AN ABSTRACT OF THE THESIS OF

Kevin Credo for the degree of Master of Science in Sustainable Forest Management presented on August 24, 2017.

Title: Assessing Alternatives for Fuel Reduction Treatment and Pacific Marten Conservation in the Southern Cascades and Northern Sierra Nevada.

Abstract approved: ____

John D. Bailey

Forest managers are challenged to restore resilience to forests with an elevated risk of stand-replacing fire by using mechanical thinning and prescribed fire. Implementation of these methods can be constrained by mandates to conserve sensitive wildlife species like the Pacific marten (*Martes caurina*). Martens avoid simplified forest stands created by these fuel reduction treatments, and populations in the northern Sierra Nevada and southern Cascades are already fragmented. Implementing fuel reduction treatments may therefore threaten forest-dependent species like the Pacific marten by reducing available habitat and habitat connectivity.

A crucial question is whether reserving marten habitat from fuel treatment results in an elevated probability of large, intense wildfires compared to permitting treatment in these areas. I used a simulation framework to compare the potential for large fires when fuel treatments were implemented with and without marten habitat reserves, defined as areas where all treatment was prohibited. I also assessed wildfire risk to the dense, high elevation forests typical of marten habitat. I simulated fuel treatments in three watersheds (mean area: ~8,000 ha) and then used randomly-placed ignitions to measure each watershed's capacity for fire spread. For each watershed, I varied the amount of area treated (10%, 20%, and 30% of the watershed) and the

type of marten reserve (none; partial, where some marten habitat was reserved; and complete, where all marten habitat was reserved). I further expanded my simulations to a larger study area (~50,000 ha) to provide a complementary depiction of the relationship between habitat reserves and fuel treatment efficacy at the landscape scale.

Prohibiting fuel treatment within marten habitat had no significant effect on the ability of treatments to control the spread of fire. I observed an increase in the capacity for fire spread only in one instance, when the most restrictive reserve strategy reduced the total area eligible for treatment below the target 30%, the most ambitious treatment scenario. These results are optimistic for managers— effective fuels management was possible with simultaneous retention of large blocks of marten habitat. In contrast, wildlife risk to marten habitat increased when fuel reduction treatments were allowed only outside of predicted habitat areas. My simulations provide evidence that allowing some fuel treatment in the margins of predicted habitat could largely eliminate this increased risk, without incursion into core areas where martens are most likely to reside. Silvicultural prescriptions that can retain canopy cover and elements of old forest structure, while also increasing resilience to fire, would be preferable for reducing wildfire risk while mitigating the short-term effects of management action in these areas.

Assessing Alternatives for Fuel Reduction Treatment and Pacific Marten Conservation in the Southern Cascades and Northern Sierra Nevada

by
Kevin Credo

A THESIS

submitted to

Oregon State University

in partial fulfillment of
the requirements for the
degree of

Master of Science

Presented August 24, 2017
Commencement June 2018

Master of Science thesis of Kevin Credo presented on August 24, 2017

APPROVED:

Major Professor, representing Sustainable Forest Management

Head of the Department of Forest Engineering, Resources and Management

---

Dean of the Graduate School

I understand that my thesis will become part of the permanent collection of Oregon State University libraries. My signature below authorizes release of my thesis to any reader upon request.

Kevin Credo, Author

ACKNOWLEDGEMENTS

This research would not have been possible without the invaluable assistance of colleagues and collaborators near and far. I have benefitted greatly from the support of my major professor, John Bailey, who introduced me to this project and has provided guidance throughout my time in the College of Forestry. I am also very grateful for the wealth of knowledge and encouragement provided by my co-advisor, Katie Moriarty. I would further like to thank Meg Krawchuk for providing constructive commentary throughout this process and pushing me to find ways to communicate these ideas in a clearer manner.

Funding for this project was provided by the U.S. Forest Service, as well as through the generosity of Rick Strachan, the Hayes family, the Alfred W. Moltke Memorial Scholarship, and the Mary McDonald Fellowship. I am enormously grateful for the financial support that has allowed me to pursue my education, and this research project, here at Oregon State.

I would also like to express my sincere thanks to: Patrick Tweedy, who answered more than one question about marten ecology over the last two years; Liz Cole, without whom I might still be in California swinging a D-tape; Patrick Doyle, whose experience, and generosity in sharing it, were great boons to this project; Brian Wing, for sharing his data and his expertise that vastly improved the reliability of these results; the Lassen marten crew, for letting me ride shotgun and all their insights; Laura Corral, who generously provided the stand exam data that was instrumental in the early stages of this project; Ariel Muldoon, whose consultation was instrumental in analyzing the results of this endeavor; and finally Panda Express, for providing the raw calories needed to fuel this research, and then some.

Lastly, I would like to thank my parents, Peg and Bruce Credo, for the faithful breeze upon which I floated into Oregon and everything else that helped me get to where I am today.

CONTRIBUTION OF AUTHORS

Brian Wing assisted with the regression modeling of canopy fuel parameters with LiDAR data in the third chapter, *Assessing strategies for managing fuels in Pacific marten habitat at the landscape scale*.

TABLE OF CONTENTS

TABLE OF CONTENTS (Continued)

Page

TABLE OF CONTENTS (Continued)

LIST OF FIGURES

LIST OF FIGURES (Continued)

<u>Figure</u> <u>Page</u>

LIST OF TABLES

LIST OF APPENDIX TABLES

Table Page

1 | Introduction

Forest management over the past 150 years has altered forest structure and the occurrence of wildfire as a fundamental ecosystem process in the northern Sierra Nevada and southern Cascades (Collins et al. 2011, Taylor 2007). Increasingly effective fire suppression, the policy of extinguishing fires as soon as possible, has resulted in an unprecedented amount of surface litter and small diameter trees in montane forests in this region (Bekker & Taylor 2001, Taylor 2000). Forested areas are currently denser and structurally homogeneous compared to the pre-suppression era, and have higher vertical and horizontal fuel continuity (McIntyre et al. 2015, Van de Water & Safford 2011, Bekker & Taylor 2010, Miller & Urban 2000). Growing incidence of large, intense fires has underlined this departure from more resilient forest structure, a departure that may be exacerbated by anticipated higher temperatures and reduced precipitation due to global climate change (McKenzie et al. 2004, Miller et al. 2009, Westerling et al. 2006).

Reducing the quantity and spatial distribution of forest vegetation can be effective at mitigating extreme fire behavior at stand and landscape scales. Thinning and removal or compaction of surface fuel accumulation, with mechanical treatment or prescribed fire, reduces fire-induced mortality and decreases incidence of crown fire (Agee & Skinner 2005, Safford et al. 2012, Richie et al. 2007). While a substantial body of research has examined the ability of treatments to modify fire behavior and reduce fire intensity, their effect on other ecological processes is not always clear (see Kalies & Kent 2016). For instance, some vulnerable wildlife species depend on complex forest structures like downed logs and multi-story canopies; mechanical interventions that simplify forest structure may be particularly problematic and indeed have proven controversial (Collins et al. 2010, Sierra Nevada Protection Campaign vs. Tippin 2006).

Previous research has examined the effect of standard fuel reduction treatments on sensitive, forest-specialist wildlife in the Sierra Nevada, including the fisher (*Pekania pennanti*), the California spotted owl (*Strix occidentalis occidentalis*), and the Pacific marten (*Martes caurina*) (Sweitzer et al. 2015, Stephens et al. 2014, Moriarty et al. 2016). For the first two species, further investigation has sought to understand the tradeoffs between short-term negative effects of fuel reduction treatment and potential long-term benefits of increased resilience to fire (Scheller et al. 2011, Tempel et al. 2015), but less attention has focused on assessing the risks attached to active management in and near Pacific marten habitat in California.

Pacific martens are associated with high-elevation forests that receive persistent snow (Spencer 1983). These areas may be less departed from pre-settlement fuel structure at the stand scale, as their fire return intervals were historically longer compared to drier sites at lower elevations (Taylor 2000, Beaty & Taylor 2008). For this reason, some have argued these areas should be lower priorities for management intervention, based on the assumption that greater departure from historical conditions is indicative of a higher risk of atypical fire behavior and reduced resilience (Hardy et al. 2001, Hann 2004, Keane et al. 2006). To address this assumption, I quantified the capacity for large fire spread and wildfire risk to marten habitat after prohibiting fuel reduction treatment in marten habitat in favor of prioritizing areas at lower elevations.

1.1 Pre-settlement fire regimes in the northern Sierra Nevada and southern Cascades

Wildfire has long been an integral part of ecosystem function in the Sierra Nevada, largely due to the mediterranean climate of dry summers and thunderstorms that allows lightning to start fires (Minnich 2006, Pyne et al. 1996). Fire acts as an ecological process that controls forest structure and composition, regulating the accumulation of biomass by removing debris and fire-prone vegetation (Agee 1993, Sugihara et al. 2006, North et al. 2009). Vegetation and animal species in this region evolved with wildfire as a periodic disturbance, and species distributions were controlled in part by fire as a driver of succession and seral diversity (Shaffer & Laudenslayer Jr. 2006, Fontaine & Kennedy 2012). Biodiversity in this region was dynamic and perpetuated by the heterogeneous structures regularly produced by fire (Kennedy & Fontaine 2009, Swanson et al. 2010). The pervasive effects of fire extend to even the most basic ecological processes, including productivity, nutrient cycling, air and water quality, and available soil moisture (Kilgore 1973, Agee 1993, Chang 1996).

Fire regimes in the montane forests of this region historically varied with elevation, aspect, and slope position. At lower elevations (1600 – 2000m) and on drier, west- and south-facing slopes, white fir (*Abies concolor*) forms mixed stands with Jeffrey pine (*Pinus jeffreyi*), sugar pine (*Pinus lambertiana*), and ponderosa pine (*Pinus ponderosa*) (Beaty & Taylor 2001, Bekker & Taylor 2001). These drier mixed-conifer forests experienced fire frequently, with estimates for median composite fire return interval ranging from 4 to 12 years (Bekker & Taylor 2010, Taylor 2000, Skinner & Chang 1996). Fires reduced surface fuel accumulation and understory cover, favoring fire-resistant pine species over shade-tolerant true firs (Taylor 2000, North et al. 2005, Taylor 2007). This removal of biomass on the forest floor and in lower canopy layers opened growing space for diverse plant species, resulting in increased structural heterogeneity at multiple scales and recruitment of habitat features like snags and downed logs (North et al. 2009, Shaffer & Laudenslayer, Jr. 2006). While low-severity surface fire in Sierran mixed conifer stands is well documented, stands at lower elevations also experienced patches of high-severity stand-replacing fire in areas with higher fuel, further contributing to the diverse forest structures present across the landscape (Bekker & Taylor 2001, Beaty & Taylor 2001, Bekker & Taylor 2010, Collins & Stephens 2010). This mixed-severity fire regime, a combination of low- and high-severity fire, created a complex mosaic of patches at different stages of succession and with diverse fuel structures (Perry et al. 2011, Agee 1998, Collins et al. 2007).

At middle and upper elevations (1800 – 2200m) and on more mesic sites, mixed-conifer stands are increasingly dominated by true fir species, white fir and red fir (*Abies magnifica*). These forests experienced a historical fire regime characterized by variability in frequency, extent, and severity. Estimates of fire return intervals for this forest type range from 9 years to 64 years for red fir-dominated stands at higher elevations (Bekker & Taylor 2001, Agee 1993, Skinner & Chang 1996, Agee 1991). Fine surface fuels at higher elevations and less exposed aspects were sufficiently dry to ignite for shorter periods of each fire season compared to lower elevations (Bekker & Taylor 2001, Taylor 2000). Further, the sparse and compact litter produced by fir forests is less conducive to rapid fire spread than the less dense litter found at lower elevations, where pine species are present in greater proportions (Skinner & Chang 1996, Van Wagtendonk & Fites-Kaufman 2006, Agee 1993). Areas where surface fuels separated by rock outcroppings or rugged topography experienced longer periods without fire, while those with more continuous litter cover often burned with greater frequency (Skinner & Chang 1996, Bekker & Taylor 2010, Meyer 2013).

Lodgepole pine (*Pinus contorta*) stands are found in mid-elevation mesic flats, valley bottoms, high-water tables, and areas of cold-air drainage in this region (Taylor & Solem 2001, Taylor 2000). Typically strongly dominated by lodgepole pines, these communities had mean fire return intervals between 35-80 years (Taylor 2000, Agee 1993). While the lodgepole pine subspecies found in the Sierras does not have serotinous, or fire-dependent, cones (Skinner & Chang 1996), stand-replacing fires likely were the dominant disturbance in this forests where fuel moisture was rarely low enough to allow fires to burn (Bekker & Taylor 2010).

1.2 Fire exclusion and legacy of forest management

The pattern of fire as a natural disturbance has fundamentally changed as a result of forest management over the last 150 years. In the northern Sierra Nevada, the difficulty and expense of transporting logs by flume and wagon confined early timber harvest to lower elevations near railroad lines (Strong 1982, Cermak 2012). For this reason, timber harvest within the current boundaries of Lassen National Forest may have been limited before 1900 (McKelvey & Johnston 1992). However, livestock grazing profoundly altered forest structure across California before the turn of the century (Gruell 2001, Norman & Taylor 2005). Sheep consumed understory vegetation and regenerating seedlings, while shepherds made extensive use of fire to clear obstacles and increase the abundance of forage plants (Cermak 2012, Coville 1898). Lassen Peak Forest Reserve (later renamed Lassen National Forest) was established in 1905 with fire control as a primary objective (Strong 1982, Cermak 2012). Primitive at first, the organization and technology of fire suppression steadily increased over the 20th century, gradually becoming more effective at controlling wildfire across the landscape (Cermak 2012).

The fire suppression era has since produced forests that are dense and structurally homogenous compared to the pre-settlement period, with large numbers of small trees (McIntyre et al. 2015, Van de Water & Safford 2011, Bekker & Taylor 2010). In the absence of fire, ingrowth of seedlings in the understory has led to a reduction in average diameter and a compositional shift from shade-intolerant to shade-tolerant species (Bekker & Taylor 2001, Collins et al. 2011). Increased surface fuel accumulation is also present in some mixed-conifer forests in the Sierra Nevada (Parsons & DeBenedetti, Keifer et al. 2006).

Tree in-growth in the understory and intermediate canopy layers creates an unnatural level of fuel continuity, horizontally and vertically, allowing fires to more readily spread both over greater distances and from the surface to the canopy (Parsons & DeBenedetti 1979, Husari et al. 2006). This in-growth effect is exacerbated by the compositional shift from more fire-resistant pine species to less fire-resistant true firs (Taylor 2000, Keane et al. 2006). These effects are most extreme in forests that historically experienced frequent fire (Taylor 2000, Taylor 2007, Skinner & Chang 1996). For example, drier mixed-conifer stands burned on average every 4– 12 years; with over 100 years of fire suppression some stands may have missed ten or more fire cycles and converted to dominantly true fir species. In red fir and lodgepole pine stands, with longer fire return intervals, departure from natural fuel conditions is present over the landscape but may not be as significant at the scale of an individual stand (Taylor 2000, Meyer 2013, Miller & Safford 2012).

Increased wildfire activity since the 1980s has been documented across the western United States as well as in the northern Sierra Nevada and southern Cascades (Westerling et al. 2006, Miller et al. 2009, Dennison et al. 2014). While current area burned per year may still be significantly lower than pre-settlement levels (Mallek et al. 2013, Stephens et al. 2007), increases in the proportion of high-severity area or concentration of high-severity area in patches has raised concerns about the potential ecological impacts of fires that are larger or more severe than expected for a given ecosystem (Miller et al. 2009, Miller & Safford 2012, Keane et al. 2008, Reilly et al. 2017). While fire-adapted ecosystems are by definition resilient to a given regime of fire disturbance, unusually large and intense wildfires may have permanent adverse effects on these ecosystems (Collins & Roller 2013, Goforth & Minnich 2008). Recent research in the northern Sierra Nevada has shown that conifer regeneration in large high-severity patches can be minimal (Crotteau et al. 2013, Coppoletta et al. 2016). Resprouting shrubs and increased distances to seed trees may prevent conifer regeneration for long periods (Chappell and Agee 1996, Donato et al. 2009). The propensity for areas with high shrub cover to reburn at high severity suggests the possibility of a feedback loop that perpetuates these altered landscapes (Coppoletta et al. 2016).

Given the complex suite of factors that regulate fire occurrence and behavior, it is difficult to definitively attribute cause to the increased wildfire activity in northern California. While climate, and in particular drought, are associated with larger and more numerous fires (Westerling et al. 2006, Littell et al. 2009), greater incidence of high-severity fire compared to the pre-settlement period suggest twentieth century management and fuel accumulation have played a role in altering fire regimes (Harris & Taylor 2015, Keane et al. 2008, Reilly et al. 2017).

1.3 Fuel treatment

In light of the pervasive impact of fire suppression and projected increases in fire activity due to climate change, forest managers are challenged to restore fire resistance and resilience through active management. In the Sierra Nevada, reducing fuel levels to aid fire suppression efforts and prevent uncharacteristically large, stand-replacing disturbances has emerged as a significant management goal (USDA 2004, North et al. 2009). While allowing wildfires to burn can reduce fuel accumulation and help restore resilient forest structure, such a policy is impractical in many regions due to safety, legal, and political constraints (Van Wagtendonk

2007, Collins & Stephens 2007). Strategies for mitigating fire hazard have therefore focused on removing fuels through mechanical treatment and prescribed fire (Agee & Skinner 2005, Stephens et al. 2012).

Fuel treatments typically target some combination of three fuel types: (a) surface fuels that consist of leaf litter, duff, and understory vegetation; (b) ladder fuels of moderate height that allow surface fires to transition to crown fires, and (c) crown fuels, consisting of branches and leaves of trees and large shrubs that, if dense enough, permit fire to spread through the canopy (Husari et al. 2006). Types of mechanical treatments include thinning of trees to reduce ladder fuels and/or the continuity of crown fuels; mastication, the grinding or chipping of the surface fuel layer to leave a compact fuel bed on site; and grazing, the use of livestock to reduce surface fuel loading (Husari et al. 2006). A substantial body of literature has explored the efficacy of prescribed fire, or controlled burning under carefully monitored conditions, and mechanical treatments on fire behavior and effects, using both real-world and modeled fire (e.g. Agee & Skinner 2005, Safford et al. 2012, Richie et al. 2007). A combination of the two may be most effective at modifying fire behavior at the stand scale, compared to treatments that employ one method alone (Kalies & Kent 2015, Stephens et al. 2009).

Fuel treatments can temper fire behavior within a landscape. Strategically-placed treatments on 20% of a real-world landscape were effective at reducing average fire growth rate (Finney 2007). That rate is equivalent to 2% treatment area per year if treatment longevity is 10 years (Finney et al. 2007). Similar modeling efforts on planned and constructed fuel treatment projects in the Sierra Nevada have found that similar treatment levels (19% and 25%) significantly reduced conditional burn probabilities, indicating such management was successful at modifying fire behavior in a landscape (Moghaddas et al. 2010, Collins et al. 2011b). The efficacy of a given treatment was not independent of its spatial arrangement, meaning that fire growth could be reduced by employing methods for optimizing fuel treatment placement (Finney 2004, Schmidt et al. 2008).

In the Sierra Nevada, the “fireshed” has emerged as a useful planning unit for fire management activities. A fireshed is a contiguous geographic area, with an area of approximately 100–500 km^2, that shares a common fire regime, fire risk factors, and has the appropriate scale for coordinated suppression activities (Ager et al. 2006). Since a formal fireshed assessment has not been conducted for all federal lands in California, hydrologic units like watersheds have been used as a substitute, under the assumption that the topographic features that control water drainage can similarly anchor wildfire suppression activities (North et al. 2015). Thus, watersheds can provide boundaries to simulate potential treatments, both for optimizing locations for planning as well as for predicting their effectiveness in reducing wildfire risk.

1.4 Pacific marten

The Pacific marten (*Martes caurina)* is a small carnivore in the Mustelid famiyl and a forest specialist, associated with dense canopy cover and old forest structures at stand and landscape scales (Buskirk & Powell 1994, Spencer 1987). In this region, martens occupy forest stands at high elevations that experience persistent snow, and are associated with stands dominated by red fir, white fir, and lodgepole pine (Spencer 1983, Purcell et al.2012).

At the stand scale, Pacific martens require structural features like snags and downed logs for denning and resting (Spencer 1987, Martin & Barrett 1991). Forest structure surrounding resting sites is typically complex, with diverse tree sizes and a dense, multi-layer canopy (Purcell et al. 2012). Martens require numerous structures of this type to be distributed across the landscape (Purcell et al. 2012). As such, resting sites and connectivity between such locations may be a limiting habitat requirement in the northern Sierra Nevada.

Martens occupy large home ranges for being the size: average territory sizes for males and females are 572 ha and 355 ha respectively (Purcell et al. 2012). Martens select home ranges with disproportionately high levels of canopy cover than available on the landscape (Moriarty et al. 2016, Spencer 1983). Within a home range, martens further exhibit a strong preference for areas with complex forest structure. This preference is likely related to increased foraging efficiency and improved predator avoidance in these stands (Andruskiw et al. 2008, Moriarty et al. 2015). Martens avoid stands with simplified structure that results from forest management activity, even when canopy cover is left intact (Moriarty et al. 2015, 2016, but see Payer and Harrison 2000).

Marten populations may decline quickly in response to fragmentation and loss of mature forest habitat (Chapin et al. 1998, Hargis et al. 1999, Moriarty et al. 2011). As a result, this species was designated a Management Indicator Species for late-seral, closed-canopy forests for all National Forests across the Sierra Nevada (USDA 2007), and is currently considered a vulnerable species by the state of California (CA Dept. of Fish and Wildlife).

The current distribution of Pacific martens is reduced compared to historical records (Zielinski et al. 2005, Grinnell 1937). Populations in the northern Sierra Nevada and southern Cascades are more fragmented than elsewhere in California (Spencer & Rustigian-Romsos 2012, Kirk & Zielinski 2010), a situation that may be due to timber harvest and forest management during the 20th century (Zielinski et al. 2005). Further fragmentation and loss of habitat is predicted in the next century due to climate change, due to expected reductions in snow pack, increases in temperature, and ensuing changes in vegetation structure (Spencer et al. 2015).

1.5 Assessing wildfire risk to sensitive wildlife species

Balancing fire management with the conservation of sensitive animal species in the western United States has created the need for methods to assess the degree to which fire threatens a given animal population. One approach integrates modeling of forest growth, fire behavior, and population dynamics to simulate the effect of alternative fuel treatment strategies on the number of reproductive individuals or viable habitat areas (Tempel et al. 2015, Scheller et al. 2011, Lee & Irwin 2005). A related method, landscape trajectory analysis, has likewise used simulation of forest growth over time to track changes in the amount or arrangement of wildlife habitat under different treatment scenarios (Thompson et al. 2011, Cushman & McGarigal 2007). While the capacity to capture both the effects of fuel treatments implemented over time and evaluate the response in terms of species demographics is valuable, a shortcoming of this temporal simulation approach is a limited ability to control for variation in potential fire location and behavior; simulating post-fire forest growth becomes increasingly computationally expensive with each additional fire that is simulated (Tempel et al. 2015, Thompson et al. 2011).

An alternative method is the extension of risk analysis to wildland fire, where risk is defined as the expectation of loss or benefit from fire and is based on the likelihood of a fire event, the expected intensity, and the effect of that event on a value or set of values (Finney 2005, Scott 2006, Miller & Ager 2013). Risk, a quantification of expected loss or benefit, is distinguished from fire hazard, which describes the potential for loss of a value should a fire occur without regard to the likelihood of fire or the relative importance of the value (Miller &

Ager 2013). For example, a stand with heavy surface and ladder fuel loads may have a high fire hazard, but contribute little to overall fire risk if it is sheltered by bare ground or recent fuel treatment and contains no high-value resources.

A risk-based or related approach has been applied to sensitive wildlife species like the northern spotted owl (*Strix occidentalis caurina*) (Ager et al. 2007, Roloff et al. 2011), as well as broadly implemented to integrate expected losses or benefits to multiple valued resources (Scott et al. 2013, Ager et al. 2010). The development of the minimum travel time algorithm, a faster and more efficient method for simulating fire spread, has made it possible to simulate large numbers of ignitions moving over a landscape (Finney 2002, Miller & Ager 2013). This ability to simulate a large number of fires under a suite of potential weather conditions, has allowed for more comprehensive fire risk assessments that can integrate a greater proportion of potential fire behaviors.

1.6 Research goals and objectives

Simulation- and risk-based approaches have been used to examine interactions between fuel treatments and the conservation of sensitive forest wildlife in California, including the fisher and the California spotted owl. However, less attention has been given to the spatial arrangement of fuel treatments in proximity to the Pacific marten population of the southern Cascades and northern Sierra Nevada, a population that appears more fragmented than elsewhere in this species' California range. The broad goals of this research effort were to apply a risk-based framework to inform the planning and spatial allocation of fuel treatment projects in this region, and to work towards establishing a scientific basis for balancing the goals of marten conservation and restoration of fire resilience.

Consequently, this investigation explored the costs and benefits of active fuels management in marten habitat areas in Almanor Ranger District, Lassen National Forest, California in terms of both the ability of landscape-scale fuel treatment to alter fire behavior and the expected loss of marten habitat from wildfire events. I compared the effect of two levels of marten habitat reserves, where management is prohibited, at two spatial scales. The project scale (~8,000 ha, Chapter 2) explored the effect of habitat reserves at the spatial scale of a single fuel treatment project, while the landscape scale (~50,000 ha, Chapter 3) allowed for a more holistic view of how reserves affect fuels management and marten conservation efforts.

1.7 Chapter 1 | Literature cited

Agee, J.K. 1993. Fire Ecology of Pacific Northwest Forests. 1st ed. Island Press. Washington, DC, USA.

Agee, J.K. 1998. The landscape ecology of Western forest fire regimes. Northwest Science **72**: 24-34.

Agee, J.K., and Skinner, C.N. 2005. Basic principles of forest fuel reduction treatments. For. Ecol. Manag. **211**(1-2): 83–96. doi:10.1016/j.foreco.2005.01.034.

Ager, A.A., Bahro, B., and Barber, K. 2006. Automating the fireshed assessment process with ArcGIS. *In* Fuels Management-How to Measure Success, Portland, OR, 28-30 March 2006. *Edited by* Patricia L. Andrews, Bret W. Butler. pp. 163-168.

Ager, A.A., Finney, M.A., Kerns, B.K., and Maffei, H. 2007. Modeling wildfire risk to northern spotted owl (Strix Occidentalis Caurina) Habitat in Central Oregon, USA. For. Ecol. Manag. **246**(1): 45–56. doi:10.1016/j.foreco.2007.03.070.

Ager, A.A., Vaillant, N.M., Finney, M.A., and Preisler, H.K. 2010. A comparison of landscape fuel treatment strategies to mitigate wildland fire risk in the urban interface and preserve old forest structure. Forest Ecology and Management **259**(8): 1556-1570.

Andruskiw, M.J., Fryxell, J.M., Thompson, I.D., and Baker, J.A. 2008. Habitat-mediated variation in predation risk in the American marten. Ecology **89**(8): 2273-2280.

Beaty, M.R., and Taylor, A.H. 2001. Spatial and temporal variation of fire regimes in a mixed conifer forest landscape, Southern Cascades, California, USA. J. Biogeogr. **28**(8): 955–66. doi:10.1046/j.1365-2699.2001.00591.x.

Beaty, R.M., and Taylor, A.H. 2008. Fire history and the structure and dynamics of a mixed conifer forest landscape in the northern Sierra Nevada, Lake Tahoe Basin, California, USA. For. Ecol. and Manag. **255**(3-4): 707-719.

Bekker, M.F., and Taylor, A.H. 2001. Gradient analysis of fire regimes in montane forests of the Southern Cascade Range, Thousand Lakes Wilderness, California, USA. Plant Ecol. **155**(1): 15–28. doi:10.1023/A:1013263212092.

Bekker, M.F., and Taylor, A.H. 2010. Fire disturbance, forest structure, and stand dynamics in montane forests of the Southern Cascades, Thousand Lakes Wilderness, California, USA. Ecoscience **17**(1): 59–72. doi:10.2980/17-1-3247.

Buskirk, S.W., and Powell, R.A. 1994. Habitat ecology of fishers and American martens. *In* Martens, sables, and fishers: biology and conservation. *Edited by* Buskirk, S.W., Harestad, A.S., Raphael, M.G., and Powell, R.A. Comstock Publishing Associates, Cornell University Press. Ithaca, NY.

California Department of Fish and Wildlife. 2017. Natural Diversity Database: Special Animals List. Periodic publication. 51 pp.

Carr, S.M., and Hicks, S.A. 1997. Are there two species of marten in North America? Genetic and evolutionary relationships within Martes. *In* Martes: Taxonomy, Ecology, Techniques, and Management. *Edited* by G. Proulx, H.N. Bryant, P.M. Woodard. University of Alberta Press, Edmonton, Alberta. pp. 15-28.

Cermak, R.W. 2005. Fire in the Forest: A History of Forest Fire Control on the National Forests in California, 1898-1956. USDA Forest Service, Pacific Southwest Region, California, USA.

Chang, Chi-Ru. 1996. Ecosystem responses to fire and variations in fire regimes. *In* Sierra Nevada Ecosystem Project: Final report to Congress, vol. II, Assessments and scientific basis for management options. University of California, Centers for Water and Wildland Resources, Davis, CA.

Chapin, T.G., Harrison, D.J., and Katnik, D.D. 1998. Influence of landscape pattern on habitat use by American marten in an industrial forest. Conserv. Biol. **12**(6): 1327–37.

Chappell, C.B., and Agee, J.K., 1996. Fire severity and tree seedling establishment in Abies magnifica forests, Southern Cascades, Oregon. Ecol. Appl. **6**(2): 628–40. doi:10.2307/2269397.

Collins, B.M., and Roller, G.B.. 2013. Early forest dynamics in stand-replacing fire patches in the northern Sierra Nevada, California, USA. Landscape Ecol. **28**(9): 1801–13. doi:10.1007/s10980-013-9923-8.

Collins, B.M. and Stephens, S.L. 2007. Managing natural wildfires in Sierra Nevada wilderness areas. Frontiers in Ecology and the Environment **5**(10): 523-527.

Collins, B.M., and Stephens, S.L. 2010. Stand-replacing patches within a 'mixed severity' fire regime: quantitative characterization using recent fires in a long-established natural fire area. Landscape Ecology **25**(6) 927-939.

Collins, B.M., Kelly, M., van Wagtendonk, J.W., and Stephens, S.L. 2007. Spatial patterns of large natural fires in Sierra Nevada wilderness areas. Landscape Ecology **22**(4): 545-557.

Collins, B.M., Stephens, S.L., Moghaddas, J.J., and Battles, J. 2010. Challenges and approaches in planning fuel treatments across fire-excluded forested landscapes. J. Forest. **108**(1): 24 – 31.

Collins, B.M., Everett, R.G., and Stephens, S.L. 2011a. Impacts of fire exclusion and recent managed fire on forest structure in old growth Sierra Nevada mixed-conifer forests. Ecosphere **2**(4): 1–14. doi:10.1890/ES11-00026.1.

Collins, B.M., Stephens, S.L., Roller, G.B., and Battles, J.J. 2011b. Simulating fire and forest dynamics for a landscape fuel treatment project in the Sierra Nevada. Forest Science **57**(2): 77-88.

Coppoletta, M., Merriam, K.E., and Collins, B.M. 2016. Post-fire vegetation and fuel development influences fire severity patterns in reburns. Ecol Appl. **26**(3): 686-699.

Coville, F.V. 1989. Forest growth and sheep grazing in the Cascade mountains of Oregon. USDA Forest Division Bulletin. No. 15. Government Printing Office, Washington, DC.

Crotteau, J.S., Varner, J.M., and Ritchie, M.W. 2013. Post-fire regeneration across a fire severity gradient in the Southern Cascades. For. Ecol. Manag. **287**(2013): 103–12. doi:10.1016/j.foreco.2012.09.022.

Cushman, S.A., and McGarigal, K. 2007. Multivariate landscape trajectory analysis: an example using simulation modeling of American marten habitat change under four timber harvest scenarios. *In* Temporal Dimensions of Landscape Ecology. *Edited by* J.A. Bissonnette and I. Storch. Springer, US. pp. 119-40.

Dawson, N.G., and Cook, J.A. 2012. Behind the genes: diversification of North American martens (*Martes americana* and *M. caurina*). *In* Biology and conservation of martens, sables, and fishers: a new synthesis. *Edited by* Aubry, K.B., Zielinski, W., Raphael, M.G., Proulx, G., and Buskirk, S.W. Cornell University Press, Ithaca, New York. pp. 23-38.

Dennison, P.E., Brewer, S.C., Arnold, Moritz, M.A. 2014. Large wildfire trends in the western United States, 1984-2011. Geophys. Res. Letter. **41**: 2928-2933.

Donato, D.C., Fontaine, J.B., Campbell, J.L., Robinson, W.D., Kauffman, J.B., and Law, B.E. 2009. Conifer regeneration in stand-replacement portions of a large mixed-severity wildfire in the Klamath-Siskiyou Mountains. Canadian Journal of Forest Research **39**(4): 823-838.

Finney, M.A. 2002. Fire growth using minimum travel time methods. Canadian Journal of Forest Research **32**: 1420-1424.

Finney, M.A. 2004. Landscape Fire Simulation and Fuel Treatment Optimization. *In* Methods for Integrated Modeling of Landscape Change: Interior Northwest Landscape Analysis System. Gen. Tech. Rep. PNW-GTR-610. *Edited by* J.L.

Hayes, A.A. Ager, J.R. Barbour. USDA Forest Service, Pacific Northwest Research Station, Portland, OR. pp. 117–31.

Finney, M.A. 2005. The challenge of quantitative risk analysis for wildland fire. For. Ecol. Manag. **211**: 97-108.

Finney, M.A. 2007. A computational method for optimizing fuel treatment locations. International Journal of Wildland Fire **16**(6): 702-711.

Finney, M.A., Seli, R.C., McHugh, C.W., Ager, A.A., Bahro, B., and Agee, J.K. 2007. Simulation of long-term landscape-level fuel treatment effects on large wildfires. International Journal of Wildland Fire **16**(2007): 7612-727.

Fontaine, J.B., and Kennedy, P.L. 2012. Meta-analysis of avian and small-mammal response to fire severity and fire surrogate treatments in U.S. fire-prone forests. Ecological Applications **22**(5): 1547-1561.

Goforth, B.R. and Minnich, R.A. 2008. Densification, stand-replacement wildfire, and extirpation of mixed conifer forest in Cuyamaca Rancho State Park, southern California. Forest Ecology and Management **256**(1-2): 36-45.

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

Gruell, G.E. 2001. Fire in Sierra Nevada Forests: A Photographic Interpretation of Ecological Change Since 1849. 1st ed. Mountain Press Publishing Co., Missoula, MT.

Hann, W.J. 2004. Mapping fire regime condition class: a method for watershed and project scale analysis. *In* Proceedings of the 22nd Tall Timbers Fire Ecology Conference: Fire in Temperature, Boreal, and Montane Ecosystems. *Edited by* R.T.Engstrom, K.E.M. Galley, and W.J. de Groot. Tall Timbers Research Station. Tallahassee, Fl.

Hardy, C.C., Schmidt, K.M., Menakis, J.P., and Sampson, R. Neil. 2001. Spatial data for national fire planning and fuel management. International Journal of Wildland Fire **10**: 353-372.

Hargis, C.D., Bissonette, J.A., and Turner, D.L. 1999. The Influence of Forest Fragmentation and Landscape Pattern on American Martens. J. Appl. Ecol. **36**(1): 157–72.

Harris, L. and Taylor, A.H. 2015. Topography, fuels, and fire exclusion drive fire severity of the Rim Fire in an old-growth mixed-conifer forest, Yosemite National Park, USA. Ecosustems **18**(7): 1192-1208.

Husari, S.H., Nichols, T., Sugihara, N.G., and Stephens, S.L. 2006. Fire and Fuel Management. *In* Fire in California's Ecosystems. *Edited by* N.G. Sugihara. University of California Press, Berkeley, CA. pp. 431–43.

Kalies, E.L. and Kent, L.L.Y., 2016. Tamm Review: Are fuel treatments effective at achieving ecological and social objectives? A systematic review. For. Ecol. Manag. **375**(2016) 84-95. http://dx.doi.org/10.1016/j.foreco.2016.05.021

Keane, R.E., Agee, J.K., Fule, P., Keeley, J.E., Key, C., Kitchen, S.G., Miller, R., and Schulte, L.A. 2008. Ecological effects of large fires on US landscapes: benefit or catastrophe? Intern. Journal of Wildland Fire. **17**(2008): 696-712.

Keane, R.E., Rollins, M., and Zhu, Z-L. 2006. Using simulated historical time series to prioritize fuel treatments on landscapes across the United States: The LANDFIRE prototype project. Ecological Modelling **204**(3-4): 485-502.

Keane, R., Ryan, K., Veblen, T., Allen, C., Logan, J., and Hawkes, B. 2002. Cascading Effects of Fire Exclusion in Rocky Mountain Ecosystems: A Literature Review. Gen. Tech. Rep. RMRS-GTR-91. USDA Forest Service, Rocky Mountain Research Station, Fort Collins, CO.

Keifer, M., van Wagtendonk, J.W., and Buhler, M. 2006. Long-term surface fuel accumulation in burned and unburned mixed-conifer forests of the central and southern Sierra Nevada, CA (USA). Fire Ecology **2**(1): 53-72.

Kennedy, P.L., and Fontaine, J.B. 2009. Synthesis of knowledge on the effects of fire and fire surrogates on wildlife in U.S. dry forest, Special Report 1096. Oregon State Agricultural Experiment Station, Corvallis, OR.

Kilgore, B.M. 1973. The ecological role of fire in Sierran conifer forests: Its application to National Park management. Quaternary Research **3**(3): 496 – 513.

Kirk, T.A., and Zielinski, W.J. 2009. Developing and testing a landscape habitat suitability model for the American marten (Martes americana) in the Cascade Mountains of California. Landscape Ecol. **24**(6): 759–73. doi:10.1007/s10980-009-9349-5.

Kirk, T.A., and Zielinski, W.J. 2010. Functional habitat connectivity of the American marten in northeastern California using least-cost corridor modeling. USDA Forest Service, Lassen National Forest and Pacific Southwest Research Station, Redwood Sciences Lab, California, USA.

Lee, D.C., and Irwin, L.L. 2005. Assessing risks to spotted owls from forest thinning in fire-adapted forests of the western United States. Forest Ecology and Management **211**(1-2): 191-209.

Littell, J.S., McKenzie, D., Peterson, D.L., Westerling, A.L. 2009. Climate and wildfire area burned in western U.S. ecoprovinces, 1916-2003. Ecological Applications **19**(4): 1003-1021.

Mallek, C., Safford, H., Viers, J., and Miller, J. 2013. Modern departures in fire severity and area vary by forest type, Sierra Nevada and southern Cascades, California, USA. Ecosphere **4**(12): 153.

Martin, S.K., and Barrett, R.H. 1991. Resting site selection by marten at Sagehen Creek, California. Northwester Naturalist **72**:37 – 42.

McIntyre, P.J., Thorne, J.H., Dolanc, C.R., Flint, A.L., Flint, L.E., Kelly, M. and Ackerly, D.D. 2015. Twentieth-century shifts in forest structure in California: denser forests, smaller trees, and increased dominance of oaks. P. Natl. Acad. Sci. USA. **112**(5): 1458–63. doi:10.1073/pna.s.1410186112.

McKelvey, K.S., and Johnston, J.D. 1992. Historical perspectives on forests of the Sierra Nevada and the transverse ranges of southern California at the turn of the century. General Technical Report PSW-GTR-133. USDA Forest Service. Berkeley, CA.

McKenzie, D., Gedalof, Z., Peterson, D.L., and Mote, P. 2004. Climate change, wildfire, and conservation. Conservation Biology **18**(4): 890-902. doi:10.1111/j.1523-1739.2004.00492.x

Meyer, M.D. 2013. Natural range of variation of red fir forests in the bioregional assessment area. Unpublished report. USDA Forest Service, Pacific Southwest Region. Albany, CA.

Miller, C. and Ager, A.A. 2013. A review of recent advances in risk analysis for wildfire management. Int. J. of Wildland Fire **2013**(22): 1-14. http://dx.doi.org/10.1071/WF11114

Miller, C. and Urban, D.L. 2000. Connectivity of forest fuels and surface fire regimes. Landscape Ecology **15**(2): 145-154. doi: 10.1023/A:10008181313360

Miller, J.D., and Safford, H. 2012. Trends in wildfire severity: 1984 to 2010 in the Sierra Nevada, Modoc Plateau, and Southern Cascades, California, USA. Fire Ecology **8**(3): 41–57. doi:10.4996/fireecology.0803041.

Miller, J. D., Safford, H.D., Crimmins, M. and Thode, A.E. 2009. Quantitative evidence for increasing forest fire severity in the Sierra Nevada and Southern Cascade Mountains, California and Nevada, USA. Ecosystems **12**(1): 16–32. doi:10.1007/s10021-008-9201-9.

Minnich, R.A. 2006. California Climate and Fire Weather. *In* Fire in California's Ecosystems. *Edited by* Sugihara, N.G., Van Wagtendonk, J.W., Shaffer, K.E, Fites-Kaufman, J., and Thode, A.E. University of California Press, Berkeley and Los Angeles, CA.

Moghaddas, J.J., Collins, B.M., Kurt Menning, Moghaddas, E.E.Y., and Stephens,S.L. 2010. Fuel treatment effects on modeled landscape-level fire behavior in the northern Sierra Nevada. Can. J. For. Res. **40**(9): 1751–65. doi:10.1139/X10-118.

Moriarty, K.M., Epps, C.W., Betts, M.G., Hance, D.J., Bailey, J.D., and Zielinski, W.J. 2015. Experimental evidence that simplified forest structure interacts with snow cover to influence functional connectivity for Pacific martens. Landscape Ecol. **30**(10): 1865–77. doi:10.1007/s10980-015-0216-2.

Moriarty, K.M., Epps, C.W., and Zielinski, W.J. 2016. Forest thinning changes movement patterns and habitat use by Pacific marten. Jour. Wild. Mgmt. **80**: 621-633. doi:10.1002/jwmg.1060.

Moriarty, K.M., Zielinski, W.J., and Forsman, E.D. 2011. Decline in American marten occupancy rates at Sagehen Experimental Forest, California. J. Wildlife Manage. **75**(8): 1774–87. doi:10.1002/jwmg.228.

Norman, S.P., and Taylor, A.H. 2005. Pine forest expansion along a forest-meadow ecotone in northeastern California, USA. Forest Ecology and Management **215**(1-3): 51-68.

North, M., Brough, A., Long, J., Collins, B., Bowden, P., Yasuda, D., Miller, J., and Sugihara, N. 2015. Constraints on mechanized treatment significantly limit mechanical fuels reduction extent in the Sierra Nevada. *Journal of Forestry* **113**(1): 40–48. doi:10.5849/jof.14-058.

North, M, Hurteau, M., Fiegener, R., Barbour, M. 2005. Influence of fire and El Nino on tree recruitmkent varies by species in Sierran mixed conifer. Forest Science **51**(3) 187-197.

North, M., Stine, P., O'Hara, K., Zielinski, W., and Stephens, S.L. 2009. An ecosystem management strategy for Sierran mixed-conifer Forests. Gen. Tech. Rep. PSW-GTR-220. USDA Forest Service, Pacific Southwest Research Station, Albany, CA.

Parsons, D.J., DeBenedetti, S.H. 1979. Impact of fire suppression on a mixed-conifer forest. Forest Ecology and Management **2**(1979): 21-33.

Payer, D.C., and Harrison, D.J. 2000. Structural differences between forests regenerating following spruce budworm defoliation and clear-cut harvesting: implications for marten. Canadian Journal of Forest Research **30**(12): 1965-1972.

Perry, D.A., Hessburg, P.F., Skinner, C.N., Spies, T.A., Stephens, S.L., Taylor, A.H., Franklin, J.F., McComb, B., and Riegel, G. 2011. The ecology of mixed severity fire regimes in Washington, Oregon, and Northern California. Fore. Ecol. and Manag. **262**(2011): 703-717.

Purcell, K.L.,Thompson, C.M., Zielinsk, W.J. 2012. Chapter 4: Fishers and American martens. *In* Managing Sierra Nevada Forests. Gen. Tech. Rep. PSW-GTR-237. *Edited by* M. North. USDA Forest Service, Pacific Southwest Research Station, Albany, CA.

Pyne, S.F., Andrews, P.L, and Laven, R.D. 1996. Introduction to wildland fire. John Wiley and Sons, New York, NY.

Reilly, M.J., Dunn, C.J., Meigs, G.W., Spies, T.A., Kennedy, R.E., Bailey, J.D., and Briggs, K. Contemporary patterns of fire extent and severity in forests of the Pacific Northwest, USA (1987-2010). Ecosphere **8**(3):e01695.

Richie, M.W., Skinner, C.N., and Hamilton, T.A. 2007. Probability of tree survival after wildfire in an interior pine forest of northern California: Effects of thinning and prescribed fire. For. Ecol. and Manag. **247**: 200-208.

Roloff, G.J., Mealey, S.P., and Bailey, J.D. 2012. Comparative hazard assessment for protected species in a fire-prone landscape. For. Ecol. Manag. **277**: 1–10. doi:10.1016/j.foreco.2012.04.015.

Safford, H.D., Stevens, J.T., Merriam, K., Meyer, M.D., and Latimer, A.M. 2012. Fuel treatment effectiveness in California yellow pine and mixed conifer forests. For. Ecol. and Manag. **274**: 17-28.

Scheller, R.M., Spencer, W.D., Rustigian-Romsos, H., Syphard, A.D., Ward, B.C. and Strittholt, J.R. 2011. Using stochastic simulation to evaluate competing risks of wildfires and fuels management on an isolated forest carnivore. Landscape Ecol. **26**(10): 1491–1504. doi:10.1007/s10980-011-9663-6.

Schmidt, D.A., Taylor, A.H., and Skinner, C.N. The influence of fuels treatment and landscape arrangement on simulated fire behavior, Southern Cascade range, California. Forest Ecology and Management **255**(8-9): 3170-3184.

Scott, J.H. 2006. An analytical framework for quantifying wildland fire risk and fuel treatment benefit. *In* Fuels Management-How to Measure Success: Conference Proceedings. 28-30 March 2006; Portland, OR. Proceedings RMRS-P-41. *Edited by* Andrews, P.L. and Butler, B.W. USDA Forest Service, Rocky Mountain Research Station. Fort Collins, CO.

Scott, J.H., Thompson, M.P., Calkin, D.E. 2013. A wildfire risk assessment framework for land and resource management: General Technical Report RMRS-GTR-315. USDA Forest Service, Rocky Mountain Research Station. Fort Collins, CO.

Shaffer, K.E, and Laudenslayer, Jr., W.F. 2006. Fire and animal interactions. *In* Fire in California's Ecosystems. *Edited by* Sugihara, N.G., Van Wagtendonk, J.W., Shaffer, K.E, Fites-Kaufman, J., and Thode, A.E. University of California Press, Berkeley and Los Angeles, CA.

Sierra Nevada Protection Campaign vs. Tippin. 2006. No. CIV. S 06-00351 FCD DAD. US District Court, Eastern District of California.

Skinner, C.N., and Chang, C. 1996. Fire Regimes, Past and Present. *In* Sierra Nevada Ecosystem Project: Final Report to Congress, Vol II: Assessments and Scientific Basis for Management Options. Wildland Resources Center Report No. 37. Centers for Water and Wildland Resources, University of California, Davis. pp. 1041–69.

Spencer, W.D. 1983. Marten habitat preferences in the northern Sierra Nevada. J. Wildlife Management. **47**(4): 1181-1186.

Spencer, W.D. 1987. Seasonal rest-site preferences of pine martens in the northern Sierra Nevada. J. Wildlife Manage. **51**(3): 616–21. doi:10.2307/3801279.

Spencer, W.D. and Rustigian-Romsos, H.L. 2012. Decision-support maps and recommendations for conserving rare carnivores in the interior mountains of California. Conservation Biology Institute, Corvallis, OR.

Spencer, W.D., Rustigian-Romsos, H., Ferschweiler, K., and Bachelet, D. 2015. Simulating effects of climate and vegetation change on distributions of martens and fishers in the Sierra Nevada, California, Using Maxent and MC1. *In* Global Vegetation Dynamics: Concepts and Applications in the MC1 Model. *Edited by* D. Bachelet, D. Turner. American Geophysical Union. pp. 135–49.

Stephens, S.L., Martin, R.E., Clinton, N.E. 2007. Prehistoric fire area and emissions from California's forests, woodlands, shrublands, and grasslands. Forest Ecology and Management **251**(3): 205-216.

Stephens, S.L., Moghaddas, J.J., Edminster, C., Fiedler, C.E., Haase, S., Harrington, M., Keeley, J.E., Knapp, E.E., McIver, J.D., Metlen, K., Skinner, C.N., Youngblood, A. 2009. Fire treatment effects on vegetation structure, fuels, and potential fire severity in western U.S. forests. Ecological Applications **19**(2): 305-320.

Stephens, S.L., McIver, J.D., Boerner, R.E.J., Fettig, C.J., Fontaine, J.B., Hartsough, B.R., Kennedy, P.L. and Schwilk, D.W. 2012. The Effects of forest fuel-reduction treatments in the United States. BioScience **62**(6): 549–60. doi:10.1525/bio.2012.62.6.6.

Stephens, S. L., Agee, J.K., Fule, P.Z., North, M.P., Romme, W.H.,Swetnam, T.W., and Turner, M.G. 2013. Managing forests and fire in changing climates. Science **342**(6154): 41–42.

Stephens, S.L., Bigelow, S.W., Burnett, R.D., Collins, B.M., Gallagher, C.V., Keane, J., Kelt, D.A., North, M.P., Roberts, L.J., and Stine, P.A., Van Vuren, D.H. 2014. California spotted owl, songbird, and small mammal responses to landscape fuel treatments. Bioscence **64**(10): 893-906.

Strong, D. 1982. Those Happy Hunting Grounds: A History of the Lassen Region. Loomis Museum Association, Mineral, CA.

Sugihara, N.G, Van Wagtendonk, J.W., and Fites-Kaufman, J. 2006. Fire as an ecological process. *In* Fire in California's Ecosystems. *Edited by* Sugihara, N.G., Van Wagtendonk, J.W., Shaffer, K.E, Fites-Kaufman, J., and Thode, A.E. University of California Press, Berkeley and Los Angeles, CA.

Swanson, M.E., Franklin, J.F., Beschta, R.L., Crisafulli, C.M., DellaSala, D.A., Hutto, R.L., Lindenmayer, D.B., and Swanson, F.J. 2010. The forgotten stage of forest succession: early-successional ecosystems on forest sites. Fronters in Ecology and the Environment **9**(2): 117 – 125.

Sweitzer, R.A., Furnas, B.J., Barrett, R.H., Purcell, K.L., and Thompson, C.M. 2015. 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. For. Ecol. and Manag. **361**(2016): 208-225.

Taylor, A.H. 2000. Fire regimes and forest changes in mid and upper montane forests of the southern Cascades, Lassen Volcanic National Park, California, U.S.A. Journal of Biogeography **27**(1): 87-104. doi:10.1046/j.1365-2699.2000.00353.x

Taylor, A.H. 2007. Forest changes since Euro-American settlement and ecosystem restoration in the Lake Tahoe Basin, USA. *In* Gen. Tech. Rep. PSW-GTR-203: Restoring fire-adapted ecosystems: proceedings of the 2005 national silviculture workshop. *Edited by* R.F. Powers. USDA Forest Service, Pacific Southwest Research Station. Albany, CA.

Taylor, A.H., and Solem, M.N. 2001. Fire regimes and stand dynamics in an upper montane forest landscape in the Southern Cascades, Caribou Wilderness, California. J. Torrey Bot. Soc. 128(4): 350–61. doi:10.2307/3088667.

Tempel, D.J., Gutiérrez, R.J., Battles, J.J., Fry, D.L., Su, Y., Guo, Q., Reetz, M.J. 2015. Evaluating short- and long-term impacts of fuels treatments and simulated wildfire on an old-forest species. Ecosphere **6**(12): 1–18. doi:10.1890/ES15-00234.1.

Thompson, C.M., Zielinski, W.J., and Purcell, K.L. 2011. Evaluating Management Risks Using Landscape Trajectory Analysis: A Case Study of California Fisher. J. Wildlife Manage. **75**(5): 1164–76. doi:10.1002/jwmg.159.

USDA Forest Service. 2004. Sierra Nevada Forest Plan Amendment: final supplemental environmental impact. USDA Forest Service, Pacific Southwest Region. Vallejo, CA.

USDA Forest Service. 2007. Sierra Nevada Forests Management Indicator Species Amendment: Record of Decision. USDA Forest Service, Pacific Southwest Region. Vallejo, CA.

Van de Water, K.M., and Safford, H.D.. 2011. A Summary of Fire Frequency Estimates for California Vegetation before Euro-American Settlement. Fire Ecology **7**(3): 26–58. doi:10.4996/fireecology.0703026.

Van Wagtendonk, J.W. 2007. The history and evolution of wildland fire Use. Fire Ecology **3**(2): 3–17.

Van Wagtendonk, J.W., and Fites-Kaufman, J.F. 2006. Sierra Nevada Bioregion. Book chapter. *In* Fire in California's Ecosystems. *Edited by* Sugihara, N.G., Van Wagtendonk, J.W., Shaffer, K.E, Fites-Kaufman, J., and Thode, A.E. University of California Press, Berkeley and Los Angeles, CA.

Westerling, A.L.. Hidalgo, H.G., Cayan, D.R., Swetnam, T.W. 2006. Warming and earlier spring increase western U.S. forest wildfire activity. Science **313**(5789): 940–43. doi:10.1126/science.1128834.

Zielinski, W.J., Truex, R.L., Schlexer, F.V., Campbell, L.A., and Carroll, C. 2005. Historical and contemporary distributions of carnivores in forests of the Sierra Nevada, California, USA. J. Biogeography **32**(8): 1385–1407. doi:10.1111/j.1365-2699.2005.01234.x.

2 | The effects of marten habitat reservation on fuel treatment efficacy at the project scale

2.1 Introduction

Forest managers in much of the western United States are tasked with restoring resilience to forests with excessive fuel accumulation and elevated fire hazard. Mechanical treatments and prescribed fire are two management tools commonly used to achieve this goal (North et al. 2009, Agee & Skinner 2005). Fuel reduction treatments can be effective at modifying fire behavior within a managed area (Stephens et al. 2012, Kalies & Kent 2016) and over greater spatial extents, such as watersheds (Moghaddas 2010, Collins et al. 2011). Unfortunately, implementation of fuel treatment projects is often limited by budgetary and infrastructure considerations, or by conflicting management objectives that may preclude treatment in some areas (Reinhardt 2008 et al., Barros et al. 2017, Calkin & Gebert 2006).

Conservation of Pacific marten (*Martes caurina*) is one objective that may be in conflict with preferable fuel treatment strategies in higher elevation forests. The Pacific marten, a small forest carnivore in the Mustelid family, is designated a vulnerable species by the state of California (CA Dept. of Fish and Wildlife 2017) and was designated a Management Indicator Species by the United States Department of Agriculture Forest Service (hereafter Forest Service) for closed-canopy forests in ten National Forests in the state (USDA Forest Service 2007). Pacific marten populations in the southern Cascade and northern Sierra Nevada Mountain Ranges are more fragmented than elsewhere in this species' range (Spencer & Rustigian-Romsos 2012, Kirk & Zielinski 2010). Martens avoid simplified forest stands created by standard fuel reduction treatments, suggesting that such treatments can have a negative effect on marten fitness by reducing habitat connectivity (Moriarty et al. 2015, 2016). This evidence, and analysis of the closely related American marten (*M. americana*) elsewhere in North America, suggest marten populations may decline quickly in response to fragmentation of their habitat (Chapin et al. 1998, Hargis et al. 1999, Moriarty et al. 2011).

The need to reconcile the goal of restoring fire resilience with budgetary considerations and other forest management objectives like marten conservation has motivated the development of tools to assess the efficiency of fuel treatment projects (Finney 2005, Scott 2006, Ager et al. 2006). For instance, potential fire behavior in treated landscapes can be compared by simulating thousands of individual wildfire ignitions and summarizing their outcomes (Moghaddas et al. 2010, Collins 2011). The application of risk analysis, a method for concisely describing the potential for highly stochastic events and their associated costs or benefits, provides the ability to relate simulated fire intensity to management assets like species' habitats (Ager et al. 2007, Miller & Ager 2013).

A crucial question for forest managers in the northern Sierra Nevada is whether reserving marten habitat from fuel reduction treatment represents a sacrifice in terms of the ability of such treatments to regulate fire behavior across a landscape. Also of interest is whether prohibiting treatment in marten habitat exposes these areas themselves to higher wildfire risk. This study used probabilistic simulation to compare the potential for large fires on simulated landscapes where fuel reduction treatments were allocated with and without marten habitat reserves. Fuel treatments were simulated on three watersheds (mean size: 7,864 ha) with treatment location identified by an optimization procedure designed to minimize fire spread rate. A series of random ignitions were then simulated under a suite of weather scenarios based on prevailing local conditions during the fire season. Conditional burn probability, or the probability of a location burning given an ignition in the landscape, was averaged over each watershed to provide a measure of the capacity of the simulated landscape to support large fire growth. A second objective was to examine expected habitat loss, a measurement of wildfire risk that integrates the context of the surrounding landscape, in order to compare the risk to marten habitat of destruction by fire under each scenario.

2.2 Methods

2.2.1 Study location

Three sixth-level, USGS hydrological units (hereafter watersheds) were chosen for analysis based on the location of previous marten monitoring efforts, a majority of Forest Service ownership, and current management interest (Figure 2.1, Table 2.1). The watersheds are all located within Almanor Ranger District, Lassen National Forest, California. These units were chosen for analysis in order to capture patterns of fire behavior on a landscape of which a substantial portion could be treated in a short period as part of a single management project. The "fireshed", a contiguous area 10,000 – 50,000 ha in area that shares common fire risk factors and over which coordinated suppression effort may be possible, has been used as a planning unit for fire management activities in this region (Ager et al. 2006, Bahro et al. 2007). While the units in this analysis are smaller than this fireshed designation, this study follows the example of North et al. (2015) who argued these sixth-level watersheds represent a reasonable approximation of the fireshed concept, and here serve as a compromise between landscape-scale analysis and the real-world constraints on fuel treatment implementation.

2.2.2 Study design

A factorial arrangement of fuel treatment scenarios was simulated with three possible amounts of treatment area, 10%, 20%, and 30% of the total watershed area, and three possible amounts of area reserved. Possible areas reserved included (1) none; (2) partial, where 20% of the watershed area with the highest probability of occupancy was reserved; and (3) complete, where all predicted marten habitat was reserved. Thus, I simulated a total of nine scenarios per watershed. In each scenario, stands were considered eligible for treatment if they were a) forested with greater than 40% canopy cover, b) owned by the Forest Service, c) outside riparian areas, and d) outside the reserve area designated for that scenario.

2.2.3 Vegetation mapping

The LANDFIRE database provided the basic raster data needed for fire modeling, at 30m resolution (Table 2.2, LANDFIRE 2014). The raw LANDFIRE data was resampled to 90m resolution in order to reduce processing time for fire simulation and risk analysis. Because such pixels do not represent meaningful units for management activity, and treating an assortment of pixels scattered across the landscape would not be feasible, it was also necessary to group pixels into stand units that would be assigned a simulated treatment together. Each watershed was delineated into stands using mapping data created for Lassen National Forest by Ward Associates and VESTRA, and supplemented by data from CALVEG, a Landsat-based map product produced by the Forest Service (USDA Forest Service 2014). Stands less than 2.02 ha (5 acres) were merged with adjacent stands to produce a map of areas that were a reasonable size for management activity. A marten habitat suitability model (Rustigian-Romsos & Spencer 2010, Zielinski et al. 2015) was used to rank stands by the estimated probability of marten occupancy in order to identify stands that would be reserved from treatment in each scenario. Stands were labeled as “marten habitat” for the complete reserve scenarios if the probability of occupancy exceeded 50% (Rustigian-Romsos & Spencer 2010).

2.2.4 Treatment Simulation

Fuel reduction treatments were simulated by modifying the values of pixels within the treated stand boundary based on a set of modifications designed to approximate the effect of a thinning from below followed by a prescribed burn, a typical fuel treatment methodology (Table 2.3). These modifications were based on silvicultural prescriptions described in a current U.S. Forest Service project and simulation of their effects on the fuel parameters needed for fire modeling, using the Forest Vegetation Simulator (Dixon 2002) and plot inventory data from the project area (USDA Forest Service 2015). This process generated a simplistic lookup table that dictated how simulated treatments would modify the initial pixel values (Table 2.3).

Stands were assigned treatment using the Treatment Optimization Method (TOM), an algorithm decided to select treatment locations that will have the greatest effect on controlling the spread of large fires, given anticipated fire weather conditions and a proportion of area available for treatment (Finney 2006). This algorithm selects optimal treatment locations by iterating through parallel rows of cells that are perpendicular to the prevailing wind direction, identifying locations where treatment would result in the greatest reduction in fire spread rate (Finney 2006). Weather data was pooled from the three nearest Remote Automated Weather Stations (Table 2.4) to calculate 97th percentile wind speed for use with the TOM algorithm (23 mph). The wind direction was set to the dominant wind direction during fire season (225 degrees). Fuel moistures for live and dead fuels were set to the “very dry” scenario described in

Scott & Burgan (2005) in order to optimize treatment placement under extreme fire weather conditions. Because TOM assigns individual pixels to treatment and not stands, an additional step was needed to translate the optimization method's output into workable units from a management perspective. For each treatment allocation, treatment was simulated on stands that had the highest proportion of TOM-assigned pixels until the total treatment area needed was met.

2.2.5 Fire simulation

For each of the nine treatment scenarios, I executed the fire model four times in order to obtain estimates of variability in the responses. For each model run I simulated 500 randomly-placed ignitions, with the probability of ignition assumed to be equal for each pixel in the landscape. Ignitions and subsequent fire spread were simulated using FConstMTT, a command-line version of the minimum travel time method for fire simulation (Finney 2002). I used a 2 km buffer around each watershed to allow for ignitions to move into the analysis area from portions of the exterior landscape. Wind speed for each ignition, and associated wind direction, were drawn randomly from the distribution of 75th percentile and higher wind speeds from the nearest Remote Automated Weather Stations stations, in order to capture variability in the conditions likely to support large fire growth (Figure 2.1, Table 2.4, Table 2.5). Ignitions were allowed to spread for 10 hours simulation time (approximately two burn days assuming an active burning period of 5 hours per day), in order to capture the capacity of each ignition to spread rapidly while maintaining a feasible processing time. Preliminary analysis showed that the larger fires produced by ignitions allowed to burn for this time period were roughly equivalent in size to the larger two day runs during the Chips Fire, a large fire that occurred in this region in 2012 (USDA Forest Service 2013).

I calculated three outputs from each model: (1) treated habitat area (ha), (2) average conditional burn probability, and (3) expected habitat loss (ha). Conditional burn probability describes the probability a pixel will burn for a given number of ignitions (in this case 500 ignitions for each model run). Because larger fires will burn more pixels, average conditional burn probability (hereafter burn probability) summarized across each watershed represents an index for the potential of each scenario to support large fire growth and rapid fire spread.

Expected habitat loss describes the risk to remaining marten habitat from wildfire and was calculated using the following formula adapted from Finney (2005):

$$Expected\ habitat\ loss = \sum_{i=1}^{6} \sum_{j=1}^{n} \left(p(F_{ij})\left[-L_{ij}\right]\right)$$

where:

i	= indicator variable for the six possible flame length values: 1 = 0-2 ft, 2 = 2-4 ft, 3 = 4-6 ft, 4 = 6-8 ft., 5 = 8-12ft, 6 = 12+ft
n	= total number of pixels within the remaining habitat.
j	= indicator variable referring to each pixel in the remaining habitat.
$p(F_{ij})$	= Probability of the ith flame length in the jth pixel
L_{ij}	= loss to the nth value from the ith fire behavior; here the area of each pixel or 0.81 ha.

Expected habitat loss can be interpreted as the amount of habitat we expect to be removed by wildfire given a random ignition within the watershed under the modeled of weather conditions. Expected habitat loss is equal to the sum of the probability of habitat loss for each pixel times the pixel area. Habitat was considered “lost” if the midpoint of the flame length category exceeded the canopy base height of the forest in that pixel, under the assumption that martens are associated with high levels of canopy cover, and flame lengths above the canopy base height likely results in crown fire and subsequent removal of that canopy cover. While risk analysis of this type typically involves a cost-benefit calculation, here only the potential for wildfire to result in habitat loss is included. While it is reasonable to assume that fire could enrich or create marten habitat in the long term by recruiting structures like snags and downed logs or increasing small mammal abundance (e.g. Fontaine & Kennedy 2012), such considerations are outside the scope of this analysis.

2.2.6 Statistical analysis

Preliminary analysis indicated that the amount of reserves and the amount of area treated did not act independently, and the nature of their interactions depended on the watershed where fuel treatment simulation was performed. Therefore, I used the following statistical model, which included parameters for each of the treatment and reserve categories as well as all interactions, for analysis. This model was applied separately for each watershed and each response in order to test for differences between fuel treatment strategies with and without reserves:

$$Y_{response} = \alpha_0 + \alpha_1 I.t20 + \alpha_2 I.t30 + \alpha_3 I.rP + \alpha_4 I.rC + \alpha_5 I.t20I.rP + \alpha_6 I.t20I.rC + \alpha_7 I.t30I.rP + \alpha_8 I.t20I.r30 + \varepsilon_t$$

where:

Y_t	= response for the t[th] model run, where t = 1 – 36 (4 runs * 9 scenarios).
α_0	= mean response with 10% fuel treatment and no reserves.
α_1	= incremental effect of fuel treatment on 20% of the watershed.
α_2	= incremental effect of fuel treatment on 30% of the watershed.
α_3	= incremental effect of partial reserves of the watershed.
α_4	= incremental effect of complete reserves of the watershed.
α_5	= combined effect of treating 20% of the watershed and partial reserves.
α_6	= combined effect of treating 20% of the watershed and complete reserves
α_7	= combined effect of treating 30% of the watershed and partial reserves.
α_8	= combined effect of treating 30% of the watershed and complete reserves
I.t20	= 1 when 20% treated, 0 otherwise.
I.t30	= 1 when 30% treated, 0 otherwise.
I.rN	= 1 when no reserves, 0 otherwise.
I.rP	= 1 when partial reserves, 0 otherwise.
I.rC	= 1 when complete reserves, 0 otherwise.
ε_t	= random error associated with the t^{th} observation, where $\varepsilon t \sim N(0, \sigma^2)$ and εt and $\varepsilon t+1$ are independent.

I reduced the likelihood of making false conclusions with multiple comparisons by using the Benjamini-Hochberg procedure and assumed significance at 5% likelihood (Benjamini & Hochberg 1995).

2.3 Results

2.3.1 Marten habitat treated

The placement of treatments relative to marten habitat varied between the three watersheds using the Treatment Optimization Method (TOM, Figure 2.2). In Colby Creek (Figure 2.3), prioritized treatment locations were primarily located in lower elevation areas on the western, windward side of the watershed. This spatial arrangement only included a small amount of predicted marten habitat (hereafter marten habitat), which is concentrated on the leeward, northeastern side of the watershed. Given this result, we might expect habitat reserves to have a limited effect in the vicinity of Colby Creek, as the TOM algorithm indicated that the majority of preferred fuel treatment locations are not within marten habitat. Alternatively, in Lower Yellow Creek (Figure 2.4) and Chips Creek (Figure 2.5), marten habitat made up a substantial portion of the preferred fuel treatment area according to the TOM algorithm. When fuel treatment was permitted within marten habitat and treatment was simulated on 10% of the total watershed area, 8 of that 10% of the watershed consisted of areas with a high probability of marten occupancy (Table 2). In these two watersheds, the optimization procedure indicated allowing fuel treatment in marten habitat would be advantageous for controlling the spread of large fires.

2.3.2 Burn probability

Restricting fuel reduction treatments in marten habitat had no substantial negative effect on burn probability across nearly all treatment scenarios and watersheds (Table 2.6, Figure 2.6). I observed a significant increase in burn probability, an estimated increase of 1.5%, in only one scenario: Chips Creeks, when 30% of the watershed was set for fuel treatment and complete reserves prohibited treatment inside any predicted marten habitat. Because of the high proportion of predicted marten habitat in Chips Creek, this was also the only case where the designated treatable area was exhausted before the desired amount was met— only 18% of the watershed area was available for treatment, so I was unable to simulate the full 30% treatment outside of reserves.

Surprisingly, four reserve scenarios resulted in statistically significant decreases in burn probability in Colby Creek (Figure 2.6), suggesting that treatments performed better in this unit when they were forced outside of designated marten habitat. This finding indicates the TOM algorithm for optimizing fuel treatment location does not necessarily result in the ideal allocation of treatments for the full suite potential weather conditions.

2.3.3 Expected habitat loss

I estimated substantial increases in expected habitat loss for three scenarios: those in Chips Creek with complete reserves, where fuel treatment was prohibited in all predicted marten habitat (Table 2.7, Figure 2.7). Estimated increases in expected habitat loss were 19.7 ha, 21.4 ha, and 27.4 ha respectively for each treatment level (10%, 20%, and 30% of the watershed). Thus, implementing complete reserves significantly elevated the risk of habitat removal due to wildfire compared to scenarios without this constraint on treatment location. Less pronounced but statistically significant increases were also observed for the complete reserve scenarios in Lower Yellow Creek (Figure 2.7). Reserves appeared to have no effect on expected habitat loss in Colby Creek, though results were more variable in that watershed than elsewhere (Figure 2.7). Partial reserves resulted in smaller, but significant, increases in expected habitat loss in only two scenarios: Chips Creek when 10% of the watershed was treated and Lower Yellow Creek when 30% of the watershed was treated.

2.4 Discussion

The effects of prohibiting fuel reduction treatments inside predicted marten habitat depended on the characteristics of each individual watershed, and may not be easily generalized at the watershed scale. Specifically, the location of predicted marten habitat relative to the prevailing wind direction and other forested area in the watershed appeared responsible for distinctive behavior predicted in this study. This finding underlines the importance of tailoring fuel and fire management efforts to individual areas that have different fuel configurations, weather, topography, and values at stake (Ager et al. 2006, Bahro et al. 2007).

The availability of treatable area had a strong influence on the effectiveness of fuel reduction treatments. Creating habitat reserves led to a significant increase in burn probability in only one scenario: 30% treatment area in Chips Creeks. That scenario was also the only one in which treatable area was exhausted well before the target treatment amount was reached (only 18% of the watershed was treatable, of the desired 30% for this scenario). A large proportion of the watershed either fell inside the reserves or lacked the 40% minimum canopy cover required for fuel treatment; this area had a prior wildfire in 2012. Thus, with few exceptions, these simulations indicate treating marten habitat was not crucial to controlling large wildfire growth as long as other treatable area was available. Treating areas only outside predicted marten habitat resulted in equivalent reductions in burn probability.

Broadly, my study suggests prioritizing fuel treatments in lower elevation montane forests, where historically fires burned more frequently, can provide a viable balance between retaining marten habitat and reducing fire risk. Lower elevation forests are likely more departed from pre-settlement fuel conditions given their shorter fire return intervals (Hardy et al. 2001, Hann 2004, Keane et al. 2006). Conversely, fuel accumulation in the higher elevation forests, dominated by true fir species and occupied by Pacific martens, may be less severe (Meyer 2013). My simulations suggest ignitions in these true fir forests behaved similar to historical models, where surface fire spread was limited by a more compact fuel bed (Skinner & Chang 1996, Bekker & Taylor 2001, Taylor 2000), and thus treatment in such areas is less crucial. Nonetheless, the potential for large fire growth as simulated represents a snapshot in time: I analyzed how potential fuels treatments interrupt fire behavior under current vegetation conditions. In time, without management intervention or fire, continued fuel accumulation in these high elevation forests will increase the potential for large fire growth (Chappell and Agee 1996).

Forest managers have substantially more constraints on potential fuel treatment locations than the relatively simplistic minimum canopy cover and non-riparian designations used in these simulations (North et al. 2015, Barros et al. 2017). As additional constraints on treatment placement are added and less treatable area is available, the potential negative effects of limiting management in marten habitat will increase, as I experienced in Chips Creek when available area for treatment was exhausted. While treating as much as 30% of a landscape area may not be necessary to achieve meaningful reductions in burn probability (Moghaddas et al. 2010, Collins et al. 2011, Finney 2007), ultimately the effect of habitat reserves may depend on their integration into a larger system of constraints on management activity. For instance, in this area the mandate to conserve Pacific marten is accompanied by similar protections for the California spotted owl and northern goshawk, as well as more standard restrictions related to economics and logistics (USDA Forest Service 2004).

Decreases in burn probability in Colby Creek suggested marten reserves actually improved the allocation of fuel treatments in these simulated landscapes, though creating reserves clearly limits the options for fuel treatment placement. Clearly, the Treatment Optimization Method did not identify the ideal fuel treatment design before reserves were created; a number of factors may be responsible for this counterintuitive result. First, though weather conditions vary throughout the fire season, the Treatment Optimization Method selects treatment locations using only one set of likely weather conditions (Finney 2006). In contrast, my fire simulations used a larger suite of possible fire weather scenarios (Table 2.5). Second, applying the TOM results to a real-world landscape required an intermediate step of translating the pixel output of the algorithm to the forest stands mapped in the study area. I observed more variable results in Colby Creek and, although the cause is not clear, my results nevertheless provide no evidence that allowing treatment in predicted marten habitat was essential to controlling fire spread. Instead these findings emphasize managers should be cautious when applying optimization algorithms to landscapes in the real world; information derived from such tools would be best applied using local knowledge of forest structure and observed fire behavior.

In specific watersheds, reductions in wildfire risk to marten habitat were possible by managing fuels outside these areas, emphasizing the benefits of landscape level planning and location-specific treatment designs. For instance, in Colby Creek fuel treatments effectively reduced the threat of wildfire to marten habitat areas without active management within them, while in Lower Yellow Creek wildfire risk to marten habitat was not reduced unless treatment was allowed in areas used by martens. Generally, treating areas upwind and adjacent to high value habitat can achieve effective reductions of wildfire risk without disruptive incursion into these habitat areas (Ager et al. 2007). If treatable non-habitat is not available in the direction of the prevailing wind, fuel reduction in marginal predicted habitat may be justified where crown fire is a major concern. This approach is further supported by the notable difference in wildfire risk between scenarios with partial and complete reserves (Figure 2.7). Allowing treatment in marginal predicted habitat (outside the highest-rated 20% of each watershed) led to significant reductions in wildfire risk compared to complete prohibition of treatment in any predicted habitat.

My simulations revealed little justification for treating the highest quality predicted marten habitat— prohibiting treatment in these areas did not represent a sacrifice in terms of overall burn probability or wildfire risk to habitat (Figures 2.6 and 2.7). Partial reserves performed as well as no reserves except in two instances: restricting treatment in the highest rated habitat resulted in slightly higher expected habitat loss in both Chips Creek at 10% treatment area and Lower Yellow Creek at 30% treatment area. As above, the effects of prohibiting fuel treatment in marten habitat reserves may become more significant as marten conservation is integrated into a larger system of management constraints. However, the ability to restrict treatment on as much as 20% of the watersheds in these simulations with little impact on burn probability suggests fuel reduction treatments can successfully modify fire behavior in this landscape while retaining large patches of high elevation forests associated with marten use.

While analysis at this scale may be appropriate for consideration of real-world project implementation (North et al. 2015), analyzing treatment allocation over sixth-level hydrological units such as these may create bias by removing the analysis area from its landscape context. While I used 2 km buffer to allow fires to enter and exit each watershed from the exterior, fuel reduction treatments on the leeward side of the analysis area still may be undervalued, as fires there are limited by the small buffer of adjacent burnable area, regardless of whether treatment is present (Collins et al. 2011). At the same time, the buffer also cannot fully capture the effects of ignition sources outside the analysis area and potential fires migrating in (Ager et al. 2007). Though these issues will always be present in simulation frameworks that necessarily isolate portions of the landscape from their surroundings, a larger-scale analysis that encompasses multiple watersheds could provide a more holistic view of how prohibiting treatment in reserves for forest dependent species can affect the overall efficiency of landscape fuel reduction efforts. As survival of this marten population likely depends on the larger landscape context beyond a single watershed unit, and perhaps also connectivity to other populations in the region, it would be appropriate to also adopt a wider view of the interplay between active fuel management and reserve creation.

2.5 Conclusions

Prohibiting all fuel reduction treatments in predicted marten habitat had little impact on the occurrence of large fires in these simulations, providing evidence that marten habitat should not be prioritized for such treatments above other portions of the landscape. However, this result was only consistent when reserves did not reduce the total treatable area below the desired level. As marten conservation is balanced with other management objectives and constraints, the impact of reserves will depend on where conservation of Pacific marten habitat is ranked in relation to other priorities.

Prohibition of fuel reduction treatments in all predicted habitat may not be desirable for long term marten conservation, where high-quality predicted habitat is plentiful and fuel accumulation is extreme. Allocating fuel reduction treatments to marginal areas, particular those to the windward side of core areas, may be beneficial in reducing the risk of habitat loss from intense fire. Strategically selecting areas for fuel reduction may also become more desirable from a conservation standpoint if silvicultural prescriptions can balance their ability to moderate fire behavior with the retention of old-forest structures and sufficient canopy cover to promote marten use within treated zones. Ultimately, the decision to allow active management in these areas may depend on weighing the anticipated marten use of treated stands versus those affected by wildfire.

Given the documented short-term negative effect of fuel treatments on Pacific marten (Moriarty et al. 2015, 2016), fuel reduction treatments in the highest-quality predicted habitat (up to 20% of the landscape) should be avoided— my simulations showed overall burn probability and wildfire risk to habitat can currently be controlled to the same degree without treatment in these areas as when they are made available for active management. However, in the long term restricting fuel treatment to outside all predicted marten habitat may not be ideal; my simulations show wholesale restrictions of that magnitude would lead to increased risk of habitat loss from fire. With continued fire suppression, such restriction could lead to a more homogeneous forested landscape in contrast to the diverse landscapes to which Pacific marten are adapted. Related simulation studies have shown that increased homogeneity at the landscape scale can have a negative effect on forest-dependent species (Roloff et al. 2012, Tempel et al. 2015, Thompson et al. 2011). Retention of old forest structures like snags and downed logs, high levels of canopy copy, and large contiguous patches of predicted habitat are strategies that can minimize the negative impacts of fuel treatments to Pacific marten (Chapin et al. 1998, Hargis et al. 1999, Moriarty et al. 2011).

2.6 Chapter 2 | Figures and Tables

Table 2.1: Names and descriptions of three watersheds in Almanor Ranger District, Lassen National Forest, California.

Name	Area (ha)	Avg. elevation (m)	Elevation range (m)	USFS ownership
Colby Creek	7,736	1,830	1,484 – 2,187	96.3%
Chips Creek	4,595	1,637	675 – 2,163	97.8%
Lower Yellow Creek	9,818	1,670	675 – 2,172	94.5%

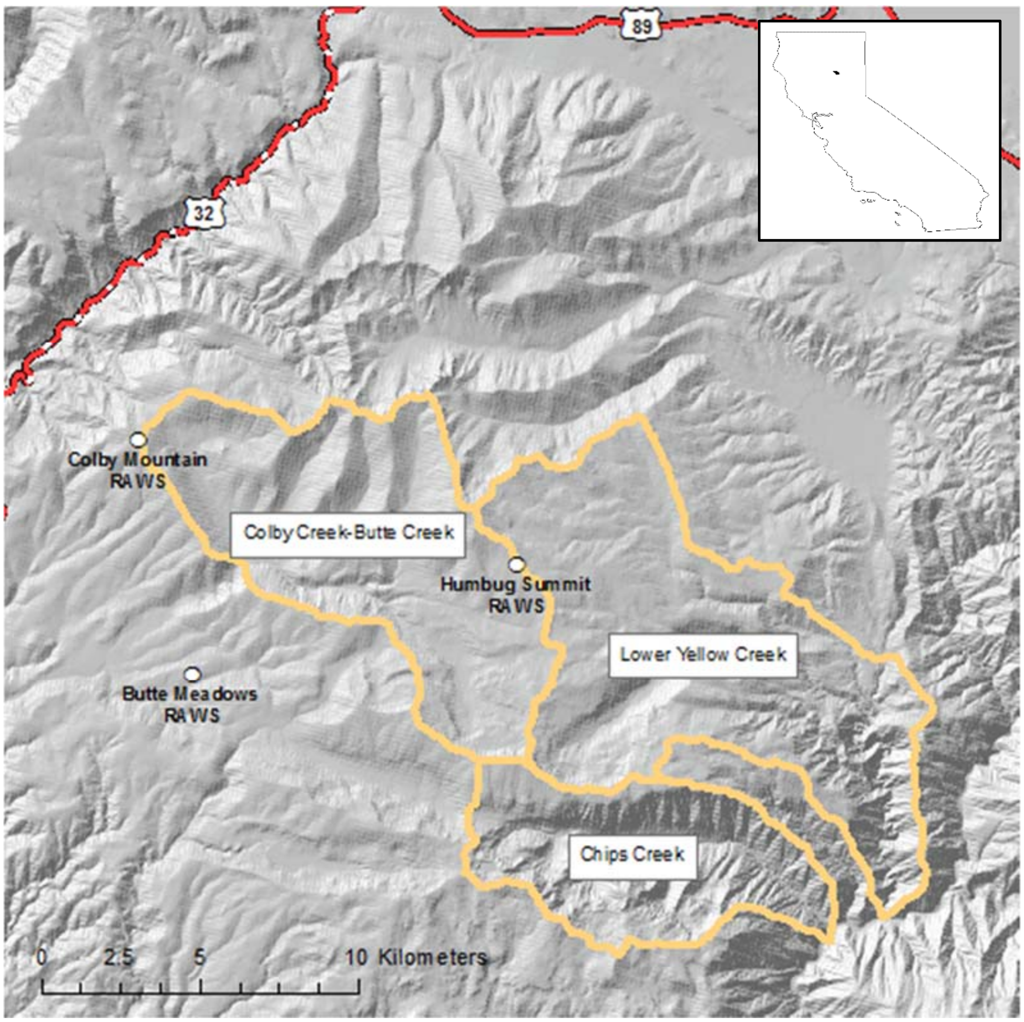


Figure 2.1: The study area, consisting of three sixth-level watersheds in Almanor Ranger District, Lassen National Forest, California. Inset shows the location of the study in northeastern California.

Table 2.2: Fire modeling inputs and their units.

Variable	Unit
Elevation	Meters
Slope	Degrees
Aspect	Azimuth degrees
Fuel model	Scott & Burgan (2005)
Canopy cover	Percent
Canopy height	Meters
Canopy base height	Meters
Canopy bulk density	Kilograms per cubic meter

Table 2.3: Simulated treatment effects on fire modeling inputs.

Variable		Change when treated	Description
Fuel model:			Sets the stand's fuel model to Timber Litter 1, a fuel model characterized by a low spread rate and flame length.
	Stands < 60% CC:	Set to TL1	
	Stands > 60 % CC:	Set to TL1	
Canopy cover:			Reduces canopy cover, representing removal trees the treatment.
	Stands < 60% CC:	Set to 40%	
	Stands > 60% CC:	Set to 50%	
Canopy height:			According to the assumptions of FVS, a typical treatment has no effect on overall canopy height, as the dominant trees are not removed.
	Stands < 60% CC:	No change	
	Stands > 60% CC:	No change	
Canopy base height:			Fuel treatment raises the canopy base height by removing smaller seedlings and saplings in the lower and intermediate canopy layers.
	Stands < 60% CC:	+ 14 ft.	
	Stands > 60% CC:	+ 19 ft.	
Canopy bulk density:			Fuel treatment lowers the canopy bulk density by removing foliage mass from the stand.
	Stands < 60% CC:	- 0.015 kg/m^3	
	Stands > 60% CC:	- 0.039 kg/m^3	

Table 2.4: RAWS stations used to calculate wind speed and direction for fire simulations

Station	Elevation (m)	Years	Months
Butte Meadows	1,511	1998 - 1999	July - September
Humbug Summit	2,039	2013 - 2016	July - September
Colby Mountain	1,907	2015 - 2016	July - September

Table 2.5: 75th percentile and higher weather conditions used for fire simulation.

20 ft. wind speed (mph)	Wind direction (azimuth degrees)	Probability
15	225	.31
20	225	.18
15	270	.12
25	225	.11
10	225	.08
10	180	.06
15	180	.06
20	270	.03
30	225	.03
30	180	.02
10	270	.01
10	315	.01
15	45	.01
10	90	.01
10	45	.01

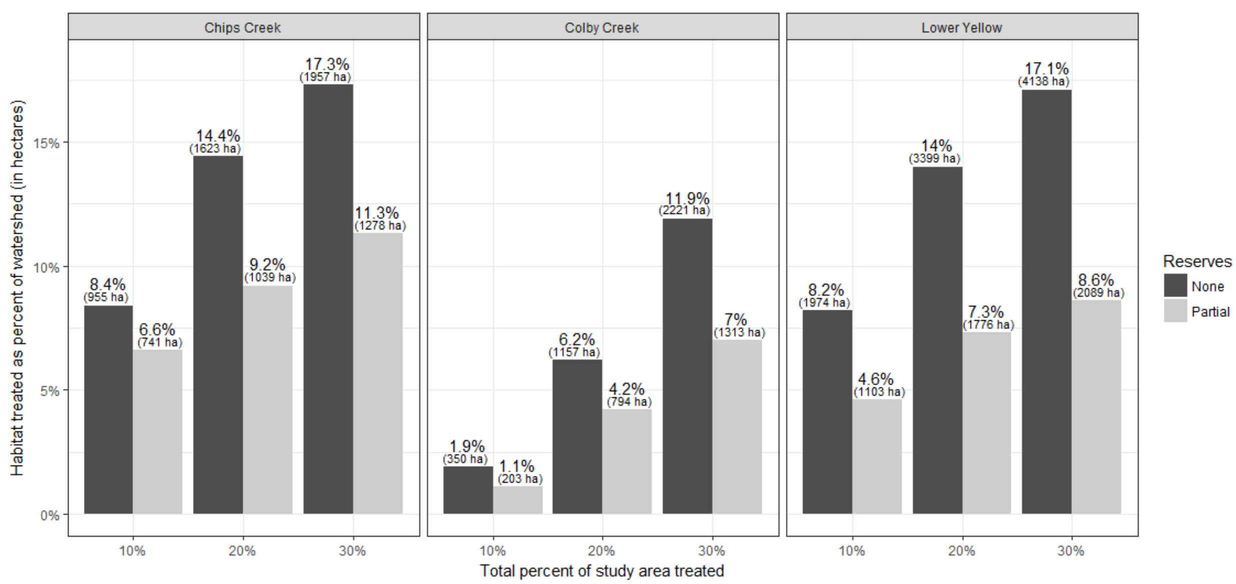
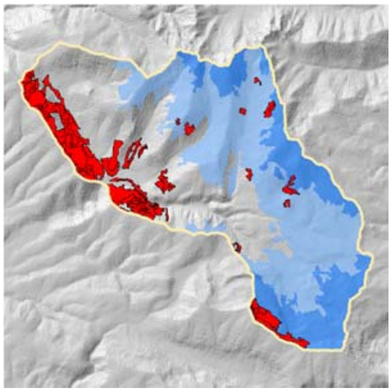
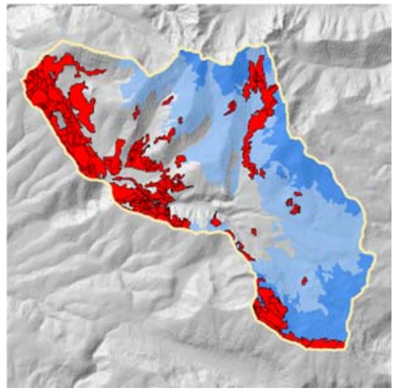


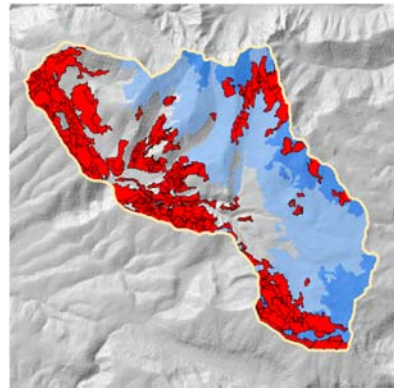
Figure 2.2: Marten habitat treated as percent of the total watershed area for the no reserves and partial reserves scenarios. No marten habitat was treated under the complete reserves scenarios. Numbers in parentheses indicate the area of habitat treated in hectares. In Lower Yellow Creek (see Figure 2.4) and Chips Creek (Figure 2.5), marten habitat makes up much of the preferred treatment area, 8 of the first 10% treated. Conversely, in Colby Creek (Figure 2.3) only 2 of the first 10% treated identified by TOM was designated marten habitat. The TOM algorithm assigned treatment to marten habitat to varying degrees among the watersheds.



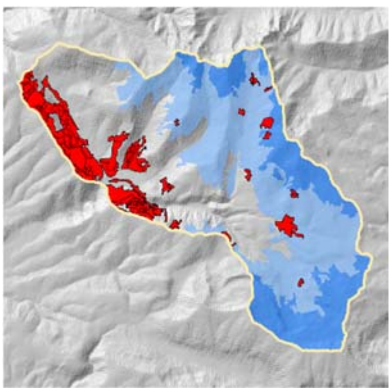
a. No reserves, 10% treated



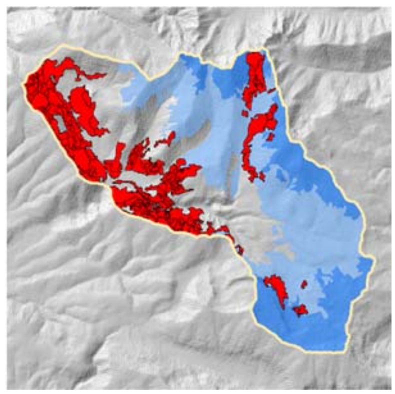
b. No reserves, 20% treated



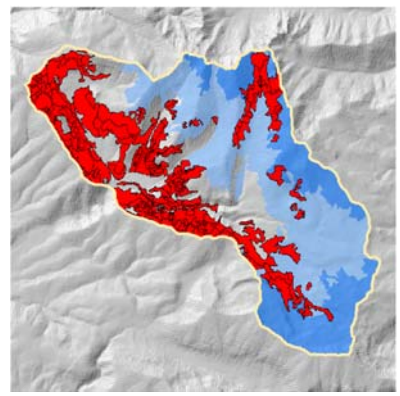
c. No reserves, 30% treated



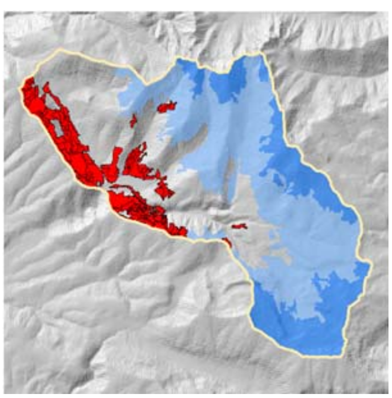
d. Partial reserves, 10% treated



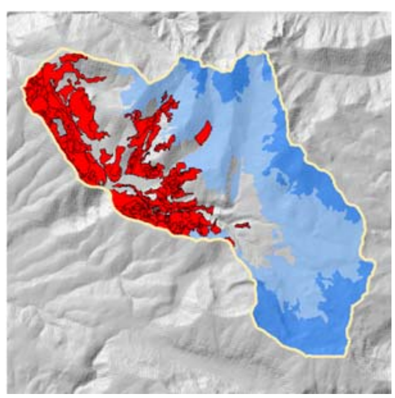
e. Partial reserves, 20% treated



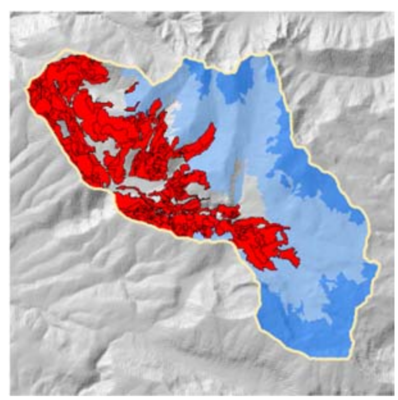
f. Partial reserves, 30% treated



g. Complete reserves, 10% treated

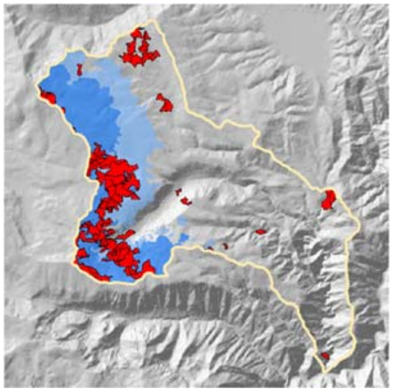


h. Complete reserves, 20% treated

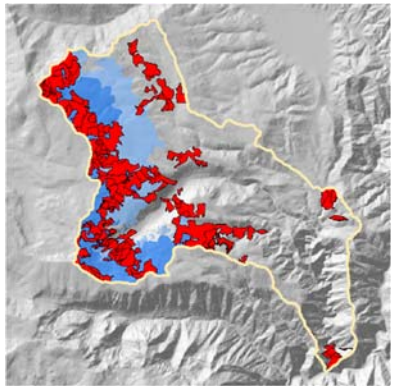


i. Complete reserves, 30% treated

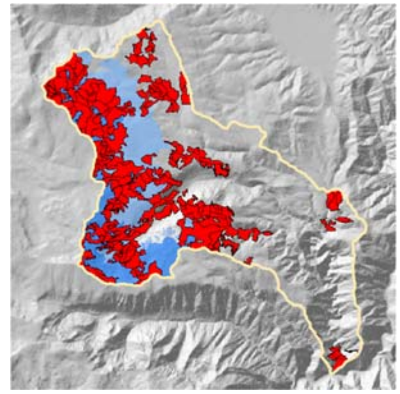
Figure 2.3: Simulated treatments and marten reserves in the Colby Creek watershed. Red indicates stand units treated under each scenario. The darker blue color indicates top 20%-rated marten habitat (areas reserved under the partial reserve scenarios) and the lighter blue indicates all potential marten habitat in the watershed.



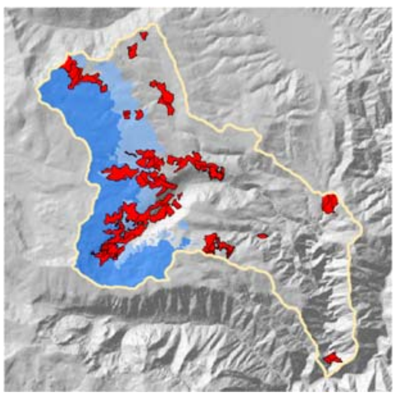
a. No reserves, 10% treated



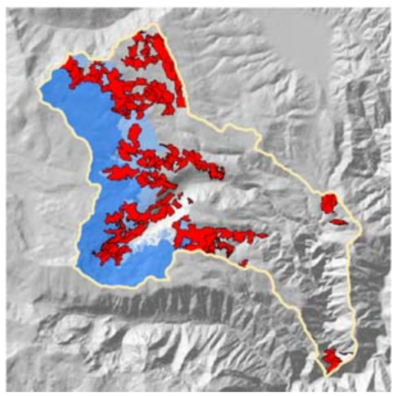
b. No reserves, 20% treated



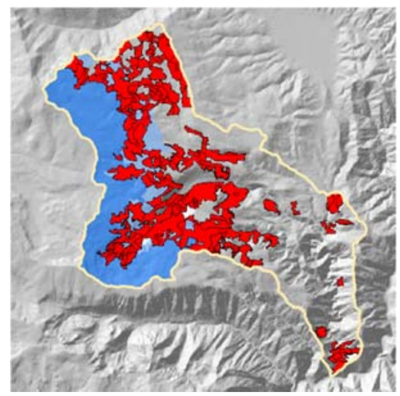
c. No reserves, 30% treated



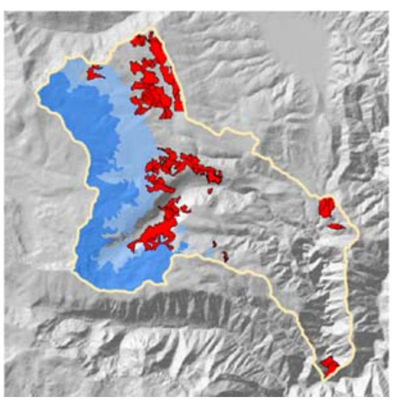
d. Partial reserves, 10% treated



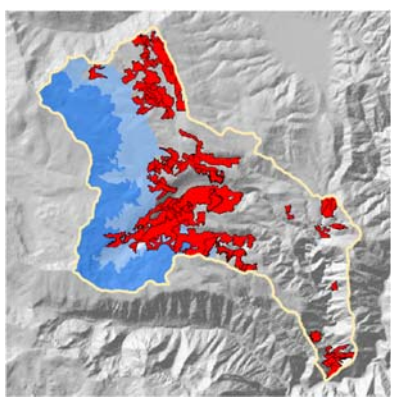
e. Partial reserves, 20% treated



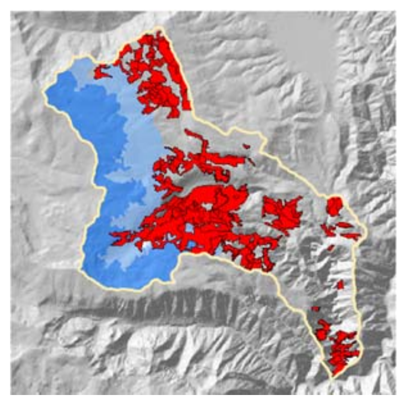
f. Partial reserves, 30% treated



g. Complete reserves, 10% treated

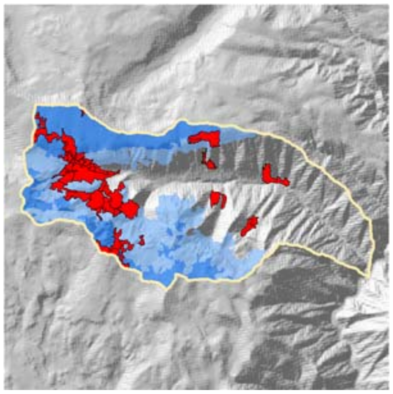


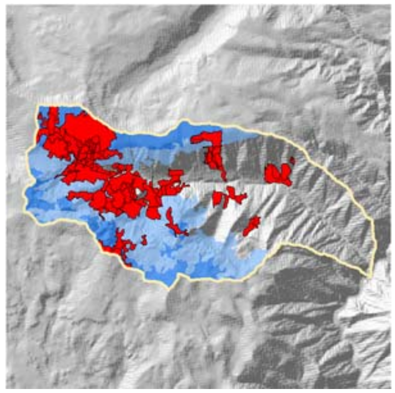
h. Complete reserves, 20% treated
treated

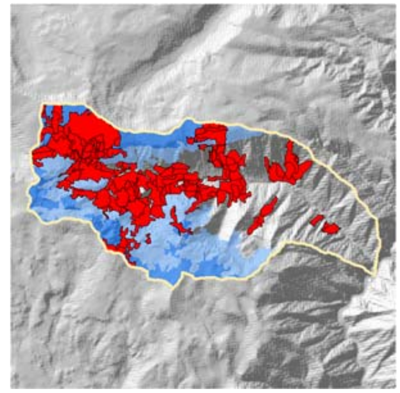


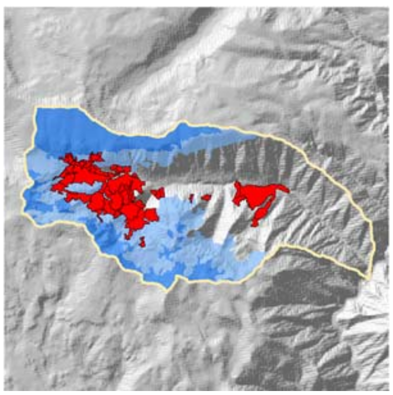
i. Complete reserves, 30%

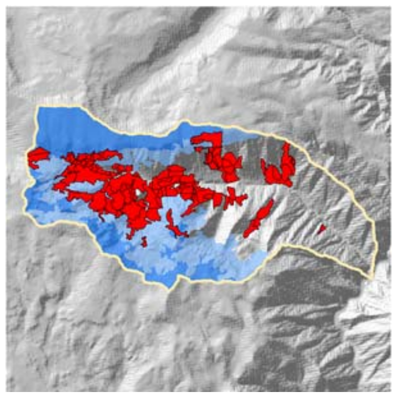
Figure 2.4: Simulated treatments and marten reserves in Lower Yellow Creek watershed. Red indicates stand units treated under each scenario. The darker blue color indicates top 20%-rated marten habitat (areas reserved under the partial reserve scenarios) and the lighter blue indicates all potential marten habitat in the watershed.

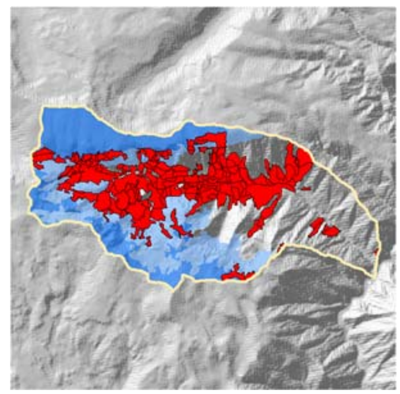

a. No reserves, 10% treated

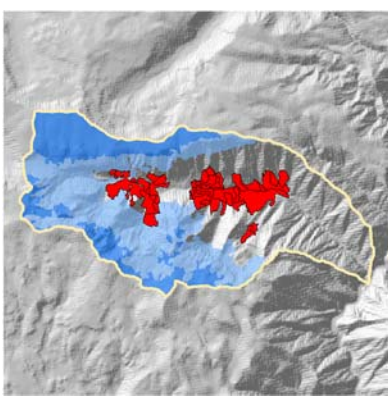

b. No reserves, 20% treated

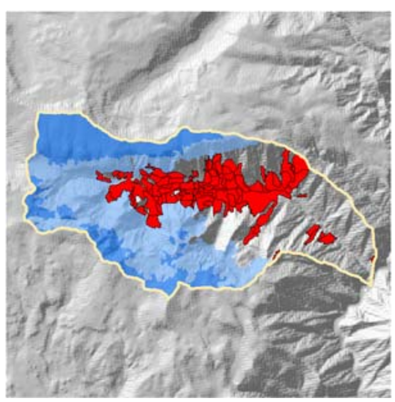

c. No reserves, 30% treated


d. Partial reserves, 10% treated


e. Partial reserves, 20% treated


f. Partial reserves, 30% treated


g. Complete reserves, 10% treated


h. Complete reserves, 20% treated*

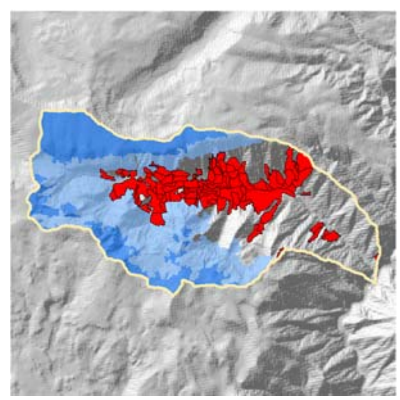

i. Complete reserves, 30% treated*

Figure 2.5: Simulated treatments and marten reserves in Chips Creek watershed. Red indicates stand units treated under each scenario. The darker blue color indicates top 20%-rated marten habitat (areas reserved under the partial reserve scenarios) and the lighter blue indicates all potential marten habitat in the watershed. Asterisks indicate scenarios where the target treatment area was not achieved: under the complete reserves scenarios, a maximum of 18% of the watershed was eligible for treatment (h, i).

Table 2.6: Estimated changes in average conditional burn probability for all treatment scenarios. Each scenario was compared to the equivalent scenario with the same treatment level and no reserves. Shading indicates a statistically significant difference. P-values shown have been adjusted to control the false discovery rate at 5%, following procedures outlined by Benjamini and Hochberg (1995), and were considered significant if less than 0.05.

Watershed	Treatment/reserve scenario	Estimate	Standard error	Adjusted p-value
Chips Creek	10% treatment, partial reserves	0.00014	0.00216	0.948
	20% treatment, partial reserves	-0.00117	0.00216	0.749
	30% treatment, partial reserves	0.00052	0.00216	0.859
	10% treatment, complete reserves	0.00177	0.00216	0.749
	20% treatment, complete reserves	0.00421	0.00216	0.157
	30% treatment, complete reserves	0.0153	0.00216	< . 001*
Colby Creek	10% treatment, partial reserves	0.00111	0.00155	0.749
	20% treatment, partial reserves	-0.00508	0.00155	0.013*
	30% treatment, partial reserves	-0.00462	0.00155	0.022*
	10% treatment, complete reserves	-0.00081	0.00155	0.749
	20% treatment, complete reserves	-0.0056	0.00155	0.007*
	30% treatment, complete reserves	-0.00562	0.00155	0.007*
Lower Yellow	10% treatment, partial reserves	-0.00196	0.00092	0.126
	20% treatment, partial reserves	-0.00037	0.00092	0.778
	30% treatment, partial reserves	-0.00128	0.00092	0.396
	10% treatment, complete reserves	-0.00046	0.00092	0.749
	20% treatment, complete reserves	0.00093	0.00092	0.638
	30% treatment, complete reserves	-0.00052	0.00092	0.749

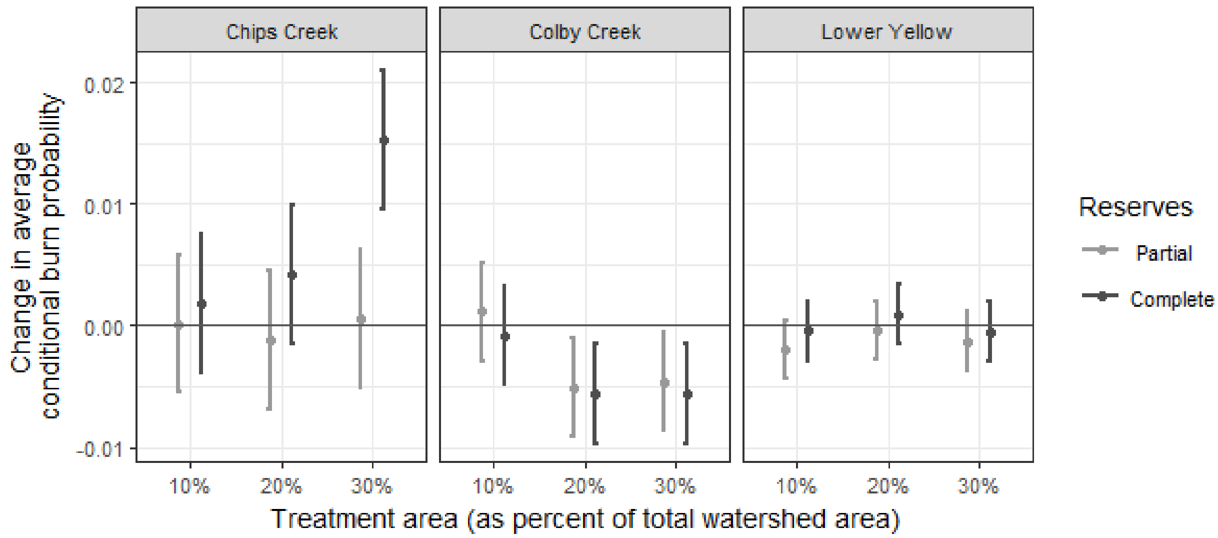


Figure 2.6: Changes in average conditional burn probability for treatment scenarios with partial and complete reserves (that prohibit fuel reduction treatment in some or all marten habitat). Each scenario was compared to the equivalent scenario with the same treatment level and no reserves. Estimates are indicated by the black dot and brackets indicate confidence 98.6% confidence intervals (the alpha level generated by performing the Benjamini-Hochberg procedure (1995) and following procedures for adjusting confidence intervals described in Benjamini and Yekutieli 2005). Complete reserves (where fuel treatment was restricted in all predicted marten habitat) only resulted in a significant increase in burn probability at 30% treatment in Chips Creek (left). Reductions in burn probability in Colby Creek (center) indicate treatments performed slightly better when moved out of predicted marten habitat.

Table 2.7: Estimated changes in expected marten habitat loss for all treatment scenarios, compared to the equivalent scenario with the same treatment level and no reserves. Shading indicates a statistical significant difference. P-values shown have been adjusted to control the false discovery rate at 5%, following procedures outlined by Benjamini and Hochberg (1995). In Chips Creek and Lower Yellow Creek, prohibiting all fuel treatment in marten habitat resulted in significant increases in expected habitat loss. Alternatively, partial reserves, where treatment was allowed in some marginal habitat areas, rarely resulted in increases in expected habitat loss. In Colby Creek, creating reserves counterintuitively improved treatment efficacy, which may have been due to the location of marten habitat in that watershed and/or the optimization method employed to identify treatment locations.

Watershed	Treatment/reserve scenario	Estimate (ha)	Standard error	Adjusted p-value
Chips Creek	10% treatment, partial reserves	6.83	1.76	0.002*
	20% treatment, partial reserves	2.42	1.76	0.203
	30% treatment, partial reserves	3.64	1.76	0.071
	10% treatment, complete reserves	19.37	1.76	<.001*
	20% treatment, complete reserves	21.15	1.76	<.001*
	30% treatment, complete reserves	27.23	6	<.001*
Colby Creek	10% treatment, partial reserves	3.90	4.31	0.395
	20% treatment, partial reserves	-10.60	4.31	0.041*
	30% treatment, partial reserves	-9.40	4.31	0.069
	10% treatment, complete reserves	-3.61	4.31	0.409
	20% treatment, complete reserves	-6.49	4.31	0.173
	30% treatment, complete reserves	-8.30	4.31	0.083
Lower Yellow	10% treatment, partial reserves	1.50	0.74	0.071
	20% treatment, partial reserves	1.51	0.74	0.071
	30% treatment, partial reserves	2.20	0.74	0.013*
	10% treatment, complete reserves	3.92	0.74	<.001*
	20% treatment, complete reserves	4.68	0.74	<.001*
	30% treatment, complete reserves	4.86	0.74	<.001*

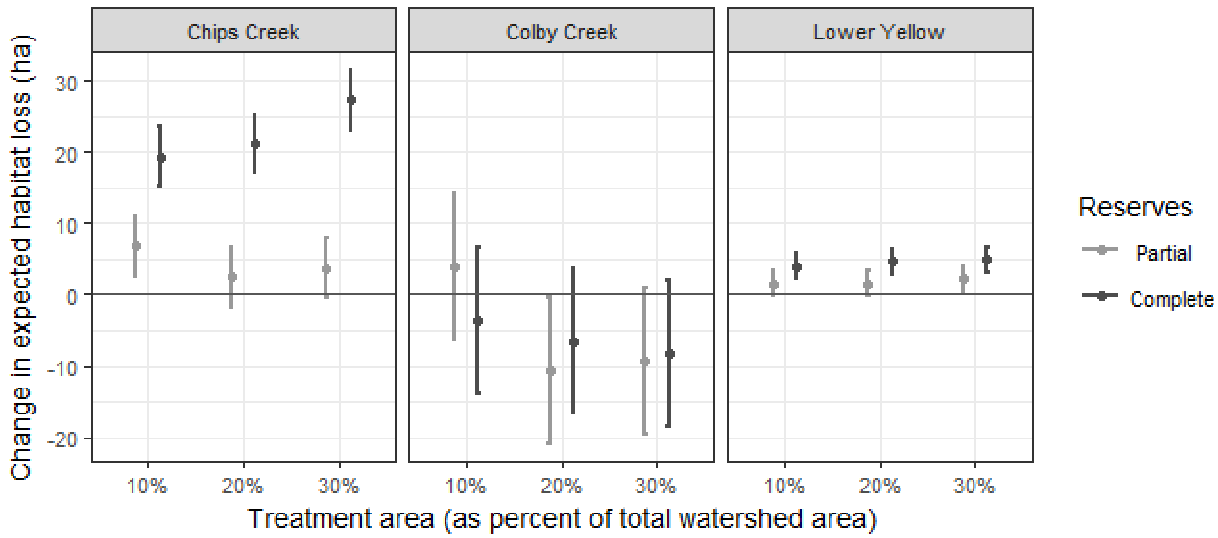


Figure 2.7: Change in expected habitat loss for treatment scenarios with reserves that prohibit fuel reduction treatment in some or all marten habitat, compared to no reserves, where treatment could occur anywhere in the watershed. Estimates are indicated by the black dot and brackets indicate confidence 97.5% confidence intervals (the alpha level generated by performing the Benjamini-Hochberg procedure (1995) and following procedures for adjusting confidence intervals described in Benjamini and Yekutieli 2005). In Chips Creek (left) and Lower Yellow (right), complete reserves resulted in significant increases in expected habitat loss, compared to the no reserves scenario where fuel treatment was allowed in predicted marten habitat. In Colby Creek (center), prohibiting treatment in predicated marten habitat had no significant positive effect on expected habitat loss, though there was more variability in this watershed than the other two.

2.7 Chapter 2 | Literature Cited

Agee, J.K., and Skinner, C.N. 2005. Basic principles of forest fuel reduction treatments. For. Ecol. Manag. **211**(1-2): 83–96. doi:10.1016/j.foreco.2005.01.034.

Ager, A.A., Bahro, B., and Barber, K. 2006. Automating the fireshed assessment process with ArcGIS. *In* Fuels Management-How to Measure Success, Portland, OR, 28-30 March 2006. *Edited by* Patricia L. Andrews, Bret W. Butler. Pp. 163-168.

Ager, A.A., Finney, M.A., Kerns, B.K., and Maffei, H. 2007. Modeling wildfire risk to northern spotted owl (Strix Occidentalis Caurina) Habitat in Central Oregon, USA. For. Ecol. Manag. **246**(1): 45–56. doi:10.1016/j.foreco.2007.03.070.

Bahro, B., Barber, K.H., Sherlock, J.W. 2007. Stewardship and fireshed assessment: A process for designing a landscape fuel treatment strategy. *In* Restoring fire-adapted ecosystems: 2005 national silviculture workshop. Gen. Tech. Rep. PSW-GTR-203. USDA Forest Service, Pacific Southwest Research Station, Albany, Ca. pp 41 – 54.

Barros, A.M.G., Ager, A.A., Day, M.A., Preisler, H.K., Spies, T.A., White, E., R.J., Pabst, Olsen, K.A., Plat,, E., Bailey, J.D., Bolte, J.P. 2017. Spatiotemporal dynamics of simulated wildfire, forest management, and forest succession in central Oregon, USA. Ecology and Society **22**(1):24. https://doi.org/10.5751/ES-08917-220124

Benjamini, Y and Hochberg, Y. 1995. Controlling the false discovery rate: A practical and powerful approach to multiple testing. J. of Royal Statistical Society: Series B (Methodological) **57**(1): 289-300.

Calkin, D., and Gebert, K. 2006. Modeling fuel treatment costs on Forest Service lands in the western United States. Western J. of Applied Forestry **21**(4): 217-221.

California Department of Fish and Wildlife. 2017. Natural Diversity Database: Special Animals List. Periodic publication. 51 pp.

Chapin, T.G., Harrison, D.J., and Katnik, D.D. 1998. Influence of landscape pattern on habitat use by American marten in an industrial forest. Conserv. Biol. **12**(6): 1327–37.

Chappell, C.B., and Agee, J.K., 1996. Fire severity and tree seedling establishment in Abies magnifica forests, Southern Cascades, Oregon. Ecol. Appl. **6**(2): 628–40. doi:10.2307/2269397.

Collins, B.M., Stephens, S.L., Roller, G.B., and Battles, J.J. 2011. Simulating fire and forest dynamics for a landscape fuel treatment project in the Sierra Nevada. Forest Sci. **57**(2): 77-88.

Collins, B.M., Stephens, S.L., Moghaddas, J.J., and Battles, J. Challenges and approaches in planning fuel treatments across fire-excluded forested landscapes. J. Forest. **108**(1): 24 – 31.

Dixon, G.E. 2002. Essential FVS: A user’s guide to the Forest Vegetation Simulator. Internal Rep. USDA Forest Service, Forest Management Service Center. Fort Collins, CO. (Revised: April 14, 2017).

Finney, M.A., 2002. Fire growth using minimum travel time methods. Canadian Journal of Forest Research: **32**(8): 1420-1424, 10.1139/x02-068

Finney, M.A. 2005. The challenge of quantitative risk analysis for wildland fire. For. Ecol. Manag. **211**: 97-108.

Finney, M.A. 2006. An Overview of FlamMap Modeling Capabilities. *In* Fuels Management- How to Measure Success: Conference Proceedings. *Edited by:* Andrews, P.L., and Butler, B.W. USDA Forest Service, Rocky Mountain Research Station. Fort Collins, CO.

Finney, M.A. 2007. A computational method for optimizing fuel treatment locations. International Journal of Wildland Fire **16**(6): 702-711.

Fontaine, J.B., and Kennedy, P.L. 2012. Meta-analysis of avian and small-mammal response to fire severity and fire surrogate treatments in U.S. fire-prone forests. Ecological Applications **22**: 1547-1561.

Hann, W.J. 2004. Mapping fire regime condition class: a method for watershed and project scale analysis. *In* Proceedings of the 22nd Tall Timbers Fire Ecology Conference: Fire in Temperature, Boreal, and Montane Ecosystems. *Edited by* R.T.Engstrom, K.E.M. Galley, and W.J. de Groot. Tall Timbers Research Station. Tallahassee, Fl.

Hardy, C.C., Schmidt, K.M., Menakis, J.P., and Sampson, R. Neil. 2001. Spatial data for national fire planning and fuel management. International Journal of Wildland Fire **10**: 353-372.

Hargis, C.D., Bissonette, J.A., and Turner, D.L. 1999. The influence of forest fragmentation and landscape pattern on American martens. J. Appl. Ecol. **36**(1): 157–72.

Kalies, E.L. and Kent, L.L.Y., 2016. Tamm Review: Are fuel treatments effective at achieving ecological and social objectives? A systematic review. For. Ecol. Manag. **375**(2016) 84-95. http://dx.doi.org/10.1016/j.foreco.2016.05.021

Keane, R.E., Rollins, M., and Zhu, Z-L. 2006. Using simulated historical time series to prioritize fuel treatments on landscapes across the United States: The LANDFIRE prototype project. Ecological Modelling **204**(3-4): 485-502.

Kirk, T.A., and Zielinski, W.J. 2010. Functional habitat connectivity of the American marten in northeastern California using least-cost corridor modeling. USDA Forest Service, Lassen National Forest and Pacific Southwest Research Station, Redwood Sciences Lab.

LANDFIRE. 2014. LANDFIRE: LF 2014. Wildland Fire Science, Earth Resources Observation and Science Center, U.S. Geological Survey. Sioux Falls, SD. Accessed: February 2, 2017.

Meyer, M.D. 2013. Natural range of variation of red fir forests in the bioregional assessment area. Unpublished report. USDA Forest Service, Pacific Southwest Region. Albany, CA.

Miller, C. and Ager, A.A. 2013. A review of recent advances in risk analysis for wildfire management. Int. J. of Wildland Fire **2013**(22): 1-14. http://dx.doi.org/10.1071/WF11114

Moghaddas, J.J., Collins, B.M., Kurt Menning, Moghaddas, E.E.Y., and Stephens,S.L. 2010. Fuel treatment effects on modeled landscape-level fire behavior in the northern Sierra Nevada. Can. J. For. Res. **40**(9): 1751–65. doi:10.1139/X10-118.

Moriarty, K.M., Epps, C.W., Betts, M.G., Hance, D.J., Bailey, J.D., and Zielinski, W.J. 2015. Experimental evidence that simplified forest structure interacts with snow cover to influence functional connectivity for Pacific martens. Landscape Ecol. **30**(10): 1865–77. doi:10.1007/s10980-015-0216-2.

Moriarty, K.M., Epps, C.W., Zielinski, W.J. 2016. Forest thinning changes movement patterns and habitat use by Pacific marten. Jour. Wild. Mgmt. **80**: 621-633. doi:10.1002/jwmg.1060.

Moriarty, K.M., Zielinski, W.J., and Forsman, E.D. Decline in American marten occupancy rates at Sagehen Experimental Forest, California. Wildlife Manag. **75**: 1774-1787. doi:10.1002/jwmg.228

North, M., Stine, P., O'Hara, K., Zielinski, W., and Stephens, S.L. 2009. An Ecosystem Management Strategy for Sierran Mixed-Conifer Forests. Gen. Tech. Rep. PSW-GTR-220. USDA Forest Service, Pacific Southwest Research Station, Albany, CA.

North, M., Brough, A., Long, J., Collins, B., Bowden, P., Yasuda, D., Miller, J., and Sugihara, N. 2015. Constraints on mechanized treatment significantly limit mechanical fuels reduction extent in the Sierra Nevada. Journal of Forestry **113**(1): 40–48. doi:10.5849/jof.14-058.

Reinhardt, E.D., Keane, R.E., Calkin, D.E., and Cohen, J.D. 2008. Objectives and considerations for wildland fuel treatment in forested ecosystems of the interior western United States. For. Ecol. Manag. **256**(12): 1997-2006.

Rustigian-Romsos, H.L. and Spencer, W.D. 2010. Predicting habitat suitability for the American marten on the Lassen National Forest. Conservation Biology Institute. Corvallis, OR.

Scott, J.H, and Burgan, R.E. 2005. Standard fire behavior fuel models: a comprehensive set for use with Rothermel's surface fire spread model. Gen. Tech. Rep. RMRS-GTR-153. USDA Forest Service, Rocky Mountain Research Station. Fort Collins, CO.

Scott, J.H. 2006. An analytical framework for quantifying wildland fire risk and fuel treatment benefit. *In* Fuels Management-How to Measure Success: Conference Proceedings. 28-30 March 2006; Portland, OR. Proceedings RMRS-P-41. *Edited by* Andrews, P.L. and Butler, B.W. USDA Forest Service, Rocky Mountain Research Station. Fort Collins, CO.

Spencer, W.D. and Rustigian-Romsos, H.L. 2012. Decision-Support Maps and Recommendations for Conserving Rare Carnivores in the Interior Mountains of California. Conservation Biology Institute, Corvallis, OR.

Stephens, S.L., McIver, J.D., Boerner, R.E.J., Fettig, C.J., Fontaine, J.B., Hartsough, B.R., Kennedy, P.L. and Schwilk, D.W. 2012. The Effects of Forest Fuel-Reduction Treatments in the United States. BioScience **62**(6): 549–60. doi:10.1525/bio.2012.62.6.6.

USDA Forest Service. 2007. Sierra Nevada Forests Management Indicator Species Amendment: Record of Decision. USDA Forest Service, Pacific Southwest Region. Vallejo, CA.

USDA Forest Service. 2014. ExistingVegNor Interior1999_2009_v1. USDA Forest Service, Pacific Southwest Region Remote Sensing Lab. McClellan, CA.

USDA Forest Service, Almanor Ranger District, Lassen National Forest. 2015. Grizzly Restoration Project: Proposed Action. USDA Forest Service. Chester, CA.

USDA Forest Service, Plumas National Forest. 2013. Chips Fire Review. USDA Forest Service, Plumas National Forest, Quincy, CA.

USDA Forest Service. 2004. Sierra Nevada Forest Plan Amendment: Final Supplemental Environmental Impact Statement: Volume 1. R5-MB-046. USDA Forest Service, Southwest Region, Vallejo, CA.

Zielinski, W., Moriarty, K.M., Baldwin, J., Kirk, T.A., Slauson, K.M., Rustigian-Romsos, H.L., and Spencer, W.D. 2015 Effects of season on occupancy and implications for habitat modeling: the Pacific marten (*Martes caurina*). Wildlife Biology **21**:56-67.

3 | Assessing strategies for managing fuels in Pacific marten habitat at the landscape scale

3.1 Introduction

Federal forest management in the northern Sierra Nevada and southern Cascades mandates restoration of fire resilience and the conservation of sensitive forest species. In this region, and elsewhere in the western United States, the conservation of wildlife species associated with mature forest structure has motivated the establishment of late seral reserves, defined as areas where management action is limited or prohibited in the interest of wildlife conservation. Where fuel structure and the fire regime have diverged significantly from pre-settlement precedents, retention of these dense, late seral patches may conflict with objectives to restore ecosystem resilience and reduce an elevated risk of stand-replacing fire (Jones et al. 2016). Making informed decisions about how to balance these two objectives requires a consideration of how potential management action may impact both these goals.

Describing the effects of forest management practices on ecological processes may depend on the spatial scale at which they are observed or analyzed (Levin 1992, Bissonette 1997). I examined the effects of varying amounts of habitat reserves for Pacific martens on the efficacy of fuel reduction treatment at the watershed scale (Chapter 2, watersheds with a mean area of approximately 8,000 ha). This scale roughly corresponds to the size of recent management projects in Lassen National Forest (USDA 2015). While analysis at that scale may be applicable to the federal planning process, the Pacific marten is a wide-ranging forest carnivore and the population in this region occupies high elevation forests across numerous such watershed units (Rustigian-Romsos & Spencer 2010, Bissonette et al. 1997). Further, wildland fire is a landscape process that is managed at larger scales, in addition to consideration at the project level (Weatherspoon & Skinner 1996, Moghaddas et al. 2010). As such, analyzing the interplay between the creation of habitat reserves and the ability of fuel treatment to moderate fire behavior on a larger scale (~50,000 ha) appropriately provides a complementary perspective to project-level analysis.

In addition to the scale of analysis and the size of reserves themselves, the effect of prohibiting treatment in some marten habitat areas on landscape fuel treatment efficacy may also depend on how treatments are prioritized. Identifying the location and extent of fuel treatments that maximize efficiency could save considerable management resources, resources that could then be dedicated to other or additional objectives (Finney 2004, Schmidt et al. 2008). Strategies for prioritizing treatments can explicitly incorporate patterns of fire movement over the landscape, and reduce risk to forest patches with high fire hazard such as patches of mature forest (*Strix occidentalis caurina*, Ager et al. 2007). Landscape analyses can identify areas most susceptible to rapid fire spread rates or intense crown fire, and can focus removal of such features in corridors of continuous fuels that parallel the prevailing wind direction (Finney 2006). Directed efforts could mitigate the effect of reserves by interrupting potential fire movement in a more economical fashion.

I explored the degree to which large-scale habitat reserves for Pacific marten (*Martes caurina*) interfere with the ability of strategic fuel treatments to moderate extreme fire behavior. I quantified wildfire risk to marten habitat areas in order to understand the extent to which reserves may expose these regions to a greater probability of crown fire. I used probabilistic simulation to simulate large numbers of ignitions on landscapes with and without marten habitat reserves where fuel treatment was prohibited. A secondary objective was to test how three treatment allocation strategies (random, prioritized for slowing fire spread, and prioritized for reducing wildfire risk to marten habitat) may reduce or amplify the effects of reserve creation. I hypothesized that 1) at the landscape scale not treating any marten habitat will have a significant effect on simulated fire behavior, as ignitions in the continuous portions of untreated marten habitat will be allowed to spread rapidly without intersecting a treated area, and 2) partial habitat reserves, where treatment is allowed in marten habitat but outside the highest-quality habitat, will allow for more even distribution of treatments over the landscape, resulting in little difference from the no reserves scenario but lower wildfire risk to marten habitat compared to complete reserves, and 3) locating treatments more strategically with an optimization method will offset the negative effect of large reserves on modeled fire behavior.

3.2 Methods

3.2.1 Study location

I simulated fuel treatments in Almanor Ranger District, Lassen National Forest, California (49,224 ha, Figure 1). This location was chosen for analysis based on the combination of marten monitoring data, management interest, and airborne LiDAR coverage, a remote sensing technology that uses pulses of light from a laser to measure topography and elements of forest structure. Almanor Ranger District is situated at the northern end of the Sierra Nevada and the southern extreme of the Cascade Range. The average elevation within the study area is 1,717 m (range: 936 to 2,187 m). Vegetation consists primarily of mixed-conifer and true fir forests, with patches of meadow and chaparral scattered across the area and pine plantations located in the northeastern section of the study area. Ownership is 82.7% USDA Forest Service, 16.4% private, and 0.9% state of California.

3.2.2 Study design

I simulated factorial arrangement of fuel treatment scenarios, with three possible values for total treatment area (10%, 20%, and 30% of the study area) and three possible values for reserve area (none, partial, and complete)(Chapter 2, sec. 2.2.2). The partial reserves scenario prohibits fuel treatment in areas where the probability of marten occupancy is greater than 65%, the threshold needed to reserve a single, contiguous patch of marten habitat across the study area (Figure 3.1). In the complete reserves scenario all potential marten habitat was reserved, where potential habitat was defined as areas where probability of occupancy is greater than 50%.

To compare treatment allocation strategies, I simulated fuel reduction treatments on 20% of the study area, corresponding to the area found to be effective at reducing landscape fire spread in previous studies (Finney et al. 2007, Collins et al. 2011, Moghaddas et al. 2010). I selected locations for simulated treatments in each scenario according to one of three allocation strategies: (1) random, (2) prioritized to reduce fire spread, and (3) prioritized to reduce risk to predicted marten habitat. Three random treatment allocations were simulated and averaged in order to capture variation in the responses due to random placement of treatments.

3.2.3 Modeling canopy fuel parameters with LiDAR data

I calculated three canopy fuel parameters necessary for fire modeling: 1) canopy height, 2) canopy base height, and 3) canopy bulk density, using a combination of LiDAR, vegetation plots, and the Forest Vegetation Simulator (FVS, Dixon 2002). Vegetation data were collected to correlate observed stand conditions with airborne LiDAR data (n = 146 vegetation plots, data collected in 2013 and 2014). Plots were circular, fixed-area plots with an area of .09 ha, within which all trees with diameter at breast height greater than 6.6 cm (2.6 in) were measured. I used the resulting tree lists to calculate canopy height, base height, and bulk density for each vegetation plot in FVS. A suite of LiDAR-derived predictor variables (Appendix A) were created use the Fusion LiDAR processing software, based on first return-only LiDAR point clouds (McGaughey 2009). I derived percent canopy cover directly from LiDAR first returns using FUSION.

My goal was to extrapolate the observed data from the field plots to the full extent of the landscape for which airborne LiDAR data was available. Since I used 146 plots to derive canopy fuel parameters for greater than 17,000 pixels in the study area, I adopted a prudent modeling approach to mitigate potential effects of overfitting the data. Canopy height, canopy base height, and canopy bulk density were each modeled using exhaustive, all-subsets linear regression with up to ten LiDAR-derived explanatory variables, in R with the 'leaps' package (R Core Team 2016, Lumley & Miller 2009, Appendix B). I used tenfold cross-validation for each candidate model to test its predictive ability and calculated variance inflation factors for each coefficient to test for collinearity between explanatory variables, with the 'DAAG' package (Maindonald & Braun 2015). For each parameter (canopy height, canopy base height, and canopy bulk density) I selected the model with the lowest mean squared error from tenfold cross-validation, and that had no variance inflation factor greater than 10 to prevent excessive collinearity among predictor variables (Table 3.1, Hair et al. 1995). While collinearity is not a major concern for prediction modeling, it can result in inaccurate prediction in cases where new data shows inconsistent relationships between explanatory variables that were highly collinear in the data used to create the model (Harrell 2001).

3.2.4 Vegetation mapping

While useful for describing gradients in topography and canopy structure, the coarse descriptions of landscape forest structure produced by LiDAR do not describe land units that are meaningful to forest managers. I used vegetation mapping data (USDA 2014) to divide the landscape into stand units of a reasonable size for management activity and silvicultural prescription. Stands less than 2.02 ha (5 acres) were merged with adjacent units to assemble a list of stands that would reasonably be assigned treatment independently. I used a marten probability of occupancy model (Rustigian-Romsos & Spencer 2010) to determine which stands would act as reserves where treatment was prohibited for each scenario. For the complete reserves scenarios, fuel treatment was prohibited in all stands with a mean probability of occupancy greater than 50%. In the partial reserves scenarios, fuel treatment was prohibited in all stands with a mean probability of occupancy greater than 65%, the minimum threshold needed to reserve a contiguous patch of marten habitat across the study area (Figure 3.1). I considered stands ineligible for treatment if they were designated as wetlands, grasslands, riparian, chaparral, or barren, according to the California Wildlife Habitat Relationships classification system (CA Dept. of Fish and Wildlife 2014).

In addition to the canopy fuel parameters mentioned above (canopy cover, canopy height, canopy base height, and canopy bulk density), simulating wildfire with the minimum travel time algorithm also requires the input of a fuel model for all areas within the study area. A fuel model is a categorical value that describes the fuel structure within that area and expected fire behavior. I used the LANDFIRE database (2014) to assign fuel models to each pixel in the study area, using Scott and Burgan's (2005) classification system. Values in this base layer were then adjusted using a combination of LiDAR data, local knowledge, and reported fire behavior during a recent large wildfire (USDA 2013) in order to reflect current conditions and potential fire behavior within the study area. Calibration steps included comparing assigned fuel models to field plots located within the study area, comparing fire sizes produced by the model to fire progressions in recent wildfire activity, and comparing fire behavior output to behavior anticipated by local management.

3.2.5 Treatment simulation

I simulated fuel reduction treatments by adjusting the values of each of the biotic fire modeling parameters (canopy cover, canopy height, canopy base height, canopy bulk density, and fuel model) for each pixel that fell within the boundary of a stand that was assigned treatment. I calculated the amount each fuel parameter would be adjusted by reviewing silvicultural prescriptions for fuel management recently produced by Lassen National Forest (USDA Forest Service 2015). I simulated an archetypal fuel reduction treatment, a thin from below followed by a prescribed burn to reduce activity and surface fuels, with the Forest Vegetation Simulator (Dixon 2002), and ran the simulation with tree list data for 89 stands collected as part of the project planning process. I tabulated simulation results and averaged the changes in each parameter to generate a simplistic guide to approximate how a standard treatment would affect the fuel parameters listed above (Table 3.2). When a treatment was simulated, the parameter values for pixels within the unit were averaged, modified by the adjustment list in the guide, and then assigned to each pixel in the treated stand. I made the assumption that a fuel treatment prescription would not modify pixels, or sections of a stand, independently, but instead would be applied uniformly within the stand. In each simulation, stands were considered eligible for treatment if they were 1) forested with greater than 40%

canopy cover, 2) owned by the Forest Service, 3) outside riparian areas, and 4) outside the defined reserve area for that scenario.

I simulated fires across the landscape with three levels of levels of fuel treatment area (10%, 20%, and 30%), using the Treatment Optimization Method (TOM), an algorithm that selects pixels for treatment that have the greatest effect on slowing fire spread rates (Finney 2006, Figure 3.2). The TOM algorithm selects treatments using constant fuel moisture, wind speed, and wind direction, values that are based on prevailing conditions during fire season. Treatment locations are therefore not necessarily optimally placed for slowing fire spread for all potential weather conditions. Weather data was pooled from the three nearest Remote Automated Weather Stations to determine the prevailing wind direction (225 degrees) and 97th percentile wind speed (23 mph). Fuel moistures for live and dead fuels were set to the "very dry" scenarios described in Scott & Burgan (2005), in order to optimize treatment placement for the most extreme fuel moisture conditions, when extensive crown fire is most probable. The pixel by pixel output of the TOM algorithm was translated into the stand units described above by selecting stands for treatment that had the highest proportion of TOM-assigned pixels, until the total treatment area needed for each scenario was met.

For the assessment of how a strategic approach to treatment planning could help offset any negative effects of retaining dense forest patches, two additional treatment allocation strategies were compared to the TOM procedure described above. First, eligible stands were randomly assigned treatment until 20% of the study area was treated. This procedure was repeated three times and responses were averaged over each repetition, in order to capture the variation inherent in random selection (Figure 3.3).

Second, stands were prioritized for treatment using a habitat protection strategy (HPS), which used the following equation derived from Ager et al. (2007):

$$Priority = \frac{Distance_{Marten}}{(abs(Azimuth_{Marten} - 225))}$$

where $Distance_{Marten}$ is the distance from the centroid of the unit being evaluated to the centroid of the nearest predicted marten habitat, $Azimuth_{Marten}$ is the azimuth (in degrees) from the centroid of the unit being evaluated to the nearest predicted marten habitat, and 225 degrees is the prevailing wind direction in the study area during fire season. Stands with the lowest Priority value are treated first, until the required 20% of the study area is reached. This method allocates treatments primarily to areas that are upwind and adjacent to predicted marten habitat, maximizing the ability of simulated treatments to interrupt and moderate the behavior of fires that spread toward habitat areas (Figure 3.4). Because this habitat strategy prioritizes stands for treatment based on their proximity to untreated habitat areas, it is not possible to simulate a no reserves scenario with this strategy.

3.2.5 Fire simulation

I conducted three fire modeling runs for each treatment/reserve scenario, in order to capture and measure variability in how ignition locations and weather conditions affect the responses. For each model run, I simulated 1000 randomly-placed ignitions, with the probability of ignition assumed to be equal for each pixel in the landscape, a methodology consistent with recent efforts to measure the effects of landscape fuel treatments (Moghaddas et al. 2010, Collins et al. 2011). These random ignitions were simulated using FConstMTT, a version of the minimum travel time method for fire simulation, an efficient algorithm for rapidly simulating fire behavior on a landscape (Finney 2002). Each ignition was allowed to burn for a simulation time of 15 hours (3 burn days), during which wind speed and direction were held constant after random selection from the pool of 90th percentile and higher wind conditions from the three nearest Remote Automated Weather Stations stations (Table 3.3). Assessing how fuel treatments affect fire behavior necessitates a compromise between more detailed simulation of individual fires, in terms of duration and changing weather conditions, and the number of fires or ignition locations. I assumed a reduced burn time at constant weather conditions would capture the capacity of a landscape to support rapid fire spread in a short time, while also controlling for the significant effect of ignition location as it interacts with the spatial allocation of treated areas.

I calculated: 1) area of habitat targeted for treatment, 2) average conditional burn probability on the landscape, 3) expected habitat loss for each model run. See Chapter 2 (p. 27-29) for a detailed description of these metrics and their formulas.

3.2.6 Statistical analysis

Preliminary analysis indicated that the level of reserves and the amount of area treated did not act independently. Therefore, the following statistical model, which includes parameters for each of the treatment and reserve levels as well as all interactions, was used for analysis. This model was applied separately for each response in order to test for differences between fuel treatment strategies with and without reserves:

$$Y_{response} = \alpha_0 + \alpha_1 I.t20 + \alpha_2 I.t30 + \alpha_3 I.rP + \alpha_4 I.rC + \alpha_5 I.t20I.rP + \alpha_6 I.t20I.rC + \alpha_7 I.t30I.rP + \alpha_8 I.t20I..r30 + \varepsilon_t$$

where:

Y_t = response for the t^th model run, where t = 1 – 27 (3 models * 3 treatment areas * 3 reserve levels)

α_0 = mean response with 10% fuel treatment and no reserves.

α_1 = incremental effect of fuel treatment on 20% of the watershed.

α_2 = incremental effect of fuel treatment on 30% of the watershed.

α_3 = incremental effect of partial reserves of the watershed.

α_4 = incremental effect of complete reserves of the watershed.
α_5 = combined effect of treating 20% of the watershed and partial reserves.
α_6 = combined effect of treating 20% of the watershed and complete reserves
α_7 = combined effect of treating 30% of the watershed and partial reserves.
α_8 = combined effect of treating 30% of the watershed and complete reserves
I.t20 = 1 when 20% treated, 0 otherwise.
I.t30 = 1 when 30% treated, 0 otherwise.
I.rN = 1 when no reserves, 0 otherwise.
I.rP = 1 when partial reserves, 0 otherwise.
I.rC = 1 when complete reserves, 0 otherwise.
ε_t = random error associated with the t^{th} observation, where $\varepsilon t \sim N(0, \sigma^2)$ and εt and $\varepsilon t+1$ are independent.

For the comparison of the three allocation strategies, preliminary exploration also presented evidence that the level of reserves and each treatment allocation strategy did not act independently when modeling both burn probability and expected habitat loss. As such, interactions of reserves and strategy were parameterized with a simple effects model in which individual parameters for each reserve/strategy combination were used for analysis.

To control the false discovery rate given the multiple comparisons necessary for each response, the Benjamini-Hochberg procedure was used to ensure a false discovery rate of no more than 5% (Benjamini & Hochberg 1995).

3.3 Results

3.3.1 Modeling canopy fuel parameters

I used LiDAR to approximate the canopy fuel parameters over the landscape, but the effectiveness of predicting canopy fuel parameters with LiDAR-derived metrics varied among canopy height, canopy base height, and canopy bulk density (Table 3.1). The canopy height model accounted for 91% of the variation in the field plots, while the canopy base height model account for 44% of variation and the canopy bulk density model accounted for 66%.

3.3.2 Vegetation mapping and treatment simulation

Based on the probability-of-occupancy model, 37.2% (18,071 ha) of the study area was initially designated marten habitat, while 29.6% (10,009 ha) had a probability of occupancy greater than 65% (Rustigian-Romsos & Spencer 2010). Without reserves, marten habitat was selected for fuel treatment on par with its distribution in the landscape, (Table 3.4, Figure 3.5). With partial reserves, approximately one fourth of the area targeted for treatment at each level consisted of predicted marten habitat. Under the most intensive treatment scenario, where 30% of the study area was targeted for fuel treatment without consideration of predicted marten occupancy, 31.3% (5,651 ha) of predicted marten habitat was identified as optimal for fuel treatment placement.

3.3.3 Average conditional burn probability

Across all treatment scenarios, the creation of marten habitat reserves had no significant impact on average conditional burn probability (Table 3.5, Figure 3.6). When fuel treatments were simulated on 20% of the study area, partial and complete reserves led to slight increases in burn probability, but those differences were not substantial or statistically significant.

3.3.4 Expected habitat loss from wildfire

Implementation of complete habitat reserves was associated with increases in expected habitat loss at all levels of fuel treatment, while partial reserves only resulted in a significant increase in expected habitat loss at the 20% fuel treatment level, compared to scenarios where treatment was allowed within all marten habitat (Table 3.6, Figure 3.7). The largest change in expected habitat loss occurred with complete reserves at 20% fuel treatment, an estimated average increase of 171.1 ha, which represents a 54% increase compared to the equivalent scenario with no reserves.

3.3.5 Comparison of treatment allocation strategies

The three treatment allocation strategies each responded differently to increasing restrictions on fuel treatment placement (Table 3.7). Compared to a random allocation of fuel treatments within the study area, the Treatment Optimization method (TOM) was associated with marginal reductions in average conditional burn probability under the no reserves and complete reserves scenarios (Table 3.8, Figure 3.8); this difference was only statistically significant when no reserves were implemented. The habitat protection strategy concentrated fuel reduction treatments in forested units upwind and adjacent to predicted marten habitat areas, which resulted in significant increases in burn probability compared to random treatment placement.

There were no significant differences in expected habitat loss between the TOM strategy and random treatment placement under any type of reserves (Table 3.9, Figure 3.9). However, the habitat protection strategy was associated with significant reductions in expected habitat loss compared to random treatment assignment. By interrupting potential fire movement into habitat areas, this strategy was effective at reducing wildfire risk, but the accompanying significant increases in burn probability suggest achieving this reduction came at the cost of controlling fire spread over the whole study area.

3.4 Discussion

Prohibiting fuel reduction treatments within predicted marten habitat had little effect on the ability of active fuel management to control large fire spread under extreme weather conditions in this landscape. While restricting fuel treatment placement did limit the options for allocating treatments within the study area, treatments were overall just as effective in controlling rapid fire spread when confined to lower elevations with insignificant probabilities of marten occupancy. While the dense, higher-elevation forests used by Pacific marten (Spencer 1983) are susceptible to crown fire, these simulations support a focus on reducing fuels in lower elevation areas whose fuel bed is more conducive to rapid fire growth in dry conditions.

The limited importance of reducing fuels in the fir-dominated forests associated with high probability of marten occupancy marten use largely due to the dense fuel structure in these areas, which was reflected in the canopy fuel parameters and fuel models assigned to these areas. Such stands often acted as impediments to fire spread in these simulations, slowing fires that moved more rapidly through drier, more open stands at lower elevations. This model behavior aligns with previous research that suggests the compact fuel bed in true fir forests limits surface fire spread rates compared to surface fuels in pine-dominated forests with higher packing ratios, though we also expect a significant amount of crown fire under the weather conditions that were simulated (Skinner & Chang 1996, Van Wagtendonk & Fites-Kaufman 2006, Agee 1993). Further, we expect that surfaces fuels in denser stands with higher canopy cover will cure more slowly compared to fuel beds in more open areas, make them less susceptible to rapid fire spread under same the weather conditions (Bekker & Taylor 2001, Taylor 2000), particularly earlier in the fire season.

In contrast to the limited effect of reserves on fire spread rate in the landscape, prohibiting fuel treatment in predicted marten habitat did have a significant effect on wildfire risk to Pacific marten habitat, especially under the complete reserves scenarios (Figure 3.7). This result was not surprising given that fuels management within habitat areas reduces surface fuels and raises canopy base heights, reducing the probability of crown fire and subsequent loss of habitat value. Further, fuel reduction treatments in predicted marten habitat also have offsite benefits, and can mitigate wildfire risk to adjacent, untreated units that are located downwind of a treated area (Ager et al. 2007). It is difficult to effectively weigh the costs and benefits of fuel treatments that on one hand reduce wildfire risk, but on the other simplify forest structure such that connectivity for Pacific martens is also reduced (Jones et al. 2016). Depending on the intensity and configuration of burned patches, wildfire may raise habitat value by recruiting structures like snags and logs that are used as resting and denning sites (Spencer 1987, Martin & Barrett 1991). These results indicate that in areas where fire is a particular threat to marten habitat— for example, where the probability of ignition may be higher due to recreational use or where forest structure is unusually homogenous— prohibiting all fuel reduction treatments in habitat areas may not be the most desirable course. Without incursion into the highest-rated habitat areas, managing fuels in regions of marginal habitat value can be effective at mitigating this wildfire risk.

The value of additional treatment area, in terms of reductions in burn probability and reductions in wildfire risk to habitat, diminished as additional treatments were added from 10% to 20%, and from 20% to 30% (Figure 3.10). An exception to this trend was the final 10% treatment implemented in the complete reserves scenario, which appeared to saturate the landscape with treatments along the margins of predicted marten habitat and resulted in a precipitous decline in expected habitat loss. It is important to note that the fuel treatment levels assessed in this study represented 10%, 20%, and 30% of the entire study area, which included private land, riparian areas, and non-forested areas that were not considered eligible for treatment in these simulations. As such, both 20% and 30% treatment area would represent ambitious fuel management strategies, the feasibility of which may be limited by current management constraints (Barros et al. 2017, North et al. 2015). However, if treatments are effective for a minimum of 10 years (Finney et al. 2007), my simulations indicate annual treatment of 2% of this study area, with no or partial habitat reserves, can yield reductions in both burn probability and wildfire risk to marten habitat that approach 50% (Figure 3.10).

The Treatment Optimization Method was more effective at placing fuel treatments to reduce fire spread than a random treatment assignment under the no reserves scenario, but this effect disappeared as additional reserve restrictions were added (Figures 3.8, 3.11). As constraints on treatment placement become more restrictive, the TOM algorithm may become more sensitive to model parameters like the maximum treatment dimension, which dictates both the maximum treatment size and the width of the strips the algorithm uses to iteratively select treatments across the landscape (Finney 2006). Another possibility is that, at these higher treatment levels, fuel treatment location becomes less important compared to the random variation of ignition locations and weather conditions. As the landscape is saturated with fuel treatments in remaining eligible locations, the precise locations of treatments may be overridden by the presence of large blocks of treated area, and treatment performance may depend more on whether ignitions find remaining pockets of untreated area within these blocks under more extreme fire weather conditions (Figure 3.2f, Figure 3.2i)

In contrast, the habitat protection strategy (HPS) represented a more significant departure from the behavior of the random treatment placement scenarios. In concentrating fuel reduction treatments to the southwest of, and adjacent to, predicted marten habitat (Figure 3.4), this strategy sacrificed the ability to control large fire spread on the landscape to produce significant reductions in wildfire risk to these habitat areas (Figure 3.11). This reduction in wildfire risk was greatest under the partial reserves scenario, where HPS targeted a substantial amount of marginal predicted habitat (where the probability of occupancy was between 50% and 65%). Under the assumption that fuel treatments have a short-term negative effect on martens by reducing connectivity, this strategy would reduce overall habitat quality in marginal habitat in order to preserve the core areas martens are most likely to use. When protective fuel treatments were pushed outside of all potential marten habitat (Figure 3.4b), the benefit of reduced wildfire risk was present but less apparent.

Current management objectives would likely preclude a fuel treatment strategy focused entirely on reducing wildfire risk to Pacific marten habitat (USDA 2004), and these results indicate such a strategy would expose the larger landscape to increased fire spread rates compared to other treatment strategies with broader goals. However, the performance of the habitat protection strategy underlines the potential benefits of purposefully managing fuels in proximity to core habitat areas. While martens avoid simplified stands created by treatment prescriptions that overly simplify forest structure (Moriarty et al. 2015, 2016), this forest carnivore is associated with fir forests that historically experienced a natural fire regime that was variable in terms of severity and frequency (Spencer 1983, Bekker & Taylor 2001). As such, if silvicultural prescriptions can more closely mimic the complexity of post-fire stand structures, and reduce fire hazard while also retaining elements important for marten cover and foraging, the tangible benefits of managing fuels in marginal habitat may be possible without sacrificing other conservation goals.

Detecting canopy height, which involves measuring laser returns at the upper extreme of the LiDAR point cloud, has long been a strength of airborne LiDAR systems (Dubayah 2000). As such, it was not surprising that the suite of LiDAR-derived variables were able to account for much of the variability in canopy heights among the field plots ($R^2 = 0.91$, Table 3.1). Two primary reasons may help account for the poor correlations between the LiDAR-based prediction models for canopy base height and canopy bulk density and the plot level values. First, deriving canopy base height and bulk density using airborne LiDAR involve extracting values from within the LiDAR point cloud (as opposed to one extreme, as is the case with canopy height). Second, and more likely here, the Fire and Fuels Extension of the Forest Vegetation Simulator (FFE-FVS) calculates the canopy fuel metrics based on tree records alone, using methodology developed by Scott and Reinhardt (2001) and derived from Sando & Wick (1972). FFE-FVS assumes crown weight for each tree is evenly distributed along the crown length, then calculates total foliage and fine branchwood weight in one foot increments. Canopy base height is the lowest height at which the three foot running mean density is greater than 0.011 kg/m^3 (30 lbs/acre-foot), whereas canopy bulk density is the highest average value of all possible 13 foot running mean densities. While the assumption of equal foliage weight along the crown length may be problematic in some cases, the larger issue is likely the exclusion of shrubs and small trees (those with DBH < 6.5 cm) from the FVS calculations. Airborne LiDAR does not make such distinctions, and thus may show a continuous canopy of intermixing shrubs, small trees, and low branches without the clear threshold identified by FVS. However, other efforts to predict canopy base height and canopy bulk density with LiDAR have achieved better results while dealing with the same issue (Andersen et al. 2005, Hall et al. 2005). Increased shrub cover or density of small trees could explain the reduced model accuracy compared to these other modeling efforts.

3.5 Conclusions

The implementation of fuel reduction treatments, as a means of restoring fire resilience to forests with an unnatural level of fuel accumulation, requires integration with a suite of other forest management objectives. This simulation framework represents an initial effort to understand how the application of landscape scale fuel treatment interacts with one other management goal, the conservation of Pacific marten, using a simplified set of additional constraints on management action in a landscape at the intersection of the northern Sierra Nevada and southern Cascades. Under these assumptions, I found no evidence that the creation of habitat reserves, where no fuel reduction treatments were permitted, interfered with the ability of large scale fuels management to reduce the landscape's capacity for rapid fire growth. The high elevation, true fir–dominated forests associated with Pacific marten did experience crown fire, but more often acted as an impediment to rapid fire spread, particularly in comparison to lower elevations with less canopy cover and a larger proportion of pine species.

While fuel reduction treatments were as effective at slowing fire growth over the landscape when placed outside of predicted marten habitat, doing so lessened the ability of the treatments to moderate crown fire behavior within areas associated with marten use. While wildfire can potentially create complex, post-fire stand structures usable by martens as cover and foraging opportunities, martens do not typically enter large openings which could result from patches of contiguous crown fire (Moriarty et al. 2015, 2016). Given that the Pacific marten population in this region is already more fragmented than elsewhere in the species' range (Spencer & Rustigian-Romsos 2012, Kirk & Zielinski 2010), the short-term negative effects of fuel reduction treatments may be tolerable in the interest of long-term conservation, where forest structure is homogenous, crown fire is of particular concern, and risk of habitat loss is high. My simulations demonstrated that significant reductions in wildfire risk to marten habitat were possible without incursion into areas most likely to be used by martens. While these high-elevation forests may currently be less departed from pre-settlement fuel structure than lower elevation stands that historically burned more frequently (Taylor 2000, Meyer 2006), without disturbance or intervention overall fuel accumulation and continuity will increase. As such, fuel treatments that can break up areas of homogeneous stand structure and fuel continuity, while also retaining logs and snags important for marten life history, could reduce wildfire risk to martens while also benefitting broader goals of ecological restoration.

3.6 Chapter 3 | Figures and Tables

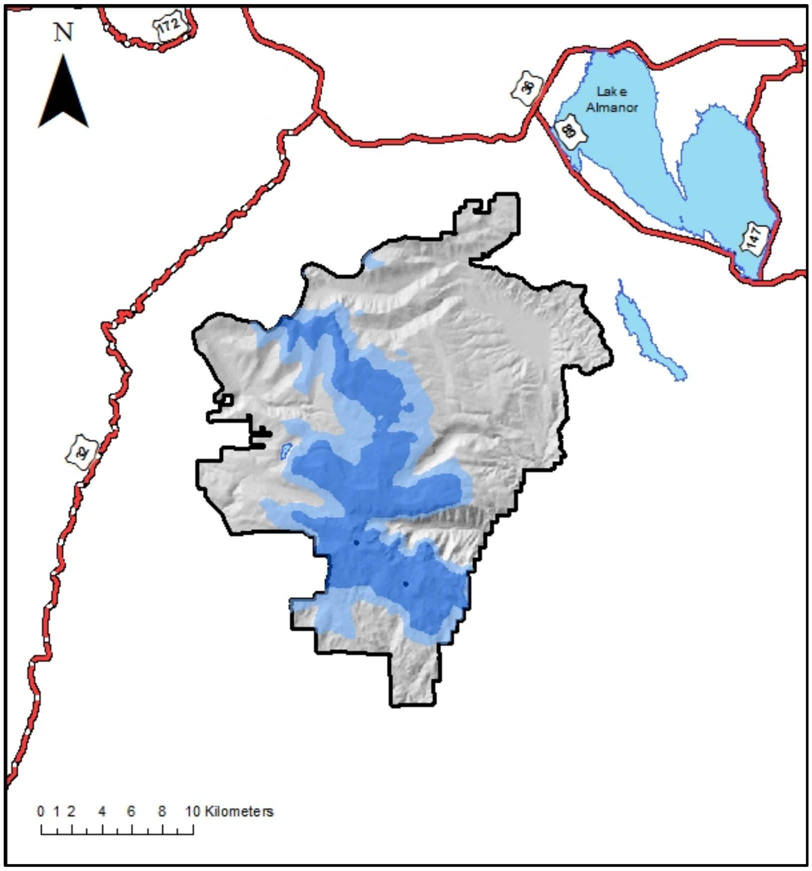


Figure 3.1: The study area located in Almanor Ranger District, Lassen National Forest, CA. The total area is 49,224 ha. Shaded blue area indicates predicted marten habitat (probability of occupancy > 50%), and darker blue indicates highest value predicted habitat (probability of occupancy > 65%) (Rustigian-Romsos & Spencer 2010).

Table 3.1: Selected regression models for canopy fuel parameters.

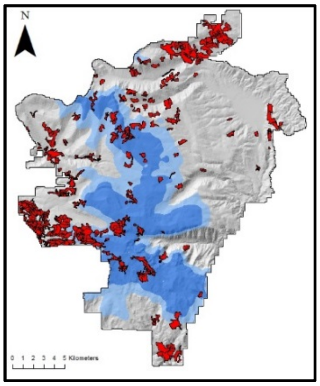
Variable	Model equation	R^2
Canopy height	19.394 + 1.026*elevmaximum – 3.175*elevkurtosis + 0.881*canopycoverfr – 1.567*strcc_225 – 0.618*strcc_612 + 0.865*strcc_3036	.93
Canopy base height	Exp(0.148 + 0.307*elevp01 + 0.074*elevp40 – 0.021*pamn + 0.028*strcc_36 – 0.040*strcc_3036)	.55
Canopy bulk density	Exp(-5.354 + .001*las13_dem30 + .001 * las13_asp30 _ .031*canopycoverfr – 0.047*volumecover)	.66

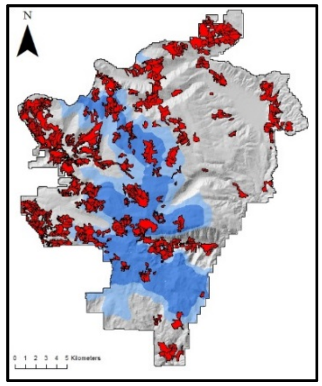
Table 3.2: Simulated treatment effects on fire modeling inputs

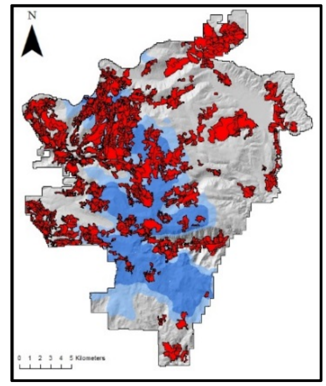
Variable		Change when treated
Fuel model:		
	Stands < 60% CC:	Set to TL1
	Stands > 60 % CC:	Set to TL1
Canopy cover:		
	Stands < 60% CC:	Set to 40%
	Stands > 60% CC:	Set to 50%
Canopy height:		
	Stands < 60% CC:	No change
	Stands > 60% CC:	No change
Canopy base height:		
	Stands < 60% CC:	+ 14 ft.
	Stands > 60% CC:	+ 19 ft.
Canopy bulk density:		
	Stands < 60% CC:	- 0.015 kg/m^3
	Stands > 60% CC:	- 0.039 kg/m^3

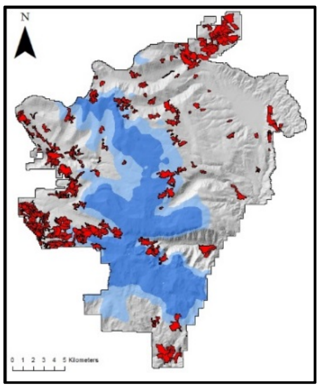
Table 3.3: 90^{th} percentile and higher weather conditions used for fire simulation.

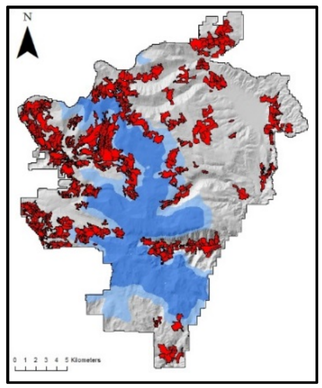
20 ft. wind speed (mph)	Wind direction (azimuth degrees)	Probability
20	225	.43
25	225	.27
15	225	.10
20	270	.06
30	225	.06
20	180	.05
15	270	.02
25	180	.01

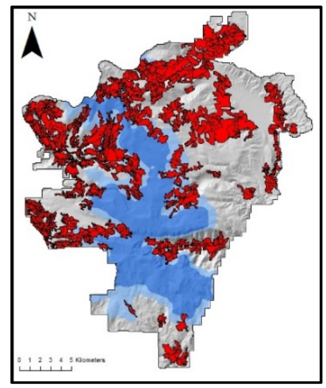

a. No reserves, 10% treated

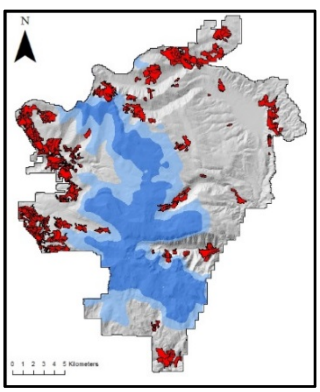

b. No reserves, 20% treated

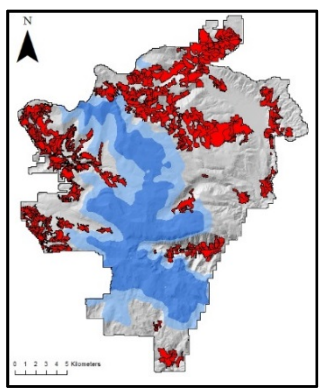

c. No reserves, 30% treated


d. Partial reserves, 10% treated


e. Partial reserves, 20% treated


f. Partial reserves, 30% treated


g. Complete reserves, 10% treated


h. Complete reserves, 20% treated

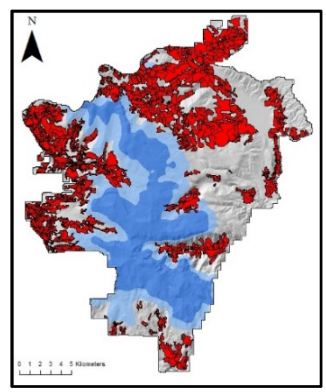
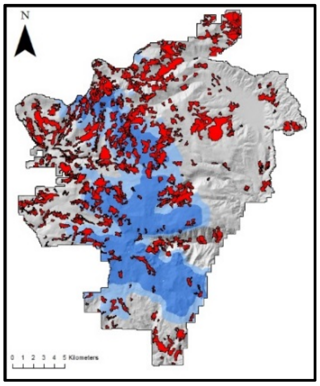
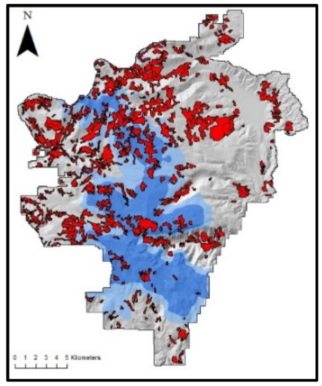

i. Complete reserves, 30% treated

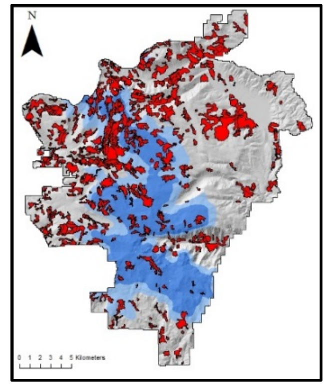
Figure 3.2: Simulated treatments and marten reserves in the study area, Almanor Ranger District, Lassen National Forest, CA. Treatments were allocated with the Treatment Optimization Method (Finney 2006). Red indicates units treated under each scenario. The darker blue color indicates the partial reserves, where the probability of marten occupancy is greater than 65%, while lighter blue indicates the area of the complete reserves, including all predicted marten habitat where probability of occupancy is greater than 50%. With no restrictions on treating fuels in marten habitat, the TOM method identified stands across the center of the study area as preferable areas for reducing fire spread rates (a, b, c). As partial reserves (d,e,f) and complete reserves (g,h,i) added restrictions on managing fuels in marten habitat, treatments became highly concentrated in the north and west.



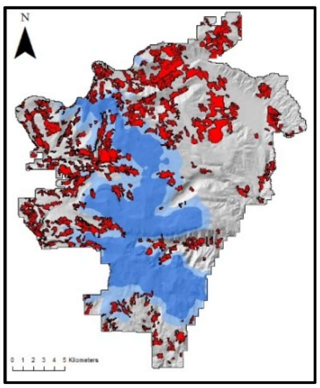
a. No reserves, iteration 1



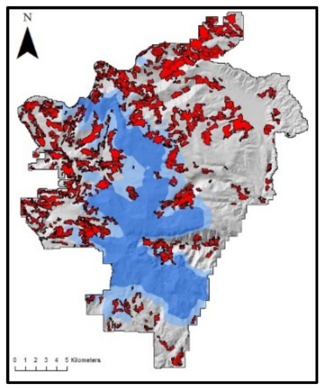
b. No reserves, iteration 2



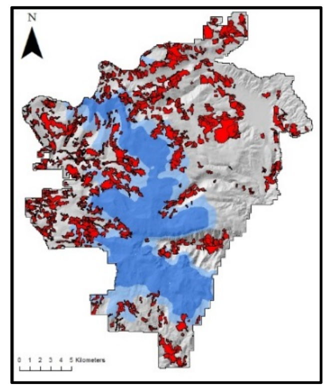
c. No reserves, iteration 3



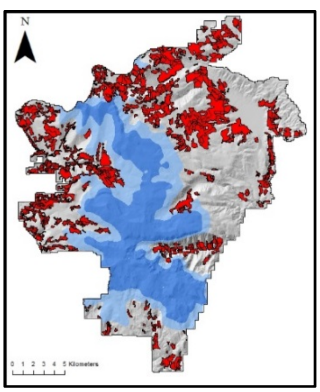
d. Partial reserves, iteration 1



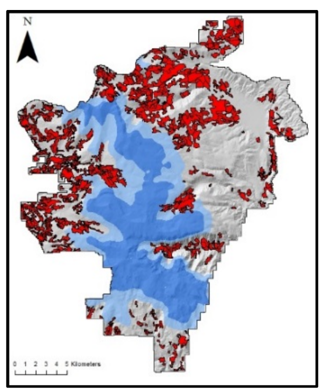
e. Partial reserves, iteration 2



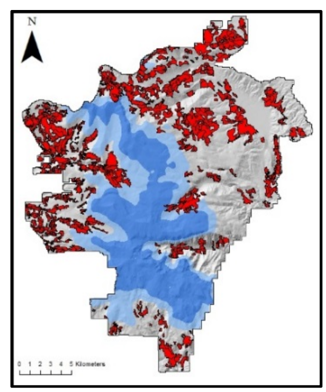
f. Partial reserves, iteration 3



g. Complete reserves, iteration 1

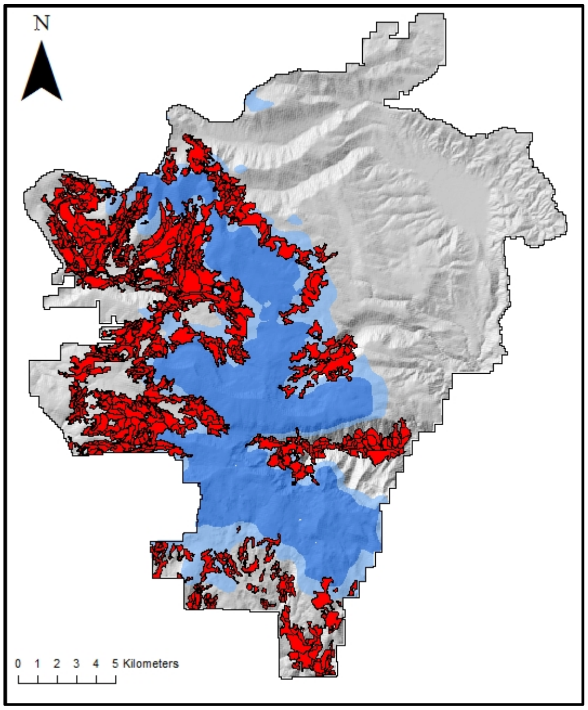


h. Complete reserves, iteration 2

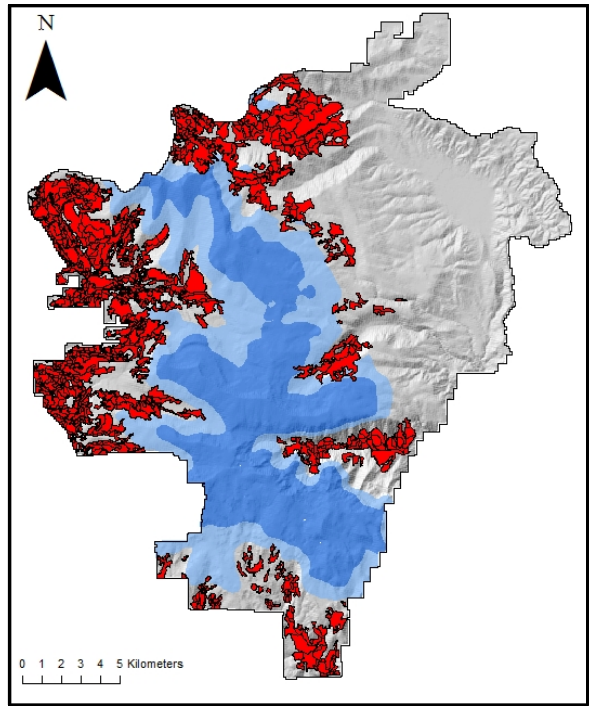


i. Complete reserves, iteration 3

Figure 3.3: Simulated treatments and marten reserves in the study area, Almanor Ranger District, Lassen National Forest, CA. Treatments were randomly placed over 20% of the study area. Random placement was repeated three times for each reserve scenario, in order to capture variation in the responses resulting from the randomization process. Red indicates stand units treated under each scenario. The darker blue color indicates the partial reserves, where the probability of marten occupancy is greater than 65%, while lighter blue indicates the area of the complete reserves, all predicted marten habitat where probability of occupancy is greater than 50%. Excluding eastern portions of the landscape that contained private land and meadows ineligible for treatment, treated stands were scattered evenly when allowed to overlap with predicted marten habitat (a,b,c,). As additional restrictions on treatment location were added, in the form of partial reserves (d,e,f) and complete reserves (g,h,i), fuel treatments were pushed out of central marten habitat to the north and west.



a. With partial reserves



b. With complete reserves.

Figure 3.4: Simulated treatments and marten reserves in the Humboldt Study Area, Almanor Ranger District, Lassen National Forest, CA. Treatments were allocated over 20% of the study area and prioritized with a habitat protection strategy, designed to reduce wildfire risk to habitat and calculated using a formula derived from Ager et al. (2007). Red indicates stand units treated under each scenario. The darker blue color indicates the partial reserves, where the probability of marten occupancy is greater than 65%, while lighter blue indicates the area of the complete reserves, all predicted marten habitat where probability of occupancy is greater than 50%. No reserves is not possible using this treatment allocation strategy, as stands are prioritized based on their ability to protect untreatable habitat. Unlike the TOM strategy (Figure 3.2) and random assignment (Figure 3.3), this method shows a clear preference for areas on the leeward side of predicted habitat, modifying fuels in these regions to best intersect potential fires moving upslope toward habitat areas.

Table 3.4: Summary of all response variables for the control and each fuel treatment scenario, averaged over all model runs.

Reserves	**Treatment amount (% of study area)**	**Predicted habitat treated (ha)**	**Average conditional burn probability**	**Expected habitat loss (ha)**
Control	0	0	0.043	624.6
None	10%	1596.2	0.032	444.5
Partial	10%	1192.9	0.031	466.2
Complete	10%	0	0.032	526.6
None	20%	3254.8	0.022	316.4
Partial	20%	2838	0.023	368
Complete	20%	0	0.023	487.5
None	30%	5651.2	0.018	295.8
Partial	30%	3370.3	0.017	321.3
Complete	30%	0	0.017	373

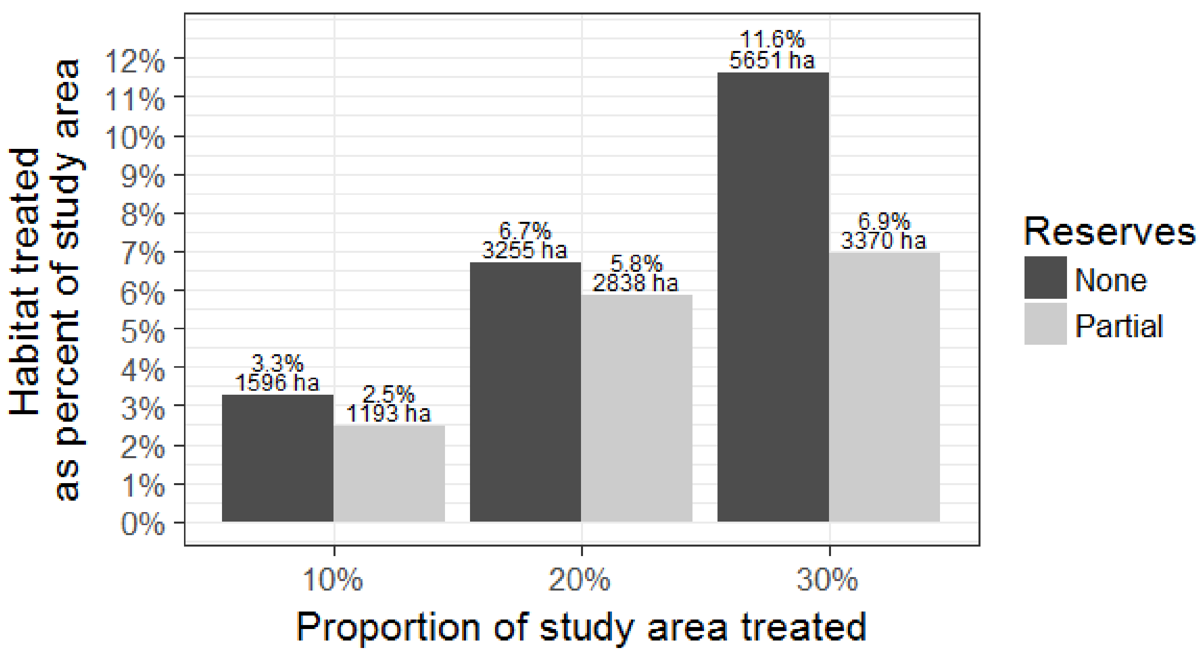


Figure 3.5: Predicted marten habitat (as a proportion of the total study area and in hectares) that was assigned fuel treatment by the Treatment Optimization Method algorithm for each scenario. Approximately one third of the treated area consisted of marten habitat under the no reserves scenarios. Approximately one fourth of the treated area was designated marten habitat under the partial reserves scenarios.

Table 3.5: Estimated changes in average conditional burn probability for all treatment scenarios, compared to the equivalent scenario with no reserves. For example, the 10% / Partial scenario was compared to the 10% / No reserves scenario. P-values shown have been adjusted to control the false discovery rate at 5%, following procedures outlined by Benjamini and Hochberg (1995), and were considered significant if less than 0.05.

Treatment/reserve scenario	**Estimate**	**Std. Error**	**Adjusted p-value**
10% / Partial	-0.00045	0.000651	0.601
20% / Partial	0.00143	0.000651	0.251
30% / Partial	-0.00079	0.000651	0.601
10% / Complete	-0.00018	0.000651	0.781
20% / Complete	0.00061	0.000651	0.601
30% / Complete	-0.00054	0.000651	0.601

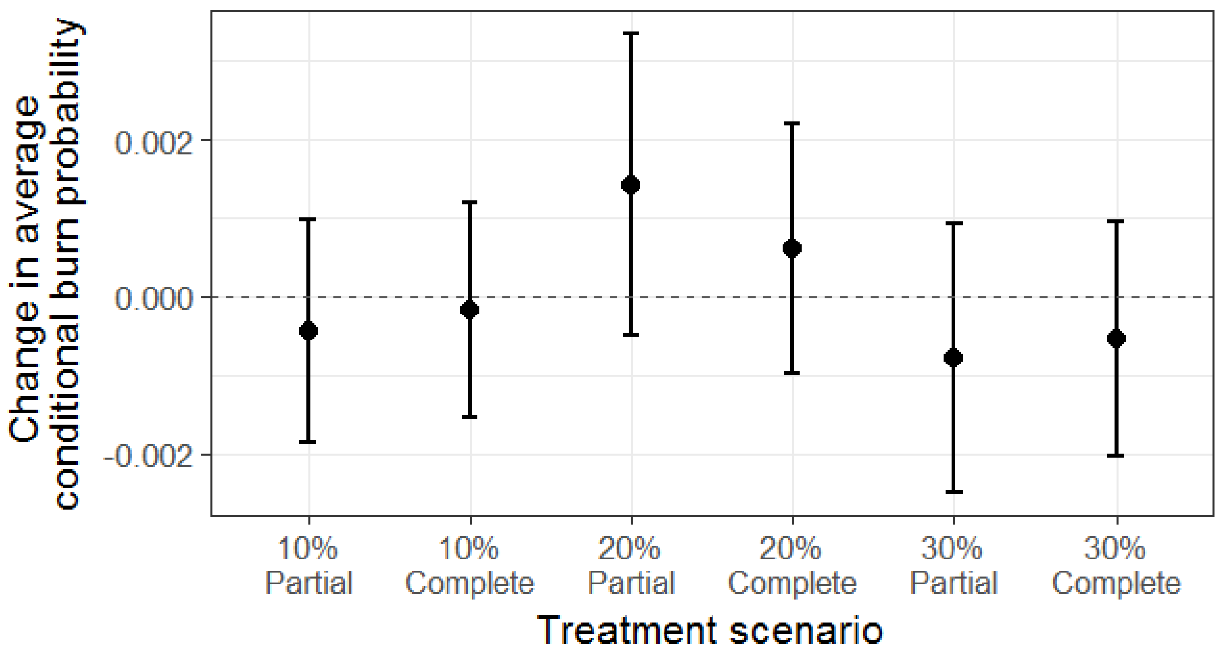


Figure 3.6: Changes in average conditional burn probability compared to the equivalent scenario with the same treatment level and no reserves. Black dots indicate point estimates while brackets indicate confidence intervals that reflect each observation's adjusted alpha level derived from the Benjamini-Hochberg procedure (1995) to control the false discovery rate at 5%.

Table 3.6: Estimated changes in expected habitat loss for all treatment scenarios, compared to the equivalent scenario with the same treatment level and no reserves. For example, the 10% / Partial scenario was compared to the 10% / No reserves scenario. P-values shown have been adjusted to control the false discovery rate at 5%, following procedures outlined by Benjamini and Hochberg (1995), and were considered significant if less than 0.05. Shading indicates a statistically significant difference. For example, a when 20% of the study area was treated and no treated was allowed in marten habitat, expected habitat loss increased by an estimated 171.1 ha compared to allowing treatment in marten habitat.

Treatment/reserve scenario	**Estimate (ha)**	**Std. Error**	**Adjusted p-value**
10% partial	21.7	16.96	0.217
20% partial	51.7	16.96	0.01
30% partial	25.5	16.96	0.18
10% complete	82.1	16.96	<.001*
20% complete	171.1	16.96	<.001
30% complete	77.2	16.96	<.001

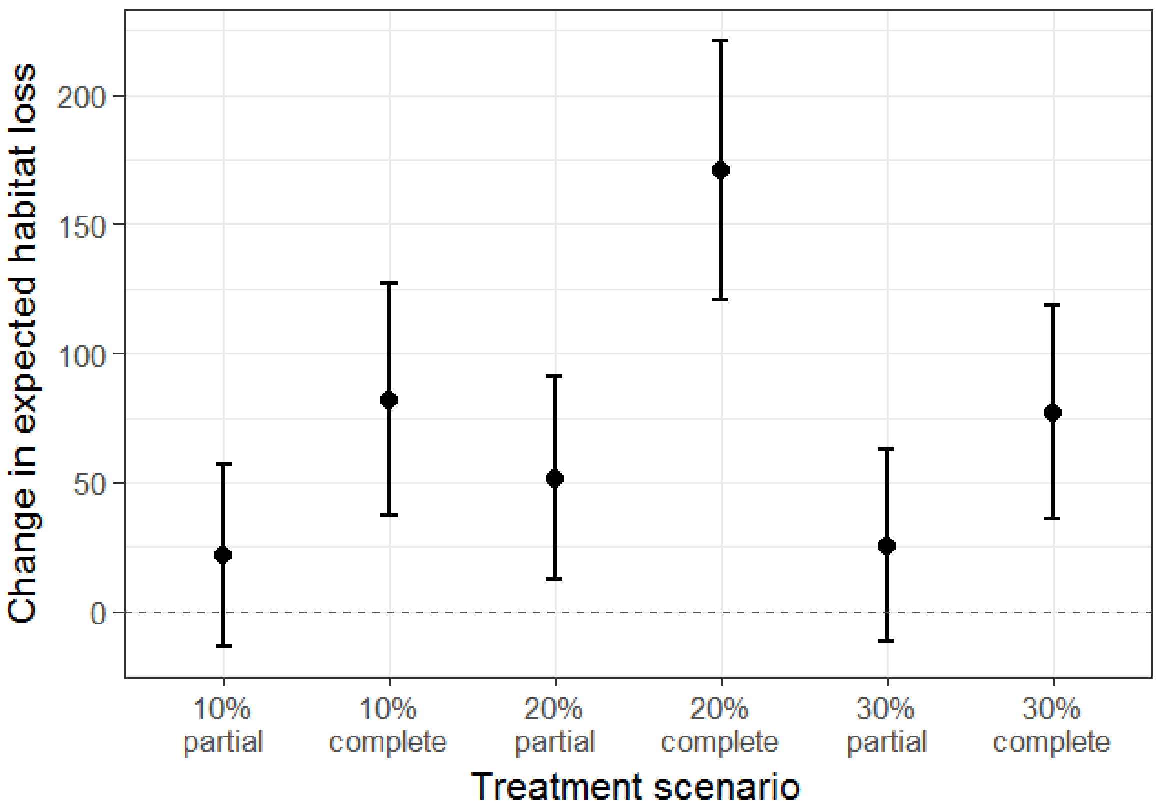


Figure 3.7: Estimated changes in expected habitat loss compared to the equivalent scenario with no reserves. Black dots indicate point estimates while brackets indicate confidence intervals that reflect each observation’s adjusted alpha level derived from the Benjamini-Hochberg procedure (1995) to control the false discovery rate at 5%.

Table 3.7: Summary of response variables for the three different treatment allocation strategies (random, the Treatment Optimization Method (TOM), and the habitat protection strategy (HPS)) for each level of reserves.

Allocation strategy	Reserves	Predicted habitat treated (ha)	Average conditional burn probability	Expected habitat loss (ha)
Random	None	3342.2	0.0234	334.6
Random	Partial	2180	0.0229	363.4
Random	Complete	0	0.0235	453.7
TOM	None	3254.8	0.022	316.4
TOM	Partial	2838	0.0234	368.1
TOM	Complete	0	0.0226	487.5
HPS	Partial	4155.5	0.0264	227.3
HPS	Complete	0	0.0258	389.3

Table 3.8: Estimated changes in average conditional burn probability for all treatment scenarios, compared to the equivalent scenario with the same treatment level and a random treatment allocation. For example, the TOM / Partial scenario was compared to the Random / Partial reserves scenario. P-values shown have been adjusted to control the false discovery rate at 5%, following procedures outlined by Benjamini and Hochberg (1995), and were considered significant if less than 0.05. Shading indicates a statistically significant difference. The Treatment Optimization Method placed treatments more efficiently than random placement only when treatments were permitted within marten habitat. The Habitat Protection Strategy (HPS) resulted in significantly higher burn probabilities compared to random placement.

Treatment/reserve scenario	**Estimate**	**Std. Error**	**Adjusted p-value**
TOM / None	-0.00139	0.00043	0.005
TOM / Partial	0.00047	0.00043	0.289
TOM / Complete	-0.00091	0.00043	0.055
HPS / Partial	0.00346	0.00043	<.001
HPS / Complete	0.00227	0.00043	<.001

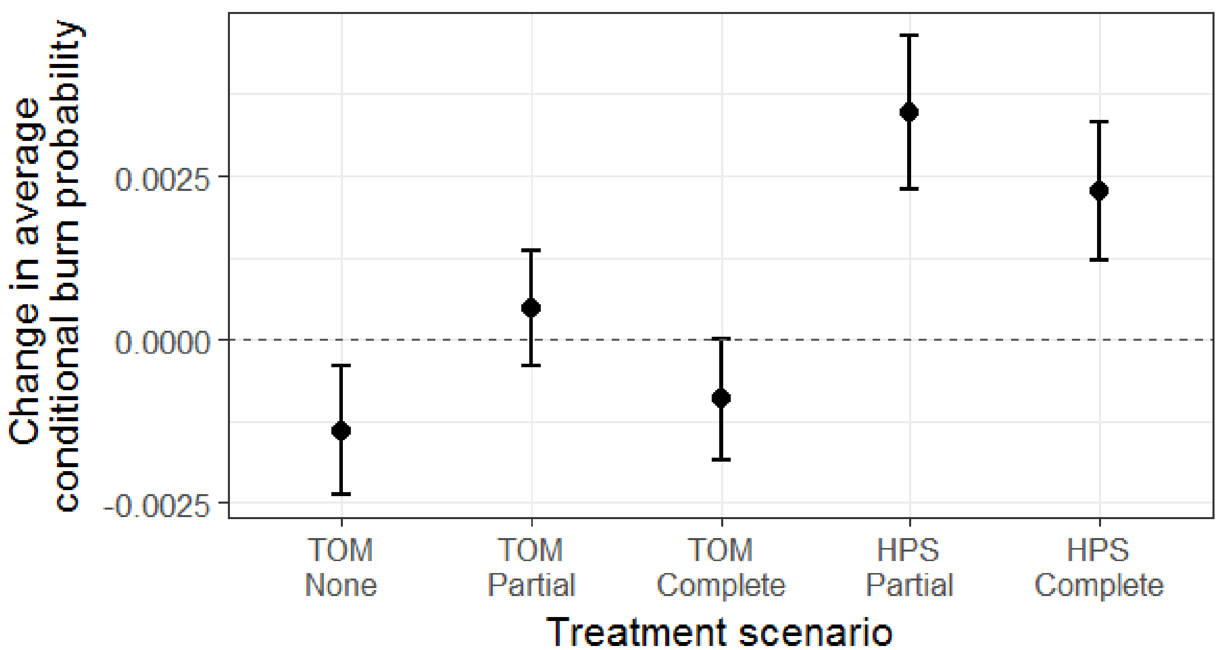


Figure 3.8: Estimated changes in average conditional burn probability compared to the equivalent scenario with the same treatment level and random treatment allocation. Black dots indicate point estimates while brackets indicate confidence intervals that reflect each observation's adjusted alpha level as derived from the Benjamini-Hochberg procedure (1995) to control the false discovery rate at 5%.

Table 3.9: Estimated changes in expected habitat loss for all treatment scenarios, compared to the equivalent scenario with the same treatment level and random treatment allocation. P-values shown have been adjusted to control the false discovery rate at 5%, following procedures outlined by Benjamini and Hochberg (1995), and were considered significant if less than 0.05. Shading indicates a statistically significant difference.

Treatment/reserve scenario	**Estimate (ha)**	**Std. Error**	**Adjusted p-value**
TOM / None	-18.3	17.18	0.368
TOM / Partial	4.7	17.18	0.786
TOM / Complete	33.8	17.18	0.095
HPS / Partial	-136	17.18	<.001
HPS / Complete	-64.3	17.18	0.002

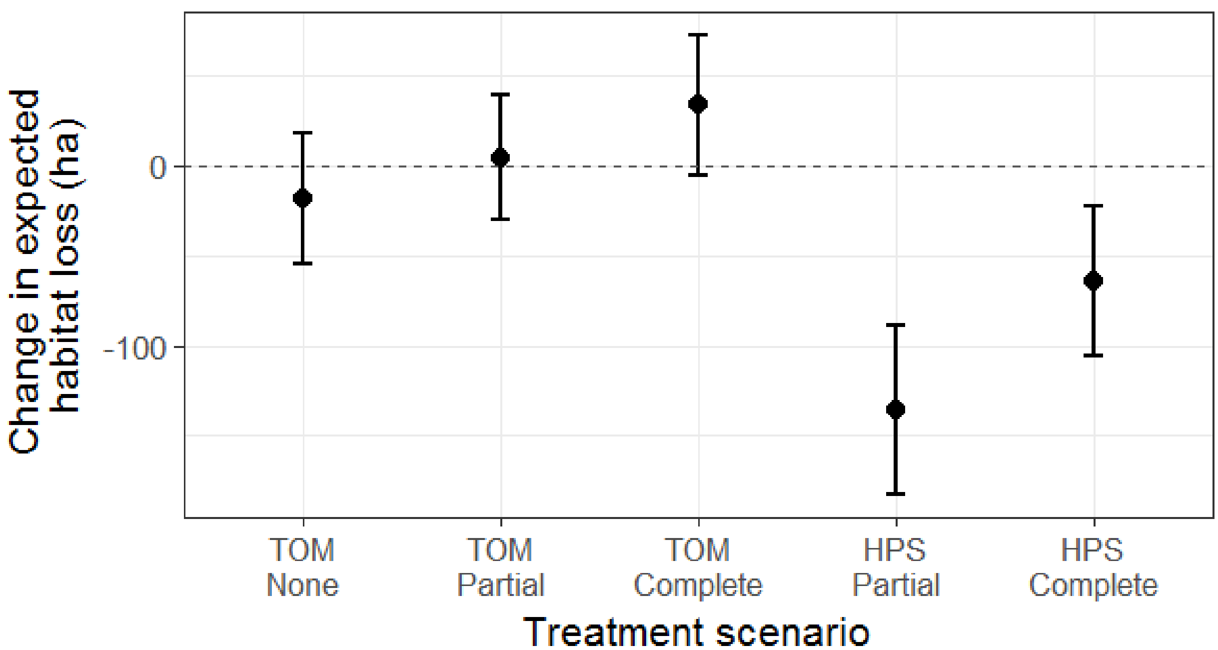


Figure 3.9: Estimated changes in expected habitat loss compared to the equivalent scenario with the same treatment level and random treatment allocation. Black dots indicate point estimates while brackets indicate confidence intervals that reflect each observation's adjusted alpha level derived from the Benjamini-Hochberg procedure (1995) to control the false discovery rate at 5%. The Treatment Optimization Method (TOM) had no significant effect on expected habitat loss compared to random treatment placement. The Habitat Protection Strategy (HPS) significantly reduced expected habitat loss compared to random treatment placement.

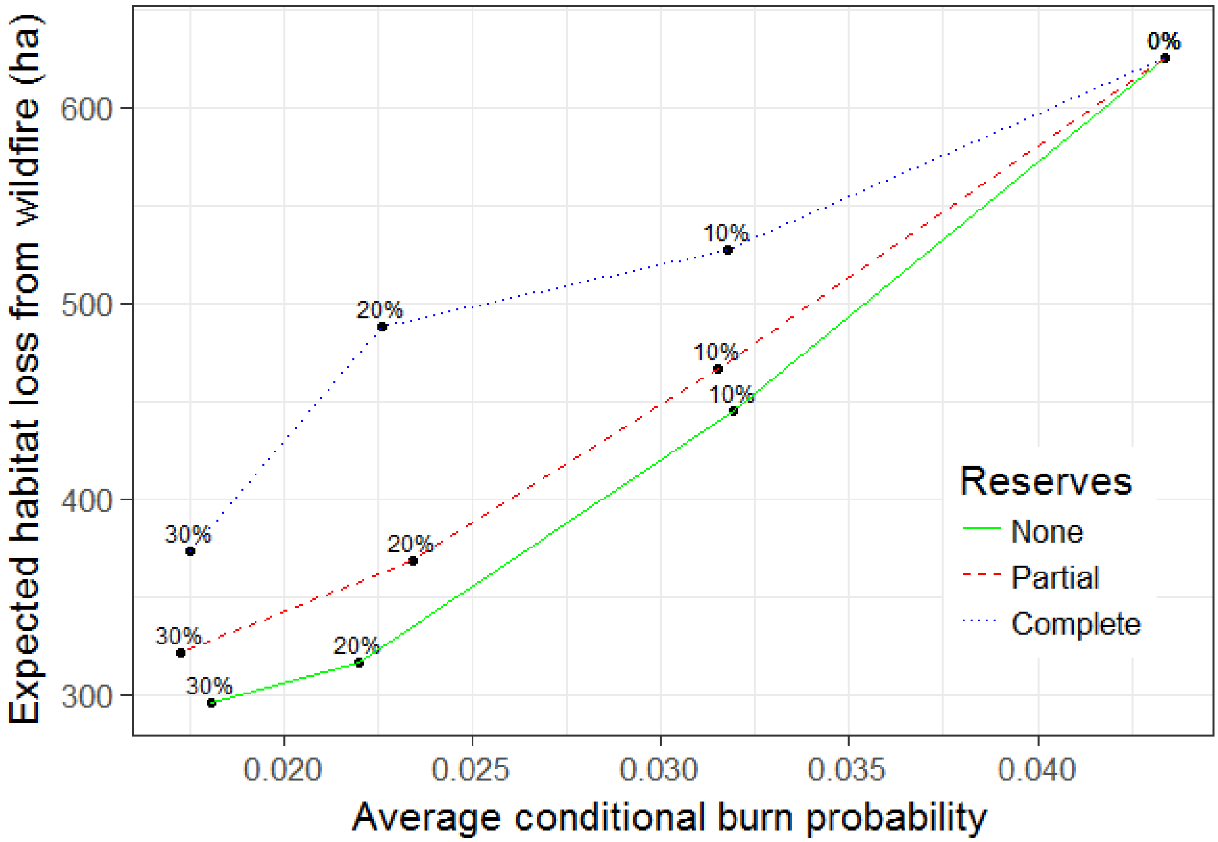


Figure 3.10: Point estimates for mean average conditional burn probability and mean expected habitat loss for the control (0% treatment) and each fuel treatment scenario. Text labels next to each point indicate the percent of the study area where fuel treatment was simulated. The lines show the progression from less fuel treatment (top right) to more fuel treatment (bottom left) for each habitat reserve strategy. Burn probabilities were similar whether or not fuel treatment was prohibited in predicted marten habitat. However, this figure shows the substantial increase in expected habitat loss when no fuel treatment was permitted in any predicted marten habitat under the complete reserves scenario (top dotted line).

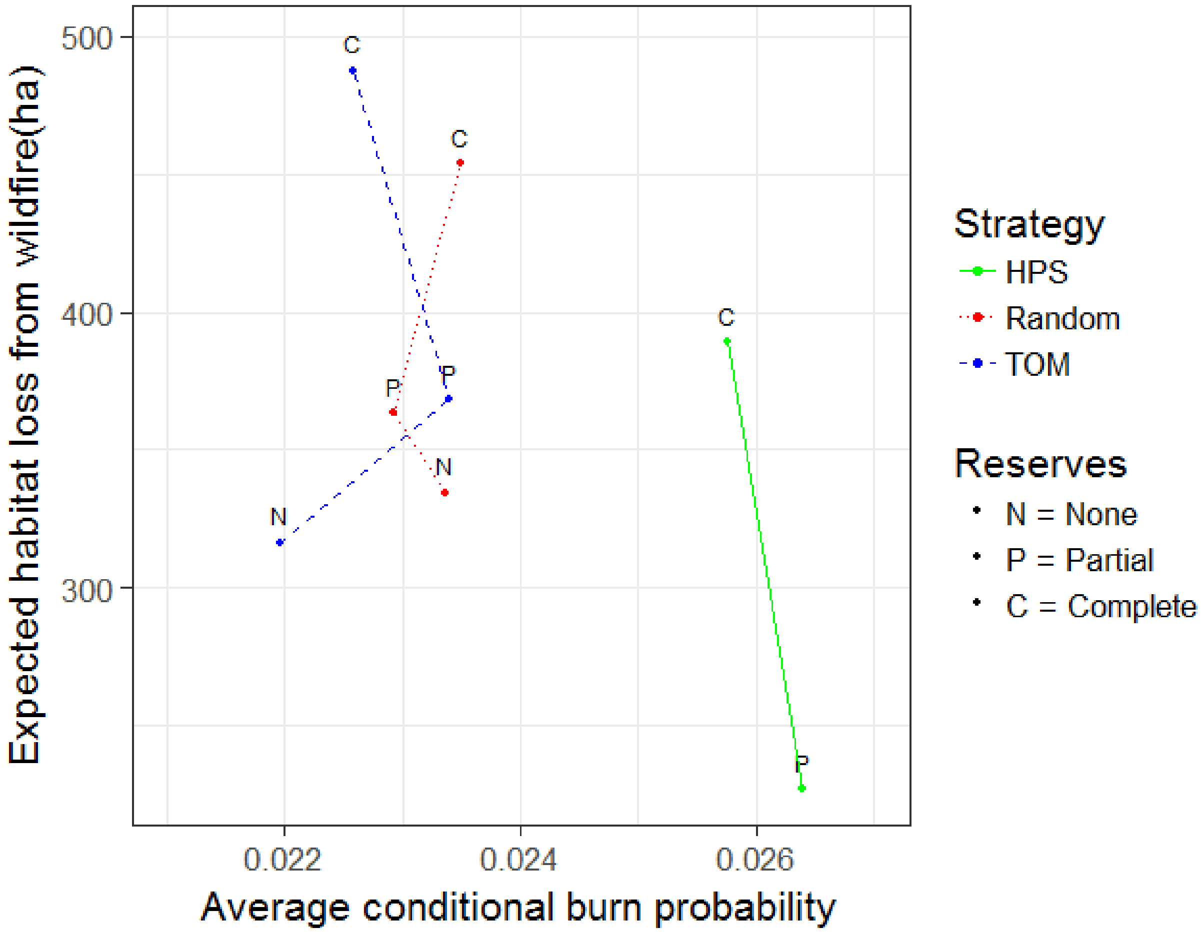


Figure 3.11: Point estimates for mean average conditional burn probability and mean expected habitat loss for each treatment allocation strategy. Since the habitat protection strategy (HPS) is based on conserving habitat areas by treating adjacent non-habitat, only partial and complete reserves were possible using that approach. The lines indicate increasing restrictions on fuel treatment placement, from no reserves (N) to partial (P) and to complete (C). The Treatment Optimization Method was marginally more effective at allocating fuel treatments to slow simulated fire spread with no reserves, while there was no significant difference from random allocations, on average, with partial and complete reserves. The Habitat Protection Strategy was clearly most effective at reducing wildfire risk to predicated marten habitat, but doing so sacrificed reductions in burn probability compared to the other strategies.

3.7 Chapter 3 | Literature Cited

Agee, J.K. 1993. Fire Ecology of Pacific Northwest Forests. 1st ed. Island Press. Washington, DC, USA.

Andersen, Hans-Erik, McGaughey, R.J., and Reutebuch, S.E. 2005. Estimating forest canopy fuel parameters using LIDAR data. Remote Sensing of Environment **94**(2005): 441-449.

Barros, A.M.G., Ager, A.A., Day, M.A., Preisler, H.K., Spies, T.A., White, E., R.J., Pabst, Olsen, K.A., Plat,, E., Bailey, J.D., Bolte, J.P. 2017. Spatiotemporal dynamics of simulated wildfire, forest management, and forest succession in central Oregon, USA. Ecology and Society **22**(1):24. https://doi.org/10.5751/ES-08917-220124

Bekker, M.F., and Taylor, A.H. 2001. Gradient Analysis of Fire Regimes in Montane Forests of the Southern Cascade Range, Thousand Lakes Wilderness, California, USA. Plant Ecol. **155**(1): 15–28. doi:10.1023/A:1013263212092.

Benjamini, Y and Hochberg, Y. 1995. Controlling the false discovery rate: a practical and powerful approach to multiple testing. J. of Royal Statistical Society: Series B (Methodological) **57**(1): 289-300.

Bissonette, J.A. 1997. Scale-sensitive ecological properties: historical context, current meaning. *In:* Wildlife and landscape ecology: effects of pattern and scale. *Edited by:* J.A. Bissonette. Pp. 3-31. Springer-Verlag, New York, New York.

Bissonette, J.A., Harrison, D.J., Hargis, C.D., and Chapin, T.G. 1997. The influence of spatial scale and scale-sensitive properties on habitat selection by American marten. *In* Wildlife and Landscape Ecooogy. *Edited by* J.A. Bissonette. Springer-Verlag, New York, NY.

California Department of Fish and Wildlife. California Interagency Wildlife Task Group. 2014. CWHR version 9.0 personal computer program. Sacramento, CA.

Collins, B.M., Stephens, S.L., Roller, G.B., and Battles, J.J. 2011. Simulating fire and forest dynamics for a landscape fuel treatment project in the Sierra Nevada. Forest Science **57**(2): 77-88.

Dixon, G.E. 2002. Essential FVS: A user's guide to the Forest Vegetation Simulator. Internal Report. USDA Forest Service, Forest Management Service Center. Fort Collins, CO. (Revised: April 14, 2017).

Dubayah, R.O., and Drake, J.B. 2000. LiDAR remote sensing for forestry. Journal of Forestry **98.6**(2000): 44-46.

Finney, M.A., 2002. Fire growth using minimum travel time methods. Canadian Journal of Forest Research: **32**(8): 1420-1424, 10.1139/x02-068

Finney, M.A. 2004. Landscape Fire Simulation and Fuel Treatment Optimization. *In* Methods for Integrated Modeling of Landscape Change: Interior Northwest Landscape Analysis System. Gen. Tech. Rep. PNW-GTR-610. *Edited by* J.L. Hayes, A.A. Ager, J.R. Barbour. USDA Forest Service, Pacific Northwest Research Station, Portland, OR. pp. 117–31.

Finney, M.A. 2006. An Overview of FlamMap Modeling Capabilities. *In* Fuels Management- How to Measure Success: Conference Proceedings. *Edited by* Andrews, P.L., and Butler, B.W. USDA Forest Service, Rocky Mountain Research Station. Fort Collins, CO.

Finney, M.A., Seli, R.C., McHugh, C.W., Ager, A.A., Bahro, B., and Agee, J.K. 2007. Simulation of long-term landscape-level fuel treatment effects on large wildfires. International Journal of Wildland Fire **16**(2007): 7612-727.

Hair, J.F. Jr., Anderson, R.E., Tatham, R.L. and Black, W.C. 1995. Multivariate Data Analysis, 3rd edition. Macmillan, New York.

Hall, S.A., Burke, I.C., Box, D.O., Kaufmann, M.R., and Stoker, J.M. 2005. Estimating stand structure using discrete-return LiDAR: an example from low density, fire prone ponderosa pine forests. Forest Ecology and Management **208**(2005): 189-209.

Harrell, F.E. Jr. 2001. Regression Modeling Strategies: With Applications to Linear Models, Logistic Regression, and Survival Analysis. Springer-Verlag, New York, New York.

Jones, G.M., Gutierrez, R.J., Tempel, D.J., Whitmore, S.A., Berigan, W.J., and Peery, M.Z. 2016. Megafires: an emerging threat to old-forest species. Frontiers in Ecology and the Environment **14**(6): 300-306.

Kirk, T.A., and Zielinski, W.J. 2010. Functional habitat connectivity of the American marten in northeastern California using least-cost corridor modeling. USDA Forest Service, Lassen National Forest and Pacific Southwest Research Station, Redwood Sciences Lab, California, USA.

LANDFIRE. 2014. LANDFIRE: LF 2014. Wildland Fire Science, Earth Resources Observation and Science Center, U.S. Geological Survey. Sioux Falls, SD. Accessed: February 2, 2017.

Levin, S.A. 1992. The problem of pattern and scale in ecology: The Robert H. MacArthur Lecture. Ecology **73**: 1943-1967. doi:10.2307/1941447.

Lumley, T., and Miller, A. 2009. leaps: regression subset selection. R package version 2.9. https://CRAN.R-project.org/package=leaps.

Maindonald, J.H. and Braun, W.J. 2015. DAAG: Data Analysis and Graphics Data and Functions. R package version 1.22. https://CRAN.R-project.org/pacakage=DAAG.

Martin, S.K., and Barrett, R.H. 1991. Resting site selection by marten at Sagehen Creek, California. Northwester Naturalist **72**:37 – 42.

McGaughey, Robert J. 2009. FUSION/LDV: Software for LIDAR data analysis and visualization. USDA Forest Service, Pacific Northwest Research Station. Seattle, WA.

Meyer, M.D. 2013. Natural range of variation of red fir forests in the bioregional assessment area. Unpublished report. USDA Forest Service, Pacific Southwest Region. Albany, CA.

Moghaddas, J.J., Collins, B.M., Kurt Menning, Moghaddas, E.E.Y., and Stephens,S.L. 2010. Fuel treatment effects on modeled landscape-level fire behavior in the northern Sierra Nevada. Can. J. For. Res. **40**(9): 1751–65. doi:10.1139/X10-118.

Moriarty, K.M., Epps, C.W., Betts, M.G., Hance, D.J., Bailey, J.D., and Zielinski, W.J. 2015. Experimental evidence that simplified forest structure interacts with snow cover to influence functional connectivity for Pacific martens. Landscape Ecol. **30**(10): 1865–77. doi:10.1007/s10980-015-0216-2.

Moriarty, K.M., Epps, C.W., and Zielinski, W.J. 2016. Forest thinning changes movement patterns and habitat use by Pacific marten. Jour. Wild. Mgmt. **80**: 621-633. doi:10.1002/jwmg.1060.

North, M., Brough, A., Long, J., Collins, B., Bowden, P., Yasuda, D., Miller, J., and Sugihara, N. 2015. Constraints on mechanized treatment significantly limit mechanical fuels reduction extent in the Sierra Nevada." *Journal of Forestry* **113**(1): 40–48. doi:10.5849/jof.14-058.

R Core Team. 2016. R: A language and environment for statistical computing. R Foundation for Statistical Computing. Vienna, Austria. URL: http://www.R-project.org/

Rebain, S.A. comp. 2010. The Fire and Fuels Extension to the Forest Vegetation Simulator: Updated Model Documentation. Internal Rep. USDA Forest Service, Forest Management Service Center. Fort Collins, CO. Revised March 23, 2015.

Rustigian-Romsos, H.L. and Spencer, W.D. 2010. Predicting habitat suitability for the American marten on the Lassen National Forest. Conservation Biology Institute. Corvallis, OR.

Sando, R.W. and Wick, C.H. 1972. A method of evaluating crown fuels in forest stands. Research Paper NC-84. USDA Forest Service, North Central Experiment Station, St. Paul, MN.

Schmidt, D.A., Taylor, A.H., and Skinner, C.N. The influence of fuels treatment and landscape arrangement on simulated fire behavior, Southern Cascade range, California. Forest Ecology and Management **255**(8-9): 3170-3184.

Scott, J.H, and Burgan, R.E. 2005. Standard fire behavior fuel models: a comprehensive set for use with Rothermel's surface fire spread model. Gen. Tech. Rep. RMRS-GTR-153. USDA Forest Service, Rocky Mountain Research Station. Fort Collins, CO.

Scott, J.H. and Reinhardt, E.D. 2001. Assessing crown fire potential by linking models of surface and crown fire behavior. Res. Pap. RMRS-RP-29. USDA Forest Service, Rocky Mountain Research Station, Fort Collins, CO.

Sierra Nevada Protection Campaign vs. Tippin. 2006. No. CIV. S 06-00351 FCD DAD. US District Court, Eastern District of California.

Skinner, C.N., and Chang, C. 1996. Fire Regimes, Past and Present. *In* Sierra Nevada Ecosystem Project: Final Report to Congress, Vol II: Assessments and Scientific Basis for Management Options. Wildland Resources Center Report No. 37. Centers for Water and Wildland Resources, University of California, Davis. pp. 1041–69.

Spencer, W.D. 1983. Marten habitat preferences in the northern Sierra Nevada. J. Wildlife Management. **47**(4): 1181-1186.

Spencer, W.D. 1987. Seasonal rest-site preferences of pine martens in the northern Sierra Nevada. J. Wildlife Manage. **51**(3): 616–21. doi:10.2307/3801279.

Spencer, W.D. and Rustigian-Romsos, H.L. 2012. Decision-Support Maps and Recommendations for Conserving Rare Carnivores in the Interior Mountains of California. Conservation Biology Institute, Corvallis, OR.

Taylor, A.H. 2000. Fire regimes and forest changes in mid and upper montane forests of the southern Cascades, Lassen Volcanic National Park, California, U.S.A. Journal of Biogeography **27**(1): 87-104. doi:10.1046/j.1365-2699.2000.00353.x

USDA Forest Service. 2004. Sierra Nevada Forest Plan Amendment: final supplemental environmental impact. USDA Forest Service, Pacific Southwest Region. Vallejo, CA.

USDA Forest Service, Almanor Ranger District, Lassen National Forest. 2015. Grizzly Restoration Project: Proposed Action. USDA Forest Service. Chester, CA.

USDA Forest Service, Plumas National Forest. 2013. Chips Fire Review. USDA Forest Service, Plumas National Forest, Quincy, CA.

USDA Forest Service. 2014. ExistingVegNor Interior1999_2009_v1. USDA Forest Service, Pacific Southwest Region Remote Sensing Lab. McClellan, CA.

Van Wagtendonk, J.W., and Fites-Kaufman, J.F. 2006. Sierra Nevada Bioregion. Book chapter. *In* Fire in California's Ecosystems. *Edited by* Sugihara, N.G., Van Wagtendonk, J.W., Shaffer, K.E, Fites-Kaufman, J., and Thode, A.E. University of California Press, Berkeley and Los Angeles, CA.

Weatherspoon, C.P. and Skinner, C.N.. 1996. Landscape-level strategies for forest fuel management.. *In* Sierra Nevada Ecosystem Project: Final Report to Congress, Vol II: Assessments and Scientific Basis for Management Options.

Wildland Resources Center Report No. 37. Centers for Water and Wildland Resources, University of California, Davis. pp. 1471-492.

4 | Conclusions and research implications

Mechanical treatments and prescribed fire are common tools used to reduce or remove accumulations of surface, ladder, and canopy fuels in forested areas and restore resilience to forests with unnatural fuel loading. However, decisions regarding the design of such silvicultural treatments, and where to execute them, require consideration of a vast array of distinct management objectives that often conflict with one another (USDA 2004, Jones et al. 2016). These objectives may include fire and fuels management, timber production, and recreation, as well as conversion of myriad wildlife species with their own habitat requirements. Projects intended to modify fuel accumulation with management intervention in the northern Sierra Nevada and southern Cascades have historically been controversial, particularly where sensitive wildlife were a concern, given that the full ecological effects of these treatments are not always well understood (Collins et al. 2010, Kalies & Kent 2016, Sierra Nevada Protection Campaign vs. Tippin 2006). The main objective of this study was therefore to provide an initial assessment of how two parameters of fuel treatment projects, their placement and extent, interact with restrictions related to one other management goal, the conservation of a sensitive Pacific marten population.

4.1 Principle findings

I used a simulation framework to analyze the effect of prohibiting fuel reduction treatments in predicted marten habitat at two scales, as well as at multiple levels of treatment area and reserves. Overall, restrictions on fuel treatment placement within marten habitat had a limited effect on their ability to control the spread of fire over an analysis area. However, at the project scale, this finding only held true if sufficient treatable area remained available after reserves were implemented— at the highest level of fuel treatment area, the most restrictive reserve strategy was capable of reducing the effectiveness of fuel reduction treatments compared to a policy without restrictions based on marten conservation. As additional constraints are added to fuel treatment placement, including protections for other sensitive wildlife species like the northern goshawk (*Accipiter* gentilis) or California spotted owl (*Strix occidentalis occidentalis*), complete prohibition of treatment in marten habitat may yield a greater influence on the efficacy of such treatments. At the landscape scale, however, implementation of habitat reserves had no significant effect on the capacity of the landscape to support large fire growth post-treatment, even under the most ambitious fuel treatment allocations. A wider view of treatment opportunities and potential fire behavior may therefore help managers resolve conflicts between objectives at the project scale by achieving a balance between them over the larger landscape.

Creation of marten habitat reserves had a substantial impact on wildfire risk to martens. Expected habitat loss increased when fuel reduction treatments were pushed out of predicted habitat areas, moving much of the benefit of reduced probability of crown fire away from the dense forests associated with marten use. Simulations showed that allowing fuel treatment in the lower quality habitat areas could largely eliminate this increased risk, without incursion into core areas where probability of occupancy was greatest. While it is difficult to weigh the short term negative impact on martens of simplifying forest structure versus the benefits of reduced wildfire risk, I found no justification for active fuels management in the core habitat of this marten population, up to 20% of the land base. Further, unnatural fuel accumulation and the homogenization of forest structure will only continue without disturbance or management intervention, processes that will increase the chances of larger patches of stand-replacing fire that would be most harmful to this marten population. Silvicultural prescriptions that can retain canopy cover and elements of old forest structure while also increasing resilience to fire will be useful for reducing wildfire risk while mitigating the short-term effects of management action. Group selection methods could allow for overall reductions in surface and ladder fuels, while also retaining dense patches or corridors of high canopy cover to facilitate marten connectivity.

4.2 Limitations

A fundamental limitation of a simulation approach is that the significance of any conclusions relies on the strength of the models that are employed and the validity of the input data. Each model used in this study has its own set of assumptions and inherent uncertainties. This research is not meant to provide specific values for management implementation; rather, I hope to identify broad trends in how landscape fire behavior interacts with treatments deployed at varying levels of habitat reserves, and establish a basis for comparing options for managing fuels where Pacific marten conservation is also a priority.

The minimum travel time algorithm for landscape fire behavior used in this study has been employed previously in similar efforts to assess fuel treatment strategies, both simulated and constructed in the real world (Moghaddas et al. 2010, Collins et al. 2011). While care was taken to validate the performance of this model, using the fuel parameters collected for this exercise as well as behavior of recent large fires in the region, field plots, and local knowledge, there still remains substantial uncertainty in how fire, and particularly crown fire, may respond to changes in forest structure. Further, ignitions were simulated over a limited burn time under constant weather conditions; in the real world, fire behavior is governed by a much larger suite of conditions that we expect to evolve given anticipated changes in climate (Westerling et al.

2006). As such, these results should always be interpreted with caution and in light of the uncertainties associated with understanding landscape fire behavior.

Another important limitation of this study is the scope of inference: can the results from these two planning units be extended to the rest of Lassen National Forest, the northern Sierra Nevada, and all Pacific marten habitat in North America? The fundamental role of local topography and site conditions must be acknowledged. The extent to which my results can be extrapolated to other localities depends on the magnitude of these potentially confounding influences. Lessons learned from this study will always need to be placed in the correct context of local fire behavior and Pacific marten ecology.

4.3 Future research opportunities

This study employed a simplistic modeling framework to explore the relationship between two potentially conflicting management objectives: Pacific marten conservation and fuels management. While a limited set of additional restrictions on treatment location were also considered (a minimum canopy cover, outside of riparian areas, and Forest Service ownership), in the real world management activities are limited by a much larger suite of competing goals and objectives, as well as logistical constraints. Further, the prioritization of management goals changes over time, evolving in response to new laws, policy shifts, and legal challenges. While the goal of this study was to understand interactions between fuel reduction treatments and conservation of marten habitat, applications of these results should ultimately consider those additional limits on management action that affect these decisions. Future efforts at understanding how habitat reserves affect landscape scale fuel management could incorporate a more robust set of additional constraints, and therefore more realistically representation current limitations on federal management action.

Another opportunity to refine these results could attempt to incorporate more realistic fire behavior, in conjunction with the post-fire configuration of burned patches. While this study focused on broad trends in fire spread rates and expected habitat losses, we know that martens respond not only to the total amount of resources available on the landscape, but also their spatial configuration. More detailed fire simulation could help identify the conditions that produce the types of fire effects most damaging to this sensitive carnivore population— namely large, contiguous patches of stand-replacing fire that disrupt current movement corridors in the landscape.

4.4 Chapter 4 | Literature Cited

Collins, B.M., Stephens, S.L., Moghaddas, J.J., and Battles, J. 2010. Challenges and approaches in planning fuel treatments across fire-excluded forested landscapes. J. Forest. **108**(1): 24 – 31.

Collins, B.M., Stephens, S.L., Roller, G.B., and Battles, J.J. 2011. Simulating fire and forest dynamics for a landscape fuel treatment project in the Sierra Nevada. Forest Science **57**(2): 77-88.

Jones, G.M., Gutierrez, R.J., Tempel, D.J., Whitmore, S.A., Berigan, W.J., and Peery, M.Z. 2016. Megafires: an emerging threat to old-forest species. Frontiers in Ecology and the Environment **14**(6): 300-306.

Kalies, E.L. and Kent, L.L.Y., 2016. Tamm Review: Are fuel treatments effective at achieving ecological and social objectives? A systematic review. For. Ecol. Manag. **375**(2016) 84-95. http://dx.doi.org/10.1016/j.foreco.2016.05.021

Moghaddas, J.J., Collins, B.M., Kurt Menning, Moghaddas, E.E.Y., and Stephens,S.L. 2010. Fuel treatment effects on modeled landscape-level fire behavior in the northern Sierra Nevada. Can. J. For. Res. **40**(9): 1751–65. doi:10.1139/X10-118.

Sierra Nevada Protection Campaign vs. Tippin. 2006. No. CIV. S 06-00351 FCD DAD. US District Court, Eastern District of California.

USDA Forest Service. 2004. Sierra Nevada Forest Plan Amendment: final supplemental environmental impact. USDA Forest Service, Pacific Southwest Region. Vallejo, CA.

Westerling, A.L.. Hidalgo, H.G., Cayan, D.R., Swetnam, T.W. 2006. Warming and earlier spring increase western U.S. forest wildfire activity. Science **313**(5789): 940–43. doi:10.1126/science.1128834.

Appendix

Table A.1: LiDAR-derived variables for predictive modeling of canopy fuel parameters.

Variable name	**Description**
las13_slp30	Percent slope at the 30m grid cell level.
las13_dem30	Elevation (m) at the 30m grid cell level.
las13_asp30	Aspect (azimuth degrees) at the 30m grid cell level.
Elevminimum	Minimum point height*
Elevmaximum	Maximum point height*
Elevmean	Mean point height*
Elevmode	Mode point height*
Elevstddev	Standard deviation of point heights*
Elevvariance	Variance of point heights*
ElevCV	Coefficient of variation of point heights*
Elevkurtosis	Kurtosis of point heights*
ElevAAD	Average absolute deviation of point heights*
ElevP01	Percentile LiDAR point height value (1)*
ElevP05	Percentile LiDAR point height value (5)*
ElevP10	Percentile LiDAR point height value (10)*
ElevP20	Percentile LiDAR point height value (20)*
ElevP25	Percentile LiDAR point height value (25)*
ElevP30	Percentile LiDAR point height value (30)*
ElevP40	Percentile LiDAR point height value (40)*
ElevP50	Percentile LiDAR point height value (50)*
ElevP60	Percentile LiDAR point height value (60)*
ElevP70	Percentile LiDAR point height value (70)*
ElevP75	Percentile LiDAR point height value (75)*
ElevP80	Percentile LiDAR point height value (80)*
ElevP90	Percentile LiDAR point height value (90)*
ElevP95	Percentile LiDAR point height value (95)*
ElevP99	Percentile LiDAR point height value (99)*
CanopyCoverFR	Number of LiDAR point heights > 2m / total number of LiDAR first returns.
Pamn	Percentage of first return points above the mean
Pamd	Percentage of first return points above the mode
VolumeCover	Mean LiDAR canopy height * (CanopyCover/100)
UnderstoryCover	(Number of points between 0.15 – 2.00m) / (Number of points between 0 and 2.00m)

Strcc_015	Percent canopy cover 0 - 0.15m (Number of points between 0 and .15m / number of points at 0)
Strcc_152	Percent canopy cover 1.5m – 2.0m (Number of points between 1.5m and 2.0m / number of points less than 2.0m)
Strcc_225	Percent canopy cover 2.0m – 2.5m
Strcc_253	Percent canopy cover 2.5m – 3.0m
Strcc_36	Percent canopy cover 3m – 6m
Strcc_612	Percent canopy cover 6m – 12m
Strcc_1218	Percent canopy cover 12m – 18m
Strcc_1824	Percent canopy cover 18m – 24m
Strcc_2430	Percent canopy cover 24m – 30m
Strcc_3036	Percent canopy cover 30m – 36m
Strcc_3642	Percent canopy cover 36m – 42m
Strcc_4248	Percent canopy cover 42m – 48m
Strcc_g48	Percent canopy cover greater than 48m

* Based on distribution of LiDAR points heights > 2m.

Table A.2: Descriptions of candidate models for canopy height using LiDAR-derived predictor variables.

Number of parameters	Explanatory variables	R-Squared	BIC	CP	Cross-validation error (tenfold-CV)	Maximum variance inflation factor
1	volumecover	0.84	-254.96	160.59	159.7	NA
2	elevmaximum, volumecover	0.9	-325.62	40.23	95.9	2.44
3	elevmaximum, elevkurtosis, volumecover	0.91	-330.92	29.98	90.6	2.47
4	elevp90, canopycoverfr, strcc_225, strcc_3036	0.91	-333.87	23.09	86.6	2.5
5	elevp90, canopycoverfr, strcc_225, strcc_612, strcc_1824	0.92	-343.95	9.5	79	10.9
6	elevmaximum, elevkurtosis, canopycoverfr, strcc_225, strcc_612, strcc_3036	0.93	-352.14	-0.88	73.4	5.38
7	elevmaximum, elevkurtosis, canopycoverfr, strcc_225, strcc_612, strcc_1824, strcc_3036	0.93	-353.22	-4.22	71.2	18.55
8	elevmaximum, elevmode, elevkurtosis, canopycoverfr, strcc_225, strcc_612, strcc_1824, strcc_3036	0.93	-351.94	-5.37	71.1	18.78
9	elevmaximum, elevmode, elevkurtosis, canopycoverfr, strcc_225, strcc_36, strcc_612, strcc_1218, strcc_1824	0.94	-350.48	-6.29	69.7	16.05
10	elevstddev, elevp10, elevp20, elevp80, elevp90, elevp99, canopycoverfr, strcc_612, strcc_1824, strcc_3036	0.94	-348.02	-6.35	70.2	193.8

Table A.3: Descriptions of candidate models for canopy base height on the log scale, using LiDAR-derived predictor variables.

Number of parameters	Explanatory variables	R-Squared	BIC	CP	Cross-validation error (tenfold-CV)	Maximum variance inflation factor
1	elevp05	0.39	-62.82	83.88	0.2	NA
2	elevp01, elevp30	0.43	-67.32	71.67	0.2	1.53
3	elevp01, elevp30, strcc_3036	0.49	-79.48	50.21	0.1	3.49
4	elevp01, elevp30, strcc_152, strcc_3036	0.53	-84.19	40.13	0.1	3.71
5	elevp01, elevp40, pamn, strcc_36, strcc_3036	0.55	-86.8	33.2	0.1	6.23
6	elevvariance, elevp01, elevp30, strcc_36, strcc_1218, volumecover	0.57	-89.16	27	0.1	10.02
7	las13_dem30, elevvariance, elevp01, elevp30, strcc_36, strcc_1824, volumecover	0.6	-92.55	20.13	0.1	11.29
8	las13_dem30, elevvariance, elevp01, elevp30, strcc_36, strcc_1218, strcc_2430, volumecover	0.61	-93.25	16.4	0.1	17.83
9	las13_dem30, elevvariance, elevp01, elevp20, elevp25, elevp30, strcc_36, strcc_1824, volumecover	0.63	-95.5	11.43	0.1	992.64
10	las13_dem30, elevvariance, elevp01, elevp20, elevp25, elevp30, strcc_36, strcc_1218, strcc_1824, volumecover	0.65	-96.45	7.95	0.1	992.65

Table A.4: Descriptions of candidate models for canopy bulk density (log scale) using LiDAR-derived predictor variables.

Number of parameters	Explanatory variables	R-Squared	BIC	CP	Cross-validation error (tenfold-CV)	Maximum variance inflation factor
1	canopycoverfr	0.51	-93.08	80.35	0.193	NA
2	las13_dem30, canopycoverfr	0.59	-113.82	46.44	0.164	1.03
3	las13_dem30, canopycoverfr, volumecover	0.64	-130.68	22.53	0.142	3.25
4	las13_dem30, las13_asp30, canopycoverfr, volumecover	0.66	-130.91	18.89	0.139	3.25
5	las13_dem30, canopycoverfr, strcc_253, strcc_1218, volumecover	0.67	-130.88	15.73	0.136	11.93
6	las13_dem30, las13_asp30, canopycoverfr, strcc_225, strcc_1218, volumecover	0.68	-130.89	12.7	0.133	10.35
7	las13_dem30, elevminimum, elevmode, canopycoverfr, strcc_253, strcc_1218, strcc_3036	0.69	-129.32	11.35	0.131	8.13
8	las13_dem30, las13_asp30, elevminimum, elevmode, canopycoverfr, strcc_225, strcc_1218, strcc_3036	0.69	-127.4	10.41	0.13	7.26
9	las13_dem30, las13_asp30, elevmaximum, elevstddev, elevp20, elevp90, pamn, strcc_36, volumecover	0.7	-127.13	8.02	0.129	90.32
10	las13_dem30, las13_asp30, elevmaximum, elevstddev, elevp20, elevp90, pamn, strcc_36, strcc_1218, volumecover	0.71	-124.67	7.72	0.128	93.56