. 2.5). Season, time of day, and vegetation class were important in both periods, with the addition of treatment size and canopy cover after treatments took place (Table 2.6 A, Fig. 2.5). Results for F08 indicated all variables affected her distance behavior in the pre-treatment period (Table 2.6 B, Fig. 2.5). The post-treatment phase results, on the other hand, found only 4 variables were significant: season, treatment type, treatment size, and canopy cover (Table 2.6 B, Fig. 2.5). Surprisingly, results for F09 showed every variable except canopy cover influenced her distance to treatments in both time periods (Table 2.6 C, Fig. 2.5). Finally, when comparing variables between the 3 females, we found 3 mutually shared factors in the pre-treatment period that would affect response distance: season, time of day, and vegetation class. Post-treatment variables eliciting a negative response and common to all 3 fishers were season and treatment size (Fig. 2.5).

DISCUSSION

Sensitivity of wildlife to anthropogenic change has long been an important topic of concern in the scientific community. Human-modified landscapes, ranging from urban encroachment and tourism development to logging operations and human-caused wildfires, impact the spatial distribution and behavior of a multitude of wildlife taxa (Zielinski et al. 2005, Tempel et al. 2014, Amaral et al. 2016, Moriarty et al. 2016). Research on this concept has primarily focused on the variable disturbance response of species at the population level, where both positive and negative associations exist. For instance, boreal wolves (*Canis lupus*) in the forests of Quebec, Canada, were found to avoid heavily logged areas (Lesmerises et al. 2012), while those in the Canadian Rockies selected for post-fire logging where foraging opportunities increased (Hebblewhite et al. 2009). Similarly, wolverines (*Gulo gulo*) were reported to be particularly sensitive to a wide range of human disruptions, including resource extraction activities (Krebs et al. 2007, Fisher et al. 2013). However, Scrafford et al. (2017), found wolverines were attracted to cutblock units because of opportunistic food resources, albeit edges were preferred over the interior of cutblocks due to predation risks. These examples reference a growing literature describing the variation of population-level responses to stochastic events across a larger landscape, but few studies have touched upon the individual responses within a population that can complicate the interpretation of disturbance-related behavior.

In our study, fishers responded to anthropogenic stressors on an individual basis, whereby each animal displayed varying tolerance levels to habitat transformations. Unexpectedly, 2 individuals (F03, M10) reacted positively to timber management activities occurring within their home range. In contrast, the remaining fishers in our study displayed varying degrees of aversion to post-treated units. A possible explanation for this individual variation could be attributed to different temperaments or personalities exhibited by fishers, which would reflect how they spatially distribute themselves on the landscape relative to changing habitat conditions (Martin and Réale 2008). For instance, an individual with a bold or explorative personality would express tolerance, or perhaps curiosity, when faced with unfamiliar circumstances, thus adapting to or even seeking out habitats replete with novel conditions. Moreover, a newly altered environment that provides exploitable resources and reduced competition may potentially attract individuals to those areas. In contrast, animals exhibiting shy or cautious tendencies might perceive risks associated with familiar places changed by human perturbations. As Blackwell et al. (2016) have suggested, predators have been known to display neophobia to deterrence stimuli within their environment. If these individuals encounter environmental stressors, such as a loss in critical structures or reduction of a prey base, they may under-utilize habitat resources if they choose to remain in place, thus resulting in costs to fitness potential (Beale 2007). Furthermore, if the individual shifts its home range as a direct result of habitat alteration, they infringe on conspecific territories, and become vulnerable to predator attacks.

In our study, we did not find any fishers permanently shifting their home ranges after treatments occurred, indicating fishers were able to tolerate disturbance-related effects by utilizing other areas of their home range. In fact, F01, F06, and F08 had adequate space within their respective home ranges to avoid treatment units. A plausible explanation for varying tolerance levels can be explained by the home range overlap between several female fishers. Although F03, F06, F08, and F09 had overlapping home ranges, their temporal location points indicated they rarely came into contact with one another. F03, for example, overlapped in home range with F06, F08, and F09, and had treatments scattered throughout the center of her home range. If contact and competition with other fishers were driving forces, she may have had no choice but to move through treatment units, thus suggesting a positive response to treatment effects. On the other hand, we noticed F01 and F08 avoided treatment areas by utilizing a shared portion of home range overlap where occasional fisher contact occurred. The area of overlap contained no treatment units and several interactions took place over a year between the females. A possible explanation is that these two fishers were related, and we speculate at some point the juvenile dispersed only a short distance from its natal range (Matthews et al. 2013). Contact between the offspring and parent, then, may be likely as home range establishment progressed.

From a management standpoint, factors that could have influenced differences in fisher behavior include the amount of area in the home range that was treated, treatment placement, management intensity level, or duration of treatments. Zielinski et al. (2013) revealed fishers can tolerate up to 2.6% of their respective habitat being treated per year. Their study design employed a 14-km² cell unit, equivalent to our average female fisher territory. Because our project comprised higher numbers of treated units for each individual home range compared to the Zielinski study (Table 2.3), we believed each fisher would avoid post-treated areas. To the contrary, we found our fishers reacted to vegetation removal in the units by variable degrees, irrespective of the percentage of home range treated. For example, F01's home range encompassed the fewest treated units and lowest proportion of area treated of all fishers (Table 2.3), yet she displayed the largest intolerant effect size to treatments than any other animal (Fig. 2.4 A and B). F08 and F09 both demonstrated similar behavioral responses according to effect size calculations. However, their home range areas included higher numbers of treated units than F01. Interestingly, F03 had a comparable number of treatment units in her home range as F08, yet she displayed the opposite response effect from all other female fishers.

In addition to percentage of area treated, spatial configuration of treated units also appeared to be an important component. We found that treatments in F01's home range were concentrated only along one edge, whereas F08 and F09 both had treatments placed near the center of their home ranges, which may have limited their movement through those areas. Additionally, we noticed that the majority of F01 and F08's point locations were grouped far from the treatment units. Spatially, GPS points observed for F01 and F08 were often clustered in areas more than 1,000 m away from post-treated units, which resulted in a higher frequency of distance counts farther from managed units, and thus a larger negative response to management activities. Other fishers had GPS locations scattered throughout their home range with fewer clusters, and our analyses showed a smaller effect size to treated units. Despite their intolerant behavior, we reasoned that F01, and to a lesser extent F08, could have utilized the untreated areas of their home ranges adequately in the post-treatment period, thus avoiding the portion of home range area that was mechanically altered.

Similarly, F09 displayed intolerant behavior to mechanical treatments in the post-treatment period as well. Treatments were centrally located in an equivalently-sized home range extent as F01 and F08, though F09's spatial points were evenly distributed throughout her area. Although F09 showed comparable evasive behavior to overall treatment activity, interesting anecdotal evidence arose regarding her tolerance to noise disturbance and logging practices during a spring denning season (unpublished data, D. Clayton, T. Smith). Timber activity within F09's home range commenced in April 2012, with the objective of cutting pine and fir trees with subsequent helicopter yarding to remove the fallen logs. Within days after felling the trees, F09 chose a standing, live conifer in the middle of the logged unit to birth her kits in a natal den. Helicopter yarding began before we discovered the den site, with high decibel noise and frequent mechanical disturbance occurring in an adjacent unit near her den tree. Upon discovery, managers postponed helicopter operations until she moved her kits to a new site further away. Yet F09 did not relocate her den for two more weeks despite a constant flow of human activity occurring nearby. Although exposure to short-term noise disturbance was not a study objective, this noteworthy event indicates that even during a reproductively sensitive time, F09 did not react negatively to sudden vegetation changes in her physical environment.

We also investigated management intensity levels (commercial versus noncommercial treatments) and found no significant differences in fisher distance behavior between the two activities. Analysis of locations found inside and outside of treatments in a temporal context did not lend any significance either, though the raw number of points in commercial versus noncommercial areas were slightly lower (52 versus 63 points, respectively). We speculate that because canopy cover change post-treatment was minimal and optimal habitat was still retained in both treatment types, fishers did not discern any threats that would minimize use in these areas. Finally, we originally intended to use treatment duration as a variable of interest, but we were unable to obtain available data for statistical testing. We suggest future analyses consider treatment duration as a potential influential factor affecting fisher behavior.

Our results exploring other covariates affecting behavior, such as environmental and vegetation components, found that in both the pooled and individual sampling, season and vegetation class influenced an individual's distance to a pre-treated unit. We reason this outcome is valid; since no change in habitat quality had yet taken place on the landscape, other ecological pressures could account for variable fisher movement (e.g., seasonal food availability, predator evasion). Two variables of significance common to the 3 fishers (pooled and individual) post-treatment were season and treatment size. Fishers were observed on a subpopulation level to be further from treated units in fall and winter, and further from treated units that ranged from very small to medium in size (<12 hectares). We speculate that larger units may have had suitable remnant habitat to travel through and use, while the smaller units were avoided or circumnavigated if habitat had been reduced. Our season covariate was significant in both periods for all fishers, and the majority of units were treated in spring or summer (63% total). Curiously, fishers were closer to treatment areas at the time they were treated, but this result may have been due to an elimination of sampling points within a 30-day treatment period, lag effects of vegetation removal, or unrelated factors such as food availability shifts.

We expected canopy cover to be the most important constituent influencing treatment response, especially given that multiple literature discussions have cited it as a principal element crucial to fisher habitat use (Davis et al. 2007, Sauder and Rachlow 2014, Niblett et al. 2015). However, descriptions of canopy cover can be subjective and difficult to interpret, as measurement methods vary between studies (Raley et al. 2012). Overhead canopy cover in our area was not manipulated in noncommercial units, though understory cover may have been reduced to a small degree. Density management in commercial units, on the other hand, reduced canopy cover as was intended in the AFR objectives. Nevertheless, relative changes in canopy cover were minimal for treatments, with a majority of units retaining >60% canopy cover post-treatment. Therefore, we surmise that for the majority of fishers, adequate overhead cover (>60%) was retained throughout most of the post-treatment home ranges, and these animals were not impacted by the change in canopy cover.

Other studies corroborate our findings of tolerance and variability in fisher response to a managed landscape. Garner (2013) speculated that fishers can tolerate a portion of their home range being treated as long as higher-quality habitat surrounds those treated areas. Indeed, we confirmed this earlier by stating that none of our fishers had shifted their home range to exclude treated areas. Most of those treated units had adequate canopy cover, and both commercial and noncommercial units continued to be utilized. In a different perspective, Sauder and Rachlow (2014) found that proximity to contiguous mature forest was the highest predictor of fisher presence in managed landscapes, and not canopy cover. Our results are consistent with these findings as well, since overhead cover did not seem to be the most critical element in fisher space use. Truex and Zielinski (2013), however, found that canopy closure was their most impacted feature resulting from management practices, but they mention this element was only tested in select stands and not extrapolated to an area the size of a fisher home range. The authors also noted that short-term effects from mechanical treatments were mitigated by the retention of larger trees for fisher use. Our study results mimicked their findings as well, since the AFR objectives were to thin only small-stemmed trees and leave larger trees and snags on the landscape, thus allowing fishers to continue using optimal habitat within their home ranges (U. S. Department of Agriculture 2008). Finally, a camera survey study concluded that extractive activities (e.g., timber harvests for commerce or hazard tree removal) had no impact on fisher use and presence, and although restorative fuel reduction practices lowered fisher occupancy, fishers continued to use the areas for multiple purposes (Sweitzer et al. 2016). Similarly, our fuel reduction practices had both positive and negative effects on our fishers, but we surmise that the treatments did not deter fishers from using a majority of their home range habitat, as evidenced by a threshold distance to treatments well within their home range and variable tolerance levels of individuals.

It should be noted that interpretation of wildlife behavior, whether applying personality traits or using distance measurements to index disturbance effects, must be used with caution. As Beale (2007) explains, an animal may leave an area due to a disturbance event, but that decision does not necessarily mean it was negatively impacted by the event. Other factors, such as environmental conditions (e.g., weather), prey resource availability, competitive interactions, or predator presence, could have influenced fishers to move around their home range independent of treatment affects. We were unable to assess all of these variables in our study due to the sheer complexity of ecological relationships. More importantly, though, other limitations hindered our ability to perform robust testing of our objectives, including data sampling and statistical approaches. Our study used a first-generation micro-GPS collar, which unfortunately included several devices that malfunctioned. We encountered other problems associated with our GPS transmitters, specifically, inaccurate spatial locations and missed fixes. These problems were namely due to terrain interference, dense vegetation cover, and infrequent scheduling of point locations, all common problems when using advanced telemetry equipment (Frair et al. 2004, Cain et al. 2005, Frair et al. 2010). Our study results also consisted of a low number of sampled individuals. A few fishers acquired enough locations over several years for a moderately strong statistical analysis (e.g., F01, F03, F09); nevertheless, a larger sample size of both sexes and better GPS-fix schedules would certainly correct some of our sampling bias. In terms of statistical power, we would suggest using a more comprehensive study design with control measures along with well-defined periods of before/after treatment experiments. Because our project started with little time between fisher captures and treatment applications, our pre-treatment period was short, and the complications of multiple treatments occurring across each home range at different periods made it difficult to create independent sampling units in a spatial and temporal context. Finally, an inclusive look at how prescribed fire affects fisher behavior is of upmost importance. We did not examine the long-term impacts that broad-scale burning or pile burns have on fisher spatial use, but seasonal timing and coverage extent of prescribed burns should be evaluated to determine whether fishers remain or return to affected areas.

MANAGEMENT IMPLICATIONS

The reform of forest fire management has reached a critical level as mega-fires become increasingly frequent and more intense throughout the western states. Management intervention to prioritize areas of high wildfire risk usually involves a four-fold treatment process: reduction of high fuel loads, retention of fire-hardy species, increasing the height to live crown ratio, and lowering canopy bulk density (Agee and Skinner 2005). The culmination of all these processes over the long term will certainly change the dynamics and behavior of wildfire, while decreasing the likelihood of stand-replacing burns. Yet, continued removal and reconfiguration of habitat elements important to fisher survival and persistence continue to be a contentious topic (Powell and Zielinski 1994, Zielinski et al. 2013).

The response of wildlife to management actions, especially those that are highly dependent on specific habitat conditions, should be considered before treatment applications begin. Fishers use a variety of structures at ground level and throughout multiple canopied-layers, and reconfiguration of home range features can alter behavioral patterns for foraging, denning, and travel paths (Sauder and Rachlow 2014). Indeed, the majority of our fishers had a weak negative association with treatments in general and many fisher locations were found at greater distances for post-treatment areas than pre-treatment areas. Results to determine which specific factors influenced a fisher to move away from treatment units were somewhat ambiguous, and it seemed decisions varied based on individual tolerance. However, we presume it will be difficult to differentiate what is occurring at the individual level and how that equates to the population as a whole, especially given that management practices spanned across home ranges of several animals. Seasonal timing of treatments, though, came out as a significant factor when investigating variables of interest. Using limited operating periods (LOPs) is a good practice that we suggest be continued in timber management plans, where forest operations are restricted during crucial wildlife reproductive periods (U. S. Department of Agriculture 2009). LOPs have been successfully implemented for other sensitive species, such as the northern spotted owl (*Strix occidentalis*) and northern goshawk (*Accipiter gentilis*). Similar proposals have been developed for the fisher, though there is some disagreement in determining an effective buffer size around known denning structures, because of multiple-use trees and large home range extent (Thompson and Purcell 2015).

Nevertheless, fundamental principles in fisher behavior can assist managers in making beneficial decisions regarding fuel modifications to forest stands while maintaining basic fisher habitat needs. In particular, identification and preservation of specific tree species that take decades to form cavities or large growth should take precedence when formulating plans for thinning projects. Legacy trees, snags, and hardwood components, used for denning and resting, can be retained in select stands where vegetation diversity and regeneration are predominant. The added benefits of retaining these components not only provides refugia sites for habitat-obligate species, but also sustain key ecological processes, such as soil stabilization, seedling protection, and nutrient recycling (Garner 2013). Configuration of mixed-forest species and structures is also essential for a couple of reasons: it creates a diverse array of resources for fishers to exploit, and historical evidence shows that fire severity through mixed-forests is lower where heterogeneity is maintained (Odion et al. 2004). We believe then, that data gathered from our research coupled with future studies might reveal new methods in mitigating the loss of stand heterogeneity at finer scales while allowing larger manipulations to occur on the landscape with minimal impact to fisher space use.

LITERATURE CITED

Agee, J. K. 1993. Fire ecology of Pacific Northwest Forests. Island Press, Washington D.C.

Agee J. K., and C. N. Skinner. 2005. Basic principles of forest fuel reduction treatments. Forest Ecology and Management 211:83–96.

Amaral, K. E., P. Michael, K. M. O’Brien, L. E. Fenderson, and A. I. Kovach. 2016.

Anthropogenic habitats facilitate dispersal of an early successional obligate: Implications for restoration of an endangered ecosystem. PLoS ONE: doi.org/10.1371/journal.pone.0148842.

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

Beale, C. M. 2007. The behavioral ecology of disturbance responses. International Journal of Comparative Psychology 20:111–120.

Blackwell, B. F., T. L. DeVault, E. Fernandez-Juricic, E. M. Gese, L. Gilbert-Norton, and S. W. Breck. 2016. No single solution: application of behavioural principals in mitigating human-wildlife conflict. Animal Behaviour 120:245–254.

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

Cai, L. 2006. Multi-response permutation procedure as an alternative to the analysis of variance: An SPSS implementation. Behavior Research Methods 38:51–59.

Cain, J. W., P. R. Krausman, B. D. Jansen, and J. R. Morgart. 2005. Influence of topography and GPS fix interval on GPS collar performance. Wildlife Society Bulletin 33:926–934.

Cohen, J. (1992). A power primer. Psychological Bulletin 112:155–159.

Cox, R. 2018. Hendrix Fire – Sunday, August 5, 2018 Morning Update, Retrieved November 23, 2018 from https://inciweb.nwcg.gov/photos/ORRSF/2018-07-18-0041-Hendrix/related_files/pict20180705-104629-0.pdf.

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

DellaSala, D. A. 2006. Fire in the Klamath-Siskiyou ecoregion: protecting and restoring the fire mosaic. Pages 132-152 in G. Wuerthner, editor. Wildfire: a century of failed forest policy. Island Press, Washington, D.C., USA.

Dussault, C., C. Réhaume, J. Ouellet, and J. Pettorelli. 2001. Influence of satellite geometry and differential correction on GPS location accuracy. Wildlife Society Bulletin 29:171–179.

Fisher, J. T., and L. Wilkinson. 2005. The response of mammals to forest fire and timber harvest in the North American boreal forest. Mammal Review 35:51–81.

Fisher, J. T., S. Bradbury, B. Anholt, L. Nolan, L. Roy, J. P. Volpe, and M. Wheatley. 2013. Wolverines *(Gulo gulo luscus*) on the Rocky Mountain slopes : natural heterogeneity and landscape alteration as predictors of distribution. Research Press 91(October):706–716.

Frair, J. L., S. E. Nielsen, E. H. Merrill, S. R. Lele, M. S. Boyce, R. H. M. Munro, G. B. Stenhouse, and H. L. Beyer. 2004. Removing GPS collar bias in habitat selection studies. Journal of Applied Ecology 41:201–212.

Frair, J. L., J. Fieberg, M. Hebblewhite, F. Cagnacci, N. J. DeCesare, and L. Pedrotti. 2010. Resolving issues of imprecise and habitat-biased locations in ecological analyses using GPS telemetry data. Philosophical Transactions of the Royal Society B: Biological Sciences 365:2187–2200.

Franklin, F. 1972. Federal research natural areas in Oregon and Washington: a guidebook for scientists and educators. Pacific Northwest Forest and Range Experiment Station, Portland, Oregon.

Fuller, A. K., and D. J. Harrison. 2005. Influence of partial timber harvesting on American martens in north-central Maine. Journal of Wildlife Management 69:710–722.

Gabriel, M. W., L. Woods, G. M. Wengert, N. Stephenson, J. M. Higley, C. M. Thompson, S. Matthews, R. A. Sweitzer, K. Purcell, R. H. Barrett, S. M. Keller, P. Gaffney, M. Jones, J. E. Foley, R. Poggenga, D. Clifford, and B. N. Sacks. 2015. Natural and human-caused mortality factors of fishers (*Pekania pennanti*) in California. PLoS One: doi:10.1371/journal. pone.0140640.

Ganey, J. L., H. Y. Wan, S. A. Cushman, and C. D. Vojta. 2017. Conflicting perspectives on spotted owls, wildfire, and forest restoration. Fire Ecology 13:146–165.

Garner, J.D. 2013. Selection of disturbed habitat by fishers (*Martes pennanti*) in the Sierra National Forest. Thesis, Humboldt State University, Arcata, California.

Gibilisco, C. J. 1994. Distributional dynamics of modern Martes in North America. Pages 59–71 in S. W. Buskirk, A. S. Harestad, M. G. Raphael, and R. A. Powell, editors. Martens, sables, and fishers: biology and conservation. Cornell University Press, Ithaca, New York.

Green, R. E. 2017. Reproductive ecology of the fisher (*Pekania pennanti*) in the southern Sierra Nevada: an assessment of reproductive parameters and forest habitat used by denning females. Ph.D. dissertation, University of California, Davis.

Haines, A. M., F. Hernández, S. E. Henke, and R. L. Bingham. 2009. A method for determining asymptotes of home-range area curves. Gamebird, (1955), 489–498. http://ckwri.tamuk. edu/fileadmin/user_upload/docs/misc_PDF/Quail/Hainesetal2009b.pdf.

Hargis, C. D., J. A. Bissonette, and D. L. Turner. 1999. The influence of forest fragmentation and landscape patterns on American martens. Journal of Applied Ecology 36:157–172.

Hebblewhite, M., R. H. Munro, and E. Merrill. 2009. Trophic consequences of postfire logging in a wolf-ungulate system. Forest Ecology and Management 257:1053–1062.

Hessburg, P. F., J. K. Agee, and J. F. Franklin. 2005. Dry forests and wildland fires of the inland Northwest USA: contrasting the landscape ecology of the pre-settlement and modern eras. Forest Ecology and Management 211:117–139.

Kalies, E. L., and L. L. Yocom Kent. 2016. Tamm Review: are fuel treatments effective at achieving ecological and social objectives? A systematic review. Forest Ecology and Management 375:84–95.

Krebs, J., E. C. Lofroth, and I. Parfitt. 2007. Multiscale habitat use by wolverines in British Columbia, Canada. Journal of Wildlife Management 71:2180–2192.

Lacy, R. C. 1997. Importance of genetic variation to the viability of mammalian populations. Journal of Mammalogy 78:320–335.

Lesmerises, F., C. Dussault, and M. H. St-Laurent. 2012. Wolf habitat selection is shaped by human activities in a highly managed boreal forest. Forest Ecology and Management 276: 125–131.

Levene, H. 1960. Robust tests for equality of variances. Pages 278-292 in I. Olkin, H. Hotelling; et al., editors. Contributions to probability and statistics: Essays in honor of Harold Hotelling. Stanford University Press, Redwood City, California.

Lewis J., J. Rachlow, E. Garton, and L.Vierling. 2007. Effects of habitat on GPS collar performance: Using data screening to reduce location error. Journal of Applied Ecology 44: 663–671.

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

Lofroth,E. C., C. M.Raley, J. M.Higley,R. L.Truex, J. S.Yaeger, J. C.Lewis, and P. J. Happe. 2010. Conservation of fishers (*Martes pennanti*) in South-Central British Columbia, Western Washington, Western Oregon, and California-Volume I: conservation assessment. U.S. Department of the Interior Bureau of Land Management, Denver, Colorado.

Martin, J. G., and D. Réale. 2008. Temperament, risk assessment and habituation to novelty in eastern chipmunks, *Tamias striatus*. Animal Behaviour 75:309–318.

Matthews, S. M., J. M. Higley, K. M. Rennie, R. E. Green, C. A. Goddard, G. M Wengert, M. W. Gabriel, and T. K. Fuller. 2013. Reproduction, recruitment, and dispersal of fishers (*Martes pennanti*) in a managed Douglas-fir forest in California. Journal of Mammalogy. 94: 100-108.

McCune, B., and J. B. Grace. 2002. MRPP (Multi-response permutation procedures) and related techniques. Pages 188-197 in B. McCune and J. B. Grace, editors. Analysis of ecological communities. MjM Software Design, Gleneden, Oregon.

McHugh, M. L. 2013. The chi-square test of independence. Biochemia Medica 23:143–149.

McIver, J. D., S. L. Stephens, J. K. Agee, J. Barbour, R. E. J. Boerner, C. B. Edminster, K. L. Erickson, K. L. Farris, C. J. Fettig, .C. E. Fiedler, S. Haase, S. C. Hart, J. E. Keeley, E. E. Knapp, J. F. Lehmkuhl, J. J. Moghaddas, W. Otrosina, K. W. Outcalt, D. W. Schwilk, C. N. Skinner, T. A. Waldrop, C. P. Weatherspoon, D. A. Yaussy, A. Youngblood, and S. Zack. 2013. Ecological effects of alternative fuel-reduction treatments: highlights of the National Fire and Fire Surrogate study (FFS). International Journal of Wildland Fire 22:63–82.

Moriarty, K. M., C. W. Epps, and W. J. Zielinski. 2016. Forest thinning changes movement patterns and habitat use by Pacific marten. Journal of Wildlife Management 80:621–633.

Niblett, M. R., S. H. Sweeney, R. L. Church, and K. H. Barber. 2015. Structure of fisher (*Pekania pennanti*) habitat in a managed forest in an interior northern California coast range. Forest Science 61:481–493.

Odion, D. C., E. J. Frost, J. R. Strittholt, H. Jiang, D. A. DellaSala, and M. A. Moritz. 2004. Patterns of fire severity and forest conditions in the Western Klamath Mountains, California. Conservation Biology 18:927–936.

Olson, D., D. A. DellaSala, R. F. Noss, J. R. Strittholt, J. Kass, M. E. Koopman, and T. F. Allnutt. 2012. Climate change refugia for biodiversity in the Klamath-Siskiyou ecoregion. Natural Areas Journal 32:65–74.

Olson, L. E., J. D. Sauder, N. M. Albrecht, R. S. Vinkey, S. A. Cushman, and M. K. Schwartz. 2014. Modeling the effects of dispersal and patch size on predicted fisher (*Pekania [Martes] pennanti*) distribution in the U.S. Rocky Mountains. Biological Conservation 169:89–98.

Powell, R. A. 1993. The fisher. 2nd ed. University of Minnesota Press, Minneapolis.

Powell, R. A., and W. J. Zielinski. 1994. The Fisher. Pages 38-73in L. F. Ruggiero, K. B. Aubry, S. W. Buskirk, L. J. Lyon, and W. J. Zielinski, editors. The scientific basis for conserving forest carnivores: American marten, fisher, lynx, and wolverine in the western United States. General Technical Report. GTR RM-254. U.S. Department of Agriculture, Forest Service, Rocky Mountain Forest and Range Experiment Station. Fort Collins, Colorado.

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

R Development Core Team. 2008. R: a language and environment for statistical computing, version 2.6.2. R Foundation for Statistical Computing, Vienna, Austria.

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

Raymond, C. L., and D. L. Peterson. 2005. Fuel treatments alter the effects of wildfire in a mixed-evergreen forest, Oregon, USA. Canadian Journal of Forest Research 35:2981–2995.

Recio, M. R., R. Mathieu, P. Denys, P, Surguey, and P. J. Seddon. 2011. Lightweight GPS-tags, one giant leap for wildlife tracking? An assessment approach. PLoS ONE doi.org/10.1371/journal.pone.0028225.

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

Scheller, R. M., W. D. Spencer, H. Rustigian-Romsos, A. D. Syphard, B. C. Ward, and J. R. Strittholt. 2011. Using stochastic simulation to evaluate competing risks of wildfires and fuels management on an isolated forest carnivore. Landscape Ecology 26:1491–1504.

Schilling, J. W. 2009. Demography, home range, and habitat selection of Northern Spotted Owls in the Ashland watershed. Thesis, Oregon State University, Corvallis, Oregon.

Scrafford, M. A., T. Avgar, B. Abercrombie, J. Tigner, and M. S. Boyce. 2017. Wolverine habitat selection in response to anthropogenic disturbance in the western Canadian boreal forest. Forest Ecology and Management 395:27–36.

Seglund, A. E. 1995. The use of resting sites by the Pacific fisher. Thesis, Humboldt State University, Arcata, California.

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

Sirén, A. P. K., P. J. Pekins, M. J. Ducey, and J. R. Kilborn. 2016. Spatial ecology and resource selection of a high-elevation American marten (*Martes americana*) population in the northeastern United States. Canadian Journal of Zoology 94:169–180.

Smirnov, N. V. 1939. Estimate of deviation between empirical distribution functions in two independent samples. Bull Moscow University 2:3–16.

Soutiere, E.C. 1979. Effects of timber harvesting on marten in Maine. Journal of Wildlife Management 43:850–860.

Sullivan, G. M., and R. Feinn. 2012. Using effect size—or why the *P* value is not enough. Journal of Graduate Medical Education 4:279–282.

Sweitzer, R. A., B. J. Furnas, R. H. Barrett, K. L. Purcell, and C. M. Thompson. 2016. Landscape fuel reduction, forest fire, and biophysical linkages to local habitat use and local persistence of fishers (*Pekania pennanti*) in Sierra Nevada mixed-conifer forests. Forest Ecology and Management 361:208–225.

Tempel, D. J., R. J. Gutierrez, S. A. Whitmore, M. J. Reetz, R. E. Stoelting, W. J. Berigan, M. E. Seamans, and M. Z. Peery. 2014. Effects of forest management on

California spotted owls: implications for reducing wildfire risk in fire-prone forests. Ecological Applications 24:2089–2106.

Thompson, C. M., and K. L. Purcell. 2015. Conditions inside fisher dens during prescribed fires; what is the risk posed by spring underburns? Forest Ecology and Management 359:156–161.

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

Truex, R. L., and W. J. Zielinski. 2013. Short-term effects of fuel treatments on fisher habitat in the Sierra Nevada, California. Forest Ecology and Management 293:85–91.

Tucker J. M., M. K. Schwartz, R. L. Truex, K. L. Pilgrim, and F. W. Allendorf. 2012. Historical and contemporary DNA indicate fisher decline and isolation occurred prior to the European settlement of California. PLoS One 7:e52803.

U. S. Department of Agriculture and U. S. Forest Service, Pacific Northwest Region. 2008. Ashland Forest Resiliency Final Environmental Impact Statement. Siskiyou Mountains Ranger District, Rogue-River-Siskiyou National Forest, Jackson County, Oregon.

U. S. Department of Agriculture and U. S. Forest Service. 2009. Final Environmental Impact Statement. Beaverside Timber Sale and Fuel Treatment Project. Mad River Ranger District, Six Rivers National Forest, Trinity County, California.

U. S. Forest Service. 2005. Ashland Forest Resiliency Draft Environmental Impact Statement. Rogue-River Siskiyou National Forest. Medford, Oregon.

Wilbert, J. C. 1992. Spatial scale and seasonality of habitat selection by martens in southeastern Wyoming. Thesis, University of Wyoming, Laramie, Wyoming.

Wisely, S. M., S. W. Buskirk, G. A. Russell, K. B. Aubry, and W. J. Zielinski. 2004. Genetic diversity and structure of the fisher (*Martes pennanti*) in a peninsular and peripheral metapopulation. Journal of Mammalogy 85:640–648.

Zielinski, W. J., R. L. Truex, G. A. Schmidt, F. V. Schlexer, K. N. Schmidt, and R. H. Barrett. 2004. Resting habitat selection by fishers in California. Journal of Wildlife Management. 68:475–492.

Zielinski, W. J., R. L. Truex, F. V. Schlexer, L. A. Campbell, and C. Carroll. 2005. Historical and contemporary distributions of carnivores in forests of the Sierra Nevada, California, USA. Journal of Biogeography 32:1385–1407.

Zielinski, W. J., J. R. Dunk, and A. N. Gray. 2012. Estimating habitat value using forest inventory data: The fisher (*Martes pennanti*) in northwestern California. Forest Ecology and Management 275:35–42.

Zielinski, W. J., J. A. Baldwin, R. L. Truex, J. M. Tucker, and P. A. Flebbe. 2013. Estimating trend in occupancy for the southern Sierra Fisher *Martes pennanti* population. Journal of Fish and Wildlife Management 4:3–19.

Zielinski, W. J., C. M. Thompson, K. L. Purcell, and J. D. Garner. 2013. An assessment of fisher (*Pekania pennanti*) tolerance to forest management intensity on the landscape. Forest Ecology and Management 310:821–826.

TABLES AND FIGURES

Table 2.1. Number of fishers captured by sex, collared by radio type or uncollared, and number of GPS-collared fishers used for analysis in the Ashland Watershed Unit, Oregon from 2010-2017.

	Number of fishers captured	Radio collar type			GPS-collared fishers used for analysis
		GPS	VHF [a]	Uncollared[b]	
Females	23	12	3	8	6
Males	17	11	1	5	2
Total	40	23	4	13	8

[a] VHF collars were only affixed when GPS collars were unavailable.

[b] Age class, weight condition, and equipment availability determined whether fishers received collars.

Table 2.2. Number of raw point locations, final point locations, collar acquisition rates, monitoring effort by day, and years monitored per individual fisher in the Ashland Watershed Unit, Oregon, from 2010-2017. "F" indicates female fisher while "M" indicates male fisher.

Fisher ID	Raw point locations	Final point locations	Acquisition rate	Days monitored[b]	Years monitored
F01	315	213	31%	148	2010-2014, 2016
F02	121	90	24%	67	2010-2011
F03	454	360	26%	258	2011-2017
F04	43	43	24%	32	2010-2011
F06	123	82	68%	56	2011-2012
F08	126	123	25%	80	2012-2014, 2017
F09	414	263	50%	167	2012-2013
M02	33	24	13%	22	2010
M04	54	70	34%	55	2014
M10	116	84	60%	40	2014
Total	1799	1352	32% [a]	925	7 years

[a] Average fix rate of GPS collars for all individuals

[b] Days monitored were calculated using final point locations instead of raw point locations.

Table 2.3. Home range size and number, size, and proportion of total treated units and mechanical-only treated units per fisher home range in the Ashland Watershed Unit, Oregon, from 2010-2017. Prescribed burns, underburns and activity fuels were not included as they did not affect overstory habitat features. Mechanical treatments were applied in both commercial and noncommercial units, which were combined for this table. "F" indicates female fisher while "M" indicates male fisher.

Fisher ID	Home range size (km^2) [a]	No. of treated units in home range	Size of all treated units (ha)	Proportion of the size all treated units to home range size	No. of mechanically treated units in home range	Size of mechanically treated units (ha)	Proportion of mechanically treated units to home range
F01	9.81	11	96	0.10	8	76	0.08
F02	14.99	85	696	0.46	50	429	0.29
F03	13.89	70	478	0.34	39	316	0.23
F04	20.69	73	834	0.40	47	464	0.22
F06	34.14	81	978	0.29	54	597	0.18
F08	12.40	61	553	0.45	46	378	0.31
F09	7.99	40	431	0.54	29	272	0.34
M02	65.42	86	1,025	0.16	60	643	0.10
M04	102.23	136	1,369	0.13	83	853	0.08
M10	40.23	76	993	0.25	48	606	0.15
Average	32.18	72	745	-- [b]	46	463	-- [b]

[a] Wilcoxon test results indicated a significant difference in home range size between the sexes (V = 55, *P* = 0.002)

[b] No average calculated for proportion columns

Table 2.4. Silviculture treatment types by number of units, size in hectares, mean area per treatment, and proportion within the managed and total watershed area in the Ashland Watershed Unit, Oregon, from 2010-2017. "F" indicates female fisher while "M" indicates male fisher.

Treatment type	No. of units	Area size (ha)	Area mean (ha)	Proportion of treatment to managed area	Proportion of treatment to watershed area [a]
Commercial[b]	59	498	8.44 ± 7.22	0.362	0.0199
Noncommercial[b]	63	640	10.15 ± 11.49	0.465	0.0256
Burn only[c]	9	185	20.61 ± 22.50	0.134	0.0074
Untreated[d]	7	53	7.62 ± 3.46	0.039	0.0021
Total	138	1376	--	1	0.0551

[a] The Ashland Watershed study area totaled 24,970 ha.

[b] Mechanically-treated units were categorized as either commercial or noncommercial type.

[c] The burn treatment category included underburns and pile burns only.

[d] Untreated units were slated for treatment in the original management plan but were not implemented during our study period.

Table 2.5. Pre and post treatment results for individual fishers using multi-response permutation procedure test (MRPP), Kolmogorov-Smirnov test (KS), and chi-squared independence test in the Ashland Watershed Unit, Oregon, from 2010-2017. The shaded cells indicate an insignificant result. "F" indicated female fisher while "M" indicates male fisher.

		MRPP			KS		Chi-square test		
Fisher ID		Delta	A	P	D	P	X^2	df	P
F01	Pre	782	0.039	< 0.001	0.280	< 0.001	187.69	7	< 0.001
	Post	807							
F02	Pre	1265	0.004	< 0.001	0.127	< 0.001	129.59	7	< 0.001
	Post	847							
F03	Pre	940	0.001	< 0.001	0.056	< 0.001	78.49	7	< 0.001
	Post	904							
F06	Pre	1752	0.002	< 0.001	0.083	0.004	21.98	7	0.003
	Post	2041							
F08	Pre	1175	0.038	< 0.001	0.344	< 0.001	303.04	7	< 0.001
	Post	1302							
F09	Pre	672	0.021	< 0.001	0.176	< 0.001	238.70	7	< 0.001
	Post	827							
M02	Pre	3711	0.003	< 0.001	0.221	0.021	10.56	7	0.159
	Post	4069							
M10	Pre	504	0.004	< 0.001	0.167	< 0.001	41.95	7	< 0.001
	Post	4704							

Table 2.6. Pre and post treatment multi-way ANOVA results as independent analyses on fishers F01, F08, and F09 in the Ashland Watershed Unit, Oregon, 2010-2017. F01 did not have appropriate data to compare treatment size as a pre-treatment variable. Shaded rows indicate significance.

F01

Pre-treatment variable	df	Sum of squares	Mean squares	F	*P*
Season	2	4.23E+07	21,148,596	57.4001	<0.001
Time of Day	3	1.44E+07	4,793,360	13.0098	<0.001
Treatment Type	1	2.11E+05	210,845	0.5723	0.450
Vegetation Class	2	1.44E+07	7,188,130	19.5095	<0.001
Canopy Cover	2	1.30E+05	64,766	0.1758	0.839
Error	690	2.54E+08	368,442		

Post-treatment variable	df	Sum of squares	Mean squares	F	*P*
Season	3	2.31E+07	7,712,300	26.8853	<0.001
Time of Day	3	1.02E+07	3,402,443	11.861	<0.001
Treatment Type	1	7.91E+04	79,081	0.2757	0.600
Vegetation Class	3	3.25E+07	10,849,983	37.8233	<0.001
Treatment Size	4	6.66E+07	16,652,598	58.0513	<0.001
Canopy Cover	2	5.98E+06	2,992,134	10.4306	<0.001
Error	1093	3.14E+08	286,860		

Table 2.6. continued.

F08

Pre-treatment variable	df	Sum of squares	Mean squares	F	*P*
Season	3	4813477	1,604,492	4.5453	0.004
Time of Day	3	3.08E+06	1,025,044	2.9038	0.034
Treatment Type	1	1.01E+07	10,104,297	28.6243	<0.001
Vegetation Class	3	3.68E+06	1,225,453	3.4716	0.016
Treatment Size	3	7.23E+06	2,411,180	6.8306	<0.001
Canopy Cover	2	1.77E+07	8,826,786	25.0053	<0.001
Error	586	2.07E+08	352,997		

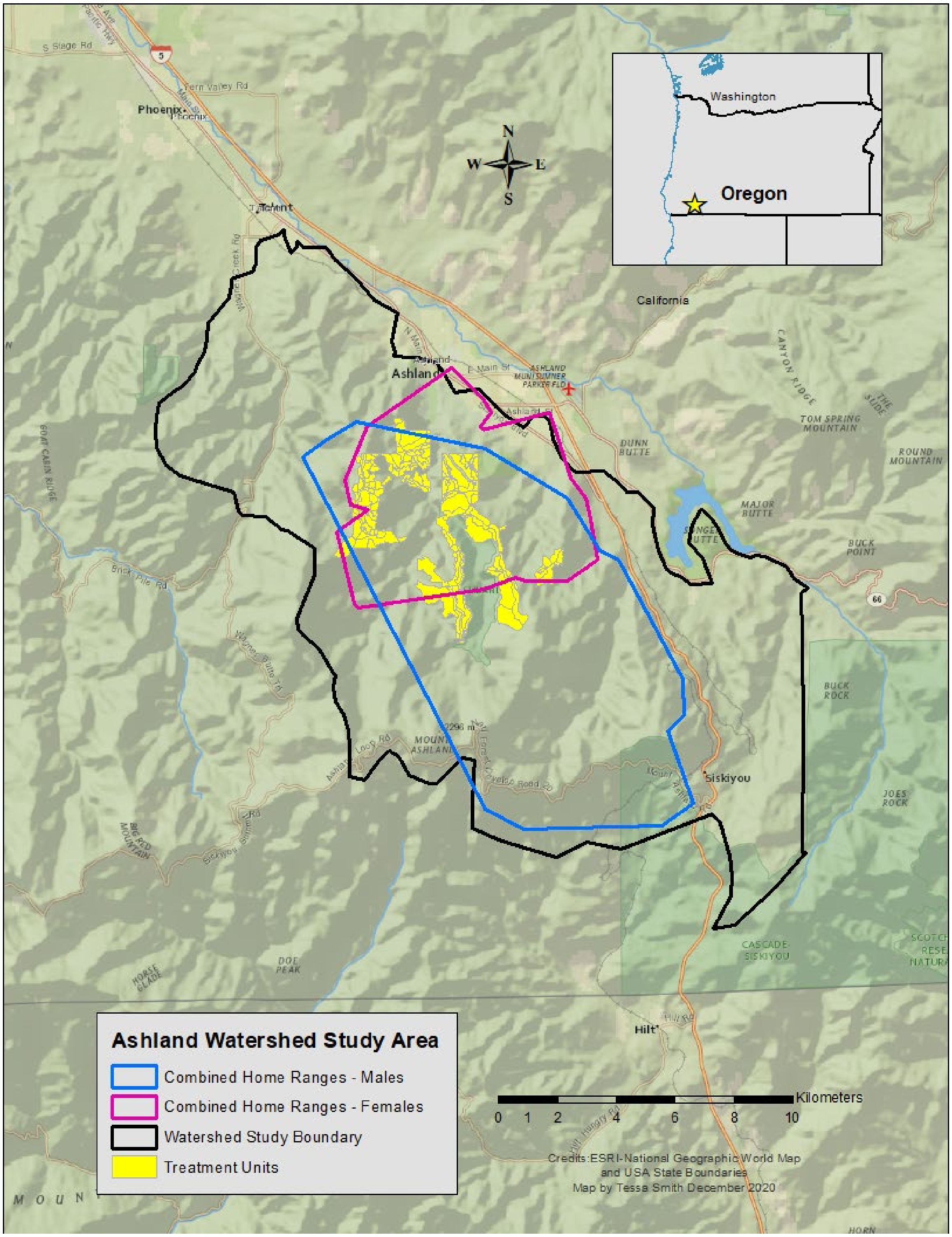
Post-treatment variable	df	Sum of squares	Mean squares	F	*P*
Season	3	1.87E+07	6,228,228	14.3848	<0.001
Time of Day	3	2.05E+06	682,085	1.5754	0.193
Treatment Type	1	3.69E+06	3,688,283	8.5185	0.004
Vegetation Class	3	2.65E+06	883,013	20.394	0.106
Treatment Size	4	2.21E+08	55,348,111	127.8325	<0.001
Canopy Cover	3	3.95E+06	1,316,485	3.0406	<0.001
Error	2542	1.10E+09	432,974		

Table 2.6. continued.

F09

Pre-treatment variable	df	Sum of squares	Mean squares	F	*P*
Season	3	37674030	12,558,010	37.8572	<0.001
Time of Day	3	1.04E+07	3,457,576	10.4232	<0.001
Treatment Type	1	2.12E+07	21,203,886	63.921	<0.001
Vegetation Class	4	1.40E+07	3,492,494	10.5284	<0.001
Treatment Size	4	9.23E+06	2,307,287	6.9555	<0.001
Canopy Cover	3	2.44E+07	8,138,593	24.5345	<0.001
Error	2233	7.41E+08	331,720		

Post-treatment variable	df	Sum of squares	Mean squares	F	*P*
Season	3	1.56E+08	51,848,104	114.1779	<0.001
Time of Day	3	4.67E+06	1,556,832	3.4284	0.016
Treatment Type	1	1.03E+07	10,275,629	22.6286	<0.001
Vegetation Class	4	4.38E+07	10,954,787	24.1242	<0.001
Treatment Size	4	1.01E+08	25,268,778	55.6459	<0.001
Canopy Cover	3	9.17E+05	305,822	0.6735	0.568
Error	4712	2.14E+09	454,099		



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Map by Tessa Smith December 2020

Figure 2.1. Map of the Ashland Watershed Study Area, Oregon, 2010-2017.

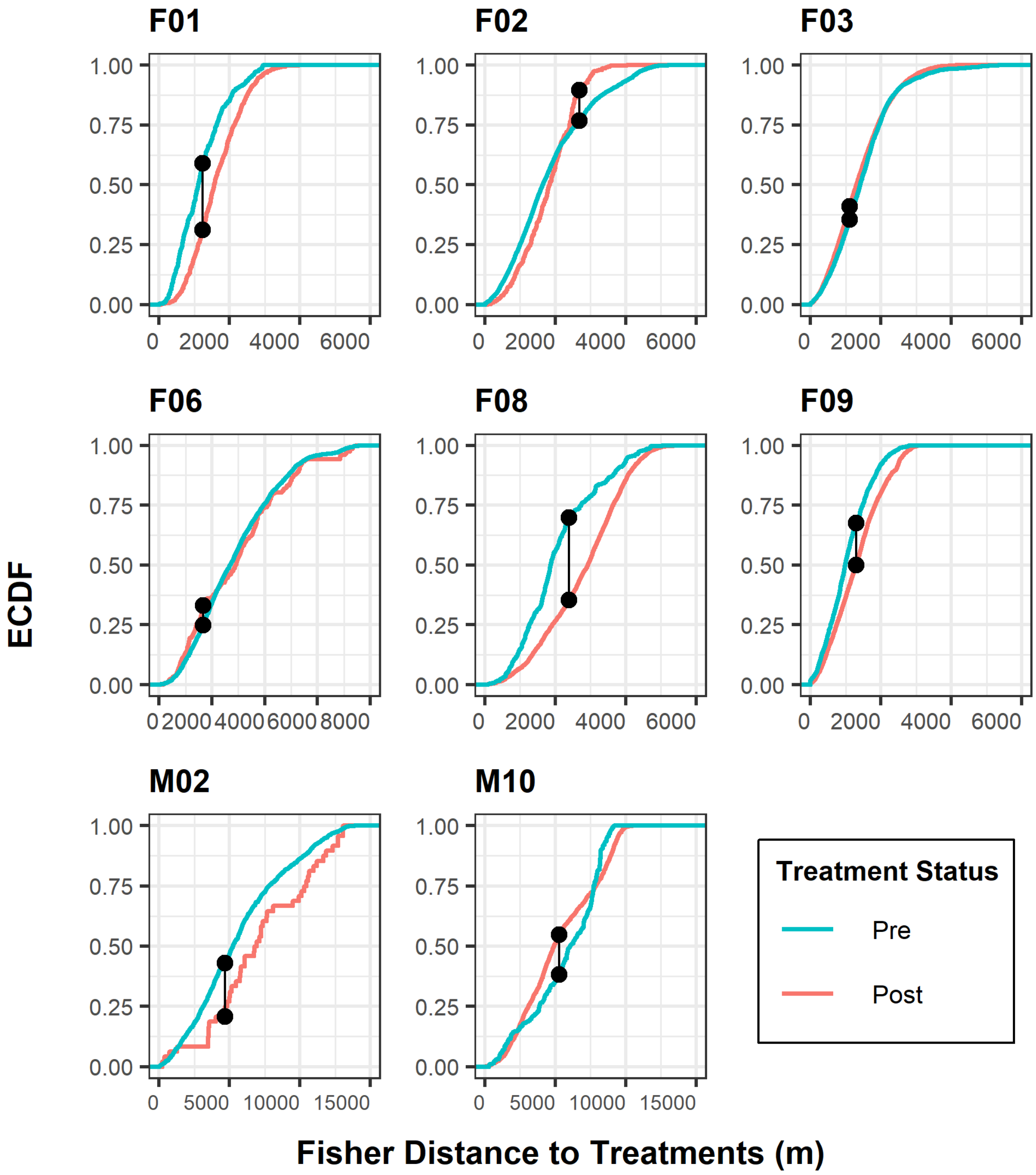


Figure 2.2. Kolmogorov-Smirnov test for individual fisher response to treatment effects in the Ashland Watershed Unit, Oregon, 2010-2017. ECDF is the empirical cumulative distribution function, defined as the probability distribution of two observed curves. The connected black points indicate where the maximum separation between each distribution curve occurs in relation to the distance (m) from a treated unit. The shifts in distribution lines correspond to tolerance levels for each period. For example, a red curve shifted to the right of a blue curve along the x-axis suggests fishers were further from treated units post-treatment.

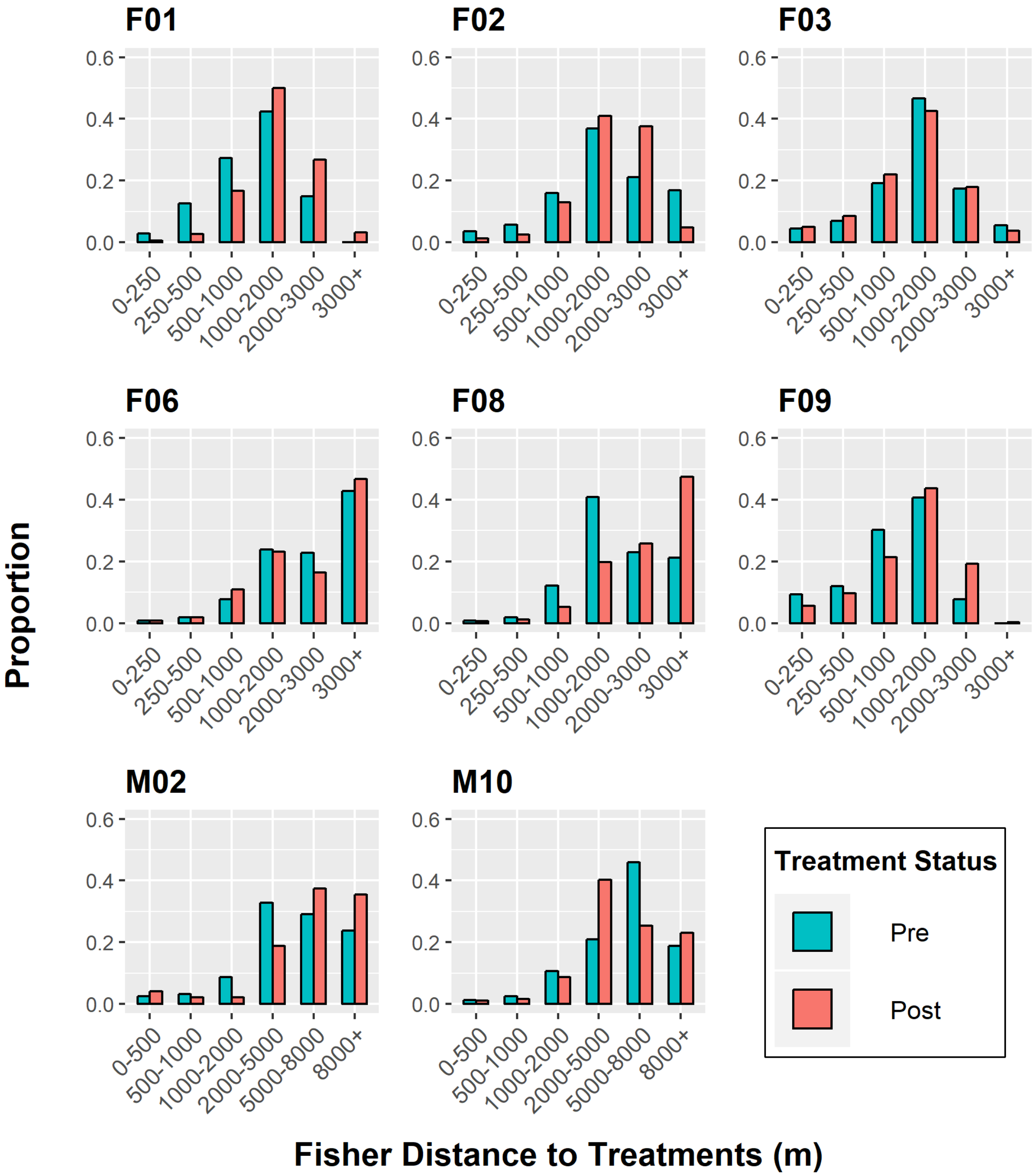
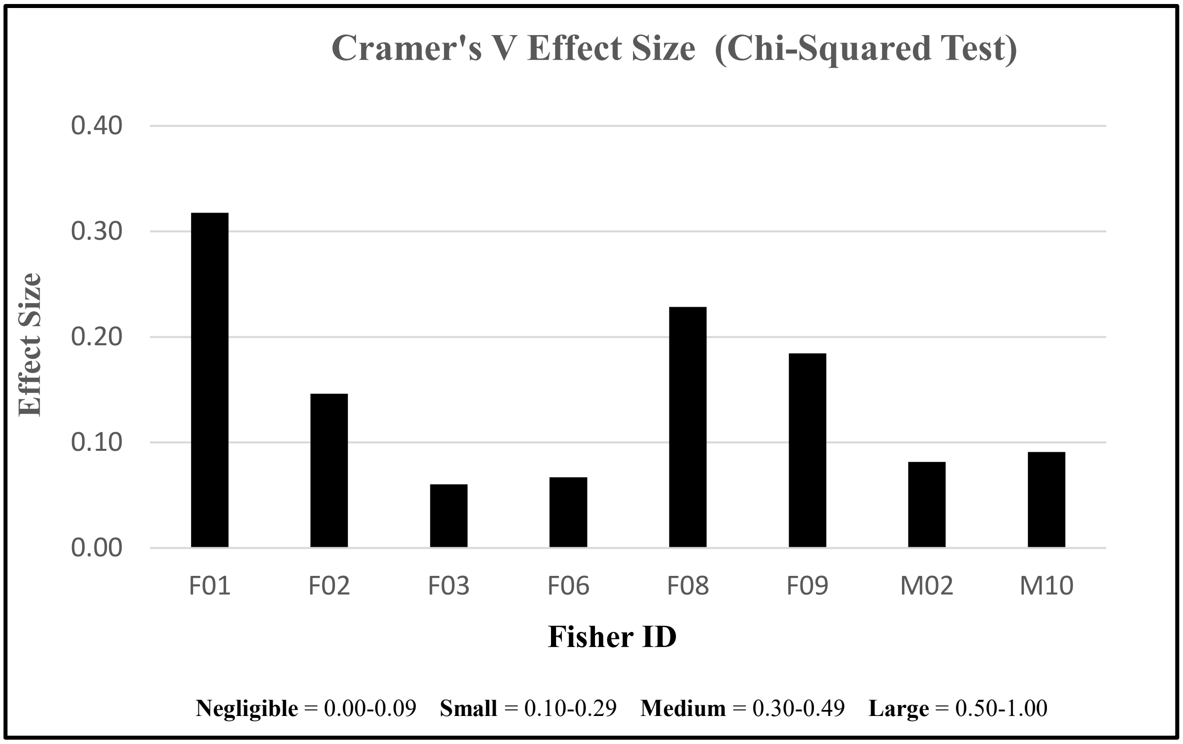


Figure 2.3. Chi-squared independence test to determine differences in individual fisher response to treatment effects in the Ashland Watershed Unit, Oregon, 2010-2017. At distances below ~1000-2000 m, fishers were found closest to treated units in the pre-treatment period (higher frequency of blue bars). In the post-treatment periods, fisher were found further from treated units, signifying a negative response until a threshold distance was reached at ~2000+ m (higher frequency of red bars). Variable responses were found for F03, F06, and M10.

A.



B.

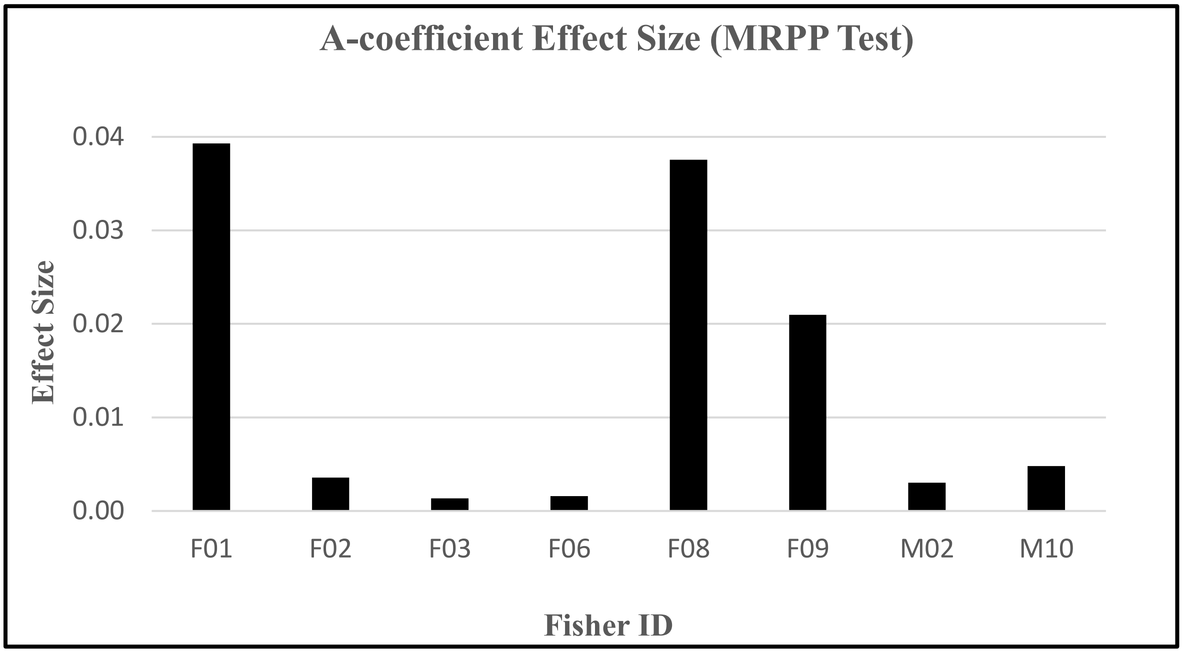
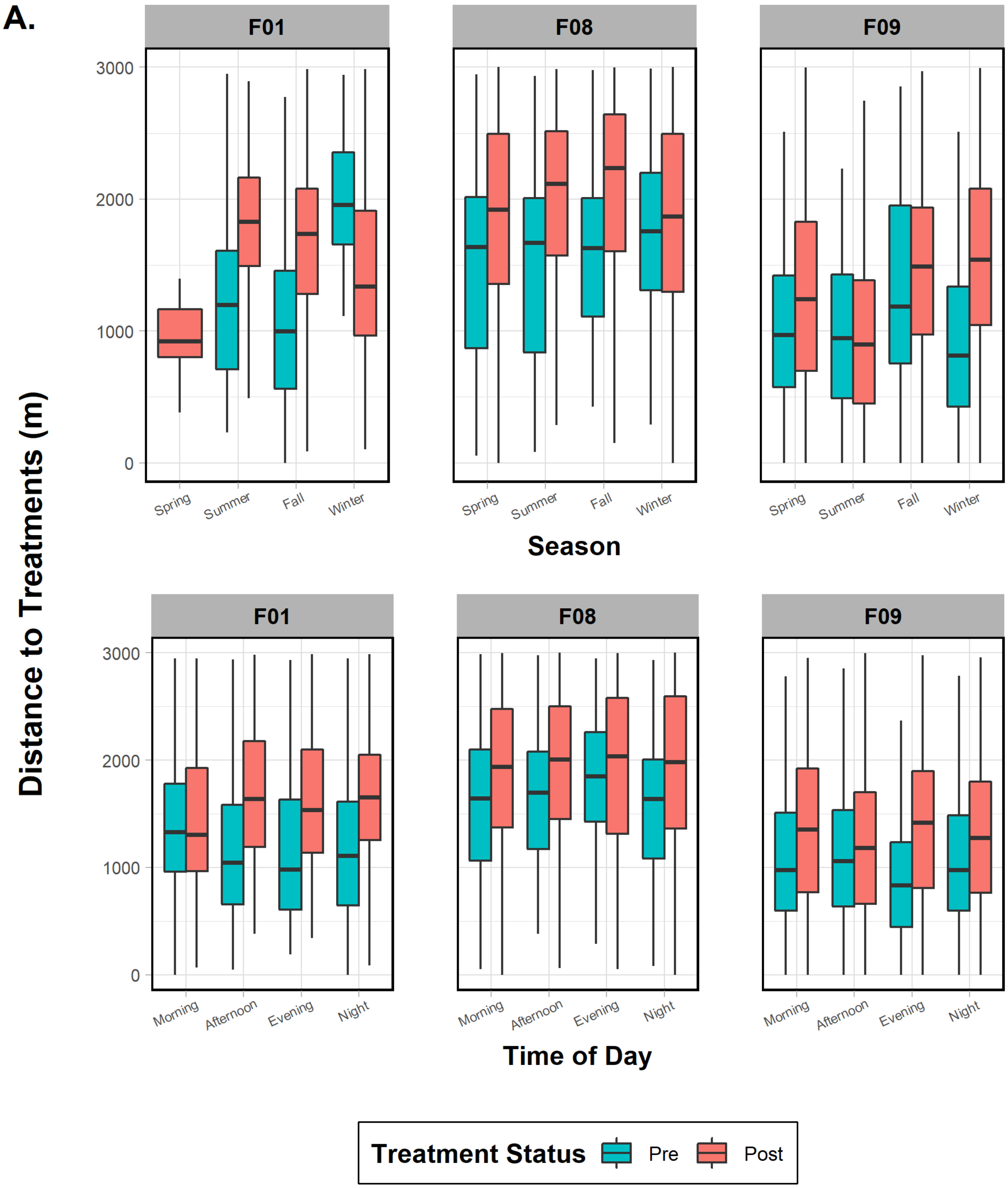
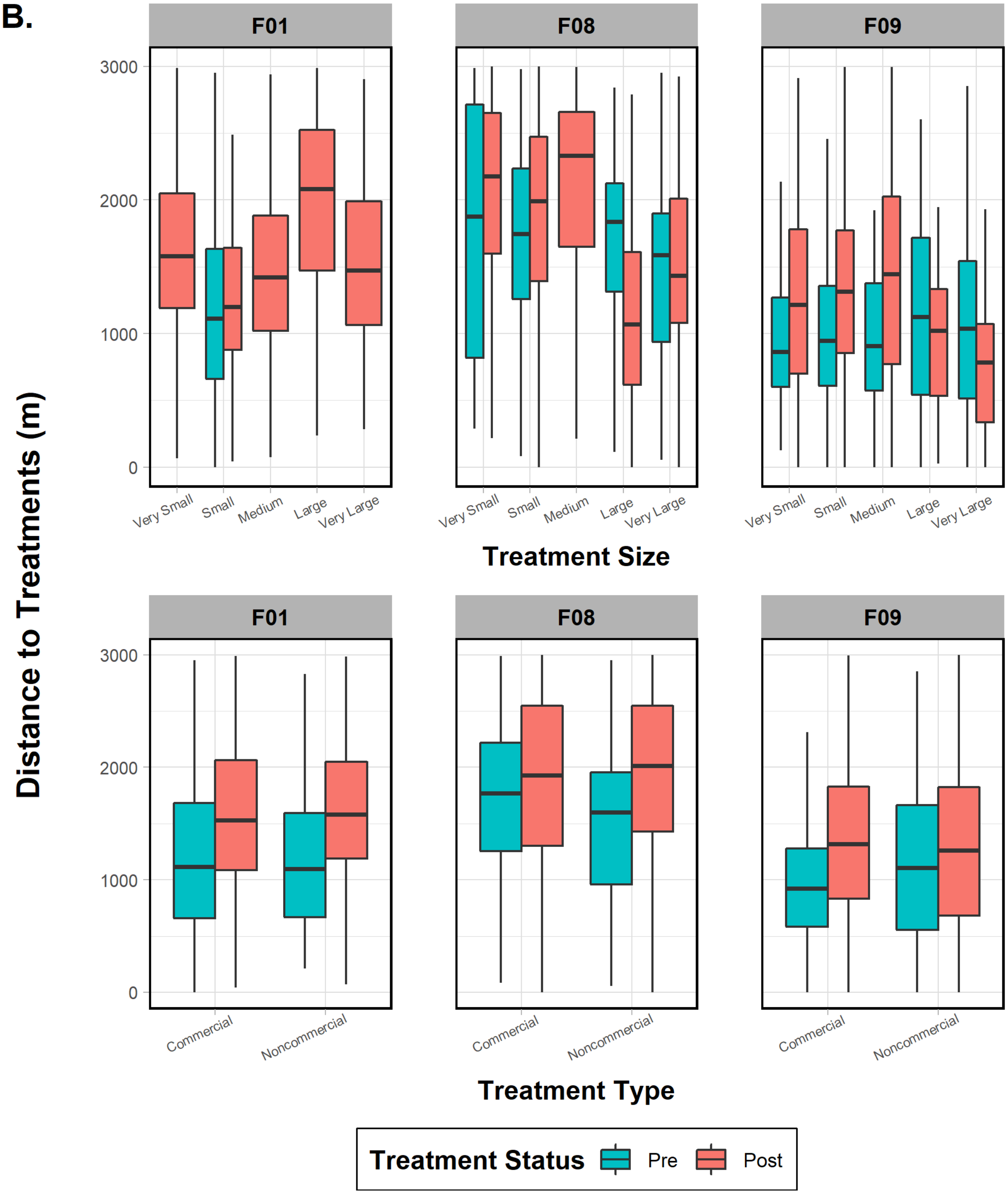


Figure 2.4. Effect size (ES) results measuring the magnitude of each fisher response between pre and post-treatment periods in the Ashland Watershed Unit from 2010-2017. Cramer's V (A) and A-coefficient (B) produced similar patterns. An index to interpret effect size is shown for Cramer's V, but no index was found in the literature for the A-coefficient (MRPP method).


A.
F01
F08
F09
3000
2000
1000
0
Spring
Summer
Fall
Winter
Season
Morning
Afternoon
Evening
Night
Time of Day
Distance to Treatments (m)
Treatment Status
Pre
Post

B.


F01
F08
F09
3000
2000
1000
0
Very Small
Small
Medium
Large
Very Large
Treatment Size
Distance to Treatments (m)
Commercial
Noncommercial
Treatment Type
Treatment Status
Pre
Post

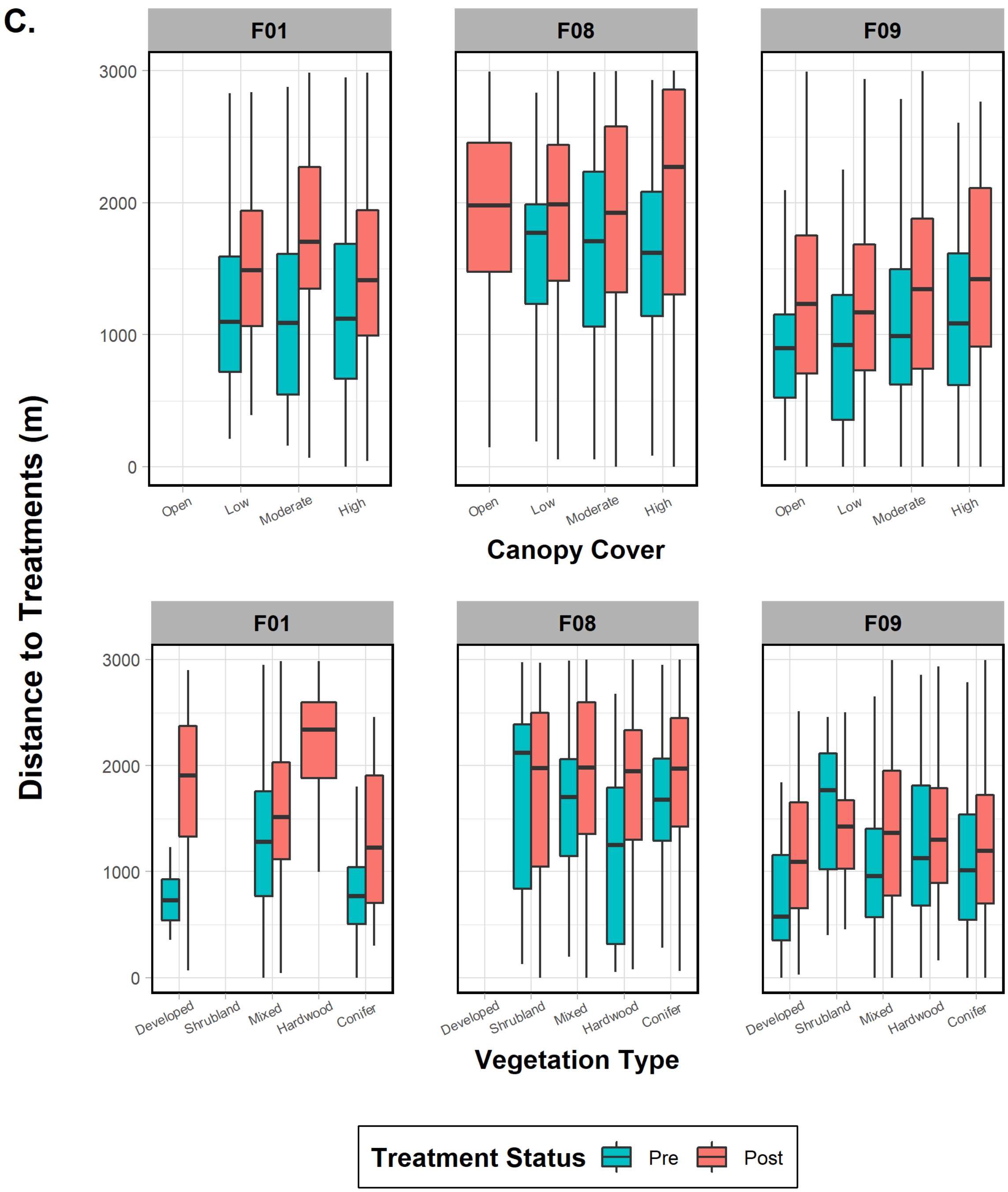


Figure 2.5. Comparison of mean, range, and error bars of environmental and treatment factors influencing fisher distance to treatments in the pre and post-treatment periods for fishers F01, F08, and F09 in the Ashland Watershed Unit, Oregon, 2010-2017. Panel A represents time variables, panel B represents treatment variables, and panel C represents floristic variables. Season, time of day, and vegetation class were significant factors driving fisher response for the pre-treatment period, whereas season, treatment type, and canopy cover were important in the post-treatment period.

CHAPTER 3

TEMPORAL CHANGES IN HABITAT PREFERENCES OF FISHERS (*PEKANIA PENNANTI*) IN A TIMBER-MANAGED LANDSCAPE

ABSTRACT

Currently, forest managers are increasing the amount of fuel reduction treatments in timber stands to mitigate the impact and frequency of high-severity wildfires. However, mechanical thinning may remove key features and alter vegetation assemblages for habitat-obligate species such as the fisher (*Pekania pennanti*). In a fisher population in southwestern Oregon, we investigated their habitat requirements where vegetation changes occurred after thinning treatments were applied across several home ranges. Using GPS collars, we followed 9 individuals (2 males, 7 females) from 2011-2016, evaluating whether resource selection differed as vegetation was incrementally removed every two years. We compared landscape features between used locations versus randomly generated locations by developing resource selection function models for 3 different time periods; one for each biennial time period (i.e., 2011-2012, 2013-2014, and 2015-2016). Topographical variables of interest included elevation, slope, ruggedness, and distance to roads and streams. We also incorporated canopy cover and vegetation classification (or type) in model analysis. When comparing the top-ranked RSF models across the three biennial time periods, elevation was in all of the top models for each RSF, while slope and ruggedness were significant in only one or two top-ranked RSF models, respectively. We believe these three topographical features were strongly correlated with habitat and site productivity, important factors in the creation of mixed-

species stands and structural complexity that fishers prefer. We found a positive association between fisher distance to roads and streams, but the relationship was weak for road distance and significant in only one model each for both variables. Consistent with prior research, canopy cover >60% was selected within home ranges in the first four years of our study, but not in the last two years. We reasoned that prior to treatments, fishers were located at disproportionate rates in densely canopied stands with >60% cover. As a result of thinning practices, the removal of small and intermediate-sized conifers reduced canopy bulk density by ~14%, and created proportional cover across fisher home ranges. However, no stands were reduced below 50% as part of the restoration plan agreement. We found no significant difference in fisher use of stands with or without canopy cover loss post-thinning. This observation suggests fishers not only tolerated a decrease in vegetative cover, but that other habitat attributes (e.g., rest sites, food resources) may be responsible for fisher presence and stand condition fidelity. Although our last biennial RSF revealed that vegetation classification, and not canopy cover, was a significant predictor of fisher habitat use, we exercised caution when interpreting this resource variable, as our sample size for the period consisted of only two individuals. Nevertheless, our research indicated that vegetation changes due to timber management activities did not negatively impact fisher habitat use in the short-term, and we observed no home range shifts during our study. We recommend foresters plan their silviculture activities with fisher resource selection in mind. Habitat mitigation measures may be achieved by retaining high-valued resource structures and sufficient canopy cover within managed units as well as preserving corridors that connect to refugia sites which may be free of treatment effects.

INTRODUCTION

Over the last several decades, the literature has increasingly referenced habitat quality as a key component in the preservation of wildlife species (Garshelis 2000, Devictor et al. 2008, Crooks et al. 2011, Gese and Thompson 2014). Optimal living accommodations for wildlife often consist of areas with diverse foraging opportunities, safe escape cover, and adequate structures to facilitate reproductive potential (Lewis et al. 2016). For some species, these needs can be met even when environmental conditions change through natural or man-made causes (Parsons et al. 2019). The animals may respond by either dispersing to adjacent areas with similar resources or developing new traits to help adapt to changing conditions. However, certain species have evolved with particular habitat arrangements tailored to their life-history traits, such as the fisher (*Pekania pennanti*). Fishers have been defined as a habitat-obligate species requiring specific features or forest configurations for reproductive survival that may be sparsely distributed on the landscape (Zielinski et al. 2005, Purcell et al. 2009, Niblett et al. 2015).

As suggested in Chapter 2, fishers use a multitude of habitat scales for various purposes. A forest-level scale includes general habitat preferences, such as contiguous forest for dispersal and predator avoidance purposes (Sauder and Rachlow 2015). However, the home range and microsite scales are where the “obligate” term is most relevant. At this scale, fishers rely on landscape elements that take decades, or even centuries, to form. Examples include mature stands of large diameter trees, decayed snags, and mixed sub-canopy communities for resting and foraging (Zielinski et al. 2004*a*, Schwartz et al. 2013, Lewis et al. 2016). Likewise, tree species that are capable of forming hollowed-out chambers are essential for females that require safe refuge to conduct annual young-rearing duties (Paragi et al. 1996, Purcell et al. 2009, Weir et al. 2011, Green 2017). However, if these habitat structures suddenly disappear, or are reconfigured (e.g., removal of trees by wildfire or patch creation due to overstory thinning), fisher individuals and populations may suffer depending on the scale and severity of the change. Although vegetation configuration and composition are important qualities, static topographical elements are equally valuable when determining fisher distribution and presence. Previous literature has indicated elevation, slope, and ruggedness influence where fishers are found across various latitudes, whereas there is little support for aspect as influencing habitat selection. Generally, fishers prefer moderate inclines, mid-elevation ranges, and rugged terrain associated with riparian drainages (Zielinski et al. 2004*a*, Purcell et al. 2009, Schwartz et al. 2013, Olson et al. 2014, Sauder et al. 2015, Facka 2017). Elevation, slope position, and terrain features are integral components shaping the configuration, composition, and microclimate of tree stands in forested ecosystems. These topographical attributes facilitate the establishment of den structures providing stable thermoregulatory properties essential to kit survival. For example, hardwoods, a preferred species for fisher den site selection, are found at lower elevations, on moderate to steep slopes, and near stream drainages where snow levels are much lower and cavity temperatures are less likely to fluctuate in spring weather (Zhao et al. 2012, Thompson and Purcell 2015). Additionally, the decreased snowpack and cool environments at these locales enable fisher to easily travel while hunting prey and avoiding their main competitor, the marten (*Martes americana*), who navigates deeper snowpack at higher elevations (Raine 1983, Pozzanghera et al. 2016).

In addition to topographic variables, fisher habitat selection can be influenced by the presence of physical features such as roads and streams. Distance to the nearest road or stream can represent a selection for or against each linear feature, and the level of use depends on anthropogenic disturbances or natural selection factors encountered there. For example, fishers were found further from logging roads, possibly to avoid heavy traffic or potential predator risks (Sweitzer et al. 2015, Lewis et al. 2016). In contrast, fisher proximity to streams indicates a preference for riparian corridors that may contain a high density of prey or mature stands with rest structures (Zielinski et al. 2004*a*, Purcell et al. 2009). Additionally, the relative distance of fishers to either feature might be contingent on factors such as dispersal routes, competitive interactions, foraging behavior, or disturbance events.

Finally, vegetation characteristics and configuration can influence where fishers are distributed. Fishers are generally associated with mid- to late-seral stands consisting of mature trees and a complex arrangement of vertical and horizontal structures (Buskirk and Powell 1994, Purcell et al. 2012). Within mixed-aged stands, varying tree and shrub species occur on both mesic and xeric sites, while multi-layered canopies and ground debris support diverse prey resources and safe travel paths for mesopredators (Manning and Edge, 2008, Perry et al. 2011). Although the core of fisher home ranges contains stands of high tree basal area and dense canopy cover, there is also evidence fishers incorporate peripheral edges of younger stands into their home range to diversify their prey base (Sauder and Rachlow 2015). Creating this mosaic of stand age and species diversity across a fisher home range requires a natural progression of vegetation growth and mortality events normally achieved by succession and episodic, mixed-severity wildfires (Spies and Franklin 1991). However the combination of historic logging practices and fire suppression has altered this landscape pattern, and the new, homogenous landscape is more likely to support large, high-severity wildfires (Collins and Roller 2013, Ager et al. 2017). These fires create a self-sustaining cycle where subsequent fires or other disturbances act uniformly across the landscape, and maintain large, homogenous patches. Not only does this new cycle threaten human communities and values through fire severity, it destroys the finer-scale mosaic that species such as fisher rely upon (Perry et al. 2011, Mallek et al. 2013).

In an attempt to break this cycle and re-establish the finer-scale mosaic, forest managers employ commercial harvest, mechanical thinning methods and prescribed burns to mimic natural disturbance events without threatening valuable resources (Hessburg et al. 2016). While not a perfect surrogate, these efforts create a landscape where fire can be safely reintroduced and allowed to play a more active role. These management actions are particularly critical near communities where fuel loads are high and fires under the current conditions could be catastrophic (Mori and Johnson 2013). However, whether or not the rate of modification and recovery of forest stands negatively affects fisher habitat selection remains unanswered, and this question has been a focal point of recent research efforts.

In the past century, episodes of aggressive timber management (e.g., clear cuts) drastically changed forest characteristics where wildlife species coexisted in rare areas such as old-growth forests (Tempel et al. 2014), leading to a decline in population demographics of habitat-obligate species. Studies in Maine, Utah, and British Columbia reported timber harvests negatively impacted marten movement in cut stands (Soutiere

1979, Hargis et al. 1999, Sullivan et al. 2017). Furthermore, even partial harvests of dense stands lowered marten presence, resulting in an increased use of neighboring secondary growth forests (Fuller and Harrison 2005). Intensive harvests also contributed to population shifts and declines for small mammals, such as snowshoe hare and flying squirrels, important prey base species for furbearers (Ferron et al. 1998, Manning et al. 2012, Sollman et al. 2016). However, few studies have reported the impacts timber harvests have on fisher habitat use or occupancy, despite considerable logging activities executed across western forests in the past century (Buck et al. 1994, Aubry and Lewis 2003). Our study did not include examination of clear-cut harvest effects on fisher use. Instead, we sought to determine whether frequent thinning projects, where canopy cover and stand density were reduced using light, restorative applications, resulted in fishers selecting for or against the same resources as habitat conditions changed through time. Our main objective was to evaluate which features were important for habitat selection explicit to a fisher population near Ashland, Oregon. Given the region's high biodiversity and unique edaphic characteristics, old-growth conditions in the Ashland watershed were considered the "gold standard" for habitat-obligate species such as the spotted owl (*Strix occidentalis*), Humboldt marten (*Martes americana humboldtensis*), northern-flying squirrel (*Glaucomys sabrinus*) and the fisher (U.S. Department of Agriculture 2008). Because fishers require specific habitat needs, we expected them to disproportionately select areas that had high-valued resources (e.g., dense canopy cover) but were in short supply across the study area, compared to other available habitat features frequently encountered on the landscape (Underwood et al. 2010, Lewis et al. 2016). Historically, fishers inhabited these valued areas across a large extent, and adapted to resource fluctuations because natural stand development was slow and minimal in scope (Franklin et al. 2002, Zielinski et al. 2013). However, the rapid pace of timber extraction accelerated potential resource changes in the environment, and may have precipitated the rate at which fishers encountered sub-optimal habitats. In our research, we attempted to uncover fisher habitat preferences relative to vegetation changes resulting from timber extraction methods occurring in the Ashland study area. We predicted fishers would be positively associated with topographical features such as mid-range elevation gradients and moderately rugged slopes, which were previously recorded as important constituents in fisher habitat suitability (Zielinski et al. 2010, LoFroth et al. 2011). Additionally, we anticipated fishers would select moderate to high canopy cover (>50%), and expected them to reside in conifer or hardwood-conifer forests rather than lowland shrub habitat.

Although previous research emphasized specific vegetation and topographic features to be suitable habitat variables for fishers, resource selection likely differs in each geographical region. Our study area contained unique biotic and abiotic factors, and we wanted to ascertain whether habitat preferences in the Ashland watershed were consistent with fisher in other areas. Consequently, our results would help fill in knowledge gaps regarding habitat selection of fisher populations in southwest Oregon and as well as the impact of critical management actions on fisher conservation plans. For instance, our findings may impact future listing initiatives by the U.S. Fish and Wildlife Service (USFWS). Recently, a proposal to list the distinct population segment (DPS) of fishers throughout California, Oregon, and Washington resulted in the southern Sierra population (CA) being given federal endangered species status (U.S. Fish and

Wildlife Service 2020). The northern California, Oregon, and Washington populations were denied protected status due to stable population levels and the lack of imminent threats to fisher in the Pacific Northwest. However, with a surge in yearly wildfires and rapid expansion of treatment applications in the region, our research can help determine if fisher population levels and habitat preferences are negatively impacted by these events and whether listing is later warranted by the USFWS.

Finally, our fisher population existed in a highly diverse biotic community, and we assumed they would select similar habitat resources (e.g., vegetation composition, elevation gradients, topographical roughness) as other populations across the western United States. If this hypothesis holds true, then the possibility of constructing habitat suitability models could be instituted on a larger geographical framework. The ramifications of connecting related habitat patches in neighboring forests would be instrumental for successful fisher conservation. The challenge of this initiative is that the mosaic of historical fisher habitat has been lost due to heavy logging and fire suppression. Forested ecosystems are now fraught with homogenous stands and undesirable conditions that are prime targets for stand-replacing wildfires in which post-impact recovery may not be feasible (Collins and Roller 2013). Nevertheless, thinning applications and fuel reduction methods may ease the repeated loss of suitable habitat needed for fisher presence (Scheller et al. 2011). Conceivably, our findings could help contribute important management considerations in forestry practices, particularly in the context of preserving sensitive areas where legacy trees and canopy cover are required components for forest-obligate species. In addition, we believe our results regarding fisher habitat selection would provide managers insight when planning applicable threshold limits for vegetation removal relative to fisher tolerance levels. Thus, timber management plans could be modified to accommodate changing fisher conservation needs at variable habitat scales and in regionally different ecosystems.

STUDY AREA

Our study area was in the Siskiyou Mountain range near Ashland, Oregon (Fig. 3.1). The 6,300 acre study site was designated in the late 1800s as a municipal watershed of the city, with various government and private entities maintaining joint ownership. A majority of our fishers resided in the area managed by the U.S. Forest Service, Rogue-River and Klamath National Forests. However, the Bureau of Land Management, City of Ashland, Lomakatsi Ecological Services, and The Nature Conservancy also managed or performed research within the watershed (U.S. Department of Agriculture 2008). The climate consisted of warm, drought-like summers and mild, wet winters. Temperatures often exceeded 38º C, with little precipitation in the summer, resulting in dry conditions favorable to wildfire from dry lightning strikes (Franklin 1972). Topography was extreme, with steep slopes and an elevation range of 600 m to 2000 m above sea level that vertically spanned only a few miles. Due to its ruggedness, productive soil types, and Mediterranean climate, the floristic composition varied widely. The site hosted some of the highest plant diversity in the western United States (DellaSala 2006). The plant association groups (Agee 1993) found in the study area included mixed-conifer zones in the higher elevations and xeric species in the lower elevations. Shrubs, grasses, and forbs dominated the forest understory, with a dense canopy of mixed-conifer or hardwood components (U.S. Forest Service 2005). Common tree species were Douglas-fir

(*Pseudotsuga menziesii*), white fir (*Abies concolor*), ponderosa pine (*Pinus ponderosa*), sugar pine (*Pinus lambertiana*), Incense cedar (*Calocedrus decurrens*), Pacific madrone (*Arbutus menziesii*), California black oak (*Quercus kelloggii*), and Oregon white oak (*Quercus garryana*). Small timber harvests occurred throughout the watershed over the last century, though extensive fire suppression efforts created a forest thick with Douglas-fir stands throughout the watershed. Fire return intervals for the region were generally 5 to 75 years; though fire regimes were hard to define due to the vegetative diversity (Agee 1993).

METHODS

Capture and Monitoring

Between 2010 and 2017, fishers were captured and fitted with radio transmitters. Trapping usually occurred in late fall and winter, with occasional targeted trapping in the summer to replace radio-collars that were old or had failed. We followed the fisher capture protocol outlined in Green (2017) that limited trapping in the spring during sensitive breeding and kit-rearing periods. Approximately 20-25 box traps (Model 108, Tomahawk Live Trap, Hazelhurst, WI) were placed in the watershed boundary area in all representative habitats during the first year. In subsequent years, we trapped in areas of known fisher presence to recapture individuals and maximize trapping success. We had limited personnel and equipment, therefore traps were placed in what was deemed “suitable fisher habitat”, usually within 100 m of roads, in mixed forest conditions, and along riparian corridors. The cage traps had an attached cubby box to provide protection from the elements and was baited with raw chicken (Seglund 1995). Scent lures were spread around trees at the site and inside the trap to attract fishers ((Hawbaker's Fisher Lure, Hawbaker and Sons, Fort Loudon, PA; Fisher Red Lure, Proline Lures, Indianapolis, IN; Cavin's Gusto, Minnesota Trapline Products, Pennock, MN; Outreach, Proline Lures, Indianapolis, IN). We checked traps once a day and processed individuals at the trapping site according to protocols set by the Kings River Fisher Project Institutional Animal Care and Use Committee (No. 2018-01), California Department of Fish and Wildlife permit (No. SC-5479, SC-2730), and Oregon Department of Fish and Wildlife permit (2017 029-17). We complied with all handling procedures using guidelines from the American Society of Mammalogists (Sikes et al. 2011).

Fishers were restrained using an aluminum or steel handling cone and anesthetized using a mixture of ketamine (22.5 mg/kg) and diazepam or midazolam (0.125 mg/kg) via a hand-held syringe. We collected morphometric, disease, and genetic samples (hair follicles and ear tissue biopsies) to determine fisher health in the population, and we inserted passive integrated transponder (PIT) tags subcutaneously between the shoulder blades to identify individuals. We took a 1-ml blood sample from the jugular or femoral vein of each fisher to assess disease prevalence in the population. Additionally, we sexed and aged individuals based on weight, sagittal crest development, and tooth wear. We placed fishers into one of three age classes: juvenile, subadult, or adult (Sauder and Rachlow 2014, Green 2017). Finally, we fitted micro-GPS collars (Quantum models, Telemetry Solutions, Concord, CA) or VHF collars (Holohil System Ltd., Carp, Ontario, Canada) to adults and subadults with a custom breakaway attachment for collar expansion as fishers grew in size (Thompson et al. 2012). Adult fishers were fitted with GPS radio collars, with the exception of two adults and a subadult receiving

VHF radio-collars. No collars were affixed to juvenile fishers or those animals where the collar weighed more than 4% of the total body weight. After collaring, we released fishers at the site of capture and monitored them for 24 hours to assess their recovery.

At the time of our study, micro-GPS collars had not been widely used on fishers or other species of equivalent size. We experimented with a variety of schedules for timed fixes and location accuracy. Other research on wildlife taxa suggested rugged topography and dense overstory vegetation might interfere with successful GPS fix rates on individual locations, and we found this to be true in our study (Frair et al. 2004, Sager-Fradkin et al. 2007). In order to account for these limitations, more fixes at shorter intervals were recommended (Frair et al. 2004, Frair et al. 2010); however, we required our collars last for several months on denning females when trapping needed to be avoided. Our solution was to attempt fixes every 10 hours in order to maximize battery efficiency and gather an appropriate sample size of points. Furthermore, we placed test collars in the field to calculate GPS-collar accuracy (i.e., error rate) for 48 hours and compared the collar locations to known Garmin GPS device locations. We located fishers to assess survival once a week using ground telemetry acquired through the VHF transmitters incorporated into each GPS collar. Location data stored on the GPS collars were downloaded every 3-4 weeks during these ground-searching telemetry events.

Statistical Analysis for Resource Selection Function

We used a resource selection function (RSF, Manly et al. 2002) to assess habitat selection for fishers in the Ashland Watershed area. The RSFs primarily compare the probability of used to available fisher location points and their respective habitat variables of interest (e.g., topographic or habitat parameters). In our RSFs, we predicted models would successfully interpret preferred habitat features relative to changing vegetation conditions as a result of thinning treatments applied in fisher home ranges. Specifically, we performed biennial RSFs at the population level for fishers to assess whether they selected for or against habitat resources where vegetation components changed due to anthropogenic practices. Moreover, we retained topographical variables in each RSF group in order to elucidate features that were static but equally important to fisher habitat suitability overall.

We started our analysis by generating home ranges for each fisher using the Geospatial Modeling Environment tool (Beyer 2015) in ArcGIS 10.3.1 (Environmental Systems Research Institute, Inc., Redlands, CA, USA). Points that were within a 10-hr time span were manually removed from our data to eliminate any temporal autocorrelation. If locations were spatially but not temporally close, we retained them in our data because they may have indicated an important resource feature was nearby (e.g., den site). The final points were used to calculate 100% minimum convex polygon (MCP) home ranges. We obtained home range estimates by calculating the required number of locations to reach an asymptote of an area-observation curve, which resulted in approximately 25 locations per fisher (Haines et al. 2006).

Within each home range, we used a matched case-control method (Whittington et al. 2005) for our RSF logistic regression. We placed a buffer around each used location that was the average distance moved between 10-hr location fixes for each fisher. Inside the buffer, we then generated 25 random points to match each used point (Northrup et al. 2013). We further constrained each random location by a 30-m buffer from the used locations so no overlap occurred among random locations and to conform to the properties of all 30x30 m raster layers in ArcGIS. We extracted variables of interest for all used and random locations using R Studio 3.5.3 (RStudio Team [2015] RStudio: Integrated Development for R. RStudio, Inc., Boston, MA, URL http://www.rstudio.com/) and ArcMap 10.3.1 (Environmental Systems Research Institute, Inc., Redlands, CA).

Parameter Design for RSFs

Although the foresters provided ground-level habitat metrics (e.g., basal area, canopy cover, log density) during their pre- and post-treatment applications, we deemed it unsuitable for our analyses due to data inconsistencies and missing information. Furthermore, we could not replicate the methods employed inside the treatment units to our random locations that were generated outside of treated units. We investigated using gradient nearest neighbor (GNN) metrics, which delivered a multitude of fine-scale information on vegetation components from Forest Inventory Plots and satellite imagery in 30-m raster cells (Ohmann and Gregory 2002); however, no data after 2012 was available for use at the time of analysis. Instead, LANDFIRE (www.landfire.gov/vegetation.php) data provided us with landscape covariates for canopy cover and vegetation classification (or type) in our models. We downloaded vegetation data, which is delivered in a standard program format of 2-year increments (i.e. 2009-2010, 2011-2012, etc.). However, our records totaled 7 years (2010-2016) and was delivered in a yearly format. Because we did not have any fisher locations collected for the 2009 portion of 2009-2010 LANDFIRE timeframe, the 2010 habitat and location data were dropped from analysis, and we grouped the GPS location data as follows:

2011-2012, 2013-2014, and 2015-2016. This also resulted in the omission of a male fisher (M02) from analysis since the only locations recorded were in the year 2010.

We imported LANDFIRE data into ArcGIS and vegetation values for canopy cover and vegetation type were extracted for each used and random point. Canopy cover values were automatically generated as a midpoint value between binned categories by LANDFIRE. For example, points that fell in the 0-10% canopy cover were given a value of 5%, points within a 10-20% canopy cover were valued at 15%, and so on. For interpretation during model analysis, we categorized canopy cover into 5 classes: open-low (0-19%), low-moderate (20-39%), moderate (40-59%), moderate-high (60-79%), and high (≥80%) canopy cover. Similarly, we reclassified vegetation types given by LANDFIRE into 4 classes, with representative proportions found throughout the study area low elevation shrubland (24%), Sierra mixed conifer (19%), conifer-hardwood (17%), and conifer (40%). Distinct vegetation assemblages existed within these categories (Table A.2), and we wanted to clarify whether certain types were of more importance to fisher selection within the home range.

We were unable to overlay the treatment polygon layer with the LANDFIRE raster layer to confirm whether LANDFIRE successfully detected canopy cover changes. This was due to the way LANDFIRE combined their data into 2 year increments, which was not comparable to our treatment polygon layers from the AFR team which were updated yearly. Instead, we detected changes in habitat conditions in successive years by calculating the difference in raster pixels between LANDFIRE biennial year data (e.g., 2011-2012 pixel values subtracted from 2013-2014 pixel values, etc.). This resulted in positive (vegetation regrowth), negative (vegetation loss), or no change in habitat conditions after a disturbance (i.e., mechanical treatment) occurred on the landscape. Errors associated with the habitat classification between biennial years in LANDFIRE may have been present because the organization remapped the landscape in 2016; though discrepancies from previous map years were corrected before they released the newest map. Despite the improved map products, we recognized that because the AFR treatments were applied lightly in order to reduce environmental impact, it was possible LANDFIRE did not detect small habitat changes that may have occurred in select stands, and should be identified as an inherent property of remote sensing technology at courser scales.

For topographical features, we obtained elevation values from a 30-m resolution digital elevation model (DEM, https://viewer.nationalmap.gov/basic/) and then derived slope, aspect, and ruggedness using code in RStudio 3.5.3 (RStudio Team [2015] RStudio: Integrated Development for R. RStudio, Inc., Boston, MA URL http://www.rstudio.com/). Aspect was given as a continuous variable in degrees but reclassified into intermediate points between the 4 cardinal directions: (North: ≥315 to <45), (East: ≥45 to <135), (South: ≥135 to <225), and (West: ≥225 to <315) for ease of model interpretation. Ruggedness in our region was defined as terrain that had steep slopes or uneven ground, and which often included descriptive features such as rock outcroppings or deep canyons (Franklin 1972). Much of our study area exemplified rugged terrain, but with varying degrees of slope gradients relative to physiographical properties. Terrain ruggedness was computed as a vector ruggedness measure (VRM, Sappington et al. 2007) which calculates ruggedness using three-dimensional dispersion of vectors orthogonal to the terrain, without slope correlation as a confounding variable.

Generalized Linear Model (GLM) Development for RSFs

For our purposes, linear regression models were appropriate since the dependent variable was binary (i.e., used and not used) and the covariates and response variable distributions were non-normal (Zuur et al. 2010). However, our RSF was limited to using a GLM framework without random effects due to small sample size. Ideally, a GLMM (generalized linear mixed models) would have included a random effect in order to explain the non-independent data points from each individual fisher (i.e., repeated measures). We initially planned on constructing GLMM models for our data; however, we encountered singular fit and convergence errors during the process, indicating our data were overfitted and contained too many parameters to support a parsimonious model.

In our GLM models, we standardized all continuous coefficients in order to eliminate convergence errors and to effectively compare all variables on the same scale. In addition, we tested for collinearity in the explanatory variables using a Pearson's statistical test (Mukaka 2012), removing covariates with a correlation coefficient >0.6. Variance inflation factors (VIF) were also calculated to assess multicollinearity, and variables with a VIF >3.0 were taken out of model runs (Whittington et al. 2005). Exploratory univariate analysis helped decide which covariates would be used in multivariate candidate models. Models within each RSF analysis were ranked using Akaike's Information Criteria with a correction for small sample size (AICc, Brewer et al. 2016). The AIC method used maximum likelihood to first assess the deviance of the models and then ranked them with AIC scores and respective delta (Δ) weights. The most parsimonious models had the lowest AIC scores, indicating a compromise between model complexity and model fit, to achieve the best predictive ability with the least information loss. According to Burnham and Anderson (2002), models with ΔAICc of ≤ 2.0 should be retained as they have the highest empirical support.

Model Validation

Model validation was completed with k-fold cross validation, a procedure that evaluated the predictive power of a model on unseen data (Boyce et al. 2002). Essentially, the data were randomly partitioned into *k* groups, or folds. One group was held as a test set and the remaining groups were training sets with a model applied to them. The training sets were evaluated on the test set to score how well the model fit the test data. We chose to use 10 folds for our cross validations as it decreased bias between the training and test sets (Kuhn and Johnson 2013). We performed further model validation by computing area under the curve (AUC) under receiver operating characteristics (ROC). AUC probability curves predicted how well the model could classify binary outcomes correctly (e.g., true versus false positives). Models with an AUC of 0.5 indicated no classification skill, whereas an AUC of 1.0 indicated perfect discrimination between true and false positive predictions (Fielding and Bell 1997). All analyses were conducted using R codes and packages in R 3.5.3.

RESULTS

Capture and Location Data

A total of 40 individual fishers (23 F, 17 M) were captured throughout the study from 2010 to 2017. The 2010 and 2017 data sets were excluded from RSF analyses as LANDFIRE was not available for those years. Nine fishers, 7 females and 2 males, had an adequate number of data points to conduct RSFs (Fig. 3.1). We followed eight fishers the first two years (7 females, 1 male), 6 fishers in the middle two years (4 females, 2 males), and 2 females the last two years. Female home ranges were significantly (Wilcoxon test, W = 0, $P = 0.02$, Table 3.1) smaller (16.27 km^2 ± 8.87) than male home ranges (69.29 km^2 ± 31.19). Acquisition rates were substantially low for the collars, with a mean fix rate of 32% for all individuals. The mean location error rate was 71.69 m ± 125.94, though more than half (55%) of fixes were within a 30 m buffer of the actual GPS location. Fisher traveling distances between 10-hr locations ranged from 635 to 3367 m with the lowest and highest values from the two male fishers ($\bar{x} = 1892$ m ± 1357.62). Female distances varied between 872 and 2113 m ($\bar{x} = 1178$ m ± 429.82). Location fixes were not always 10 hours apart and varied from 5.5 to 10 hours ($\bar{x} = 8.09$ ± 1.51 for females, $\bar{x} = 8.20$ ± 2.39 for males). Our Wilcoxon ranked sum tests did not find significant differences between males and females in the distance traveled (W = 8, $P = 0.67$) or the lag time between distance locations (W = 9, $P = 0.89$, Table 3.1).

Used points from 2011-2012 comprised the most locations from any of the study years, mainly due to the availability of collar equipment and personnel to trap fishers at that time. Locations substantially dropped off in the ensuing years of 2013-2014, with the fewest number of used points available for analysis in the last year group (2015-2016). Only two female fishers (F01, F03) retained their collars for the entire project (Table 3.2). All other collars either failed, were out of range, malfunctioned, or the fisher died or dispersed out of the study area.

Resource Selection Functions for Habitat Selection

Only in the last biennial period (2015-2016) were two covariates found to be correlated: elevation and distance to stream, and thus these covariates did not appear in the same model. Collinearity was absent among all other variables across all other years. We found high variability in the strength and significance of predictors among the RSF models. One variable, aspect, was not found to be significant in any our models. However, in exploratory data analysis, we noted that fishers preferred north and east facing slopes more frequently over south, or west-facing slopes (Fig. 3.1), perhaps due to the cooler microclimate on the north and east sides (Schwartz et al. 2013).

The top-performing model for 2011-2012 included elevation, slope, ruggedness, distance to road, and canopy cover with 48.6% weight supporting this model. The second top-performing model integrated the same parameters, but with the addition of distance to stream and explained an additional 22.9% of model weight. The 2 top-ranked models explained 71.5% of model weight in 2011-2012 (Table 3.3). In 2013-2014, the top-supported model identified elevation, distance to roads, and canopy cover as the most influential variables with a model weight of 13.4%. Of the 5 top-ranked models, canopy cover was in all 5 models, and elevation and distance from roads were in 4 of these models, with the 5 models having a combined weight of 49.8% (Table 3.3). The RSF for 2015-2016 revealed different results than the previous two RSFs where canopy cover was replaced by vegetation type. Our top model showed elevation, ruggedness, stream distance, road distance, and vegetation type as important parameters, with an AICc weight of 21.5%. There were 3 models with ΔAIC weight <2.0 for a combined weight of

47.9%. The second ranked model included ruggedness and vegetation type with a model weight of 13.5% (Table 3.3).

The coefficient signs (direction) for every variable were consistent across all years, indicating fishers continued to select or avoid the same variables through time (Table 3.4., Fig. 3.2.). In all years, our beta (β) coefficients indicated fishers selected against elevation as gradients increased, with strong selection at a range of 500-600 m, slowly decreasing as elevations increased to 2000 m (Table 3.4, Fig. 3.3). Selection for elevation held a smaller explanatory power in the 2013-2014 model from the previous RSF but was significantly larger in the first and last RSFs (2011-2012 and 2015-2016). Slope was positively selected by fishers in the first RSF but was not selected in any other RSF models (Table 3.4, Fig. 3.2). Terrain ruggedness was also significant in the first and last RSF models, with a higher β-coefficient in 2015-2016 than in 2011-2012 (Table 3.4, Fig. 3.2, and Fig. 3.6). Strangely, it was absent as a variable selected by fishers in the middle two years (2013-2014, Table 3.4, Fig. 3.2). Although road distance was selected by fishers in all models, it was a weak association given the low β coefficients (Table 3.4). In addition, the confidence intervals for distance from roads marginally overlapped zero and did not result in a P value <0.05 (Fig. 3.2). Canopy cover in the moderate to high range, and to a lesser extent, high canopy cover, were preferred by fishers in the 2011-2012 and 2013-2014 RSF models (Table 3.4, Fig. 3.2). Though it was not selected as a significant variable in the last RSF, we observed the same trend as other years where fishers were located in higher canopy cover more than expected compared to open or low-moderate cover (Fig. 3.4). Surprisingly in the last RSF (2015-2016), vegetation type was included as a significant variable; yet it was not found to be significant in previous years (Table 3.4, Fig. 3.2). The vegetation type beta coefficients suggested they preferred to be within conifer forests, and to a smaller degree conifer-hardwood vegetation types. Fishers selected against Sierra mixed conifer and shrubland vegetation zones (Table 3.4, Fig. 3.2, 3.5).

Model Validation

Variance inflation factors for all models was below 3.0, resulting in no issues with collinearity in covariates. The RSF models had k-fold test error estimates of 0.0385 (2011-2012), 0.0369 (2013-2014), and 0.0368 (2015-2016) suggesting model prediction error rates were low, just under 4%. The AUC for the 2011-2012 RSF was 0.63 (95% confidence intervals between 0.61-0.65), for the 2013-2014 RSF, 0.58 (95% confidence intervals between 0.55-0.61 and for the 2015-2016 RSF 0.66 (95% confidence intervals between 0.60-0.71 suggesting the top ranked models for each RSF analysis were relatively successful at discriminating between used and random points.

DISCUSSION

Prior to our research, scale-dependent studies emphasized fisher habitat selection at mostly broad or fine scales (Scheller et al. 2011, Truex and Zielinski 2013, Sauder and Rachlow 2014). However, research has only recently been conducted at an intermediate home-range level, with several studies evaluating how fishers are impacted by changing vegetation conditions in a temporal context (Thompson et al. 2011, Garner 2013, Sweitzer et al. 2016). Our results were similar to previous reports in which fishers were observed selecting topographical elements such as low and mid elevation ranges, low slope values, and low to moderately rugged features (Zielinski et al. 2004*a*, Purcell et al.

2009, Sauder and Rachlow 2014). Our observations also coincided with other studies that proposed canopy cover was an influential habitat variable when fisher are confronted with anthropogenic changes in their environment (Truex and Zielinski 2013, Sweitzer et al. 2015). However, canopy cover was not an important attribute several years after known thinning treatments in the study area had reduced the overstory, suggesting that this habitat requirement may be tolerated by fishers at lower thresholds than previously proposed.

Our resource selection models indicated fishers in the AFR watershed selected against sites as elevation increased, signifying they were negatively associated with high elevation areas, and selected for low to intermediate elevations. As noted in prior studies, deep snowpack levels, a smaller prey base, and interspecific competition with martens may explain why fishers evade high-elevation environments (Aubry and Lewis 2003, Fisher et al. 2012). In particular, fisher presence substantially decreases with increasing snow depth, as their heavier weight negatively affects both their travel and hunting ability in deep snow (Krohn et al. 1995, Krohn et al. 2005). However, sex ratios in our study were highly skewed towards females (7 out of 9 individuals), and this may have played a role in the elevation ranges selected by fishers. Female fishers preferred to raise their young in areas with low or moderate ruggedness and inclines, and were observed in low to mid-elevation forests which consisted of cavity-bearing hardwoods and conifers. In contrast and based on prior research, males usually travel extensively through a wide variety of landscapes, and have been found on steep slopes and at higher elevations (Lofroth et al. 2011, Olson et al. 2014). However, the majority of our male GPS points were obtained during the breeding season when their need to search for females likely biased their locations to female den sites (Zielinski et al. 2004*b*), which skewed averages towards lower elevations and less rugged inclines.

An important physiographic variable, terrain ruggedness, was selected by fishers in two out of the three RSF analyses for our study period. We speculated that fishers found in rugged locales might perceive these sites as safe havens from predators, similar to bighorn sheep using rugged slopes as escape terrain (Sappington et al. 2007). Terrain roughness can be difficult to interpret on a coarse landscape scale, especially if descriptions are limited to broad categorical features, such as valleys or ridges, rather than fine-scale variability within a home range. Additionally, the topography can interfere with collecting consistent data from GPS-collared animals that may have ventured into rocky or deep canyons (Webb et al. 2013). Our fishers selected areas with low to moderate ruggedness. Given the topographic roughness of our study area, it is possible fishers were using more rugged terrain, but our collars were unable to connect to satellites and record those locations at higher rates. In previous research, fishers selected concave terrain (e.g., riparian zones) rather than hillside slopes near ridges (Schwartz et al. 2013, Olson et al. 2014). The reasons for this positive association are not well understood. Possibly, these wet drainage systems comprise a diversity of vegetation types attracting multiple prey populations (Underwood et al. 2010), which would benefit a generalist predator such as the fisher. Alternatively, fishers could be drawn to mesic environments because they support a heterogeneous mix of forest stands with available rest and den structures as well as more temperate microclimates (Schwartz et al. 2013). Whatever the reason, the inaccessibility of these unique ecosystems offer a substantial reprieve from anthropogenic disturbances such as timber management (Fisher et al. 2013) and thereby creates refugia for sensitive wildlife species. Consistent with prior research, we also observed a positive relationship between fishers and terrain roughness in an extensive riparian corridor that was rugged, deep, and which stretched across several female home ranges. The area was utilized frequently by several fishers before and after treatment application, and though it was surrounded by mechanically treated units, the drainage itself did not experience any treatment. These refugia patches are often slated as critically protected habitat for fisher, and forest structures found here are considered important resources that need to be safeguarded from severe fire events.

Surprisingly, we also observed fishers were located closer to roads, which was counter to our original hypothesis in which we predicted they would select against road proximity. Prior studies have reported ambiguous results with fisher occupancy and use near roads, with most showing neutral effects from exposure near road systems (Davis et al. 2007, Lewis et al. 2016). Several of our fishers were within 300 m of major roads that underwent timber haul transportation and were visually seen crossing roads from open areas (T. Smith, unpublished data). Additionally, we observed denning females within 200 m of roads that tolerated frequent road traffic and noise without any negative behavioral effects. We believe fisher selection for road proximity may actually be a by-product of the heavy recreational and economic use of the area, which facilitated the creation of an extensive road system. As such, a fisher likely encountered a road often when traveling through its home range. In a different context, other research had also suggested predators, such as mountain lions or coyotes, often utilized road networks, where possible lethal interactions between fishers and potential predators may occur (Sweitzer et al. 2016, Nichols 2017). This might facilitate an avoidance of roads in fisher spatial use. However, we found little sign or sightings of fisher predators in the Ashland watershed, and recorded only two mortalities over the study period (D. Clayton, personal communication). Again, the possibility of high recreational and economic use of the watershed may discourage travel by large predators on the network of roads. Our road distance variable in the RSF models was significant in the first and last year groups, but was not significant in the middle two years. No clear explanation can be posited for the lack of significant findings in this period, as new roads were not constructed during that time and the mean distances were similar between the last two biennial year groups. Finally, because our confidence intervals overlapped zero and selection was weak for this variable, we postulated that roads may not be a hazardous feature and did not hinder travel or affect survivability of our fisher population, especially as fishers did not shift home range boundaries during timber operations.

The importance of stream distance was observed in the last years of 2015-2016, but not beforehand. Interestingly, this time period coincided with a drought period which occurred throughout California and Oregon. We surmise this variable could be indirectly related to the extreme drought conditions which started in 2014 and continued into early 2016. Drought could have had a small but important impact on the hydrological regime of riparian-rich environments, consequently affecting both flora and fauna. For instance, water deficits in riparian systems will trigger plant communities to shift from hydric to xeric species when drought conditions persist more than 30 days, reducing seedling survival and affecting tree species composition (Garssen et al. 2014). Additionally, Prugh et al. (2018) reported that long dry spells impact the producers and end consumers of the food chain the most (e.g., plant and carnivore species), either directly through water deficits, or indirectly by species replacement (e.g., tree composition) or food web interactions (e.g., prey resources). For fisher in particular, droughts can affect resource selection if the availability of structural components are impacted. For example, an extended drought period in California from 2012-2016 rendered massive tree die-offs, which negatively influenced fisher fitness, habitat selection, and reproductive fecundity in their core home ranges (Kordosky 2019). Yet, the AFR watershed may have experienced a less severe and shorter drought than that of California during our study period, and because our fishers had a positive association with stream proximity, we do not believe that water deficits negatively impacted food, water, or structural resource availability. However, we should note that our observation of fisher selection for streams occurred only in the last two years, and further investigation would be necessary to derive lag effects of drought-driven habitat conditions relative to long-term riparian use by fishers.

Canopy cover was predicted to be a significant variable in habitat selection for fisher. Indeed, it was strongly associated with fisher selection in our top RSF models for the first 2 biennial periods. Because canopy cover was directly influenced by mechanized vegetation removal we expected fishers would continue to search for areas of disproportionately high cover during and after management activities transpired. Over the first 4 years of our research period, canopy cover remained an important constituent for fisher habitat selection. However, in the last 2 years of our study, canopy cover was not influential. Possibly, fishers were using dense cover habitat in an accessible portion of their home range that was free of any anthropogenic activities (i.e., fuel reduction methods). Yet, in our LANDFIRE map for the final RSF, we discerned a perceptible habitat change in canopy cover values within the AFR watershed. Canopy cover appeared more homogenous across fisher home ranges in the last two years, and reductions in overhead cover percentages of previously dense stands coincided with the locations of timber management activities in those mapped areas, indicating a disturbance event in vegetation conditions. Given the results of thinning prescriptions in our study area, we calculated there was a 14% reduction in canopy bulk density across the study area. This is a small percentage in average canopy cover reduction, yet we speculated the less intensive management actions used subsequently propagated a rapid growth response of site-specific understory species, encouraging sapling and shrub regeneration in areas where resource competition was mitigated (Ares et al. 2010, Wagner at al. 2011). As shown by other researchers, experiments comparing no action, control plots to mechanical and/or prescribed burn plots found that mechanical-only treatments produced a strong vegetation response 7 years post-treatment compared to the alternatives (Stephens et al. 2012, Collins et al. 2014). In our watershed, canopy density in treated stands likely increased as saplings and small trees grew into the canopy, reflecting pre-treatment cover extent, especially in the moderate density range of 40-50% canopy cover. Yet, these managed stands consisted of an age-class diversity with structural differences compared to pre-treatment periods (e.g., uneven-aged versus even-aged stands, respectively). As a result, fishers in the last two years of our study continued to select dense canopy cover when available, especially in areas with high-valued structural resources. But they were not found in those areas disproportionately, which explains why canopy cover was not present as a significant variable in the last RSF. In fact, fishers were observed in habitat that experienced vegetation change through the removal and reduction of overhead cover. Thus, we believe it is plausible their resource needs were met through sufficient understory cover and new structural diversity in a recently transformed environment.

In agreement with prior literature, we believe fishers in the AFR preferred forests with a heterogeneous mix of variable canopy cover classes (Sauder and Rachlow 2015). This was due to the fact that we located fishers in stands with known density reductions throughout the study period. Furthermore, our study revealed that in the case of timber managed stands, fishers will occupy cover structure in equal proportion to randomly available cover, as long as patches of dense cover are retained throughout their home range. Finally, we propose that for specific circumstances, fishers may choose sites in low canopy cover but with higher resource value, both for foraging and reproductive purposes. For example, fishers have been found in managed stands offering hunting opportunities, such as younger forests with snowshoe hares (Weir and Harestad 2003, Happe et al. 2020, Parsons et al. 2020), or rodent-occupied slash piles (Lofroth et al. 2011, Sweitzer et al. 2016). Likewise, female fishers will select sites with low canopy cover, but where large structures exist for denning purposes (Niblett et al. 2017). Clearly, the selection of canopy cover by fishers remains a complex attribute that must be assessed at variable scales when considering restorative forest management strategies.

Fishers continued to occupy and use the same vegetation classes (or types) throughout the study, despite the fact that areas of their home ranges had experienced considerable transformations, especially near the urban edges. Similar to previous research, we found that overall, fishers preferred conifer and conifer-hardwood stands and avoided frequent use of shrubland habitat in lower elevations (Sauder and Rachlow

2014, Niblett et al. 2015). Predictably, where conifer density was reduced by mechanical treatments, the propagation of shrub and sapling cover intensified in the understory. This led to an increased availability of shrubland habitat in the final years of our study, especially in wildland-urban interfaces. Consequently, the availability of conifer and hardwood habitat decreased as the cumulative effects of thinning and successional stage dynamics continued on the landscape. Interestingly, two females (F01 and F03) in the last 2 years of our research experienced unusual circumstances in each home range relative to habitat changes. F01's home range included a large proportion of private property that was subject to its own thinning activities, and she lived closest to the urban edge. Though her location points fell mostly in conifer and hardwood forest stands, her exposure to edge effects was evident by her increased use of low-elevation shrubland habitat, perhaps to exploit new prey items. Additionally, we expected F03 would remain in the interior of conifer forests after thinning projects were complete. However, she explored the edges of disturbed areas (i.e., treatment units) multiple times as well, possibly for the same reason as F01. These two cases highlight the importance of individual variance in different circumstances, and how vegetation changes in habitat can present behavioral aberrations that may not be evident at the population level.

Finally, we must note that after examining ArcGIS layers derived from remote sensing data, we determined that LANDFIRE vegetation categories might have been reclassified between the years of 2014 and 2016. In particular, a majority of areas classified as Sierra Mixed Conifer (SMC) in 2012 and 2014 were substantially absent or reduced in size in the newer 2016 maps, and replaced with other conifer categories instead. This reclassification of a dominant plant community beginning in 2015 could explain why vegetation type was a significant variable in our last model. It was not significant before this period, and the SMC category classification remained virtually unchanged between 2010 and 2014. Whether this modification was a product of improved mapping technology or an actual difference in successional stand development was unclear since new vegetation maps were not released until 2017, after our analysis was completed. Even though we observed a disparity in SMC classification, other vegetation categories remained consistent throughout all study years. Notably, the extent of developed areas at the wildland-urban interface increased after 2014 as anthropogenic land use grew. Logging and other disturbances were also mapped both inside and outside areas where known silviculture practices had occurred (i.e., thinning treatments) and were reflected in the latest LANDFIRE maps of 2015-2016.

MANAGEMENT IMPLICATIONS

Before implementation of a cohesive silviculture plan, managers should evaluate fisher habitat requirements at landscape and local scales, as their needs differ depending on the age and sex of the animal. Male fishers can span several female home ranges, which would necessitate forest managers retain corridors of contiguous forest with variable cover. Meanwhile, adult female fishers require cavity-bearing trees at lower elevations, therefore snags and older conifers or hardwoods should be retained in mixed stand assemblages. Kits and subadults need all of the above requirements for dispersing to new home ranges, as well as ground structures such as decayed logs and shrub cover to hone their hunting skills and avoid predation. These habitat requirements are consistent with the recommendation that mature stands should be left untouched in order to achieve or maintain late-seral characteristics. Our study expanded on this concept, determining that fishers can tolerate restorative timber practices in their environment as long as stands with approximately 50% or more canopy cover and a prerequisite number of large structures for rest sites exist within their home range extent. Heterogeneous landscapes consisting of variable age class trees, moderate but variable canopy cover, and a complex arrangement of vertical and horizontal structures are preferred by fishers and their prey. Mechanical methods, like those practiced by the AFR, preserved sufficient canopy cover levels while thinning dense stands at risk for high-severity fires, accomplishing goals for both silviculture restoration and fisher habitat conservation. Additionally, regrowth of vegetation in areas after thinning likely provided adequate ground cover and new foraging opportunities for fisher as they traveled throughout their home range. Although we did not investigate the effects of prescribed fire on vegetation change relative to fisher resource selection, it is imperative we know what effects slash burning and broadcast burns have on fisher spatial use and behavior. We would suggest a waiting period of a year or more be employed between mechanical thinning and prescribed burns, which could benefit both the fisher and stand dynamics. The interlude between management activities would give fishers the opportunity to return slowly to modified habitat and hunt for small rodents in slash piles established post-harvest. Meanwhile, the interim between silviculture practices would allow decomposition of surface fuels to occur, thereby optimizing the amount of low-flammable fuels to be burned. However, periodic burn cycles may be required, and additional considerations such as weather, social and economic impacts, and resource limitations may hinder when prescribed burns can be implemented.

LITERATURE CITED

Agee, J. K. 1993. Fire ecology of Pacific Northwest Forests. Island Press, Washington D.C.

Ager, A. A., A. M. G. Barros, H. K. Preisler, M. A. Day, T. A. Spies, J. D. Bailey, and J. P. Bolte. 2017. Effects of accelerated wildfire on future fire regimes and implications for the United States federal fire policy. Ecology and Society 22:12.

Ares, A., A. R. Neill, and K. J. Puettmann. 2010. Understory abundance, species diversity and functional attribute response to thinning in coniferous stands. Forest Ecology and Management 260:1104–1113.

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

Beyer H. L. 2015. Geospatial Modelling Environment [Internet]. http://www.spatialecology.com/gme

Boyce, M. S., P. R. Vernier, S. E. Nielsen, and F. K. A. Schmiegelow. 2002. Evaluating resource selection function. Ecological Modelling 157:281–300.

Brewer, M. J., A. Butler, and S. L. Cooksley. 2016. The relative performance of AIC, AICc and BIC in the presence of unobserved heterogeneity. Methods in Ecology and Evolution 7:679–692.

Buck, S.G., C. Mullis, A. S. Mossman, I. Show, and C. Coolahan. 1994. Habitat use by fishers in adjoining heavily and lightly harvested forest. Pages 368–376 *in* S. W. Buskirk, A. S. Harestad, M. G. Raphael, and R. A. Powell, editors. Martens, sables, and fishers: biology and conservation. Comstock Publishing Associates, Cornell University Press, Ithaca, New York.

Burnham, K. P., and D. R. Anderson. 2002. Model selection and multi-model inference: a practical information-theoretic approach. Second edition. Springer, Secaucus, New Jersey, USA.

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

Collins, B. M. and G. B. Roller. 2013. Early forest dynamics in stand-replacing fire patches in the northern Sierra Nevada, California, USA. Landscape Ecology 28:1801–1813.

Collins, B. M., A. J. Das, J. J. Battles, D. L. Fry, K. D. Krasnow, and S. L. Stephens. 2014. Beyond reducing fire hazard: fuel treatment impacts on overstory tree survival. Ecological Applications 24:1879–1886.

Crooks, K. R., C. L. Burdett, D. M. Theobald, C. Rondinini, L. Boitani. 2011. Global patterns of fragmentation and connectivity of mammalian carnivore habitat. Philosophical Transactions of the Royal Society 366:2642–2651.

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

DellaSala, D. A. 2006. Fire in the Klamath-Siskiyou ecoregion: protecting and restoring the fire mosaic. Pages 132-152 *in* G. Wuerthner, editor. Wildfire: a century of failed forest policy. Island Press, Washington, D.C., USA.

Devictor, V., R. Julliard, F. Jiguet. 2008. Distribution of specialist and generalist species along spatial gradients of habitat disturbance and fragmentation. Oikos 117:507–514.

Facka, A. N. 2017. Conservation translocations as opportunities for scientific advancement: A case study with fishers (*Pekania pennanti*). PhD. dissertation, North Carolina State University, Raleigh, North Carolina.

Ferron, J., F. Potvin, and C. Dussault. 1998. Short-term effects of logging on snowshoe hares in the boreal forest. Canadian Journal of Forest Research 28:1335–1343.

Fielding, A. and J. Bell. 1997. A review of methods for the assessment of prediction errors in conservation presence/absence models. Environmental Conservation 24:38–49.

Fisher, J. T., B. Anholt, S. Bradbury, M. Wheatley, and J. P. Volpe. 2012. Spatial segregation of sympatric marten and fishers: the influence of landscapes and species-scapes. Ecography https://doi.org/10.1111/j.1600-0587.2012.07556.x

Fisher, J. T., S. Bradbury, B. Anholt, L. Nolan, L. Roy, J. P. Volpe, and M. Wheatley. 2013. Wolverines (*Gulo gulo luscus*) on the Rocky Mountain slopes : natural heterogeneity and landscape alteration as predictors of distribution. Research Press 91(October):706–716.

Frair, J. L., S. E. Nielsen, E. H. Merrill, S. R. Lele, M. S. Boyce, R. H. M. Munro, G. B. Stenhouse, and H. L. Beyer. 2004. Removing GPS collar bias in habitat selection studies. Journal of Applied Ecology 41:201–212.

Frair, J. L., J. Fieberg, M. Hebblewhite, F. Cagnacci, N. J. DeCesare, and L. Pedrotti. 2010. Resolving issues of imprecise and habitat-biased locations in ecological analyses using GPS telemetry data. Philosophical Transactions of the Royal Society B: Biological Sciences 365:2187–2200.

Franklin, F. 1972. Federal research natural areas in Oregon and Washington: a guidebook for scientists and educators. Pacific Northwest Forest and Range Experiment Station, Portland, Oregon.

Franklin, J. F., T. A. Spies, R. V. Pelt, A. B. Carey, D. A. Thornburgh, D. R. Berg, D. B. Lindenmayer, M. E. Harmon, W. S. Keeton, D. C. Shaw, K. Bible, and J. Chen. 2002. Disturbances and structural development of natural forest ecosystems with silvicultural implications, using Douglas-fir forests as an example. Forest Ecology and Management 155:399–423.

Fuller, A. K., and D. J. Harrison. 2005. Influence of partial timber harvesting on American martens in north-central Maine. Journal of Wildlife Management 69:710–722.

Garner, J. D., 2013. Selection of disturbed habitat by fishers (*Martes pennanti*) in the Sierra National Forest. Thesis, Humboldt State University, Arcata, California.

Garshelis, D. L. 2000. Delusions in habitat evaluation: measuring use, selection, and importance. Pp. 111–164 *in* L. Boitani and T. K. Fuller, editors. Research techniques in animal ecology: controversies and consequences. Columbia University Press, New York.

Garssen, A. G., J. T. A. Verhoeven, and M. B. Soons. 2014. Effects of climate-induced increases in summer drought on riparian plant species: a meta-analysis. Freshwater Biology 59:1052–1063.

Gese, E. M., and C. M. Thompson. 2014. Does habitat heterogeneity in a multi-use landscape influence survival rates and density of a native mesocarnivore? PLoS ONE 9(6): e100500. doi:10.1371/journal.pone.0100500.

Green, R. E. 2017. Reproductive ecology of the fisher (*Pekania pennanti*) in the southern Sierra Nevada: an assessment of reproductive parameters and forest habitat used by denning females. Ph.D. dissertation, University of California, Davis, California.

Haines, A. M., F. Hernández, S. E. Henke, and R. L. Bingham. 2006. A method for determining asymptotes of home-range area curves. Gamebird, (1955), 489–498. http://ckwri.tamuk.edu/fileadmin/user_upload/docs/misc_PDF/Quail/Hainesetal2009b.pdf.

Happe, P. J., K. J. Jenkins, R. M. Mccaffery, J. C. Lewis, K. L. Pilgrim, and M. K. Schwartz. 2020. Occupancy patterns in a reintroduced fisher population during reestablishment. Journal of Wildlife Management 84:344–358.

Hargis, C. D., J. A. Bissonette, and D. L. Turner. 1999. The influence of forest fragmentation and landscape patterns on American martens. Journal of Applied Ecology 36:157–172.

Hessburg, P. F., T. A. Spies, D. A. Perry, C. N. Skinner, A. H. Taylor, P. M. Brown, S. L. Stephens, A. J. Larson, D. J. Churchill, N. A. Povak, P. H. Singleton, B. McComb, W. J. Zielinski, B. M. Collins, R. B. Salter, J. J. Keanem, J. F. Franklin, and G. Riegel. 2016. Tamm Review: Management of mixed-severity fire regime forests in Oregon, Washington, and Northern California. Forest Ecology and Management 366:221–250.

Kordosky, J. R. 2019. Landscape of stress: Does drought prevail over anthropogenic activity in influencing cortisol levels and fitness in the Pacific fisher? Thesis, Utah State University, Logan, Utah.

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

Krohn W., C. Hoving, D. Harrison, D. Phillips, and H. Frost. 2005. *Martes* foot-loading and snowfall patterns in Eastern North America *in* D. J. Harrison, A. K. Fuller, G. Proulx, editors. Martens and Fishers (Martes) in human-altered environments. Springer, Boston, Massachusetts.

Kuhn, M., and K. Johnson. 2013. Applied Predictive Modeling. Springer Science + Business Media, New York. Pages 69–74.

Lewis, J. C., K. J. Jenkins, P. J. Happe, D. J. Manson, and M. McCalmon. 2016. Landscape-scale habitat selection by fishers translocated to the Olympic Peninsula of Washington. Forest Ecology and Management 369:170–183.

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

Mallek, C., H. Safford, J. Viers, and J. Miller. 2013. Modern departures in fire severity and area vary by forest type, Sierra Nevada and southern Cascades, California, USA. Ecosphere 4:153.

Manly, B. F. J., L. L. McDonald, D. L. Thomas, T. L. McDonald, and W. Erikson. 2002. Resource selection by animals: statistical design and analysis for field studies. Kluwer Academic, Norwell, Massachusetts.

Manning, J. A. and W. D. Edge. 2008. Small mammal responses to fine woody debris and forest fuel reduction in Southwest Oregon. Journal of Wildlife Management 72:625–632.

Manning, T., J. C. Hagar, and B. C. McComb. 2012. Thinning of young, Douglas-fir forests decreases density of northern flying squirrels in the Oregon Cascades. 2012. Forest Ecology and Management 264:115–124.

Mori, A. S. and E. A. Johnson. 2013. Assessing possible shifts in wildfire regimes under a changing climate in mountainous landscapes. Forest Ecology and Management 310:875–886.

Mukaka, M. M. 2012. A guide to appropriate use of Correlation coefficient in medical research. Malawi Medical Journal 24 No. 3.

Niblett, M. R., S. H. Sweeney, R. L. Church, and K. H. Barber. 2015. Structure of fisher (*Pekania pennanti*) habitat in a managed forest in an interior northern California coast range. Forest Science 61:481–493.

Niblett, M. R., R. L. Church, S. H. Sweeney, and K. H. Barber. 2017. Characterizing habitat elements and their distribution over several spatial scales: The case of the fisher. Forests 8, 186.

Nichols, B. C. 2017. Foraging ecology of mountain lions in the Sierra National Forest, California. Thesis, Utah State University, Logan, Utah.

Northrup, J. M., M. B. Hooten, C. R. Anderson, Jr., and G Wittemyer. 2013. Practical guidance on characterizing availability in resource selection functions under a use-availability design. Ecology 94:1456–1463.

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

Olson, L. E., J. D. Sauder, N. M. Albrecht, R. S. Vinkey, S. A. Cushman, and M. K. Schwartz. 2014. Modeling the effects of dispersal and patch size on predicted fisher (*Pekania [Martes] pennanti*) distribution in the U.S. Rocky Mountains. Biological Conservation 169:89–98.

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

Parsons, M. A., J. C. Lewis, J. N. Pauli, T. Chestnut, J. I. Ransom, D. O. Werntz, and L. R. Prugh. 2020. Prey of reintroduced fishers and their habitat relationships in the

Cascades Range, Washington. Forest Ecology and Management 460:117888. https://doi.org/10.1016/j.foreco.2020.117888.

Parsons, A. W., C. T. Rota, T. Forrester, M. C. Baker-Whatton, W. J. McShea, S. G. Schuttler, J. J. Millspaugh, and R. Kays. 2019. Urbanization focuses carnivore activity in remaining natural habitats, increasing species interactions. Journal of Applied Ecology 56:1894–1904.

Perry, D. A., P. F. Hessburg, C. N. Skinner, T. A. Spies, S. L. Stephens, A. H. Taylor, J. F. Franklin, B. McComb, and G. Riegel. 2011. The ecology of mixed severity fire regimes in Washington, Oregon, and Northern California. Forest Ecology and Management 262:703–717.

Pozzanghera, C. B., K. J. Sivy, M. S. Lindberg, and L. R. Prugh. 2016. Variable effects of snow conditions across boreal mesocarnivore species. Canadian Journal of Zoology 94:697–705.

Prugh, L. R., N. Deguines, J. B. Grinath, K. N. Suding, W. T. Bean, R. Stafford, and J. S. Brashares. 2018. Ecological winners and losers of extreme drought in California. Nature Climate Change 8:819–824.

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

Purcell, K. L., C. M. Thompson, and W. J. Zielinski. 2012. Managing Sierra Nevada forests. General Technical Report. PSW-GTR-237, U.S. Department of Agriculture. Forest Service. Pacific Southwest Station. Albany, California.

Raine, R. M. 1983. Winter habitat use and responses to snow cover of fisher (*Martes pennanti*) and marten (*Martes americana*) in southeastern Manitoba. Canadian Journal of Zoology 61:25–34.

Sager-Fradkin, K. A., K. J. Jenkins, R. A. Hoffman, P. J. Happe, J. J. Beecham, and R. G. Wright. 2007. Fix success and accuracy of global positioning system collars in old-growth temperate coniferous forests. Journal of Wildlife Management 71:1298–1308.

Sappington, J. M., K. M. Longshore, and D. B. Thompson. 2007. Quantifying landscape ruggedness for animal habitat analysis: A case study using bighorn sheep in the Mojave Desert. Journal of Wildlife Management 71:1419–1426.

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

Sauder, J. D., and J. L. Rachlow. 2015. Forest heterogeneity influences habitat selection by fishers (*Pekania pennanti*) within home ranges. Forest Ecology and Management 347:49–56.

Scheller, R. M., W. D. Spencer, H. Rustigian-Romsos, A. D. Syphard, B. C. Ward, and J. R. Strittholt. 2011. Using stochastic simulation to evaluate competing risks of wildfires and fuels management on an isolated forest carnivore. Landscape Ecology 26:1491–1504.

Schwartz, M. K., N. DeCesare, B. Jimenez, J. P. Copeland, W. E. Melquist. 2013. Stand- and landscape-scale selection of large trees by fishers in the Rocky Mountains of Montana and Idaho. Forest Ecology and Management 305:103–111.

Seglund, A. E. 1995. The use of resting sites by the Pacific fisher. Thesis, Humboldt State University, Arcata, California.

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

Sollmann, R., A. R. White, G. L. Tarbill, P. N. Manley, E. E. Knapp. 2016. Landscape heterogeneity compensates for fuel reduction treatment effects on Northern flying squirrel populations. Forest Ecology and Management 373:100–107.

Soutiere, E.C. 1979. Effects of timber harvesting on marten in Maine. Journal of Wildlife Management 43:850–860.

Spies, T. A. and J. F. Franklin. 1991. The structure of natural young, mature, and old-growth Douglas-fir forests in Oregon and Washington. General Technical Report PNW-GTR-285, U. S. Department of Agriculture, Forest Service, Pacific Northwest Research Station, Portland, Oregon.

Stephens, S. L., B. M. Collins, and G. Roller. 2012. Fuel treatment longevity in a Sierra Nevada mixed conifer forest. Forest Ecology and Management 285:204–212.

Sullivan, T. P., D. S. Sullivan, J. H. Sullivan. 2017. Mammalian responses to windrows of woody debris on clearcuts: Abundance and diversity of forest-floor small mammals and presence of small mustelids. Forest Ecology and Management 399:143–154.

Sweitzer, R. A., V. D. Popescu, R. H. Barrett, K. L. Purcell, and C. M. Thompson. 2015. Reproduction, abundance, and population growth for a fisher (*Pekania pennanti*) population in the Sierra National Forest, California. Journal of Mammalogy 96:772–790.

Sweitzer, R. A., B. J. Furnas, R. H. Barrett, K. L. Purcell, and C. M. Thompson. 2016. Landscape fuel reduction, forest fire, and biophysical linkages to local habitat use and local persistence of fishers (*Pekania pennanti*) in Sierra Nevada mixed-conifer forests. Forest Ecology and Management 361:208–225.

Tempel, D. J., R. J. Gutierrez, S. A. Whitmore, M. J. Reetz, R. E. Stoelting, W. J. Berigan, M. E. Seamans, and M. Z. Peery. 2014. Effects of forest management on California spotted owls: implications for reducing wildfire risk in fire-prone forests. Ecological Applications 24:2089–2106.

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

Thompson, C. M., R. E. Green, J. Sauder, K. L. Purcell, R. A. Sweitzer, and J. M. Arnemo. 2012. The use of radiotelemetry in research on Martes species: techniques and technologies. Pages 284-319 *in* K. B. Aubry, W. J. Zielinski, M. G. Raphael, G. Proulx, and S. W. Buskirk, editors. Biology and conservation of martens, sables, and fishers: a new synthesis, Cornell University Press, Ithaca, New York, USA.

Thompson, C. M., and K. L. Purcell. 2015. Conditions inside fisher dens during prescribed fires; what is the risk posed by spring underburns? Forest Ecology and Management 359:156–161.

Truex, R. L., and W. J. Zielinski. 2013. Short-term effects of fuel treatments on fisher habitat in the Sierra Nevada, California. Forest Ecology and Management 293:85–91.

Underwood, E. C., J. H. Viers, J. F. Quinn, and M. North. 2010. Using topography to meet wildlife and fuels treatment objectives in fire-suppressed landscapes. Environmental Management 46:809–819.

U. S. Department of Agriculture and U. S. Forest Service, Pacific Northwest Region. 2008. Ashland Forest Resiliency Final Environmental Impact Statement. Siskiyou Mountains Ranger District, Rogue-River-Siskiyou National Forest, Jackson County, Oregon, USA.

U. S. Fish and Wildlife Service. 2020. Endangered and threatened wildlife and plants; Endangered species status for Southern Sierra Nevada distinct population segment of fisher; Final rule. Federal Register 85:29532-29589.

U. S. Forest Service. 2005. Ashland Forest Resiliency Draft Environmental Impact Statement. Rogue-River Siskiyou National Forest. Medford, Oregon, USA.

Wagner, S., H. Fischer, and F. Huth. 2011. Canopy effects on vegetation caused by harvesting and regeneration treatments. European Journal of Forest Research 130:17–40.

Webb, S. L., M. R. Dzialak, J. P. Mudd, and J. B. Winstead. 2013. Developing spatially-explicit weighting factors to account for bias associated with missed GPS fixes in resource selection studies. Wildlife Biology 19:257-273.

Weir, R. D., and A. S. Harestad. 2003. Scale-dependent habitat selectivity by fishers in South-Central British Columbia. Journal of Wildlife Management 67:73–82.

Weir, R. D., M. Phinney, and E. C. Lafroth. 2011. Big, sick, and rotting: Why tree size, damage, and decay are important to fisher reproductive habitat. Forest Ecology and Management 265:230–240.

Whittington, J., C. C. St. Clair, and G. Mercer. 2005. Spatial responses of wolves to roads and trails in mountain valleys 15:543–553.

Zhao, F., R. A. Sweitzer, Q. Guo, and M. Kelly. 2012. Characterizing habitats associated with fisher den structures in the Southern Sierra Nevada, California using discrete return lidar. Forest Ecology and Management 280:112–119.

Zielinski, W. J., R. L. Truex, G. A. Schmidt, F. V. Schlexer, K. N. Schmidt, and R. H. Barrett. 2004*a*. Resting habitat selection by fishers in California. Journal of Wildlife Management. 68:475–492.

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

Zielinski, W. J., R. L. Truex, F. V. Schlexer, L. A. Campbell, and C. Carroll. 2005. Historical and contemporary distributions of carnivores in forests of the Sierra Nevada, California, USA. Journal of Biogeography 32:1385–1407.

Zielinski, W. J., J. R. Dunk, J. S. Yaeger, and D. W. LaPlante. 2010. Developing and testing a landscape-scale habitat suitability model for fisher (*Martes pennanti*) in forests of interior northern California. Forest Ecology and Management 260:1579–1591.

Zielinski, W. J., C. M. Thompson, K. L. Purcell, and J. D. Garner. 2013. An assessment of fisher (*Pekania pennanti*) tolerance to forest management intensity on the landscape. Forest Ecology and Management 310:821–826.

Zuur, A. F., E. N. Ieno, and C. S. Elphick. 2010. A protocol for data exploration to avoid common statistical problems. Methods in Ecology and Evolution 1:3–14.

TABLES AND FIGURES

Table 3.1. Home range size (km^2), mean distance traveled (m), and mean lag time traveled (hr) within home ranges for individual fishers and mean and standard deviation between sexes in the Ashland Watershed Unit, Oregon, from 2011-2016. “F” indicates female fisher while “M” indicates male fisher.

Fisher ID	Home Range Size (km^2) [a]	Mean distance traveled (m) [b]	Mean lag time between distances (hr) [c]
F01	9.81	1012	8.7
F02	14.99	1088	8.8
F03	13.89	964	6.3
F04	20.69	2113	10
F06	34.14	1251	8.9
F08	12.40	872	5.8
F09	7.99	946	8.1
	$\bar{x} = 16.27 \pm 8.87$	$\bar{x} = 1178 \pm 429.82$	$\bar{x} = 8.09 \pm 1.51$
M02	65.42	1613	9.8
M04	102.23	3367	9.3
M10	40.23	695	5.5
	$\bar{x} = 69.29 \pm 31.19$	$\bar{x} = 1892 \pm 1357.62$	$\bar{x} = 8.20 \pm 2.35$

[a] Wilcoxon test results indicated a significant difference in home range size between the sexes (W=0, *P*=0.02)

[b] Wilcoxon test results did not indicate significant differences in distance traveled between the sexes (W=8, *P*=0.67)

[c] Wilcoxon test results did not indicate significant differences in lag times between the sexes (W=9, *P*=0.89)

Table 3.2. Number of used and random point locations in 2011-2012, 2013-2014, and 2015-2016 per individual fisher in the Ashland Watershed Unit, Oregon, from 2011-2016. "F" indicates female fisher while "M" indicates male fisher.

	2011-2012 Points		2013-2014 Points		2015-2016 Points	
Fisher ID	Used	Random	Used	Random	Used	Random
F01	149	3725	27	675	35	875
F02	11	275	--	--	--	--
F03	175	4375	86	2150	63	1575
F04	6	150	--	--	--	--
F06	82	2050	--	--	--	--
F08	47	1175	57	1425	--	--
F09	192	4800	71	1775	--	--
M04	35	875	35	875	--	--
M10	--	--	84	2100	--	--
Total	697	17425	360	9000	98	2450

Dashed lines indicate no data was observed for the fisher individuals during these years.

Table 3.3. Top generalized linear models in two year increments of all fishers in the Ashland Watershed Unit, Oregon, 2011-2016. Only models < 2.0 delta AICc (ΔAICc) are shown for all years. Bold type signifies top model for each RSF period.

	Model Description [a]	AICc	ΔAICc	Weight	Log Likelihood
2011-2012	**Elevation + slope + ruggedness + roaddist + cc**	5792.9	0.00	0.4861	-2887.436
	Elevation + slope + ruggedness + roaddist + streamdist + cc	5794.4	1.50	0.2287	-2887.189
2013-2014	**Elevation + roaddist + cc**	3037.5	0.00	0.1338	-1511.728
	Slope + Aspect + roaddist + cc	3037.5	0.04	0.1312	-1508.741
	Elevation + cc	3038.3	0.88	0.0862	-1513.170
	Elevation + slope + roaddist + cc	3038.6	1.09	0.0774	-1511.273
	Elevation + ruggedness + roaddist + cc	3038.8	1.29	0.0703	-1511.369
2015-2016	**Elevation + ruggedness + streamdist + roaddist + veg**	815.7	0.00	0.2145	-399.838
	Ruggedness + veg	816.6	0.90	0.1354	-403.314
	Ruggedness + streamdist + veg	816.7	1.00	0.1291	-402.357

[a] cc = Canopy Cover; roaddist = Distance to Road; streamdist = Distance to Stream; veg = Vegetation Type

Table 3.4. Model covariates, beta-coefficients, and standard errors in parentheses are shown for the best supported generalized linear models in each two year increment for all fishers in the Ashland Watershed Unit, Oregon, 2011-2016.

Landscape Covariates	**2011-2012**	**2013-2014**	**2015-2016**
Elevation (m)	-0.23 (0.05) ***	-0.16 (0.18) *	-0.46 (0.22) *
Slope (degrees)	0.14 (0.04) ***	--	--
Ruggedness (VRM)	0.12 (0.03) ***	--	0.21 (0.09) *
Road Distance (m)	0.08 (0.04) *	0.10 (0.06)	0.10 (0.11)
Stream Distance (m)	--	--	0.46 (0.20) *
Low-Moderate CC [a]	0.22 (0.17)	0.13 (0.23)	--
Moderate CC	0.49 (0.18) **	0.35 (0.25)	--
Moderate-High CC	0.94 (0.15) ***	0.68 (0.20) ***	--
High CC	0.67 (0.15) ***	0.61 (0.20) **	--
Conifer Hardwood	--	--	0.81 (0.31) **
Conifer	--	--	1.06 (0.34) **
Sierra Mixed Conifer	--	--	-0.08 (0.37)
AICc	5792.9	3037.5	815.7

[a] CC=canopy cover

Standard errors in parentheses. Significance denoted as: *** $p < 0.001$; ** $p < 0.01$; * $p < 0.05$.

Dashed lines indicate the covariate was not found in the model.

Open-Low CC was used as a reference category for canopy cover.

Shrubland was used as a reference category for vegetation type.

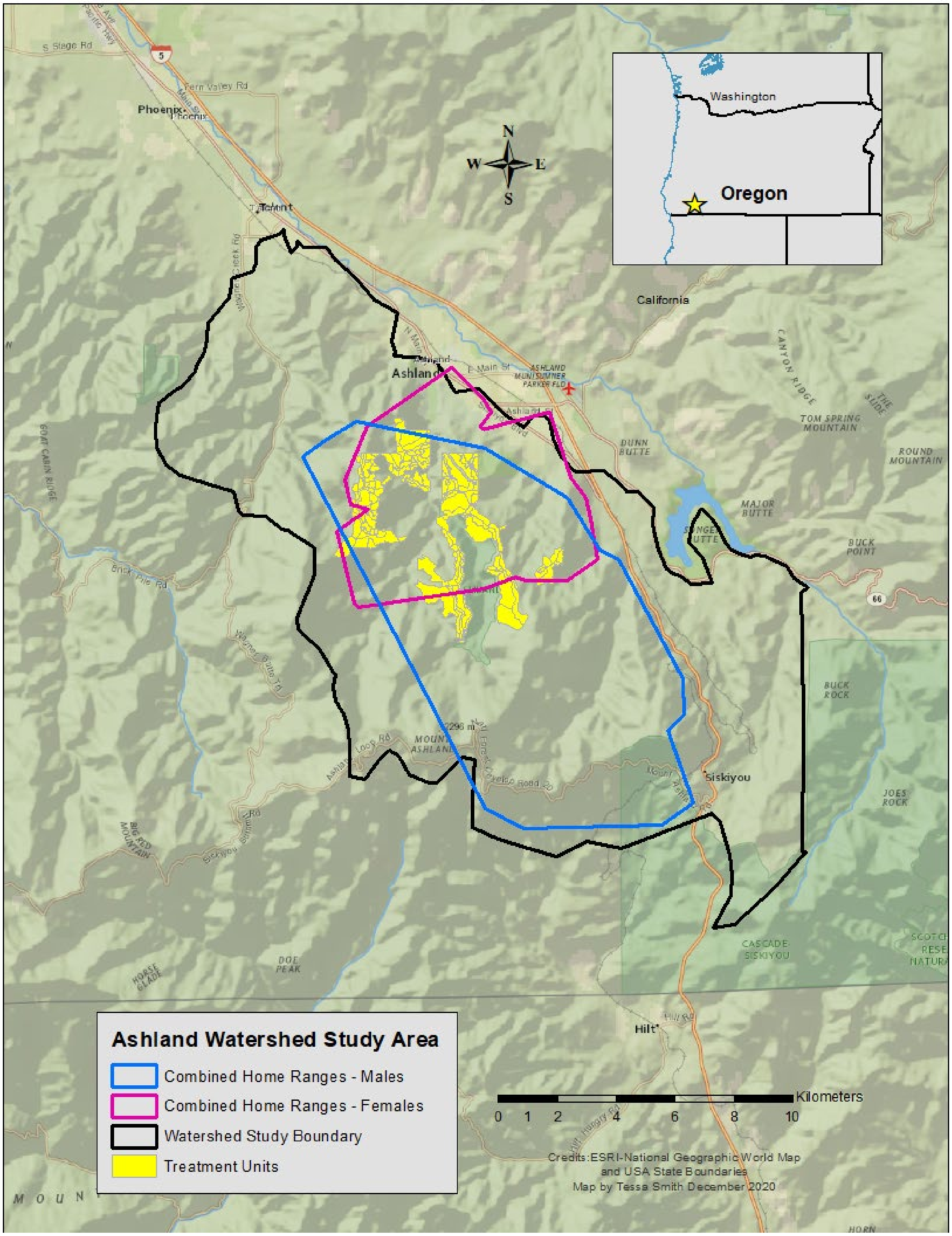


Figure 3.1. Map of the Ashland Watershed Study Area, Oregon, 2011-2016.

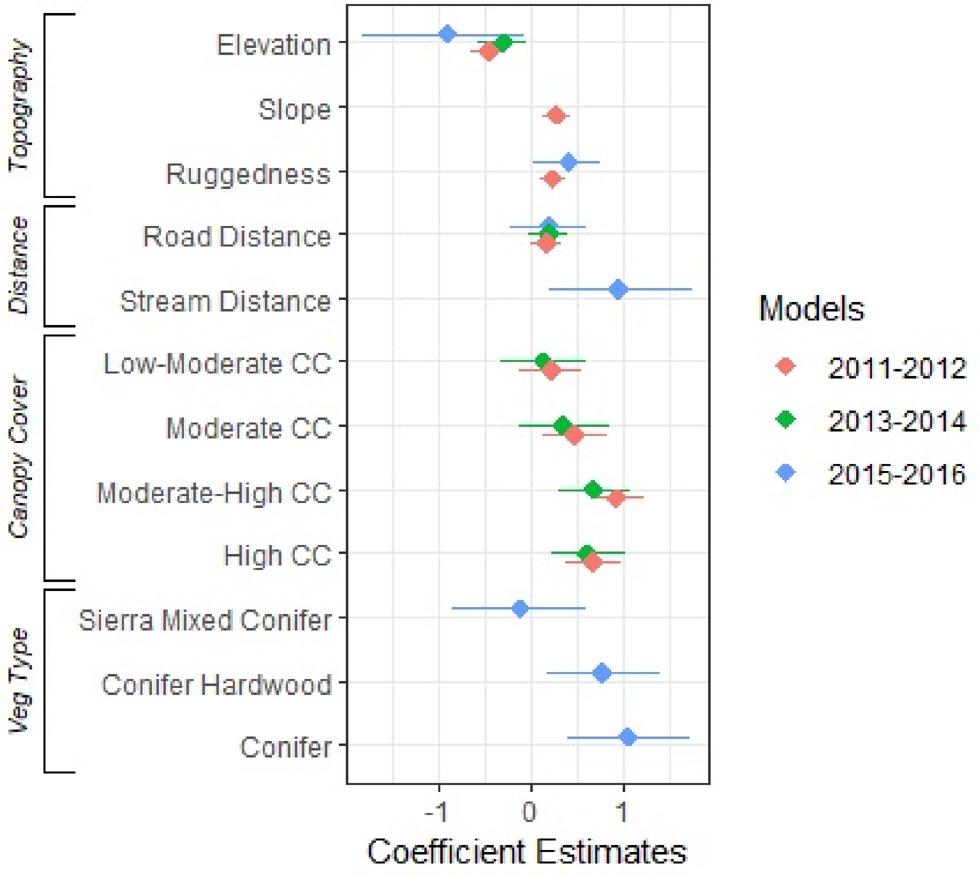


Figure 3.2. Coefficient plot comparing all biennial RSF model years for fishers in the Ashland Watershed Unit, Oregon, 2011-2016. Coefficient estimates near 1 indicates a selection for the variable while a coefficient estimate closer to -1 indicates a selection against the variable. For example, fishers selected against increasing elevation gradients and selected for moderately rugged features. Fishers were also positively associated with moderate to high canopy cover and conifer or conifer-hardwood vegetation types. A positive selection for road distance was found to be significant, though confidence intervals overlapped zero in all model years. CC = canopy cover. Open-Low CC and shrubland were used as reference categories for canopy cover and vegetation type, respectively.

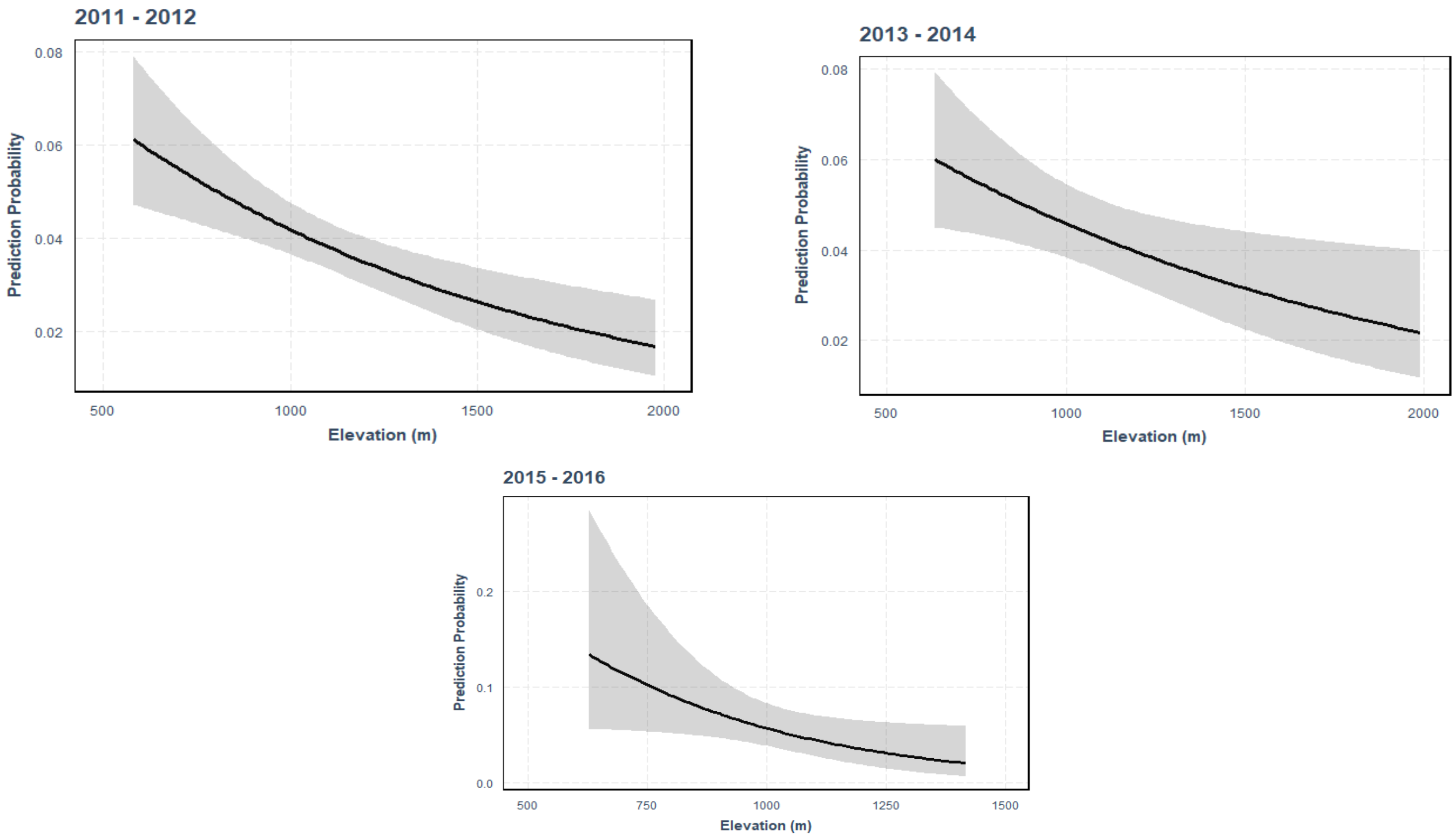


Figure 3.3. Effect plots for elevation in each RSF biennial year in the Ashland Watershed Unit, Oregon, 2011-2016. Fishers were positively associated with lower elevations at approximately 600 meters. As elevation increased, fishers selected against this variable, as indicated by a decreasing slope. The town of Ashland sits at 594 meters, where fishers were located within the elevation range.

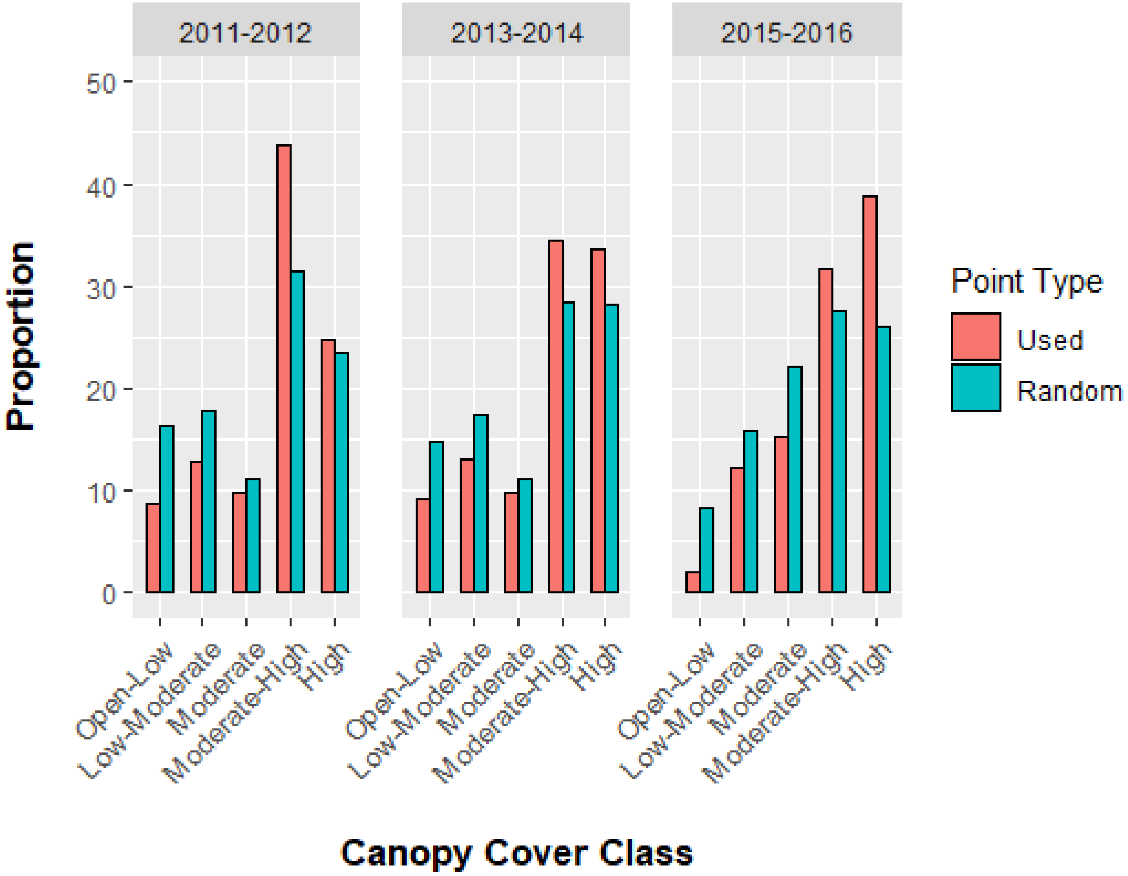


Figure 3.4. Bar plots comparing used and random canopy cover points, fishers disproportionately selected sites with moderate to high or high only canopy cover rather than moderate or low cover in the Ashland Watershed Unit, Oregon, 2011-2016. Percentages of cover type were as follows: Open-Low = 0-19%, Low-Moderate = 20-39%, Moderate = 40-59%, Moderate-High = 60-79%, High = 80+%.

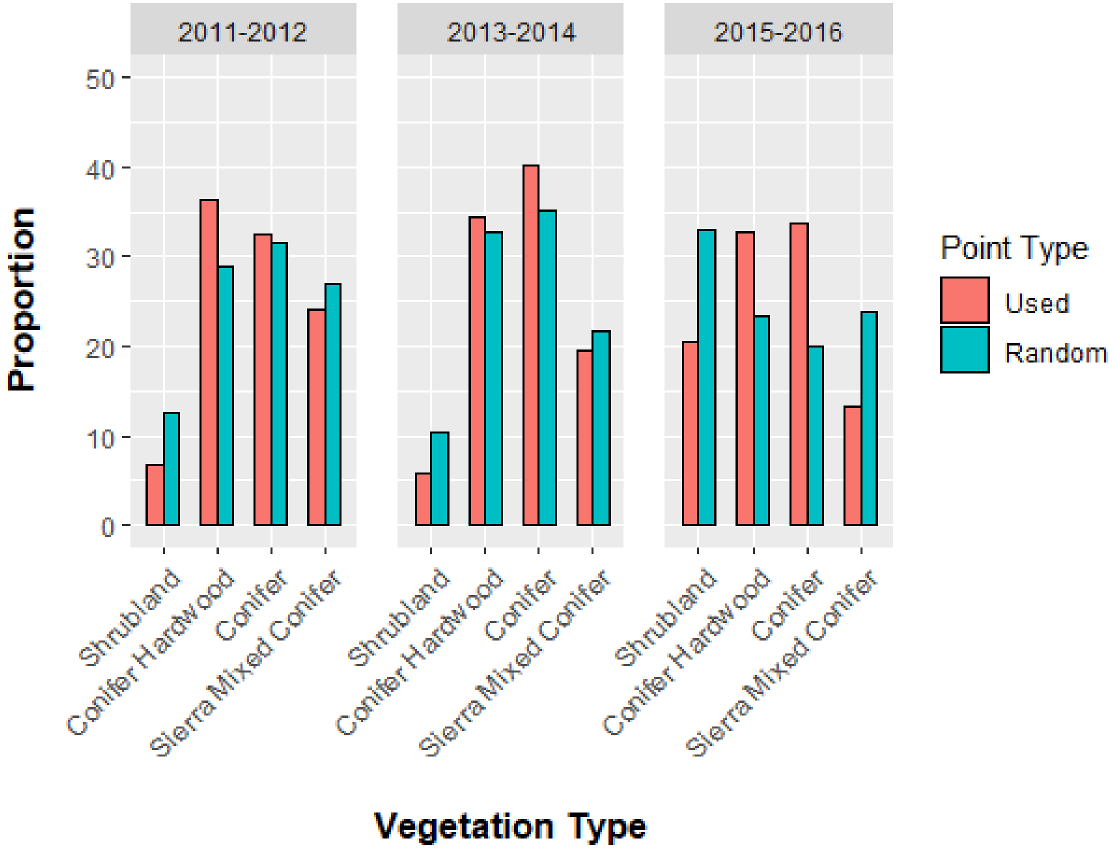


Figure 3.5. Bar plot comparison of used versus random fisher points for vegetation types in the Ashland Watershed Unit, Oregon, 2011-2016. Fishers positively selected sites in conifer and conifer-hardwood habitats more often than random, and avoided low elevation shrub areas and Sierra Mixed Conifer zones. Vegetation type was only significant in our last RSF model of 2015-2016. During this period, random point availability of low elevation shrubland sites increased while conifer and hardwood areas decreased.

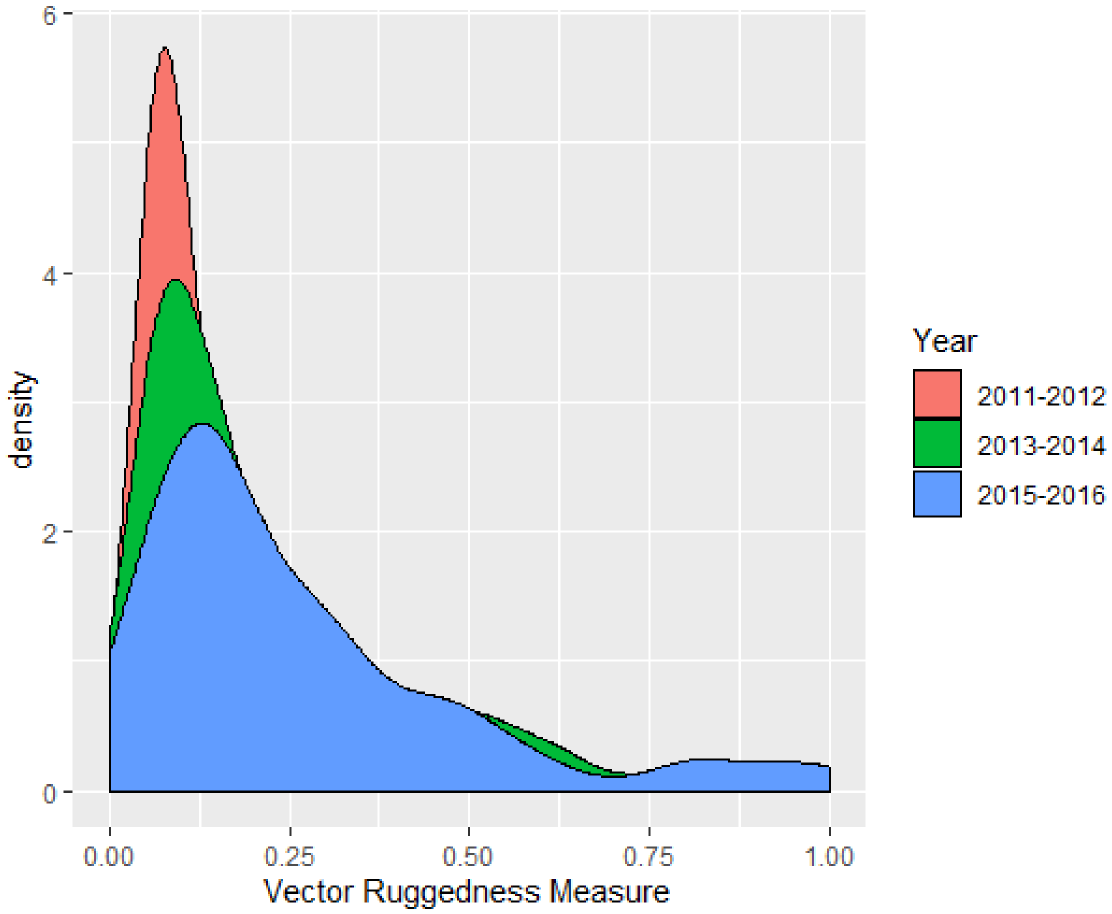


Figure 3.6. Density plot showing the level of topography ruggedness fishers selected in each RSF model. Values were normalized to a range of 0-1, where 0 indicates flat terrain and 1 indicates extremely rugged terrain. Fishers selected habitat where topography was either low or moderately rugged, avoiding areas with extreme terrain roughness.

CHAPTER 4

CONCLUSION

In the last several decades, our ideology of forest health has transformed from a long-standing philosophy of fire suppression to a new paradigm of fire support (North et al. 2015, Jones et al. 2016, Kalies and Yokom Kent 2016, Ager et al. 2017). The recognition that forests and fire have coevolved has informed our view of ecosystem sustainability and services. In order to reverse the heavy tolls logging and fire suppression have taken on stand dynamics, anthropogenic interference is now exercised in the form of holistic approaches. Silviculturists are using adaptive management strategies such as patch-based thinning and prescribed burns as surrogate methods to maintain the fire-resilient characteristics inherent in heterogenic stands (Hessburg et al. 2016). But despite the best intentions, the need for fuel reduction management will likely endure for decades due to the extensive acreage and maintenance that warrants attention across our forests (Schoennagel et al. 2017). In 2019, the U.S. Government Accountability Office released a report stating that 100 million acres of forested land could benefit from treatment prescriptions; yet, only 3 million acres have actually undergone this process (U.S. Government Accountability Office 2019). Given that management activities will likely be a long-term practice to combat severe wildfire events, monitoring how wildlife responds to treated habitat is an essential function in sustaining a part of forest ecosystem health.

As land is steadily manipulated through silvicultural treatments, forest managers should consider the ramifications thinning and prescribed burns place on the occupancy, distribution, and reproductive behavior of fishers. Our research found that the timing and size of restoration practices influenced whether or not fishers tolerated mechanical activities within a temporal and spatial context. Mechanically treated units occurred mainly in the spring and summer months. This season is a sensitive time for reproductive female fishers and kits, and their acceptance of high-level anthropogenic disturbance may be lower during these periods (Cummings 2016, Green 2017). We observed a weak negative response to a majority of these seasonal treatments, but fishers were tolerant of restoration activities at a threshold distance of ~2000 meters, well within their home ranges. Garner (2013) suggested fishers may be tolerant of treatment practices as long as accessible untreated habitat was available nearby. Similarly, our research observed fishers traveling through untreated portions of their home range in order to avoid the treated areas. We concluded that no fishers vacated their home ranges, indicating fishers could tolerate effects from low-intensity treatment methods. However, we highlight that our analyses did not incorporate high-level disturbance activities and their impacts on fisher behavior. Anecdotal evidence indicated fishers displayed a strong negative response to highly intensive operations such as helicopter yarding and tree felling. Fishers were located several kilometers away as soon as these activities commenced. However, individuals returned slowly to these original locations after treatments were completed. Additionally, they were observed using the recently altered habitat post-treatment, a positive sign in adaptive behavior to strong anthropogenic disturbances. We suggest, though, that future studies explore whether fisher tolerance levels are dependent on management intensity across larger areas and how that may impact breeding or denning behavior in the early spring.

During seasonally reproductive periods (approximately March-July), limited operating periods (LOPs) may help alleviate any stress females and kits encounter due to fuel reduction methods (U. S. Department of Agriculture 2009). Though we perceived no direct impacts to kit rearing by females relative to treatment disturbance, indirect effects in breeding behavior might have occurred within our fisher population. For example, we observed that a 3-year old adult female (F03) had not yet reproduced during the mechanical treatment phases, but once they were completed she was found to be successfully denning with kits (U.S. Forest Service 2016). Although our data suggest F03 had a positive response to treatments (i.e., she was found closer to treatment edges), we may have falsely interpreted this tolerance based on biological, topographic, or habitat-constrained circumstances. For instance, F03 was surrounded to the south and east by other female fishers, and could have been avoiding confrontations with conspecifics overlapping her territory. To the north, her movements were limited by the urban sprawl of Ashland, while the western edge consisted of high topographic relief. Along this western front, site conditions such as exposed ridgelines, deep snowpack, or abrupt vegetation changes could have deterred her from avoiding treatment effects occurring throughout the rest of her home range. Furthermore, we observed that F03 had little choice but to move through several treated areas since many of them spanned a large portion of her territory. Consequently, F03 was proximate to or inside treated units more often than other fishers, and it is possible that any unnecessary energy expenditure she spent avoiding stressful circumstances could have negatively influenced her reproductive status.

In management units, varying the number, perimeter size, spatial distribution, and resource configuration of treatments could also impact fisher movement and presence in their home ranges (Zielinski et al. 2013, Sweitzer et al. 2016). We acknowledge that stands receive different treatment prescriptions based on site conditions, topography, resource protection, and time constraints prioritized during the planning stage. These descriptive factors may explain why managers set distinctive shapes and sizes of unit boundaries across a targeted area to maximize efficiency for treatment implementation. However, the intensity and timing of treatments performed in units of variable sizes may induce stress as fishers move through them. Our results showed that treatments performed in smaller units negatively influenced the distance fishers were willing to travel or approach those areas. This behavior was pronounced if treatments were < 20 acres in size, spatially clustered, and treated within a short period of one another. By contrast, the few large scale (e.g., >30 acres) units placed within each home range had less impact on fisher avoidance patterns than the smaller units, especially in areas where important structural resources were identified for fishers (e.g., dens or dense canopy closure). Notably, the larger units were also placed on the edges of treatment blocks; and again, as Garner (2013) suggested, fisher tolerance of mechanical treatments may be due to having available untreated habitat close by. Given our results, we recommend future strategies consider the impacts these treatment parameters have on fisher movement as well as access to critical resources that may be restricted by fuel reduction activities.

Moreover, treatment effects on forest configuration appear fundamental in regard to fisher occupancy rates. Sauder and Rachlow (2014) point to forest configuration as an important parameter for fisher presence in Idaho, and a change in open areas from 5% to

10% can decrease occupancy of fisher by 39%. This could have a substantial effect on smaller isolated fisher populations if corridors with inadequate canopy cover negatively impact biological movements such as dispersal, foraging, or predator evasion. Studies have also suggested fisher may prefer heterogeneity within their core home range scale, where landscape edges and intermediate (40-60%) canopy cover is selected (Weir and Harestad 2003, Sauder and Rachlow 2015). Because their generalist diet and diversity in rest site criteria correlates with heterogeneous stands, fishers may prefer habitat with variable levels of canopy closure and edge density (Sauder and Rachlow 2015). If this is true, foresters could target density-management objectives in select stands where no more than 50% total canopy cover is reduced and where edge effects are created through selective thinning practices. This approach can help diversify plant composition in the understory, encourage prey species richness, maintain suitable canopy cover for fisher dispersal, and reduce wildfire potential (Harrington 2009, Waltz et al. 2014, Tsai et al. 2018).

Although the implications of prescribed burns were not explored in our study, prior research has shown variable side effects on habitat quality under different treatment scenarios. The timing of prescribed burns may again be a crucial factor in preserving intact habitat features. For instance, Truex and Zielinski (2013), reported that the combination of mechanical and fire prescriptions negatively affected predicted fisher resting habitat, especially in reducing average canopy closure. However, spring burns yielded less damage in habitat quality than fall burns. Thompson and Purcell (2015) likewise suggested that early-season burns benefit habitat quality by decreasing the chance of tree mortality, retaining more woody debris, and reducing the likelihood of uncontrolled fires. However, it was unknown whether low-intensity burns would adversely impact the physical integrity of fisher den cavities. In the experiment run by Thompson and Purcell (2015), internal temperatures and carbon monoxide (CO) levels were measured for actual and surrogate fisher dens during a spring period of controlled burns. They found that internal temperatures were surprisingly stable even as external temperatures soared, and only the surrogate dens chosen by the researchers contained higher CO concentrations. Undoubtedly, more information should be gathered on the criteria fishers use when choosing den sites and how prescribed burning affects those decisions through time, especially on the impact CO levels have on neonates or unborn fisher kits. But if fishers are observed selecting den structures after prescribed fires have changed part of their landscape, then spring burning could be a feasible management option with few risks and considerable habitat benefits.

As resource selection is likely tied to the individual fitness of fishers, finding a link between chosen landscape features and behavioral trends may also be critical when planning management objectives (Aldridge and Boyce 2007). Our study emphasized that important topographical features, such as intermediate elevations and terrain roughness, were positively selected by fishers. This is likely due to unique topographic combinations and site-specific factors that produce distinct plant communities at these mid-elevation ecotones (Odion et al. 2004, Olson et al. 2012). Furthermore, canopy cover was significant in fisher selection as a habitat variable, but our results suggest fishers can adapt to reduced overhead cover, as long as it remains >50%. We also hypothesize that the consequences of using an area with reduced cover may be offset by the presence of nearby ground structures deemed highly desirable by fishers, as suggested by previous research (Niblett et al. 2017). Additionally, because fishers travel through stands with differing age classes for a variety of purposes, lower landscape-scale cover may be adequate for fisher habitat use at multiple scales (Niblett et al. 2015).

The Ashland Forest Resiliency Project strategy included the preservation of diverse habitat resources for fisher conservation and other umbrella species. We believe this trend should be continued in future management plans. Our research area consisted of more restorative, rather than extractive, treatment approaches, which worked well in supporting biodiversity and resource protection objectives. This management plan allowed conifer and hardwood trees >25 cm DBH (diameter-at-breast-height) to remain in noncommercial stands for fisher and other wildlife to use. Additionally, commercial thinning was practiced in declining stands by removing a select cohort of intermediate trees ≤50 cm DBH, with canopy cover inevitably reduced during these density-management practices. However, caution should be exercised if management methods aim to remove trees or snags larger than 50 cm DBH. Whereas canopy cover can be replaced over a short period of time, large trees and snags are not easily restored on the landscape once gone, and the loss of these structures may impact fisher kit production and population persistence (Purcell et al. 2009). Female fishers are highly selective when choosing den sites with specific characteristics (Green 2017), and we recommend that priority be given to protect areas where decadent trees, cavity-bearing hardwoods, and snags exist within a fisher home range. Moreover, safeguarding known den and rest sites previously used by fishers cannot be emphasized enough, as several studies have shown repeated use of these microsites by the same individual in following years, or even use by different individuals in nonconsecutive years (Sweitzer et al. 2015, Green et al. 2018, Matthews et al. 2019). Finally, forest managers should retain several habitat patches with high-value to remain across a larger extent of the landscape, allowing legacy trees and snags the opportunity to develop and protect resources not yet identified as fisher-valuable, while creating refugia for fisher and other species during fuel reduction management.

LITERATURE CITED

Ager, A. A., A. M. G. Barros, H. K. Preisler, M. A. Day, T. A. Spies, J. D. Bailey, and J. P. Bolte. 2017. Effects of accelerated wildfire on future fire regimes and implications for the United States federal fire policy. Ecology and Society 22(4):12. https://doi.org/10.5751/ES-09680-220412.

Aldridge, C. L., and M. S. Boyce. 2007. Linking occurrence and fitness to persistence: habitat based approach for endangered Greater Sage-Grouse. Ecological Applications 17:508–526.

Cummings, C. 2016. Environmental and anthropogenic influences on fisher (*Pekania pennanti*) den attendance patterns in California. Thesis, Humboldt State University, Arcata, California.

Garner, J. D., 2013. Selection of disturbed habitat by fishers (*Martes pennanti*) in the Sierra National Forest. Thesis, Humboldt State University, Arcata, California.

Green, R. E. 2017. Reproductive ecology of the fisher (*Pekania pennanti*) in the southern Sierra Nevada: an assessment of reproductive parameters and forest habitat used by denning females. Ph.D. dissertation, University of California, Davis.

Green, R. E., K. L. Purcell, C. M. Thompson, D. A. Kelt, and H. U. Wittmer. 2018. Reproductive parameters of the fisher (*Pekania pennanti*) in the southern Sierra Nevada, California. Journal of Mammalogy 99:537–553.

Harrington, C. 2009. Let's mix it up! The benefits of variable-density thinning. Pacific Northwest Research Station. Portland, Oregon. https://www.fs.fed.us/pnw/sciencef/scifi112.pdf.

Hessburg, P. F., T. A. Spies, D. A. Perry, C. N. Skinner, A. H. Taylor, P. M. Brown, S. L. Stephens, A. J. Larson, D. J. Churchill, N. A. Povak, P. H. Singleton, B. McComb, W. J. Zielinski, B. M. Collins, R. B. Salter, J. J. Keanem, J. F. Franklin, and G. Riegel. 2016. Tamm Review: Management of mixed-severity fire regime forests in Oregon, Washington, and Northern California. Forest Ecology and Management 366:221–250.

Jones, G. M., R. J. Gutiérrez, D. J. Tempel, S. A. Whitmore, W. J. Berigan, and M. Z. Peery. 2016. Megafires: an emerging threat to old-forest species. Frontiers in Ecology and the Environment 14:300–306.

Kalies, E. L., and L. L. Yocom Kent. 2016. Tamm Review: are fuel treatments effective at achieving ecological and social objectives? A systematic review. Forest Ecology and Management 375:84–95.

Matthews, S. M., D. S. Green, J. M. Higley, K. M. Rennie, C. M. Kelsey, and R. E. Green. 2019. Reproduction dens selection and its consequences for fisher neonates, a cavity-obligate mustelid. Journal of Mammalogy 1004:1305-1316.

Niblett, M. R., S. H. Sweeney, R. L. Church, and K. H. Barber. 2015. Structure of fisher (*Pekania pennanti*) habitat in a managed forest in an interior northern California coast range. Forest Science 61:481–493.

Niblett, M. R., R. L. Church, S. H. Sweeney, and K. H. Barber. 2017. Characterizing habitat elements and their distribution over several spatial scales: The case of the fisher. Forests 8:186. doi:10.3390/f8060186.

North, M. P., S. L. Stephens, B. M. Collins, J. K. Agee, G. Aplet, J. F. Franklin, and P. Z. Fulé. 2015. Reform forest fire management: agency incentives undermine policy effective- ness. Science 349:1280–81.

Odion, D. C., E. J. Frost, J. R. Strittholt, H. Jiang, D. A. DellaSala, and M. A. Moritz. 2004. Patterns of fire severity and forest conditions in the Western Klamath Mountains, California. Conservation Biology 18:927–936.

Olson, D., D. A. DellaSala, R. F. Noss, J. R. Strittholt, J. Kass, M. E. Koopman, and T. F. Allnutt. 2012. Climate change refugia for biodiversity in the Klamath-Siskiyou ecoregion. Natural Areas Journal 32:65–74.

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

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

Sauder, J. D., and J. L. Rachlow. 2015. Forest heterogeneity influences habitat selection by fishers (*Pekania pennanti*) within home ranges. Forest Ecology and Management 347:49–56.

Schoennagel, T., J. K. Balch, H. Brenkert-Smith, P. E. Dennison, B. J. Harvey, M. A. Krawchuk, N. Mietkiewicz, P. Morgan, M. A. Moritz, R. Rasker, M. G. Turner, and C. Whitlock. 2017. Adapt to more wildfire in western North American forests as climate changes. Proceedings of the National Academy of Sciences of the United States of America 114:4582-4590.

Sweitzer, R., C. Thompson, K. Purcell, and R. Barrett. 2015. Appendix D. Fisher Team Final Report. Sierra Nevada Adaptive Management Project. University of California, Berkeley, California.

Sweitzer, R. A., B. J. Furnas, R. H. Barrett, K. L. Purcell, and C. M. Thompson. 2016. Landscape fuel reduction, forest fire, and biophysical linkages to local habitat use and local persistence of fishers (*Pekania pennanti*) in Sierra Nevada mixed-conifer forests. Forest Ecology and Management 361:208–225.

Thompson, C. M., and K. L. Purcell. 2015. Conditions inside fisher dens during prescribed fires; what is the risk posed by spring underburns? Forest Ecology and Management 359:156–161.

Truex, R. L., and W. J. Zielinski. 2013. Short-term effects of fuel treatments on fisher habitat in the Sierra Nevada, California. Forest Ecology and Management 293:85–91.

Tsai, H.-C., J.-M. Chiang, R. W. McEwan, and T.-C. Lin. 2018. Decadal effects of thinning on understory light environments and plant community structure in a subtropical forest. Ecosphere 9(10):e02464.10.1002/ecs2.2464.

U.S. Department of Agriculture and U.S. Forest Service, 2009. Final Environmental Impact Statement. Beaverside Timber Sale and Fuel Treatment Project. Mad River Ranger District, Six Rivers National Forest, Trinity County, California, USA.

U.S. Forest Service. 2016. Ashland Forest Resiliency Fisher Monitoring Interim Report. Rogue-River Siskiyou National Forest. Medford, Oregon, USA.

U.S. Government Accountability Office. 2019. WILDLAND FIRE: Federal agencies' efforts to reduce wildland fuels and lower risk to communities and ecosystems. Report to Congressional Requesters 20-52. Retrieved from https://www.gao.gov/assets/710/703470.pdf.

Waltz, A. E. M., M. T. Stoddard, E. L. Kalies, J. D. Springer, D. W. Huffman, and A. Sánchez Meador. 2014. Effectiveness of fuel reduction treatments: Assessing metrics of forest resiliency and wildfire severity after the Wallow Fire, AZ. Forest Ecology and Management 334:43–52.

Weir, R. D., and A. S. Harestad. 2003. Scale-dependent habitat selectivity by fishers in South-Central British Columbia. Journal of Wildlife Management 67:73–82.

Zielinski, W. J., C. M. Thompson, K. L. Purcell, and J. D. Garner. 2013. An assessment of fisher (*Pekania pennanti*) tolerance to forest management intensity on the landscape. Forest Ecology and Management 310:821–826.

APPENDICES

APPENDIX A

CHAPTER 2 SUPPLEMENTARY MATERIAL

Table A.1. Pre and post-treatment multi-way ANOVA results for pooled subset of fisher data analyzing factors that affect fisher distance to treatments in the Ashland Watershed Unit, Oregon, 2010-2017. Fishers combined in the analysis include F01, F08, and F09. Shaded rows indicate significance.

(A)

Pre-treatment variable	df	Sum of squares	Mean squares	F	*P*
Season	3	5.17E+07	17,241,944	41.2387	<0.001
Time of Day	3	4.00E+06	1,332,588	3.1872	0.023
Treatment Type	1	8.06E+06	80,645,488	19.2886	<0.001
Vegetation Class	4	2.59E+07	6,484,672	15.5098	<0.001
Treatment Size	4	5.76E+06	1,439,561	3.4431	<0.001
Canopy Cover	3	2.14E+07	7,121,796	17.0337	<0.001
Error	3536	1.48E+09	418,101		

(B)

Post-treatment variable	df	Sum of squares	Mean squares	F	*P*
Season	3	2.56E+08	85,197,703	161.9846	<0.001
Time of Day	3	2.16E+06	720,643	1.3701	0.250
Treatment Type	1	6.80E+05	679,606	1.2921	0.256
Vegetation Class	4	7.63E+07	19,067,905	36.2534	<0.001
Treatment Size	4	1.06E+08	26,457,317	50.3027	<0.001
Canopy Cover	3	4.25E+06	1,416,785	2.6937	0.004
Error	3536	4.41E+09	525,962		

Table A.2. Vegetation reclassification derived from LANDFIRE vegetation categories for the Ashland Watershed Unit, Oregon, 2010-2017.

Vegetation Reclassified	LANDFIRE Vegetation Category	LANDFIRE Vegetation Classification
Low Elevation	Developed	Developed
	Sparsely Vegetated	Sparsely Vegetated
	Shrubland	Deciduous Shrubland
	Shrubland	Introduced Upland Vegetation - Herbaceous
	Agriculture	Agriculture
	Barren	No Dominant Lifeform
	Barren	Non-vegetated
	Grassland	Idaho Fescue
	Grassland	Green Fescue
	Hardwood	Western Juniper-Big Sagebrush-Bluebunch Wheatgrass
	Riparian	Riparian Woodland
	Shrubland	Northern Coastal Shrub
	Shrubland	Chamise Chaparral
	Shrubland	Scrub Oak Mixed Chaparral
	Shrubland	Montane Shrubland
	Shrubland	Coastal Prairie
	Grassland	Rough Fescue-Bluebunch Wheatgrass
	Shrubland	Big Sagebrush-Bluebunch Wheatgrass
	Shrubland	Mountain Big Sagebrush
	Shrubland	Wyoming Big Sagebrush
	Shrubland	Low Sagebrush
	Grassland	Tall Forb
	Shrubland	Curlleaf Mountain-Mahogany
	Grassland	Chokecherry-Serviceberry-Rose
	Shrubland	Arizona Chaparral
Conifer	Conifer	Mountain Hemlock
	Conifer	Red Fir
	Conifer	Interior Douglas-Fir
	Conifer	White Fir
	Conifer	Grand Fir
	Conifer	Aspen
	Conifer	Lodgepole Pine
	Conifer	Sitka Spruce

	Conifer	Coastal True Fir-Hemlock
	Conifer	Pacific Douglas-Fir
	Conifer	Douglas-Fir-Western Hemlock
	Conifer	Redwood
	Conifer	Interior Ponderosa Pine
	Conifer	Jeffrey Pine
	Conifer	Knobcone Pine
	Conifer	California Mixed Subalpine
Conifer-Hardwood	Hardwood	Red Alder
	Hardwood	Black Cottonwood-Willow
	Hardwood	Oregon White Oak
	Conifer-Hardwood	Douglas-Fir-Tanoak-Pacific Madrone
	Hardwood	Cottonwood-Willow
	Hardwood	California Black Oak
	Conifer-Hardwood	Blue Oak-Digger Pine
Sierra Mixed Conifer	Sierra Nevada Mixed Conifer	Sierra Nevada Mixed Conifer

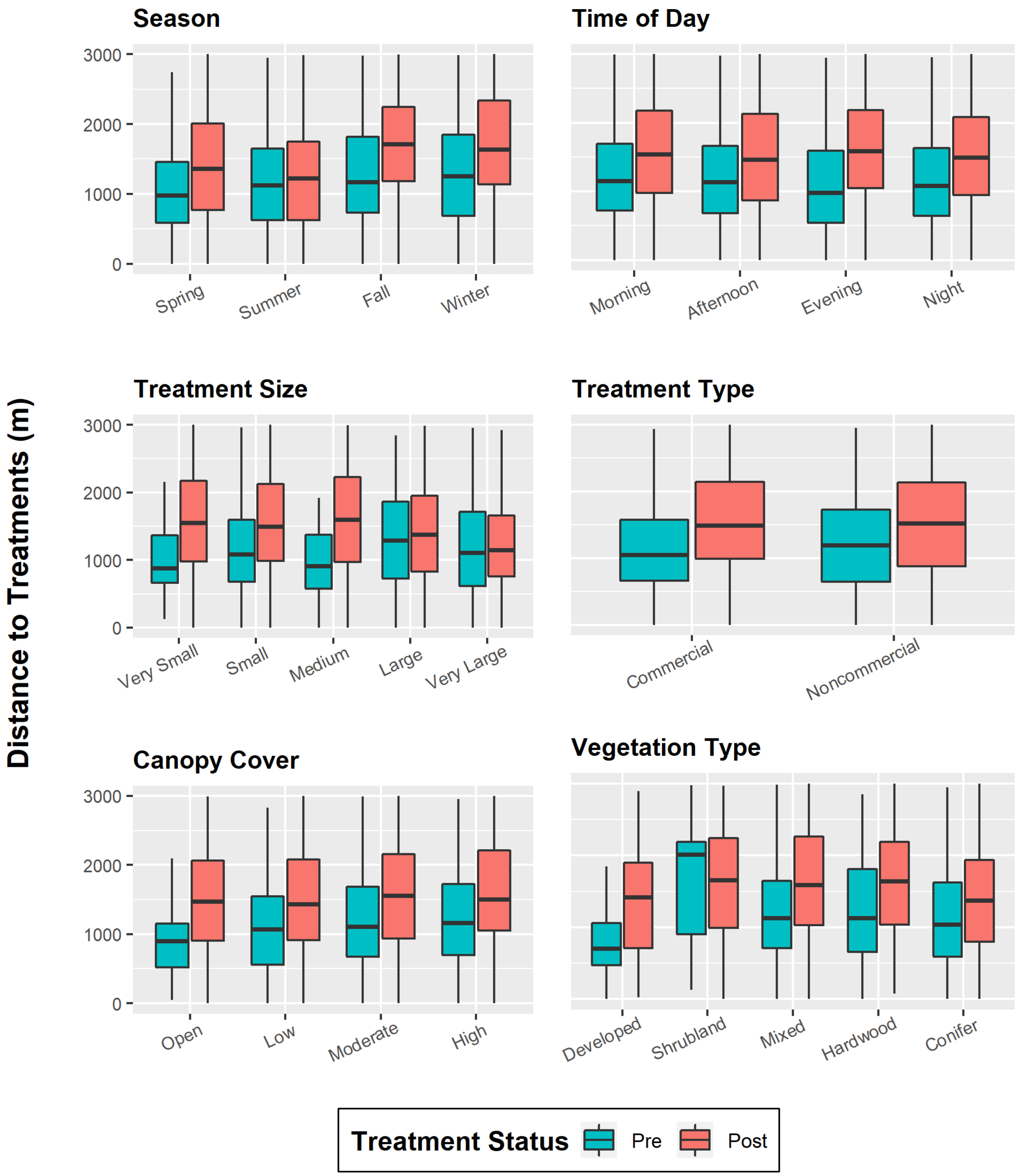


Figure A.1. Factors influencing pooled subset of fisher distances to treatments during pre and post-treatment periods, Ashland Watershed Unit, Oregon, 2010-2017. Fishers combined in the analysis include F01, F08, and F09. A maximum threshold distance of 3000 meters to treatment was used to assess response to various factors on a finer scale.

APPENDIX B

CHAPTER 3 SUPPLEMENTARY MATERIAL

Figure B.1. Aspect was not found to be a significant variable in our analysis of habitat selection for fishers in the Ashland Watershed Unit, Oregon, 2011-2016. However, barplots indicate fishers selected sites on north-facing slopes more than random, whereas other aspects were chosen less than random.