Baseline Evaluation of
Fisher Habitat and Population Status
&
Effects of Fires and Fuels Management on Fishers
In the Southern Sierra Nevada

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This Final report replaces all previous drafts and progress reports submitted for this project. Model results included herein reflect refinements to various input variables, model assumptions, and parameter estimates, so *previous maps and model results should be discarded and replaced with contents of this report*.

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Foreword

> *The important thing in science is not so much to obtain new facts as to discover new ways of thinking about them.*
>
> Sir William Bragg

Despite considerable expenditure of public funds, and the strenuous efforts of scientists over the previous decade, we still know little about the ecological requirements of fishers and how they are affected by human activities. Yes, we know the habitats that fishers are associated with, we know what they eat, and we are confident about the sizes of their home ranges. The true measure, however, of our research success – at least in the realm of conservation – is not how much we know about fishers but how much we know about the *effects of management activities* on their domain. We cannot currently forecast the effects of roads, silvicultural practices, or off-road vehicles on fishers and, therefore, we cannot provide guidance as to the amount or intensity of these disturbances that a viable fisher population can tolerate. The first generation of studies on any species focuses on the easier, descriptive questions. It isn't until we have a good foundation about the ecology of the species that we begin to tackle the thornier questions like: What is the effect of activity *X* on fishers and their habitat? How much of habitat *X* is necessary to maintain a well-distributed population of fishers? These require more resources, more creativity, and often are asked in political environments where the answers have economic consequences and affect resources.

We are just beginning to enter the second generation of fisher studies in California and the western United States, and none too soon. The fisher was found by the USFWS to be warranted for federal listing in the west coast states and a decision about listing the fisher by the state of California is pending. Thus, the stakes are high for understanding how fishers will be affected by various plans for public and private lands.

Here's the current conundrum: How do we simultaneously protect human communities and forest resources from uncharacteristically severe fire while at the same time protecting the dense forest conditions that fishers favor from potentially negative effects of thinning? Can we thin forests to the point that fire intensities are significantly reduced without also rendering them inhospitable to fishers? These are not trivial questions, and the current fisher population doesn't provide much room for mistakes: there are probably fewer than 150 female fishers in the southern Sierra Nevada population, which is isolated from other populations to the north. Answering these questions using traditional experimental methods is unlikely: landscape habitat conditions are incredibly variable, individual fishers can use over 20 mi^2 each, and sample sizes are usually statistically inadequate. Add to these constraints the fact that an individual study can cost a half a million dollars.

What to do in the face of these odds? The US Forest Service Pacific Southwest Region (Region 5) chose to contract with the Conservation Biology Institute (CBI), a quantitatively inclined non-governmental organization, to establish a baseline description of fisher habitat and population size and to use this information to evaluate various management scenarios using field data and simulations.

I could see the necessity of this choice coming for years. Planning and analysis by Region 5 staff was regularly challenged – by outside entities and by scientists in the Forest Service's own research branch. Moreover, internal analysts were viewed by some stakeholders as potentially biased. And, scientists within the research station did not have the trust of other stakeholders. The political heat was turned up when several important forest management projects in the southern Sierra were denied by a judicial decision because the effects on fishers had not been adequately considered. A reasonable, but unprecedented approach was brokered: select a non-profit research organization to attempt the challenge of reconciling fishers, forestry, and fire.

You have before you the results of this remarkable enterprise – one that would have been very difficult to achieve if led by either the management or the research branch of the Forest Service. It was remarkable in that one entity, CBI, managed the scientific input, the diverse input from stakeholders, *and* conducted much of the analysis. I watched when Jim Strittholt and Wayne Spencer first realized, with anxiety, the magnitude of the challenge. But I watched also, with equal fascination, how they strategized to define and parcel the work and recruited technical staff to address specialized requirements of their new challenge.

It was exciting. I attended many meetings and the atmosphere was rich with brainstorming, problem solving, questioning, and frustration. That this excitement came from discussions involving diverse representatives from forest management, environmentalists and scientists foreshadowed CBI's success. I had not participated in such productive sessions in any of the previous fisher-related initiatives or assessments. In the CBI effort, I believe we have an example of a new process in addition to an example of a new wildlife risk analysis in California. Read this report and share it with your colleagues; it's much more than about fishers, forestry, and fire – it is an example of how an open process in a contentious environment got the job done.

Before CBI's Baseline Evaluation, opinions about the effects of fire and fuels treatment on fishers were flying this way and that; nearly everybody had one. What was necessary was a scientific platform from which to test them. CBI helped the Forest Service with two very important goals: (1) establishing a baseline definition of fisher habitat and fisher numbers, and (2) using this information to develop a format for determining how fisher habitat could be restored, while also reducing future fire intensity. And, for the first time in the history of fisher management, I sensed a shift in the burden of proof. Previous objectives were driven by a desire to conduct management project *X*, and the effects on fishers were considered only later. The current effort puts fishers first and asks the question: Of the possible approaches to forest management, which most benefits fishers? Society has demanded this of public land managers and, in my opinion, CBI has delivered. But let's view this as a first step, and one that is not perfect. As the process of fisher conservation continues, new questions will need to be answered. Thankfully, however, the transparent process that was central to this project has resulted in new models, tools, and insights that will make addressing the new questions infinitely easier. Although this is a first step, having taken it makes the distance to our mutual goals for fisher conservation seem much shorter.

William J. Zielinski

May 30, 2008

Arcata, California

Acknowledgements

This project was a team effort, with extensive input, guidance, and review by numerous parties. Due to the large and somewhat revolving list of participants, we apologize for any inadvertent omissions from these acknowledgements.

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Richard Truex provided the fisher monitoring data and abundant insights about modeling, methods, and fisher biology. Sylvia Haultain provided vegetation plot data and Pat Lineback provided fire data for Sequoia-Kings Canyon National Park. Andrew Gray assisted with calculations of fisher resting microhabitat value using Forest Inventory and Analysis data. Karen Waddell assisted us in obtaining and understanding the FIA data.

Our technical work was overseen by an experienced group of science advisors: William Zielinski, Frank Davis, Reginald Barrett, Carl Skinner, Scott Stephens, John Vankat, Jan Van Wagtendonk, David Graber, Keith Aubry, Bob Heald, Malcolm North, and David Mladenoff. Their wisdom has added considerable value to this work (although the final selection of methods, interpretation of results, formulation of management recommendations, and any inaccuracies in this report are CBI's alone). Special gratitude goes to Bill Zielinski for his unwavering guidance and unparalleled knowledge about fishers and habitat selection; Frank Davis for clear and wise guidance to our habitat modeling efforts; Reginald Barrett for his insights about fisher habitat and populations; Keith Aubry for his constant reminders that we should strive to address the full range of fisher life requisites; David Graber for assistance in understanding forest management in the National Parks and with obtaining data for Sequoia-Kings Canyon National Park; David Mladenoff for help in understanding the world of vegetation dynamics modeling; Bob Heald for his knowledge of Sierran forests and insights on the need to factor climate, drought, and insect outbreaks into the equation; Carl Skinner and Jan Van Wagtendonk for supplying their extensive knowledge of forest and fire ecology in the Sierras; John Vankat for his insights about forest ecology and his careful reading of early drafts of this document; Scott Stephens for insights on fire ecology and for refining assumptions for modeling fire and fuel dynamics; Malcolm North for help defining old tree variables and discussions concerning forest succession, and Peter Stine for advice on data sources, uncertainties, and assumptions in modeling.

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Executive Summary

This report assesses the status of fishers (*Martes pennanti*) in the southern Sierra Nevada and presents an integrated simulation modeling approach used to assess how fisher habitat and populations may be affected by wildfires and by vegetation management intended to reduce the size and severity of wildland fires. The results provide useful information to consider in Fireshed Assessments and cumulative effects analyses for site-specific project plans.

Perhaps more importantly, the integrated simulation approach we developed for analyzing alternative futures provides a means for building and analyzing ever more refined and useful scenarios. Future scenarios could incorporate, for example, the effects of climate change, insect outbreaks, diseases, or droughts on fires, forests, and fishers — or a variety of other species for that matter. The limited range of possible future fire regimes and management approaches analyzed in this report could be greatly extended, in an adaptive management framework, to simulate effects of an unlimited array of vegetation management strategies and assumptions about ecological processes on fishers or other at-risk species. And, as we continue to learn more about fishers, forests, and fires, as new information accumulates about climate change and other critical processes, and as new or refined management approaches are proposed, these integrated simulation models could be used to test and compare alternative futures — and to work continually toward developing optimal solutions to our complex and ever-changing forest management problems.

General Approach — To assess the current status of fishers and to predict future changes in their habitat and population viability under various management and fire scenarios, we coupled a landscape change model that simulates the effects of fires (and other disturbances), forest management actions, and ecological succession on forest vegetation at regional scales (LANDIS-II) with a landscape-level fisher habitat model (LAND8) and a spatially explicit population model (PATCH). The study area focuses on the Stanislaus, Sierra, and Sequoia National Forests, but also encompasses Yosemite and Sequoia-Kings Canyon National Parks as well as miscellaneous other lands adjacent to these parks and forests.

Habitat Assessment — We predicted fisher probability of occurrence (sometimes referred to as habitat value)[1] at the landscape scale using empirically derived Generalized Additive Models (GAM) built using a wide array of environmental data layers and the regional fisher monitoring data compiled by the Pacific Southwest Region of the USDA Forest Service (Region 5). The final, best model (out of more than 250 alternative models we compared) predicts fisher probability of occurrence at the landscape scale based on three predictors averaged over 5-km^2 moving windows: latitude-adjusted elevation, average annual precipitation, and total above-ground biomass of trees. The model provides a strong statistical fit to the fisher monitoring data (more than 94% correct classification of presence vs. absence points) as well as to an independent data set and several other measures. It reaffirms and adds to existing understanding

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[1] Although we use the phrases *probability of fisher occurrence* and *fisher habitat value* interchangeably throughout this document, technically the former is more correct. Fishers are sometimes detected in areas of low habitat value, or may not be detected in areas of high value. However, in general, fishers are more likely to occur in areas of higher value.

of fisher habitat and distribution in the study area, and it demonstrates that fishers in the southern Sierra Nevada are strongly associated with mid-elevation, mixed-coniferous forests with high density and/or abundant large trees.

We also assessed the availability of fisher resting microhabitat based on an existing published model using Forest Inventory and Assessment data at FIA plots. This analysis reaffirms previous assessments suggesting that high-quality resting microhabitats are relatively rare and may be limiting to fishers. Only 5% of the FIA plots in forested areas had a suitability rating >0.5 (on a scale of 0 to 1).

Population Assessment — We next coupled the population dynamics model PATCH with the current-day probability of occurrence map to estimate an equilibrium fisher population size (or carrying capacity) and to identify potential population source, sink, and expansion areas. PATCH simulates spatially explicit population processes (births, deaths, and dispersal) that are affected by habitat quantity, quality, and arrangement, and stochastic events. This allowed us to test the relative effects of different assumptions on fisher population size, distribution, and viability.

Based on current habitat status and a biologically realistic range of assumptions, the baseline carrying capacity estimate from this simulation analysis ranged from 230 to 392 individuals (ignoring juveniles) of which 73 to 147 are adult females (the most critical class for sustaining a population).

We compared this estimate with two independent estimates of population size derived by (1) extrapolating field-measured fisher densities in the Kings River Project area[2] and (2) extrapolating fisher annual occupancy estimates based on sampling theory applied to the regional fisher monitoring dataset[3]. The density-extrapolation method estimated 276 to 359 fishers (ignoring juveniles), of which 55 to 83 are adult females. The sampling theory extrapolation estimated the total population at 160 to 250 individuals.

Based on the relative congruence of these three independent estimates, and balancing the various assumptions and uncertainties inherent to each method, we conclude that the total population of fishers in the southern Sierra Nevada is most likely *between 160 and 360 total individuals, of which only about 55 to 120 are adult females — the most important class for sustaining a population*. Isolated wildlife populations this small are at risk of extirpation from a variety of causes, including stochastic events.

Further simulation experiments suggested that northward expansion of the fisher population, from the Sierra National Forest onto the currently unoccupied Stanislaus National Forest, has the greatest potential to increase fisher population size and viability. However, there is uncertainty about the likelihood of such expansion occurring naturally, due to potential dispersal impediments (e.g., steep canyon slopes, open habitats, the Merced River, and heavily traveled roads). *We therefore recommend a focused analysis of habitat connectivity and the potential for fisher movement across the Merced River Canyon.*

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[2] From Mark Jordan's MS Thesis, 2007, UC Berkeley.

[3] Estimates provided by Richard Truex, the USDA Forest Service Carnivore Monitoring Team Leader for the Sierra Nevada.

Simulation Methods for Effects of Fires and Fuels Management on Fishers – Changes in forest composition and structure due to historical land-management actions and fire suppression have fueled concerns that Sierra Nevada forests are now prone to uncharacteristically large and severe fires, which threaten both human communities and natural resources—including fisher habitat. However, management actions proposed to counter these trends and sustain and restore fire-adapted ecosystems have been controversial, in part due to concerns that the treatments themselves may harm fishers and other resources.

Therefore, to assess the effects of fires and forest management on forest vegetation and fisher habitat, we refined, parameterized, and calibrated the LANDIS-II vegetation dynamics model to simulate Sierra Nevada fuel characteristics, forest succession, management prescriptions, and fire behavior. We defined a Baseline Fire regime based on recent fire histories, and a High Fire regime that anticipates an increase in fire size and severity in the near future due to altered forest condition, climate change, or other factors.

Because LANDIS-II tracks changes in forest biomass by tree species and age cohorts over time, we coupled its outputs with the fisher habitat model (which selected forest biomass as a strong predictor of habitat quality). This allowed us to simulate changes in fisher habitat quality over time under various assumptions about fire regimes and management actions. Coupling the changing habitat-value maps with the population dynamics model, PATCH, then allowed us to investigate how these assumptions about forest management and fires might affect fisher populations in the future.

We used an experimental design that allowed us to systematically explore the relative and absolute effects on fishers of alternative fuel treatment rates (proportion of treatable landscape treated per time step), intensities (two prescription combinations: moderate thin + prescribed fire, and light thin + prescribed fire), locations (inside fisher habitat, outside fisher habitat, or both), and fire regimes (baseline or high fire). This factorial design allowed us to investigate how these factors, alone or in combination, may affect fisher habitat and populations. For every combination of factors, we simulated 50 years of forest change using LANDIS-II and replicated each combination 10 times to account for stochastic events. Each decade, the updated landscape map was imported into the fisher habitat model, and the fisher population model was run for the ensuing 10 years. We continued the simulations for a total of 50 years of landscape change and 60 years of population change (10 years after the final landscape change). To compare the effects of different combinations of factors on fishers, we compared the ending fisher population sizes (year 60).

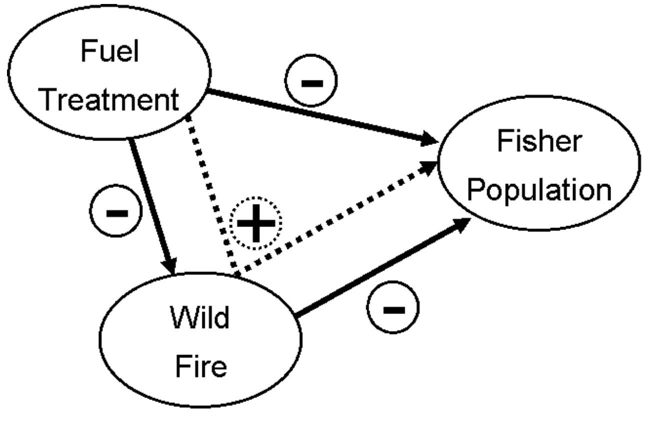

Figure ES-1. Expected interactions among fuel treatments, wildfire, and fisher populations in the southern Sierra Nevada. Solid lines represent direct effects; the dashed line represents an indirect effect.

Because fires and fuel treatments both reduce forest biomass (which is highly predictive of fisher occurrence), either might have direct negative effects on fishers. However, because fuels treatments might also reduce the negative effects of fires on fishers (by reducing the extent and severity of fire over time), they might also have an indirect, positive effect. We tested these four sets of hypotheses with our simulation experiments, illustrated in Figure ES-1, to determine the circumstances under which the overall positive effects on fishers may outweigh the potential negative effects.

In addition to tracking changes in overall habitat quality and fisher population size through these simulations, we also tracked changes in the abundance of young and old age cohorts of specific tree species that are considered important for providing fisher habitat elements (e.g., large, old black oaks, firs, and pines that provide resting and denning structures).

Results of the Simulation Experiments — Results of the simulations demonstrated that the treatments we tested may effectively reduce the extent and severity of fire on the landscape over a 50-year time span, and that, given the right combinations of treatment rate, intensity, and location, the indirect benefits to fishers can outweigh the localized, direct negative effects. In general, treatment effectiveness (for both fires and fishers) was greater under the high fire regime than under the baseline fire regime, largely because there is an increased probability of wildfires actually encountering treatments, and thus having their spread rates reduced by them, under the heightened regime. This suggests that if the anticipated increase in severe fire conditions occurs (due to changing fuel characteristics, weather conditions, and/or other factors), vegetation management may be even more important to implement sooner rather than later.

Both the area of treatment and the intensity of treatment affected fire spread rates and the total amount and severity of fire, at least when 4% to 8% of the treatable landscape was treated every 5 years[4] and for the two sets of fire prescriptions (intensities) that we modeled. However, extrapolating beyond these results is not advised. Just because more extensive or intensive treatment were more effective under certain simulated conditions does not mean that more is always better.

The ending year-60 fisher populations were always lower under the high fire regime than for the baseline fire regime. When no treatments were simulated for the high fire regime, simulated fisher populations declined substantially from baseline levels due to fires removing large amounts of biomass and fragmenting the habitat. No such dramatic declines were seen under the baseline fire regime. When fires were simulated along with the fuel treatments, the reduction of fire due to treatments compensated for the direct negative effects of treatments, at least under the high fire regime. In general, the relative influences of treatment rates, intensities, and locations on fishers were subtle relative to the difference between no treatment and some treatment.

Across the broad scales examined, and the assumptions we were provided about how treatments affect fuel characteristics and hence fire spread rates and severity, and within the finite combinations of fire regimes and treatments tested, we conclude that the longer-term positive effects of fuel treatments (due to the reduction of fire hazard) outweigh the short-term negative

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[4] Pilot simulations over a subset of the study area treating only 2% every 5 years proved ineffective at reducing fire. We cannot extrapolate to higher rates of treatment (>8% every 5 years) because this was not tested.

effects of fuel treatments (due to immediate loss of biomass) on fisher. Across all the simulations, managing fuels generally had positive effects on the simulated fisher populations, especially under the heightened fire regime. However, if actual fuel treatments remove substantially more biomass than those we simulated, or if actual treatments do not reduce fire spread rates and severity to the degree assumed in our models, then we would expect to see more parity between the positive and negative effects of fuel treatment on fisher. Conversely, if a greater reduction in fire spread rate and fire severity could be achieved with more strategic removal of less biomass, the positive effects of fuel treatment could be greater than our results suggest. Less area could be treated if the fuel treatments were strategically clustered in areas at high risk of fire, and in and near fisher habitat (as opposed to far outside fisher habitat).

Under the baseline fire regime, there was no significant benefit to fishers of the simulated treatments, at least considering average trends across the landscape as a whole. However, when treatment effects were parsed out separately by fire region (using structural equation modeling), there was a weak but significant positive effect of treatments within the higher elevation region. This may be due in part to the generally faster fuels there (and hence greater relative effectiveness of treatments in reducing spread rates). Also for the baseline fire regime, treatments located inside fisher habitat had a significant, positive effect on fisher numbers that more than compensated for the slight negative effects of those treatments. Treatments outside fisher habitat under the baseline regime had no significant effect on simulated fishers. This is apparently because the treatment effects on fire spread are relatively local, such that treatments far outside fisher habitat had little influences on fires inside fisher habitat.

The benefits of treatments on fishers were stronger under the high fire regime than under the baseline fire regime. Because the high fire regime produced larger fires on average than the baseline regime, treatments outside fisher habitat had a greater potential to influence fires that might burn into fisher habitat from the outside, and they did so without a strong, direct, negative effect on fisher habitat. Therefore, while treatments inside fisher habitat may be overall more effective in protecting fisher habitat, treatments outside fisher habitat could yield some additional benefits to fisher, while having little direct, negative impact on fishers. In weighing trade-offs between positive and negative effects of treatments, strategic placement outside of fisher habitat appears most beneficial under the high fire regime, where we observed the best ratio of positive to negative effects.

We were not able to realistically simulate strategic placement of treatments (i.e., clustered treatment arrays or “Finney patterns”) that could respond dynamically to the locations of previous treatments or fires, and thereby maximize landscape-level effectiveness over time. However, it may be possible to treat less area if treatments are located strategically to maximize effectiveness.

Management Recommendations – The integrated simulation approach we developed for analyzing alternative possible futures provides a means for building and analyzing ever more refined and useful scenarios and for testing specific hypotheses about the effects of various disturbance regimes and management approaches. Future analyses should incorporate effects of climate change, insect outbreaks, diseases, and droughts on fires, forests, and fishers. As more information accumulates from monitoring and research, these models should be refined and used to test additional hypotheses in an adaptive management framework. The ultimate goal should be to work toward optimal solutions that best balance the array of management goals that must be met on National Forests, including sustaining fishers and other at-risk species while minimizing fire hazards to human communities and natural resources.

In the meantime, the results of our analyses support the following recommendations for forest management:

- The fisher population in the southern Sierra Nevada is small (160 to 360 individuals) and at risk of extirpation by stochastic events, including uncharacteristically large or severe wildfires that could fragment habitat and isolate fishers in smaller areas. Fuels management should prioritize treatments in areas at highest risk of large, severe wildfire that could move through the relatively narrow band of fisher habitat or widen gaps between current habitat segments (e.g., near major river canyons).
- Fisher resting microhabitats appear to be rare. Management should maintain and if possible increase the distribution and abundance of forest stands that provide high-value resting sites, especially within predicted fisher occupancy areas. We recommend deriving and analyzing spatial rules for the size and continuity of dense forest patches that may best retain fisher resting habitat value while minimizing risks of crown fire.
- Increasing the amount or value of fisher habitat south of the Merced River may increase fisher population size and viability and should not be precluded by management decisions. Northward expansion of the population onto the Stanislaus National Forest could significantly increase population size and viability. We recommend a focused analysis of habitat connectivity across the Merced River Canyon.
- Treating 4% to 8% of treatable area every 5 years can significantly reduce fire sizes and fire severity and thereby benefit fishers. Strategic placement of treatments may decrease the necessary treatment area. The Stewardship and Fireshed Assessment process should continue identifying and evaluating areas of high fire risk that overlap with fisher habitat to improve treatment strategies.
- Fuel treatments should use site-specific analyses that consider fisher habitat value in and near the treatment. Within fisher habitat, treatments should balance desired fuel conditions with maintaining sufficient overstory and habitat elements to sustain or encourage occupancy by fishers. Removing larger trees and other essential habitat elements should generally be avoided within fisher habitat, to the degree feasible while meeting fuel reduction and landscape vegetation management goals.
- Placing treatments inside fisher habitat is not necessarily detrimental to fisher populations, although they may adversely affect some individuals locally. The positive indirect effect of treatments in reducing fire size and severity can help protect fisher habitat. Treatment effects on fire spread are local, so treatments inside fisher habitat may better protect fisher habitat than treatments outside habitat. However, treatments in fisher habitat should maintain overstory canopy and avoid removing large trees that may provide resting structures. Treatments outside of fisher habitat may benefit fishers by reducing the spread of large fires from outside into fisher habitat, especially if fire conditions worsen in the near future, as anticipated.

Technical Abstract[5]

This report assesses (1) the current status of fisher habitat and fishers (*Martes pennanti*) in the southern Sierra Nevada, California, and (2) how fisher habitat and the fisher population may respond in the future to potential forest management practices and wildfires. The ultimate goals are to help the three southern Sierran Forests (Sierra, Sequoia, and Stanislaus) improve landscape-level fuels management plans intended to reduce the risk of unplanned and unwanted wildland fire to human and natural communities, to restore and maintain fire-adapted ecosystems, and to conserve habitat for at-risk species.

The fisher is one at risk species whose habitat and population in the Sierra Nevada may be threatened by unnaturally large and severe wildfires; however, they may also be harmed by management efforts intended to reduce wildfire threats. This report assesses these competing threats and applies the results to recommending approaches for maximizing Fireshed Assessment goals, including to conserve and enhance habitat value for fishers to ensure their continued persistence, and perhaps expansion, in the Sierra Nevada.

This Final report replaces all previous drafts and progress reports submitted for this project. Model results included herein reflect refinements to various input variables, model assumptions, and parameter estimates, so *previous maps and model results should be discarded and replaced with contents of this report.*

Methods

Study Area – The study area includes the Sierra, Sequoia, and Stanislaus National Forests and Yosemite and Sequoia-Kings Canyon National Parks, plus a 10-km buffer. We shrank this study area for modeling purposes by omitting areas unlikely to support fishers and unlikely to contribute wildfires that may encroach into fisher habitat. This makes the models more sensitive to detecting changes in habitat quality and fire effects within the region of greatest interest, while also decreasing computation time.

Baseline Habitat Assessment – We assessed current fisher habitat at two spatial scales to account for how fishers select habitat at the home-range scale (which for fishers encompasses landscape heterogeneity over about 1 to 10-km^2; Davis et al. 2007) and how they select resting microhabitats at the fine scale (about 1-ha resolution). This is essential because fires and forest management actions could influence structural aspects of fisher habitat at the fine scale with no observable effects at the coarser scale, or they could affect habitat configuration at the home-range scale without affecting the underlying microhabitat suitability.

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[5] Due to the great length and complexity of this report, we present this extended technical abstract for those wishing sufficient explanation to understand the analytical process and important results without getting lost in technical details. Those wishing to delve deeper into the assumptions, statistics, methods, citations, etc., should read the full report or appropriate sections.

At the coarse or home-range scale[6], we built and compared more than 250 Generalized Additive Models (GAM) using fisher occurrence data from the Forest Service's fisher monitoring program (USDA 2006 and R. Truex personal communications) and a wide array of environmental predictor variables derived at 1-ha resolution from GIS data. The GAM models combine multiple predictor variables to rank the probability of fisher occupancy from zero to one based on environmental conditions averaged over a 5-km^2 circle centered on points where fishers have been detected (presence) or not detected (presumed absence). We built these multivariate statistical models using a conservative interpretation of the fisher monitoring data (referred to as MAPE2), which omits monitoring locations surveyed only once or detecting fishers in only one year. We tested the resultant models using (1) cross-validation techniques, (2) a more liberal interpretation of the fisher data that includes all monitoring locations (referred to as MAPE) and (3) a temporally independent set of fisher survey data for the same area (Zielinski et al. 2000). Candidate models were compared using a variety of statistical measures, including AIC weights[7], AUC values[8], percent deviance explained, and mean and standard error in AUC based on five-fold cross validation[9].

At the microhabitat scale, we applied an existing, published model of fisher resting microhabitat value, developed empirically within the same study region, using variables derived from Forest Inventory and Analysis (FIA) plots (Zielinski et al. 2006). Variables used were maximum tree dbh, basal area of small trees (<51 cm dbh), percent slope, average canopy cover, maximum conifer snag dbh, and hardwood dbh. We applied the model to FIA plot locations between 1,100 and 2,300 meters elevation[10] on the west slope of the Sierra Nevada as a representative spatial sample of microhabitat value and to compare with home-range scale models of habitat value.

Baseline Population Assessment – We used both static and dynamic approaches to estimating fisher population size and distribution. One static approach was to extrapolate fisher density estimates (as measured by Jordan 2007 on the Kings River study area) over the area predicted to be suitable by habitat models, excluding areas north of the Merced River, which are not known to support fishers. A second approach (supplied by R. Truex, personal communication) was to estimate population size by extrapolating preliminary occupancy estimates of Region 5's fisher

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[6] We use the terms "coarse scale," and "home-range scale" somewhat interchangeably to reflect the scale over which individual animals appear to integrate information about environmental heterogeneity when selecting and utilizing their home ranges. Following Davis et al. (2007) we assume this encompasses landscape heterogeneity over about 1-10 km^2 as opposed to the regional (population) scale (1,000-10,000 km^2) and the microscale (0.1-1 ha).

[7] Akaike's Information Criterion (AIC) measures how well an estimated statistical model fits a set of data (e.g., species presence-absence data). It balances the complexity of the model (i.e., the number of variables used) against how well it fits the data. In comparing a set of alternative models, the one having the lowest AIC value (or highest AIC weight) theoretically optimize this balance between simplicity and fit.

[8] The AUC (or Area Under Curve) score is a measure of a model's predictive skill that combines sensitivity (correct classification of presence points) and specificity (correct classification of absence points) into a single index. An AUC of 0.5 indicates no predictive skill; an AUC of 1.0 indicates perfect predictive skill; and an AUC >0.8 is generally considered a "good fit."

[9] Cross validation provides a measure of uncertainty around model predictions. It involved sampling the full data set five times, using 4/5 of the data to build the model each time, applying the results to the remaining 1/5, calculating the mean and standard deviation of the AUC, and comparing this to the AUC for the full dataset.

[10] The Zielinski et al. (2006) resting microhabitat model was built using FIA data collected within this elevation range west of the Sierra Nevada crest, and so was applied only within these limits to avoid extrapolation errors.

monitoring program to the entire southern Sierra population and derive estimates of population size based on the number of fishers presumed to be detected at each sample unit.

The dynamic approach applied the spatially explicit population model PATCH (Schumaker 1998) to estimate the equilibrium population size (or carrying capacity) of fishers in currently occupied habitat areas (south of the Merced River), and to identify likely source, sink, and population expansion areas. PATCH couples population processes (birth, death, dispersal) with spatially explicit habitat values to model territory occupancy by females over time. Coupling population dynamics with habitat models in a spatially explicit way is a useful approach for revealing which areas may be most important to sustaining the fisher population, such as core (source) areas or critical landscape linkages or movement corridors (Carroll et al. 2002, 2003a, 2003b; Carroll 2006). We parameterized the model based on previous fisher population modeling studies (Lamberson et al. 2000, Lewis and Hayes 2004) and on field studies in the study area (Zielinski et al. 2004b, Mazzoni 2002, Jordan 2007). We performed sensitivity analyses by varying territory size, initial number of fishers, maximum dispersal distance, and age-specific survival and fecundity rates and observing effects on the predicted equilibrium population size and distribution.

To estimate current population size, we did not allow model fishers to disperse north of the Merced River, and compared equilibrium population sizes among models using different parameter values. Source and sink areas are identified by territory-specific net value, or the annualized difference between births and deaths within each territory. To look at the potential for range expansion given current habitat conditions, we allowed fishers to disperse north of the Merced River for 20 years into the future.

Changing Fire and Landscape Conditions – We estimated landscape change due to the interactions among succession, wildfire, and fuels management using the spatially dynamic forest landscape model, LANDIS-II (Scheller et al. 2007). LANDIS models emphasize spatial variation of the abiotic template and spatially dynamic processes, such as wildfire and seed dispersal. Wildfires were simulated using the Dynamic Fire extension (Sturtevant et al. 2008) and a new biomass fuels extension developed specifically for this project (Scheller 2008). Fuels management was simulated with the new Biomass Harvest extension which extends the functionality of the Base Harvest extension (Gustafson et al. 2000) to allow partial removal of cohorts. LANDIS-II was chosen because of its history of successful application across a diversity of forest types worldwide (Mladenoff 2004), its extensibility (which allows such additional processes as climate change, insect mortality, and land-use change to be added in the future), because it was conceived and designed to be applied at regional scales, and because it can be used to run replicated simulations that incorporate stochastic events.

We parameterized LANDIS-II using a combination of available vegetation data (Forest Inventory and Analysis [FIA] data, California EVEG), abiotic data (slope, elevation, aspect), historic fire occurrence data, weather station data, outputs from the Forest Vegetation Simulator (FVS), and from expert opinion. LANDIS-II initial conditions were calibrated by comparison with FVS estimates of successional trends for individual FIA plots. Fuel treatments were designed by USFS personnel to approximate potential treatments (reductions in the biomass of ladder fuels and some larger trees) and their effects on fire spread rates and fire severity.

We tested the efficacy of fuel treatments for reducing area burned and fire severity by simulating two fire regimes (a Baseline Fire regime based on the last 20 years of fire trends, and High Fire regime designed to approximate larger, more severe fires) and fuel treatments that varied by their rate of application (area treated per model time step) and their intensity (the amount and age distribution of biomass removed). All simulations were run for 50 years with 10 replications for each combination of treatments and fire regime.

Effects of Fires and Fuels Management on Fishers — We estimated the effects of wildfire and fuel treatments on fisher habitat and population by size by combining LANDIS-II landscape-change simulations with the best overall landscape-level habitat model (called LAND8, see below) and the population dynamics model (PATCH). For this we used a factorial experimental design that allowed us to systematically explore the relative and absolute effects on fishers of alternative fuel treatment rates (proportion of treatable landscape treated per time step), intensities (two prescription combinations: moderate thin + prescribed fire, and light thin + prescribed fire), locations (inside fisher habitat, outside fisher habitat, or both), and fire regimes (baseline or high fire). This factorial design allowed us to investigate how these factors, alone or in combination, may affect fisher habitat and populations, including interactions among the factors.

For each combination of factors we tested (14 total combinations), we ran 10 replicates using LANDIS-II for 50 years (140 total LANDIS-II runs). After each simulation decade, the total above-ground live-tree biomass outputs from LANDIS-II were exported into LAND8 to change predicted habitat value in response to vegetation dynamics, for a total of 840 simulated habitat maps. These decadal habitat maps, in turn, served as inputs to PATCH, which assigned appropriate demographic parameter values based on the spatially explicit habitat values. The population dynamics were then run for the ensuing decade, when a new habitat map was introduced and the parameter values again adjusted by PATCH. We continued these simulations for a total of 50 years of landscape change and 60 years of population change (10 years after the final landscape change). Because, like LANDIS-II, PATCH has stochastic components, we ran 10 PATCH replicates for each LANDIS-II replicate to assess variation in the predictions about population change, for a total of 1400 population simulation runs.

The resulting database of simulation runs contained the fisher population estimates at the end of each PATCH replicate (year 60), the location and area burned within each of three fire regions, fire severity by fire region, the rate of fuel treatment application (proportion of treatable area treated per 5-year period), and the total aboveground live biomass removed by fuel treatment. In addition, biomass removed by treatments was divided by whether the treatments were performed only inside of fisher habitat, only outside of fisher habitat, or both.

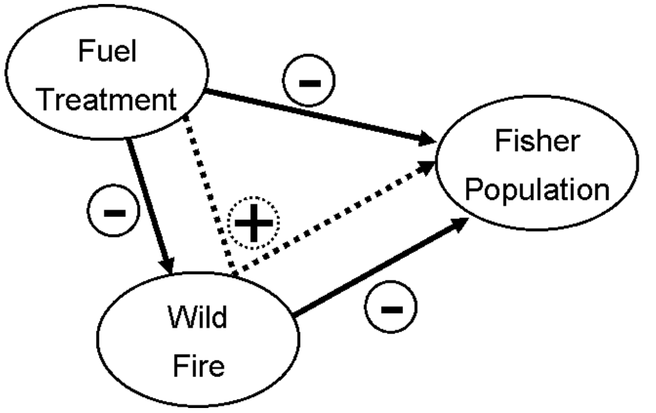

Figure A-1. Expected interactions among fuel treatments, wildfire, and fisher populations in the southern Sierra Nevada. Solid lines represent direct effects; the dashed line represents an indirect effect.

We applied a variety of statistical tests to this database to investigate correlations among wildfire, the various fuel treatment factors, and ultimate fisher population estimates. Analyses included analysis of variance (ANOVA), multiple regression, and structural equation modeling (SEM). Because both fires and fuel treatments reduce forest biomass, either might have direct negative effects on fishers. However, because fuels treatments might also reduce the negative effects of fires on fishers, they might have an indirect, positive effect. Figure A-1 illustrates four sets of hypotheses we therefore tested with these simulation experiments to determine the relative strength of these interactions under the various sets of assumptions, and to determine under what circumstances the overall positive effects on fishers may outweigh the potential negative effects. The design of these simulation experiments allowed us to assign causation to the trends and correlations revealed by the simulations.

Finally, in addition to tracking habitat and population changes using the multivariate habitat model and PATCH, we also tracked changes in the abundance of individual variables that may represent availability of important habitat elements for fishers (e.g., large, old black oaks, firs, or pines that provide resting and denning structures).

Results and Discussion

Home-Range-Scale Habitat Models – The highest ranking GAM model, named LAND8, predicts fisher distribution using three variables: latitude-adjusted elevation, average annual precipitation, and total above-ground biomass of trees. This model produced a 94.1% fit to the fisher detection and non-detection points (using MAPE2), explained 53.3% of deviance, and was nearly 3 AIC units above any other model[11]. It also had the highest statistical fit of all the models using the more liberal MAPE interpretation of fisher presence (83.1% correct classification) and the five-fold cross validation test (90.5%). Figure A-2 maps the probability of fisher occurrence using LAND8.

All top-ranking models included two abiotic variables (elevation and either insolation index or precipitation, which combinations appear to establish physical conditions favorable to fishers and fisher habitat at the regional scale) along with one or more biotic variables that reflect vegetation characteristics associated with fishers. The results reaffirm that fishers are closely associated with dense, old, large forests in mid-elevation zones of the southern Sierra Nevada, which is already firmly established in the scientific literature (e.g., Powell and Zielinski 1994, Zielinski et

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[11] The recommended rule-of-thumb for selecting models other than the top model is that it should be less than 2.0 AIC units away from the top model (Burnham and Anderson 2002, W. Zielinski personal communication).

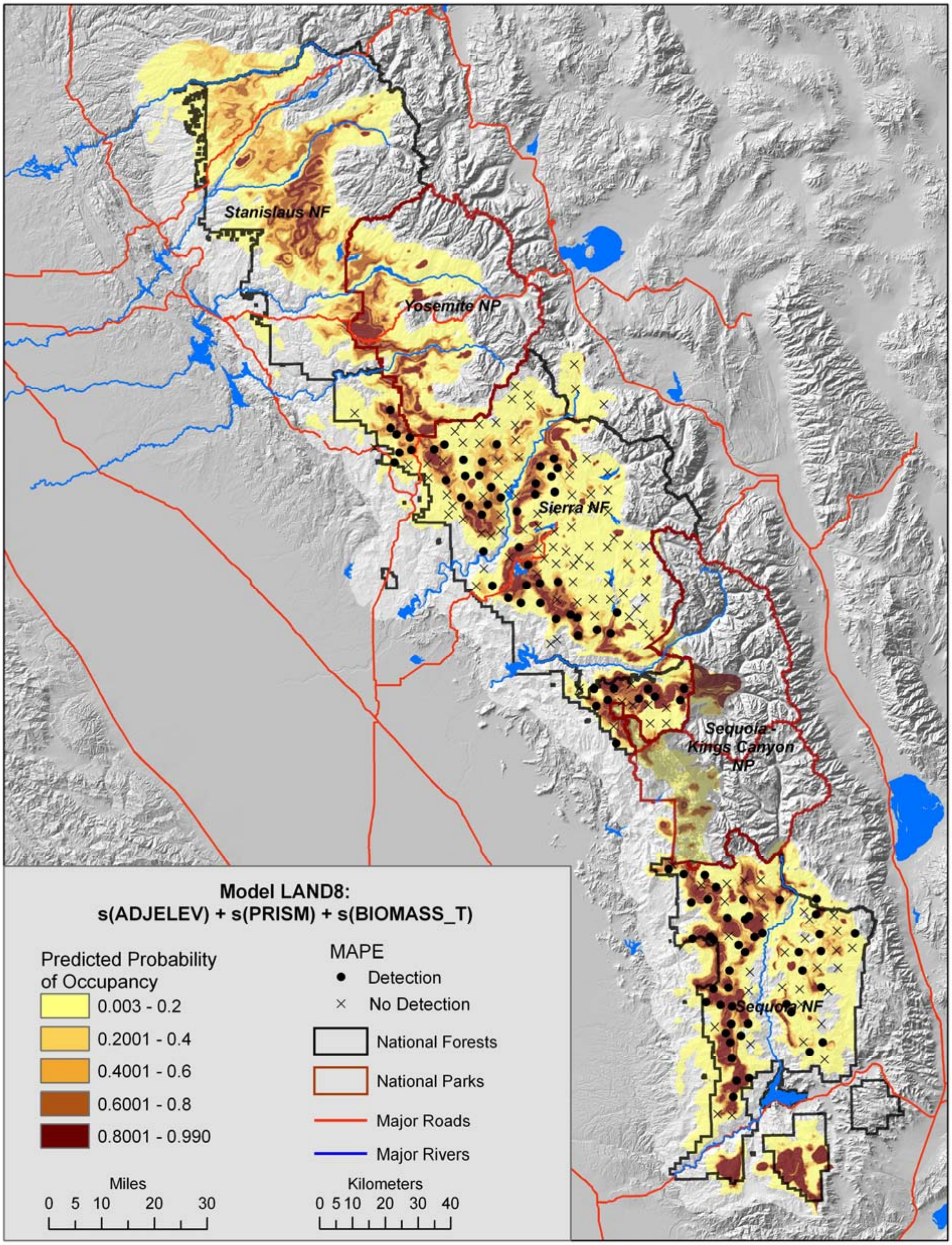

Figure A-2. Predicted fisher habitat value (probability of occupancy) from model LAND8 using adjusted elevation, annual precipitation, and total tree biomass within a 5-km2 moving window and compared with fisher detection-nondetection monitoring data. (Data gaps for tree size result in under-prediction of habitat value in portions of Sequoia Kings Canyon NP.)

al. 2004a, 2004b, 2005, Davis et al. 2007). The results also suggest that most of the potential habitat on the Sierra National Forest and Sequoia National Forest is occupied by fishers, but that potential habitat on the Stanislaus National Forest is not.

Resting Microhabitat Model – Figure A-3 illustrates resting microhabitat value at approximate FIA point locations overlaid on the predicted coarse-scale habitat map using model LAND8. Resting microhabitats with high predicted value are rare in the study area, and the overall correlation between predicted resting microhabitat value and predicted coarse-scale habitat value is modest (0.290). Only 15 (5%) of the 250 FIA plots that were at least partially forested had microhabitat suitability values greater than 0.5. Most of the high-value resting plots are clustered within areas also predicted to have high habitat value at the home-range scale; but there are also many plots with predicted low resting value within areas of predicted high value at the coarse scale. Suitable coarse-scale habitat must also contain sufficient resting microhabitat to support fishers (Zielinski et al. 2004a, 2004b, 2006), but we don't yet know how much is enough or how it must be distributed.

Baseline Habitat Assessment – Our analyses corroborate existing published descriptions of the current state of fisher habitat in the southern Sierra Nevada, and provide some new insights and additional tools for assessing current as well as future conditions.

Fisher habitat is restricted to a relatively narrow band of mid-elevation forests in the southern Sierra Nevada, mostly on the western slope of the range, particularly in areas having high total above-ground biomass of trees (i.e., abundant larger trees). Model results also suggest that, given these conditions, fishers are associated with more mesic topographic positions (i.e., slopes experiencing relatively low annual exposure to solar radiation), but with relatively low annual precipitation. Together with visual inspection of maps and discussions with fisher experts, these results suggest that fisher habitat is concentrated in and near larger and older stands of mixed conifers, sequoia, and ponderosa pine, especially in and near areas supporting black oaks, which provide resting structures as well as food for fisher prey. The selection of areas having lower annual precipitation (all else being equal) appears to support the hypothesis that fishers tend not to be found in areas that support the deepest and most persistent snow cover (W. Zielinski, personal communications).

Predicted coarse-scale fisher habitat is broken into somewhat discrete segments, with gaps generally corresponding to major river canyons. Some of these segments are occupied (south of the Merced River) whereas others (north of the Merced) are not. The zone of predicted suitable habitat appears to broaden toward the north (Stanislaus National Forest), although average predicted probability of occurrence does not appear as high there as on the other two forests.

Highly suitable resting microhabitats are relatively rare even within areas of predicted suitable habitat at the coarse scale. Much of the predicted suitable habitat on Sequoia and Sierra National Forest (roughly 204,000 ha total) appears to be occupied by fishers, whereas large areas of predicted suitable habitat on the Stanislaus National Forest (about 60,400 ha) are not. LAND8 also predicts some high-value habitat at the southern extreme of the study area, in the Piute Mountains and on Breckenridge Mountain, that probably does not support fishers.

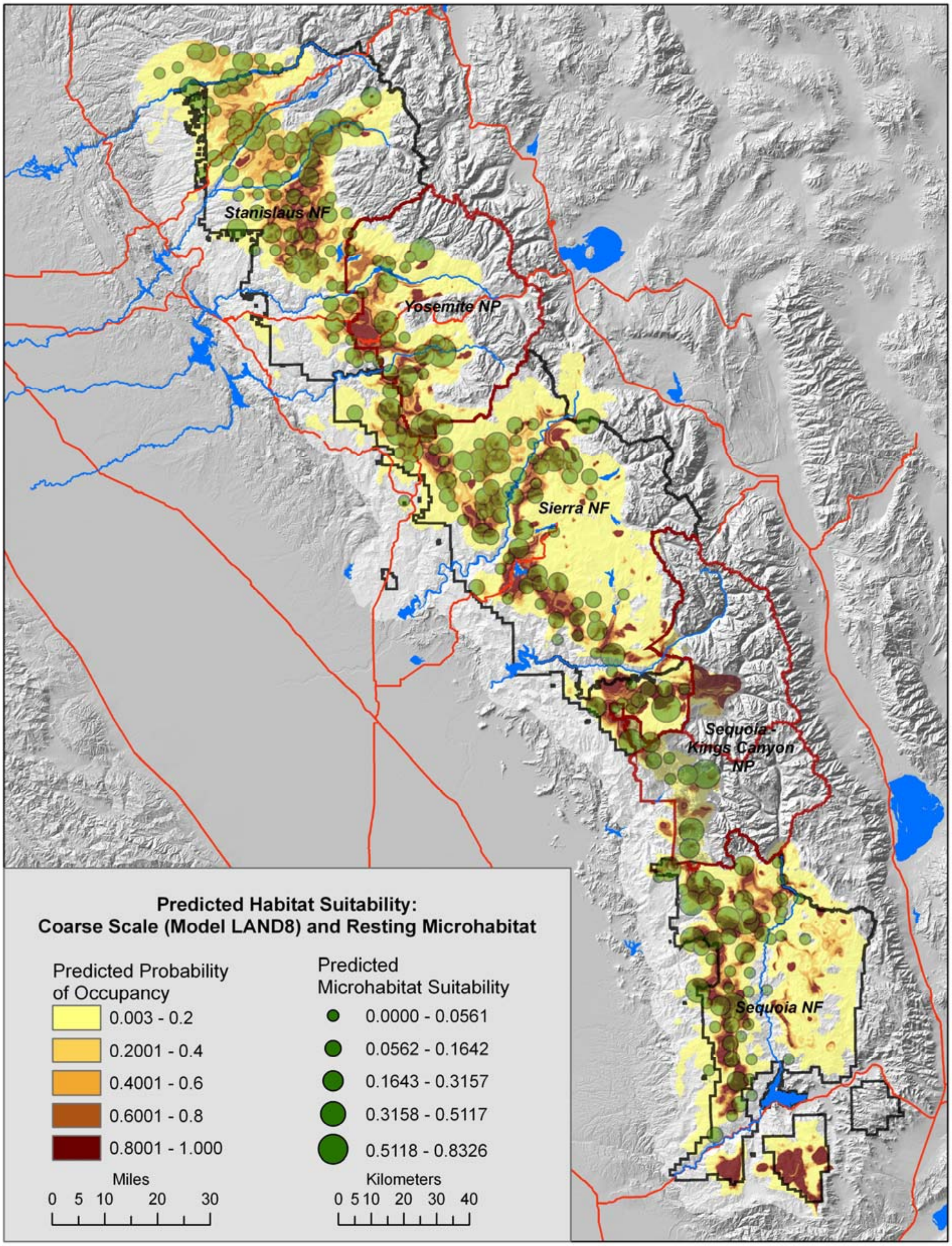

Figure A-3. Predicted fisher resting microhabitat value at approximate FIA plot locations overlaid on coarse scale predicted habitat value (LAND8). (Data gaps for tree size result in under-prediction of coarse scale habitat value in portions of Sequoia Kings Canyon NP.)

Most of the home-range-scale models we tested did not predict as much highly suitable habitat on the Kern Plateau (the southeastern portion of the study area) as is suggested by annual fisher detections there, although the best model, LAND8, appeared to perform better there than any other. Difficulties in modeling habitat in that portion of the study area could be due to one or a combination of factors, including poor-resolution or inaccurate environmental data layers, or unique environmental conditions that make model extrapolation poor there. The Kern Plateau is ecologically distinct from other portions of the study area, with less precipitation, gentler and less incised terrain, and perhaps different vegetation composition and structure (Miles and Goudey 1998). The potential under-prediction of fisher occurrence on the Kern Plateau by our habitat model should be considered when interpreting the following results concerning population size and viability, and may be worth further research and development of a separate habitat model for that area, perhaps using higher resolution or updated environmental data layers.

Baseline Population Assessment – Equilibrium population estimates from the population dynamics model PATCH were relatively insensitive to assumptions about initial population size, dispersal distance, and most age-specific fecundity and survival rates, but were highly affected by territory size and survival rates for adult females. Based on the most biologically defensible ranges of parameter values, we estimate that currently occupied habitat areas in the southern Sierra Nevada can support about 73-147 adult females (the most critical sex-age class for sustaining a population). Assuming a 1:1 sex ratio (which probably overestimates the number of adult males) yields an estimate of 146-294 adults fishers. Accounting for subadult animals provides a rough estimate of 230-392 total fishers in the southern Sierra Nevada. Although this estimate accords reasonably well with other estimates of the Sierra Nevada fisher population, it has several important sources of uncertainty. We suspect that several assumptions may tend to inflate this estimate (e.g., assumed 1:1 sex ratio, average territory sizes of 860-1,200 ha, optimistic age-specific fecundity and survival rates, no accounting for influences of other environmental factors on fishers like competitors or prey, and no accounting for increased mortality during dispersal). In addition, model LAND8 projects some high habitat value in the Piute Mountains and on Breckenridge Mountain at the southern extreme of the study area and outside the geographic range of fisher as delineated by Grinnell et al. (1937). On the other hand, the under-representation of habitat value for Kern Plateau and Sequoia-Kings Canyon NP may tend to slightly underestimate the total population.

Extrapolating fisher density estimates supplied by Jordan (2007) over our modeled habitat area south of the Merced River (276,077 ha, using the 0.5 cutpoint in probability of occupancy for model LAND8) yielded population estimates of 276 to 359 fishers (ignoring juveniles), of which 55 to 83 are adult females. Although these estimates accord well with those from PATCH, it is an uncertain calculation, because (1) we are extrapolating density from only a single sample location, which may not be representative of density over the entire area; (2) using a single probability cutpoint to define occupiable habitat is overly simplistic; and (3) the extrapolation area includes small and isolated areas of predicted habitat that are probably not occupied.

Finally, the preliminary estimate supplied by R. Truex (personal communication) by applying sampling theory to the fisher monitoring data is 160-250 total fishers in the study area. This also overlaps with the low range of our estimates using PATCH (230-392). The relative concordance between these three independently derived population estimates gives us some confidence that the number of fishers in the southern Sierra Nevada, ignoring juveniles, is *between 160 and 360 total individuals, of which about 55 to 120 are adult females, which comprise the most important class for sustaining a population*.

Our model test for the potential of northward expansion onto the Stanislaus National Forest from occupied habitat south of the Merced River suggests that there is strong potential (Figure A-4). However, this prediction may be overly optimistic because the model does not account for increased mortality during dispersal and may not adequately account for barriers or filters to dispersal, such as steep and sparsely vegetated river canyons and heavily traveled highways. We recommend continued monitoring of the fisher population, with special attention to its northern frontier, roadkill along Highway 140 and other roads, and dispersal movements of fishers to better determine the potential for natural northward expansion versus active translocations of fishers. We also recommend performing a focused study of habitat connectivity across the Merced River Canyon.

Changing Fire and Landscape Conditions – Overall, our results indicate that the fuel treatments we simulated may effectively reduce the extent and severity of fire across a large, heterogeneous landscape during a 50-year time span. Our simulated fuel treatments caused a more pronounced reduction in fire size than in fire severity. However, the cumulative impacts of fuel reduction on fire severity would likely be expressed over a longer period than was simulated.

Both the area of treatment and the intensity of treatment were effective at reducing fire on the landscape. The effectiveness of fuel treatments was greatest at higher elevations and under the high fire regime, but was also significant at mid elevations. Although chaparral and oak fuel types at lower elevations had the fastest fire spread rates, there were fewer treatments simulated in those areas. Under the more severe fire regime, greater fuel treatment effectiveness reflected the greater probability of fires encountering fuel treatments. Therefore, at least in the forests represented in our simulations, and using the fuel treatment rates and intensities that we evaluated, fuel treatment may become more beneficial if fire weather conditions become more severe, as is anticipated.

Effects of Fires and Fuels Management on Fishers – In general, aboveground live biomass was a useful currency for understanding the interactions among fuel treatments, wildfire, and fisher populations at the very broad scales we examined. Although the amount of biomass removed by fuel treatments was largely controlled by the rate of fuel treatment, both the rate and intensity of treatment are important in determining area burned, depending on local conditions.

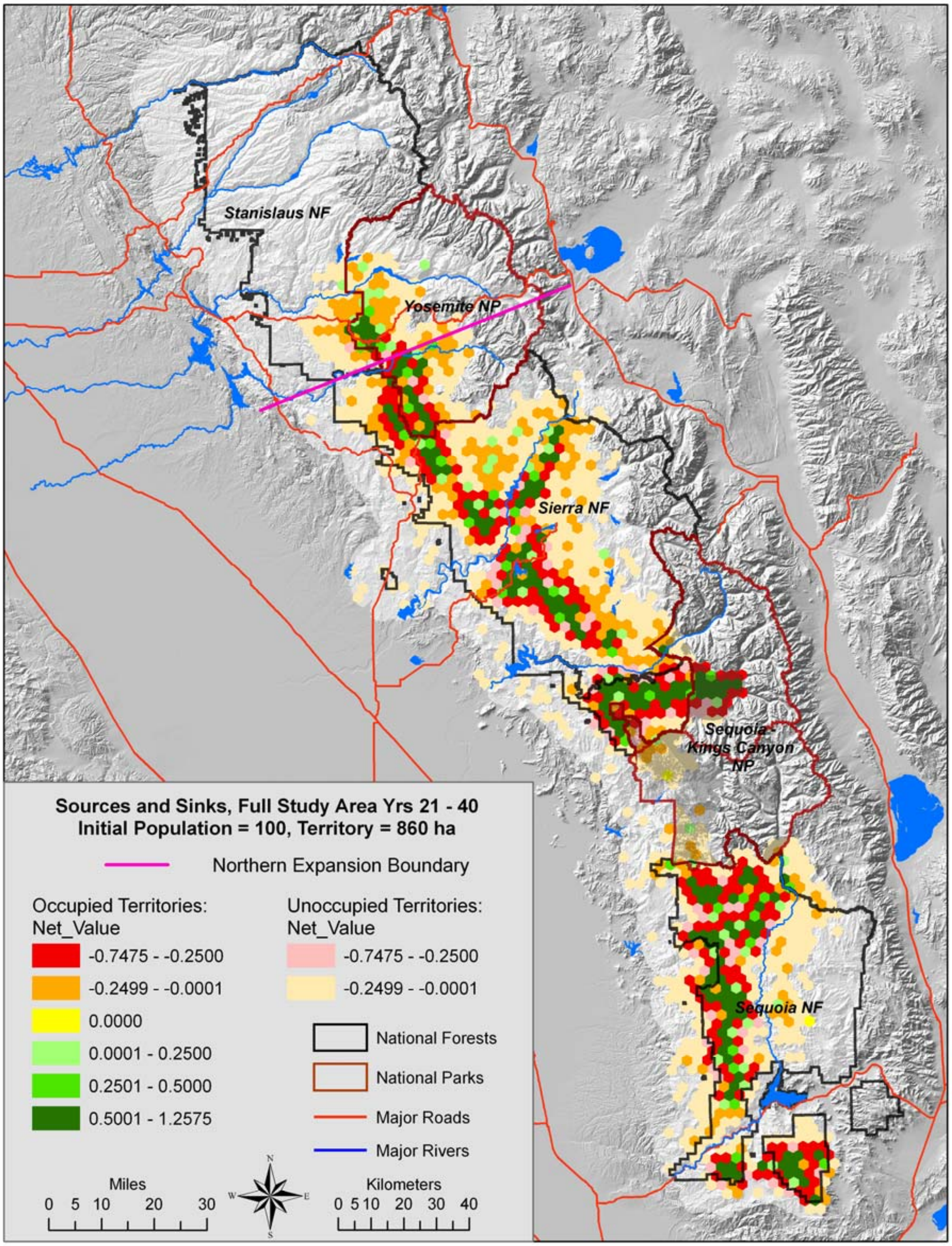

Figure A-4. Net value of territories (annualized births – deaths) observed over 20 years in program PATCH (territory size = 860 ha; maximum dispersal = 50 km; 100 initial females). Source territories are in greens (births > deaths) and sink territories (deaths > births) in reds. Data gaps may result in under-representation of habitat value and hence territory value in Sequoia-Kings Canyon NP. Note predicted population expansion north of the Merced River (magenta line).

Across the broad scales we examined, and using assumptions provided by the Forest Service about how treatments affect fuel characteristics and hence fire spread rates and severity, and within the finite combinations of fire regimes and treatments tested, we conclude that the longer-term positive effects of fuel treatments (due to the reduction of fire hazard) outweigh the short-term negative effects of fuel treatments (due to immediate loss of biomass) on fisher. Across all the simulations, it appeared that managing forest structure had positive effects on the simulated fisher populations, especially under the heightened fire regime, and that differences among management approaches were more subtle than the differences between managing and not managing. If actual fuel treatments remove substantially more biomass than those we simulated or if actual treatments do not reduce fire spread rates and severity to the degree modeled, then we would expect to see more parity between the positive and negative effects of fuel treatment on fisher. Conversely, if a greater reduction in fire spread rate and fire severity could be achieved with more strategic removal of less biomass, the positive effects of fuel treatment could be larger than our results indicated. Specifically, less area could be treated if the fuel treatments were strategically clustered in areas at high risk of fire, and in and near fisher habitat (as opposed to far outside fisher habitat).

Management Implications and Recommendations

The results of our analyses should be considered just one form of input to forest management and planning decisions, with all their assumptions and uncertainties kept firmly in mind. Management of Sierra Nevada forests must consider all scales of interest, from site-specific effects of fuel treatments on fine-scale habitat elements, to the broad, landscape-level patterns we've investigated here. While the interacting system of models we have used here provide a landscape context for management decisions concerning fuels, forests, and fishers, local, on-ground information is just as essential for evaluating the risks of habitat loss due to fires as well as how best to influence future fire outcomes and sustain forest conditions that support fishers and other at-risk species. It is critical that any strategy for managing fire risks and fisher habitat constantly ground itself by ensuring it considers both the fine scale as well as the landscape scale.

The analytical process and results described in this report also emphasize the need to carefully consider the temporal scales of management effects on fires and resources, from immediate, often short-lived, direct effects, to longer-term cumulative effects that may ripple through vegetation succession patterns and species population dynamics over many years or decades. Due to the complexity of nature and the uncertainty it brings to any forecasting exercise, it is essential that we continually adapt to new findings about fisher habitat use and biology as well as for changing landscape conditions through management actions and environmental changes.

Based on the results from our broad-scale analyses, we offer the following recommendations, which should be refined in consultation with fuels management experts and fisher biologists.

Maintaining Habitat Quality and Continuity – The fisher population in the southern Sierra Nevada is small (160 to 360 individuals) and at risk of extirpation by stochastic events, including uncharacteristically large or severe wildfires that could fragment habitat and isolate fishers in smaller areas. Management should strive to sustain and increase the area and continuity of fisher habitat at the landscape scale. Fuels management should prioritize treatments in areas at highest risk of large, severe wildfire that could move through the relatively narrow band of fisher habitat or widen gaps between current habitat segments (e.g., near major river canyons).

Maintaining Resting Site Microhabitat Value – Fisher resting microhabitats appear to be rare. Management should maintain and if possible increase the distribution and abundance of forest stands that provide high-value resting sites, especially within high-value macro-habitat areas. However, it may not be necessary to maintain dense canopies over large areas. We recommend deriving and analyzing spatial rules for the size and continuity of dense forest patches that may best retain fisher resting habitat value while minimizing risks of crown fire.

Maintaining or Improving Potential for Population Expansion – Increasing the amount or value of fisher habitat south of the Merced River may increase fisher population size and viability and should not be precluded by management decisions. Northward expansion of the population onto the Stanislaus National Forest could significantly increase population size and viability. We recommend a focused analysis of habitat connectivity across the Merced River Canyon.

Fuel Treatment Rates – Treating 4% to 8% of treatable area every 5 years can significantly reduce fire sizes and fire severity and thereby benefit fishers. Strategic placement of treatments may decrease the necessary treatment area. The Stewardship and Fireshed Assessment process should continue identifying and evaluating areas of high fire risk overlapping with fisher habitat to improve treatment strategies.

Fuel Treatment Intensity – Fuel treatments should use site-specific analyses that consider fisher habitat value in and near the treatment. Within fisher habitat, treatments should balance desired fuel conditions with maintaining sufficient overstory and habitat elements to sustain or encourage occupancy by fishers. Removing larger trees and other essential habitat elements should generally be avoided within fisher habitat, to the degree feasible while meeting fuel reduction and landscape vegetation management goals.

Fuel Treatment Location – Placing treatments inside fisher habitat is not necessarily detrimental to fisher. The positive indirect effect of treatments in reducing fire size and severity can help protect fisher habitat. Treatment effects on fire spread are local, so treatments inside fisher habitat may better protect fisher habitat than treatments outside habitat. However, treatments in fisher habitat should still strive to maintain overstory canopy and avoid removing large trees that may provide resting structures. Treatments outside of fisher habitat may also benefit fishers by helping reduce the spread of large fires from outside into fisher habitat, especially given the anticipated increase in severe fire conditions in the near future.

1.0 Introduction

The fisher (*Martes pennanti*) is a large member of the weasel family associated with dense, structurally complex, low- to mid-elevation forests (Powell and Zielinski 1994, Zielinski et al. 2004a, 2004b, Davis et al. 2007). Remaining populations in the western U.S. are small, disconnected from one another, and threatened by habitat modification and fragmentation (Powell and Zielinski 1994, Zielinski et al. 2005). In 2004, the U.S. Fish and Wildlife Service (2004) found that the west coast population of fisher warranted protection under the Endangered Species Act, but that listing this distinct population was precluded by higher priorities.

In the Sierra Nevada, fishers currently occupy less than half their historic range, with a small population persisting south from Yosemite National Park through the Greenhorn Mountains in southern Tulare County (Zielinski et al. 2000, 2005). This region is also experiencing increasing risks of large, stand-replacing wildfires due to previous forest management actions (e.g., fire suppression, logging), climate change, and increased human-caused ignition rates (Agee and Skinner 2005, Van Wagtendonk and Fites-Kaufman 2006, Westerling et al. 2006,Syphard et al. 2007). However, vegetation management plans to reduce the size and severity of wildland fires, increase the resilience of forest stands, and conserve habitat for at-risk wildlife species, have been controversial, in part due to uncertainties about the relative short-term, long-term, and cumulative effects of these actions on rare species like the fisher. Fuels-management actions (e.g., forest thinning) may reduce fisher habitat value locally and in the short term, but they may also reduce the probability of large, stand-replacing fires that could have larger and longer-term effects on fisher habitat and population viability. Moreover, the effects of many local fuels-management projects on fisher habitat and populations could cumulatively be substantial if they are not sited, timed, and implemented strategically.

The Conservation Biology Institute (CBI) was commissioned by the Pacific Southwest Region of the USDA Forest Service (Region 5) to help disentangle how these complex and probabilistic factors affect fishers so that an effective management strategy can be devised to minimize fire hazards and maximize fisher habitat potential. We were asked to compile available data on fisher populations and habitat in the southern Sierra Nevada, assess their status, and predict how they are likely to respond to various alternative forest management actions, as well as unplanned events like wildfire. To meet the project goals, we coupled a spatially explicit landscape change model with fisher habitat suitability and population dynamics models. Coupling these dynamic models enabled us to examine how various factors—like the amount, type, and placement of fuels treatments—may interact to affect fires and fishers under various assumptions about future fire regimes. What we've learned can serve as a basis for developing more effective fuels management strategies to minimize fire hazards while maximizing fisher habitat potential. Future scenarios could incorporate, for example, the effects of climate change, insect outbreaks, diseases, or droughts on fires, forests, and fishers—or a variety of other species for that matter. The limited range of possible future fire regimes and management approaches analyzed in this report could be greatly extended, in an adaptive management framework, to simulate effects of an unlimited array of vegetation management strategies and assumptions about ecological processes on fishers or other at-risk species. And, as we continuing learning more about fishers, forests, and fires, as new information gathers about climate change or other critical processes, and as new or refined management approaches are proposed, these integrated simulation models could be used to test and compare alternative futures — and to work continually toward optimal solutions to our complex and ever-changing forest management problems.

The project involves extensive collaboration with other scientists, stakeholders, and agencies. A group of independent science advisors (Appendix A) has overseen technical aspects of the analyses and advised on use of best available science. Advisors were chosen for expertise in fisher biology, forest ecology, silviculture, fire ecology and management, and landscape dynamics modeling. Stakeholder groups have been kept informed via a group of stakeholder representatives comprising individuals from the timber industry, conservation organizations, the U.S. Fish and Wildlife Service, California Department of Fish and Game, and the various National Parks and Forests in the study area. Finally, although CBI's work products are produced independent of the U.S. Forest Service to ensure objectivity, we collaborate closely with various experts from the Pacific Southwest Research Station and Region 5 to obtain and interpret data. However, the contents of this report, and any inaccuracies in reporting and interpretation, are solely the responsibility of CBI.

1.1 Study Area

The study area was originally delineated to include the Sierra, Sequoia, and Stanislaus National Forests and Yosemite and Sequoia-Kings Canyon National Parks, plus a 10-km buffer around these federal lands (Figure 1.1). This area comprises all known occupied fisher habitat in the southern Sierra Nevada plus additional potential but presumably unoccupied habitat which may be important to sustaining or expanding the current fisher population. The study area ranges in elevation from 31 to 4,409 m (102 to 14,456 feet) and encompasses the entire Southern Sierra Fisher Conservation Area (SSFCA) plus considerable surrounding lands. Although composed primarily of federal (National Forest and Park) lands, the study area also includes some tribal lands (Big Sandy, Cold Springs, Tule River, and Tuolumne) and some private lands. Industrial timber lands are concentrated to the north, interspersed with Stanislaus National Forest.

Based on early analyses of the distribution of fishers, fisher habitat, and wildfires, we modified the study area somewhat for modeling purposes (Figure 1.1). By omitting areas highly unlikely to support fishers now or in the future, and areas unlikely to contribute wildfires that may encroach into fisher habitat, we effectively shrank the modeling universe in a way that makes the habitat models more sensitive to changes in habitat quality, while also decreasing computation time. This allows us to do more sensitivity and calibration runs and to analyze more variations in future management and fire scenarios with minimal sacrifice of important results. Note that in shrinking the modeling universe, we did not remove broad swaths of lower-elevation vegetation in the western foothills of the Sierra, because this area of high fire frequency is essential in understanding how fires start and spread. Fuels manipulations in those lower elevations may affect fishers as well as human communities in nearby (generally upslope) areas. Higher elevation areas and portions of the east slope of the Sierra Nevada, in contrast, could be removed from the modeling universe with little adverse effect on our ability to detect changes in fisher habitat quality and hazards of wildfires within the area of greatest interest.

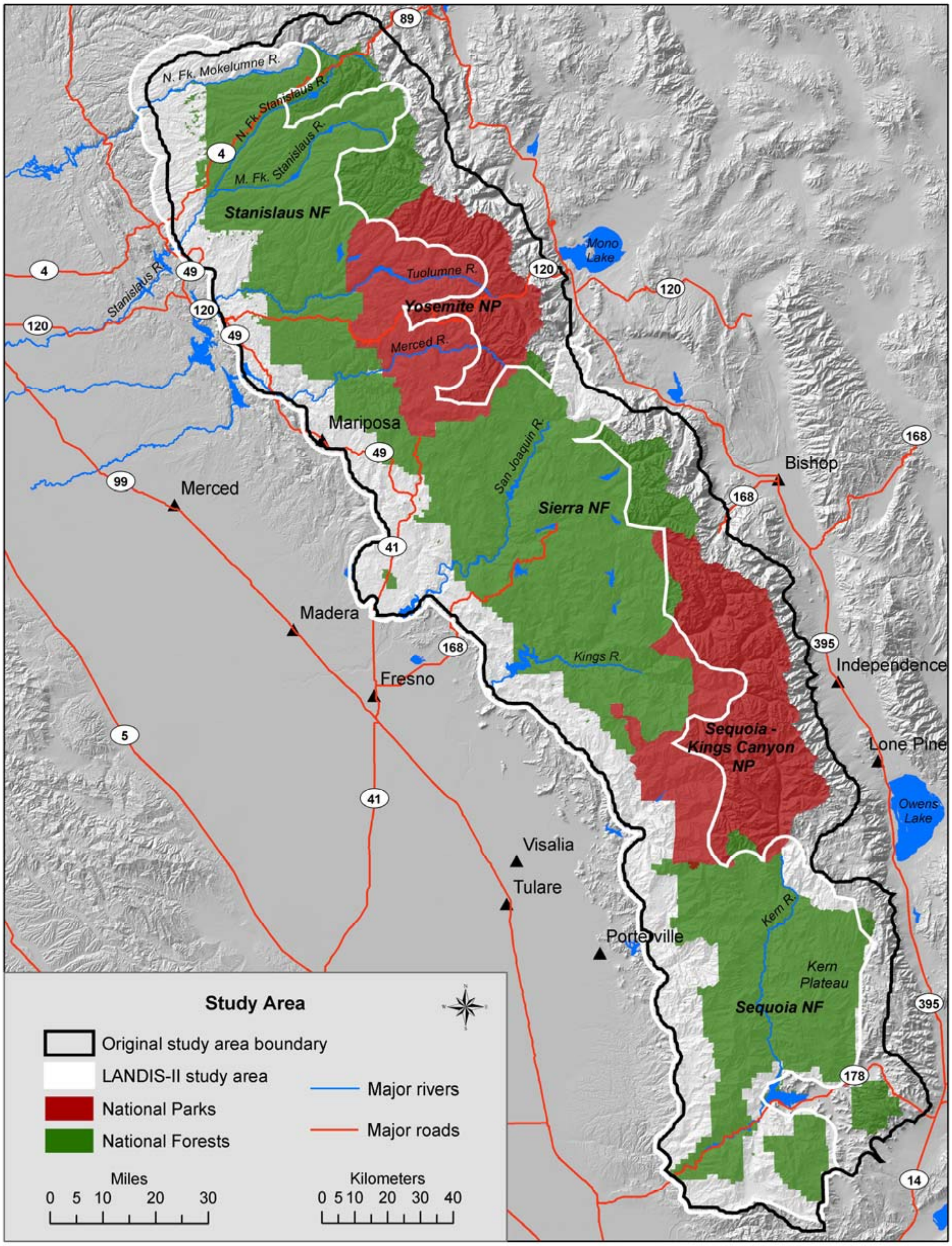

Figure 1.1. Original study area boundary (black outline) and smaller modeling study area (white outline).

1.2 Study Limitations and Constraints

Throughout this project, we strived to be clear and transparent in stating assumptions that may lead to uncertainties in our methods and interpretations of results. For this report, we think it is important to summarize some general concerns right up front. The following limitations and constraints should be kept in mind while reading the rest of this report, because they inject some uncertainties into the assessment. Many of these uncertainties could be reduced by results of continued monitoring and ongoing or future research.

1.2.1 Past vs. Present Habitat Conditions

Current patterns of fisher detections are assumed to represent areas suitable for supporting fishers, but we do not know how current habitat quality relates to the historic or prehistoric habitat quality – i.e., how the best available habitat today compares with what may have been best habitat prior to logging, trapping, fire suppression, porcupine eradication, and other human influences. Understanding the natural landscape prior to these impacts would help to establish the proper context for current resource challenges.

In recent decades, fishers have only been detected south of the Stanislaus National Forest (south of the Merced River) although historically fishers occurred throughout the length of the Sierra Nevada (Grinnell et al. 1937, Zielinski et al. 2005). Although fisher absence north of the Merced could be due in part to inadequate habitat conditions, it is also highly likely, in the opinion of fisher experts, that fishers may be absent there due to historical extirpation (by logging, trapping and other factors) and insufficient time, dispersal ability, or source of potential colonizers for the population to expand back into that area (Zielinski et al. 2005). Because our empirical habitat models are built using fisher detection-nondetection data south of the Merced River, and then extrapolated north of the Merced, there is a degree of uncertainty about baseline habitat value projections on the Stanislaus National Forest. This deserves further investigation; but in the meantime, it is prudent to *ensure that management actions do not preclude maintenance and recovery of habitat value on the Stanislaus nor disrupt the potential for fisher expansions toward the north*.

1.2.2 Models as Simplifications

All scientific models are necessarily oversimplifications of reality. Statistical models, such as those used here to predict fisher habitat quality, rely on correlations between variables to make predictions, often without full understanding of the cause and effect relationships underlying these correlations. As pointed out by D. Graber and C. Hanson (personal communications) knowledge of what specific elements are actually required by fishers within their habitats remains substantially unknown. Thus, predictions by models relying on various explanatory variables should be seen as functional hypotheses rather than absolute truth. Fitting a model to characterize the distribution of a species over a large and heterogeneous region, such as the southern Sierra Nevada, risks oversimplifying how the species actually selects habitat on a finer scale, which may vary somewhat from place to place. For example, none of the habitat models we tested appeared to accurately characterize fisher distribution on the Kern Plateau (the southeastern portion of the study area). This could be due to limited fisher representation in the monitoring dataset (fisher typically account for <20% of the annual detections recorded by the monitoring program), fundamental ecological differences between the west slope of the southern

Sierra Nevada and the typically drier, higher elevation Kern Plateau (R. Truex, personal communication), or poor ability of a regionally derived model to extrapolate over every portion of such a heterogeneous study region (see Davis et al. 2007 for more detailed treatment of this issue).

Likewise, our landscape change modeling is limited by the large spatial extent (> 2,000,000 ha of forest) that we modeled. At this scale, both the inputs to the model and the processes modeled must be simplified. For example, although our landscape model is capable of representing finer resolutions and greater environmental heterogeneity, we were limited by the data available for parameterization across the entire landscape. And although the capability exists to model the growth and mortality of individual trees, modeling such mechanistic detail across a large extent is not computationally feasible. As a result of simplification, our landscape estimations of change lack great precision. However, given the stochastic nature of fire on the landscape and the uncertainty about future fire trends, maximal precision would not greatly decrease uncertainty.

1.2.3 Analytical Scale versus Site-Specific Interpretations

The coarse-scale habitat models included in this document should not be used for fine-scale planning or management decisions, and cannot reflect on-ground habitat conditions at finer resolution than the moving window size at which the variables are integrated (5 km^2). Because the models average variables over a 5-km^2 circle and assign the average value to the central pixel (1-ha resolution), the actual on-ground conditions in this central pixel may be quite different than the average. Consequently, although the models may be effective at depicting gross patterns in habitat distribution and population dynamics, they should not be used to interpret finer-scale habitat attributes or impacts of specific projects on fishers. This would require finer-scale analyses, preferably informed by field assessment.

1.2.4 Data Inaccuracies and Gaps

Our habitat models use predictor variables derived from available GIS environmental data layers, such as land cover maps that include information on tree size and density classes. Although these data layers are quite complete and up to date for much of the study area, there are always inherent inaccuracies. Models are only as good as the data that drive them, and using multiple themes can easily compound the errors. The best available data were used throughout this project, but the reader should consider these important limitations.

In addition, missing data on portions of Sequoia and Kings Canyon National Parks (SEKI) led to under-representation of habitat quality there (and within 1.3 km of the park boundaries). Most maps in this report therefore show SEKI with dark shading to remind the reader to keep this uncertainty in mind when interpreting maps and results.

1.2.5 Importance of Prey Habitats for Sustaining Fisher Populations

Habitat quality evaluated using empirical, statistical models based on fisher detection-non-detection data do not account for possible off-site contributions to fisher habitat value from areas not actually occupied by fishers. For example, vegetation conditions following large, mixed severity fires may support high populations of potential fisher prey, which may contribute value to other areas occupied by fishers, even if fishers rarely if ever venture into the burned areas (C.T. Hanson, and D. Graber, personal communications). It is extremely difficult to account for this potential effect using statistical models that make predictions based on environmental conditions at locations where fishers are detected or not. Since we do not have environmental data layers depicting prey type and abundance in the Sierra Nevada, and have no evidence that prey availability is limiting to fisher populations in the study area, we have no choice but to assume that a sufficient prey base exists in areas having similar characteristics to those where fishers have been detected (W. Zielinski and F. Davis, personal communications).

1.2.6 Other Important Disturbance Factors

Repeatedly, our Science Advisors have emphasized the importance of climate change and the role of insects and disease as strong agents of forest change that interact intimately with both management and fire. We have included high fire regime(s) in our scenario testing phase of the project (Syphard et al. 2007), which addresses one aspect of the expected climate change in the region. However, simulations performed to date have not addressed other implications of climate change – including effects on the growth and survival of various tree species. Likewise, we have not yet addressed the effects of insect outbreaks, which may cause both fine-scale reduction in overstory density and broad-scale mortality in concert with the influences of climate change, drought, and invasive species. The degree to which these factors may influence our vegetation change and fisher habitat models is unknown and deserving of additional analysis.

1.3 Report Organization

The analyses are presented in three major sections:

- Section 2 (Baseline Assessment of Fisher Habitat and Population Status) presents our analysis of the status of fisher habitat and the fisher population in the study area using landscape-level habitat modeling, a resting habitat microhabitat model, a spatially explicit population model, and other techniques.
- Section 3 (Simulation of Changing Landscape Conditions) presents our application, calibration, and testing a spatially-explicit landscape change model (LANDIS-II) for simulating interactions between fires, fuel treatments, and forest succession.
- Section 4 (Effects of Fires and Fuels Management on Fishers) presents projections about the potential future status of fisher habitat and the fisher population by coupling the LANDIS-II model described in Section 3 with the landscape-level habitat model and the spatially explicit population model developed in Section 2, as well as by tracking future trends in specific variables that are considered important to fishers and fires, such as the abundance of mature black oaks and other tree species.

Following these separate analytical sections, Section 5 synthesizes their pertinent results into a general discussion of their implications for forest management and for future analyses.

2.0 Baseline Assessment of Fisher Habitat and Population Status

2.1 Methods

2.1.1 Data Sources and Data Gaps

Existing data on environmental factors and management actions were compiled into a Geographic Information System (GIS, Appendix B). These data layers were used to derive potential predictor variables for the creation of multivariate fisher habitat models (Appendix C). In some areas the USDA Region 5 Existing Vegetation (EVEG) data were updated using supplemental data sources, including maps of recent clearcuts on private lands near Stanislaus National Forest and locations of giant sequoia groves.

Limited size and density data were available in EVEG for portions of the study area that fall within Sequoia and Kings Canyon National Parks (SEKI). While some vegetation plot data provided by the NPS fell within these areas, sampling within some vegetation polygons was insufficient for confident extrapolation of size and density attributes to them. For polygons in SEKI missing density data, we therefore used canopy density obtained from the National Land Cover Database (NLCD) Tree Canopy Layer (2001). This may result in overestimating density in places that have burned since 2001. Polygons in SEKI missing size data in EVEG were not altered, which likely resulted in underestimating tree size in fisher predictor layers estimated over 5-km^2 moving windows[12]. Because of the uncertainties caused by these data gaps for any habitat models relying on tree size or density classes, we show modeled habitat quality within SEKI using shaded hues in our results, and urge readers to interpret habitat and population model results with caution for that area.

2.1.2 Home-Range-Scale Habitat Models

We used an empirical, Resource Selection Function (RSF; Manley et al. 1993, Boyce et al. 2002) approach to modeling habitat value at the coarse or home-range scale. An RSF is a form of statistically rigorous habitat suitability model that produces index values proportional to the probability of use by an organism. In this case, the units are 1-ha pixels associated with environmental variables (such as forest vegetation attributes) that may be correlated with the actual resource elements required by fishers (e.g., prey items and resting sites). Probability of use by fishers is derived using presence-absence (or more correctly, detection-nondetection) data for fishers from Region 5's fisher monitoring program.

Model Type – We selected Generalized Additive Models (GAM) as the most suitable model type from a variety of approaches we investigated (Appendix D) given the nature of the available data, exploratory tests, and project objectives. A GAM is a semi-parametric form of regression

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[12] This would reduce estimated habitat value within SEKI as well as within 1.3 km buffer outside of SEKI (the radius of the 5-km^2 moving window). The area of predicted habitat affected by this potential decrement in value is very small relative to the area of predicted habitat in the entire study area, but it may result in slight under-representation of total fisher population size.

analysis which uses a link function to establish a relationship between presence/absence survey data and a smoothed function of the explanatory variables. GAM models produced the best fits to the fisher monitoring data, take best advantage of the detection/non-detection nature of the data, are highly interpretable, provide useful diagnostics, and accommodate complex, non-linear relationships between predictor and response variables. Davis et al. (2007) also tested a variety of multivariate model types for fisher in California (and several regions of the state, including the southern Sierra Nevada) and found GAM models most useful due to greater performance or easier interpretation than alternatives.

Model Scale – Based on exploratory analyses, discussions with science advisors, and the particular goals of this study, we settled on using 5 km^2 as the moving window size over which coarse scale habitat value is integrated. This resolution is biologically defensible, because it approximates a small to average female fisher home range in the study area (Zielinski et al. 2004b), and it provides good statistical fit to the fisher detection-nondetection data. It is also fairly discriminating of fine-scale bottlenecks or patterns in habitat configuration, at a scale meaningful to management decisions[13], which might be lost using larger window sizes. Although Carroll et al. (1999) found predictive ability of habitat models to break down for fishers in northern California at smaller moving window sizes, Davis et al. (2007) found slightly stronger statistical associations at 3 km^2 than 10 km^2 (they did not test 5 km^2) in the southern Sierra Nevada. They believed that their 10-km^2 window size may have been too coarse given the greater spatial heterogeneity and finer-grained nature of vegetation in the southern Sierra Nevada than other regions.

Potential Predictor Variables – Potential environmental predictor variables included numerous abiotic factors (e.g., precipitation, snow depth, elevation, relief, solar radiation indices), biotic factors (e.g., numerous forest type, size, and density variables), spatial configuration variables (using program FRAGSTATS), vegetation diversity indices (using Shannon diversity applied to vegetation data classified in various ways), management variables (e.g., from Region 5's FACTS database), and fire history variables. During refinement and calibration of the LANDIS-II model for vegetation dynamics (see Section 3) we also created some new, continuous variables within LANDIS-II (including maximum age of trees and mean above-ground biomass of trees) which outperformed other biotic variables in GAM models. Appendix B lists data layers used in modeling and Appendix C defines potential predictor variables that were derived from these layers

Fisher Data – We used data from the US Forest Service, Region 5, carnivore monitoring program (USDA 2006 and R. Truex personal communications) to derive different measures of fisher presence and absence (or detection-nondetection) to use as response variables for habitat models. Fixed survey arrays (N = 290 within the original study area, 276 in the reduced modeling area), co-located with FIA plots and sufficiently spaced (~5 km apart) to represent independent samples of detected fishers, are repeatedly surveyed, generally every other year. On Sequoia and Sierra National Forests, most monitoring sites have been sampled between two and

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[13] We urge recognition, however, that these landscape-scale models should not be used for site-specific planning or management at finer resolution than the chosen window size. The models average values over the entire window and assigns the average value to the 1-ha center point. Consequently, actual value of any variable at the central pixel may be quite different from on-the-ground conditions at that point.

four times since 2001, while on Stanislaus National Forest sites have generally only been sampled once or twice[14]. It is possible that the monitoring program failed to detect fishers at certain sites even though they were truly present, although the survey protocol is estimated to have a ~96% probability of detecting a fisher if one is present (W. Zielinski, personal communication).

The repeated sampling design used by the monitoring program allowed us to investigate whether models can discriminate "reliably occupied" habitat from "occasionally occupied" or "never occupied" for those arrays that have been sampled more than once. We therefore defined two different fisher detection response variables for model building and testing, as follows:

- MAPE (short for *MArtes PEnnanti*): This includes all survey points south of the Merced River[15] regardless of the number of sample years or fisher detections. Points with at least one detection (in any year) were classified as presence points; points with zero detections across all years were classified as absence points.
- MAPE2: This omits any points that were sampled only once, or sampled repeatedly but with only one detection. Thus, presence points are defined here as those yielding detections in at least two survey years; absence is defined as points sampled at least twice but with zero detections.

Models built using MAPE2 as the response variable always provided better statistical fit to fisher detection-nondetection data than those using MAPE (CBI March 2007) despite the reduction in sample size that results from eliminating single survey and single detection points (from 230 locations to 169 locations). Models built using MAPE2 are therefore considered potentially more discriminating of fisher habitat value, and of more reliably occupied fisher habitat, than models built using MAPE. As pointed out by W. Zielinski (personal communication) using only those sites that had at least 2 years with detections produces a model more likely to identify the most important areas for long-term fisher tenure than models based on single samples, "which are influenced by detections of transient individuals as well as those that occupy home ranges repeatedly." Nevertheless, models built using MAPE to define the response variable may be more liberal in identifying potentially suitable habitat areas, so may be more appropriate for contexts where errors of omission (discounting truly suitable habitat as unsuitable) are most important. Based on these considerations, statistical results of our exploratory analyses, and recommendations of science advisors, we built the statistical models using MAPE2 as the response variable, but also tested more than 30 of the highest performing models against the MAPE data to see how well they perform relative to this larger and more liberal detection-nondetection data set.

Candidate Habitat Models – Based on the exploratory analyses and recommendations from science advisors, we originally (CBI March 2007) used a broadly inclusive, information-theoretic approach to creating and comparing candidate habitat models (Burnham and Anderson

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[14] The database we used included results through 2006, since which time additional data have been collected.

[15] Including the numerous absence points north of the Merced River might force models to spuriously discriminate against suitable habitat in areas where the species may be absent due to historical influences rather than current habitat conditions (W. Zielinski, F. Davis, R. Truex, R. Barrett, personal communications).

2002)[16]. This approach involves creating a potentially large set of *a priori* candidate habitat suitability models by listing different combinations of potential predictor variables that may together explain fisher distribution.[17] Candidate models were based on existing published or unpublished models of fisher habitat selection (e.g., Davis et al. 2007, Carroll et al. 1999) as well as other plausible models based on ideas or hypotheses provided by Science Advisors, agencies, stakeholder representatives, Forest Service employees, and other interested parties, and our own interpretations of the literature concerning the factors likely to affect fisher distribution. CBI (March 2007) provided examples of specific hypotheses that various models were intended to test. As mentioned above, during the process of building and testing these models, we discovered that new biotic variables created within LANDIS-II proved to be good predictors and added them into the pool of models. This somewhat ad-hoc process varies from the strict information–theoretic approach, but as our goal was to derive the most useful and highly predictive model possible, we felt it was justified for pragmatic and schedule reasons.

Model Testing and Refinement – All candidate models were next "competed" (compared statistically) against one another to objectively determine which model(s) best fit the empirical data (using the MAPE2 response variable) and explain the pattern of fisher presence and absence. Appendix E lists the more than 250 candidate models we originally created and tested (CBI March 2007). We used several diagnostics to determine which models produced the best fit to the fisher detection-nondetection data: Akaike Information Criterion (AIC)[18], AIC weights, percent deviance explained, residual deviance, and Area Under Curve (AUC) of the Receiver-Operating Characteristic (ROC) [19].

We further tested the 34 highest-ranking models (as ranked by AIC weights) to see how well they predicted fisher detection data derived from a relatively independent fisher survey data set provided by William Zielinski (Zielinski et al. 2000). These test data were obtained using similar protocols, and a subset of the same locations, as monitored for the Region 5 data set, but during a different time period (1998-1999 rather than 2001-2005; Zielinski et al. 2005). We also tested the top 34 models using the MAPE response variable, which includes those additional

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[16] Note that for various pragmatic reasons, we have not strictly followed the information-theoretic philosophy to model building and testing. These reasons include the emergence of new predictive variables and insights about variable interactions gained during analyses. More importantly, the peculiar requirements of this project required us to select models that are useful for detecting habitat changes using variables that are responsive to modeled fires and fuels treatments. This "ad hoc" approach to model selection and refinement is counter to the strict interpretation of the information-theoretic approach (Burnham and Anderson 2002; R Truex and W. Zielinski, personal communications). We nevertheless maintain it is justified for pragmatic reasons and due to schedule constraints.

[17] We restricted models to including no more than four, or in a few cases five variables, due to high correlations between many variables, the potential to "overfit" models with many variables (Burnham and Anderson 2002, Hastie and Tibshirani 1990), and our inability to conceive how more than about four variables might interact with each other to affect predictive success (W. Zielinski, personal communications).

[18] Akaike's Information Criterion (AIC) measures how well an estimated statistical model fits a set of data (e.g., species presence-absence data). It balances the complexity of the model (number of variables included) against how well it fits the data. In comparing a set of alternative models, the one having the lowest AIC value (or highest AIC weight) theoretically optimize this balance between simplicity and fit.

[19] The AUC (or Area Under Curve) of the ROC (Receiving Operator Characteristic) is a measure of a model's predictive skill which combines sensitivity (correct classification of fisher presence points) and specificity (correct classification of absence points) into a single index. An AUC of 0.5 indicates no predictive skill; an AUC of 1.0 indicates perfect predictive skill; and an AUC >0.8 is generally considered a "good fit."

Region 5 monitoring points that had been surveyed only once or that had yielded only one fisher detection—thus representing a potentially more liberal interpretation of fisher habitat suitability.

Finally, for the top 19 models, we evaluated their predictive ability using five-fold cross-validation. For this, the MAPE2 dataset was randomly divided into five subsets of approximately the same size and detection/nondetection ratio. The models were estimated from 4/5 of the data and applied to the remaining 1/5. This was repeated for the other four subsets. The mean and standard deviation ROC AUC were calculated from the five prediction sets for each model and compared to ROC AUC derived from the full dataset. This gives a measure of the relative degree of uncertainty around model predictions.

2.1.3 Resting Microhabitat Model

An existing parametric version of a non-parametric model was used to determine fisher resting microhabitat suitability using data from Forest Inventory and Analysis (FIA) plots (Zielinski et al. 2006). The model variables, in decreasing order of importance, were maximum tree dbh, basal area of small trees (<51 cm dbh), percent slope, average canopy cover, maximum conifer snag dbh, and hardwood dbh. Spatially accurate FIA plots and associated data were provided for the study area by the USDA Forest Service, Pacific Norwest Research Station. We applied the model to FIA plot locations between 1100 and 2300 meters elevation[20] on the west slope of the Sierra Nevada as a representative spatial sample of microhabitat value and to compare with home-range scale models of habitat value. Resting microhabitat suitability was thus calculated at 267 FIA plots, mapped, and compared to predicted coarse-scale habitat value across the same locations.

2.1.4 Baseline Population Assessment[21]

We used the spatially explicit population model PATCH (Program to Assist Tracking Critical Habitat; Schumaker 1998) to estimate a realistic range of equilibrium population sizes (or carrying capacities) for fishers within the currently occupied habitat areas of the southern Sierra Nevada, and to identify likely source, sink, and population expansion areas. This approach is very similar to that that used by other authors on fishers and other species for conservation planning purposes (e.g., Lewis and Hayes 2004, Carroll et al. 2002, Carroll 2007). We compared population estimates derived using PATCH to estimates derived by (1) extrapolating field measures of population density (Jordan 2007) over the area of predicted suitable habitat and (2) estimates derived by extrapolating the fisher monitoring data and making assumptions about the number of fishers detected at each monitoring location (R. Truex, personal communications).

PATCH models occupancy of territories by females over time within hexagons set to average female territory size based on mean habitat value within each hexagon, species' dispersal

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[20] The Zielinski et al. (2006) resting microhabitat model was built using FIA data collected within this elevation range west of the Sierra Nevada crest, and so was applied only within these limits to avoid extrapolation errors.

[21] The sections of this report concerning population estimation and population dynamics were not included in our August 2007 Draft Report. However, these new sections have been reviewed, commented on, and approved for inclusion in this Final Report by Science Advisors R. Barrett, W. Zielinski, and K. Aubry, as well as by R. Truex. The approach used herein to assess effects of habitat configuration on population dynamics (using program PATCH) replaces the spatial configuration analyses from our August report (using program FRAGSTATS) which was pointed out by Advisor F. Davis to be problematic (see Appendix B).

characteristics, and age-specific survival and fecundity rates as functions of habitat value. Accounting for males (based on assumed sex ratio) one can also estimate total population size. As a dynamic model that couples population processes (birth, death, dispersal) with spatially explicit habitat values, PATCH is more useful than static models for assessing population viability and for revealing which areas may be most important to sustaining a population or metapopulation, such as core (source) areas or critical landscape linkages or movement corridors (Carroll et al. 2002, 2003a, 2003b; Carroll 2006). This approach also addresses shortcomings identified for deterministic population viability models (e.g., Lamberson et al. 2000).

Territory Size – We used three estimates of fisher home range[22] or territory size (hexagon size) that we assume bracket the range of actual territory sizes for female fishers within the study area, based on Zielinski et al. (2004b) and Mazzoni (2002):

- 500 ha: This is rounded down from the 527 ±65.1 SE found by Zielinski et al. (2004b) using radio telemetry on seven female fishers on Sequoia National Forest, Tulare County.
- 1,200 ha: This is rounded up from the 1,192 ha reported by Mazzoni (2002) using radio telemetry and capture locations on seven female fishers in the Kings River Administrative Study Area, Sierra National Forest.
- 860 ha: This is the median of 527 and 1,192 ha.

Table 2.1. Cross walk of mean probability of occurrence (habitat value) calculated using habitat model LAND8 and integer habitat scores applied to territories in PATCH.

Predicted Probability of Occurrence	**Habitat Score in PATCH**
0	0
0.001 – 0.1	1
0.101 – 0.2	2
0.201 – 0.3	3
0.301 – 0.4	4
0.401 – 0.5	5
0.501 – 0.6	6
0.601 – 0.7	7
0.701 – 0.8	8
0.801 – 0.9	9
0.901 – 1.0	10

Habitat Associations – Habitat scores in PATCH were assumed to be directly related to probability of fisher occurrence calculated with the coarse-scale habitat model we selected (LAND8) (Table 2.1). Each 1-ha pixel within a territory hexagon was provided an integer score from 0 to 10 based on Table 2.1, and the mean habitat score across all pixels was used as the territory score.

Demography – PATCH uses age-specific survival and fecundity rates in a population projection matrix (a stage-based Leslie matrix). Age-specific fecundity is the product of mean annual survival rate, mean percent of females producing young, mean litter size, and proportion of females in the litter (assumed 0.5). The demographic parameters are input as those expected in optimal habitat, and scaled appropriately to reflect site-specific habitat value in each model territory. We based the maximum demographic parameter values on the "high" values used by Lamberson et al. (2000) and assumed a 4-stage Leslie matrix (Table 2.2). These high survival and fecundity values were assigned to hexagons having territory scores ≥ 7.5 (the "ceiling" value in PATCH) with values discounted linearly from this ceiling with declining habitat value (x

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[22] The terms territory and home range are treated as synonymous for adult female fishers. PATCH assumes that reproductive females use home ranges that are exclusive of other reproductive females, which is the case for female fishers and many other carnivores, which exhibit intra-sexual territoriality (minimal overlap between same-sex home ranges; Powell and Zielinski 1994).

intercept = 0; exponent = 1). A ceiling value of 7.5 was also used by Lewis and Hayes (2004) and assumes that fitness benefits from habitat do not increase further beyond about 75% of maximum predicted habitat occupancy value. The lambda value (intrinsic rate of population growth) for the matrix of high values is 1.19, indicating that territories receiving scores >7.5 are expected to be source areas (not accounting for landscape context). The discounting of demographic parameters with declining habitat scores below the ceiling moderates the maximum parameter values across the landscape, such that moderate-value territories are relatively neutral (lambda near 1.0) and poor territories are likely sink areas (lambda < 1.0).

Table 2.2. The 4-stage Leslie matrix showing maximum fecundity and survival values based on the "high" values developed by Lamberson et al. (2000). These maximum values apply within the highest value territories (territory score > 7.5). The finite rate of change (lambda) of a population characterized by this matrix would be 1.19 if all territories were high value.

Stage	0 (kits, 0-12 mo)	1 (subadults, 13–24 mo)	2 (subadults, 25–36 mo)	3 (adults, >36 mo)
Fecundity	0	0.315	0.49	1.62
	0.5			
Survival		0.7		
			0.7	0.9

To assess sensitivity of predicted equilibrium population size to these parameter values, we independently decreased each stage-specific value by 25% and observed the percent change in predicted population size. We also simultaneously decreased fecundity across all age classes, and survivorship across all age classes by 25%.

Site Fidelity – Site fidelity in PATCH determines the likelihood of an individual remaining on (versus abandoning) a territory from one year to the next. We set this to high for Stages 2 and 3 (animals > 24 months old) because female fishers, like other mustelids, appear to maintain stable territories once established (W. Zielinski personal communication). In PATCH, individuals never give up their territories (barring mortality) when site fidelity is set to high (Schumaker 1998). We set fidelity to medium for juveniles and yearlings (Stages 0 and 1), although young are forced to leave territories anyway as only one female can breed per hexagon. When site fidelity is set to medium, individuals will decide to move depending on territory quality and occupancy (leaving expected sink hexagons and remaining in neutral or source hexagons if unoccupied by another fisher).

Dispersal – As also assumed by others for fishers and other carnivores (e.g., Lewis and Hayes 2004, Carroll et al. 2004) we used the directed random walk option to model dispersal, in which movement decisions combine some degree of randomness with a tendency to continue in the direction of the last step. Animals are attracted to higher quality habitat but have no knowledge about habitat quality beyond territories immediately adjacent to their current location. Variability in the distance and direction moved provides stochasticity. The maximum distance moved determines when the animal must settle into a territory, even if it is of poor quality, but in many simulations the animal will encounter a suitable territory and settle before reaching the maximum distance. We tested maximum dispersal distances of 25, 50, and 100 km and found little or no effect on population estimates (see Results). For most subsequent runs, we therefore used the intermediate value of 50 km, which was also used by Lewis and Hayes (2004) based on available data on maximum observed dispersal and mean weighted dispersal distances from several studies.

Initialization – We initialized simulations using the same three starting numbers of adult females as Lamberson et al. (2000) – 50, 100, and 200 – which represented their "best guesses as to the range of number of females that currently exist in the southern Sierra population." Across all runs at different territory sizes, the initial population size had virtually no effect on equilibrium population size (as might be expected unless populations go extinct), so most runs used the intermediate starting size of 100 females.

At time zero, all individuals are started at Stage 3 (PATCH does not allow initializing with mixed stages, so the model must be run for a number of generations before an equilibrium in age structure is established). Individuals were placed on the landscape at the highest value hexagons. All hexagons having fisher detections in the monitoring data (MAPE = 1) were given an initial habitat value of 10 (highest value) on Year 1 to force placement of the starting population where we already know fishers occur. After Year 1, all hexagons reverted to their modeled habitat value as described above, so that simulated fishers could redistribute from this initial distribution to fill areas of suitable habitat.

Simulations – For each set of territory sizes, dispersal distances, number of initial females, and demographic parameters, we ran 20 replicates for 40 years. Because at Year 1 all animals are started at Stage 3, it takes at least 10 years to reach an equilibrium in population age structure and size (see Results), so the first 20 years are discarded, and results were reported from Years 21 to 40. Regardless of initial population size (50, 100, or 200 females), runs at any given territory size converge on a concordant range of equilibrium population sizes by Year 40 except for runs where extinction occurred due to low survival rates (see Results). We therefore recorded the mean, standard deviation, median, minimum, and maximum number of females at Year 40 to compare results among different models. For Years 21 – 40, we also mapped and calculated the total number of territories (and total area) occupied at least once, the average yearly occupancy of each territory (total number of females per territory divided by 20 replicates and 20 years)[23], percent of available territories (defined as territory score > 0) occupied per year, and mean births minus deaths per territory (standardized by number of years and replicates tallied). The last metric, called net value, is an index of source and sink territories.

To first estimate equilibrium population size within the currently occupied habitat areas, we set habitat value for areas north of the Merced River to zero to avoid having model fishers disperse and establish territories there. To assess the potential for the fisher population to expand northward under current habitat conditions, we next allowed habitat values to revert to their modeled value north of the Merced River at Year 20 (once the model population approaches equilibrium within the currently occupied region) and observed territory occupancy patterns for years 21-40. (In essence, model Year 20 represents the present time, and Years 21-40 represent 20 years into the future.) To estimate total fisher population size from model results, we multiply the mean number of adult females at Year 40 by 2, assuming a 1:1 adult sex ratio (Jordan 2007, Powell and Zielinski 1004).

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[23] Tallies include territorial females plus non-territorial floaters and any juveniles residing in their mother's territory.

2.2 Results

2.2.1 Home-Range-Scale Habitat Models

Table 2.3 summarizes characteristics of the top 10 performing habitat models (and Appendix F lists all candidate models we compared) in order of AIC weights, from highest to lowest. All of the top models included a combination of two abiotic variables—adjusted elevation plus either insolation index or annual precipitation—with one or more biotic (vegetation) variables. Together, elevation and either annual precipitation or insolation index (which integrates slope and aspect into an index that correlates strongly with the annual exposure of land surface to solar radiation; Gustafson et al. 2003) are known to explain a high degree of variation in potential vegetation characteristics (Pierce et al. 2005). Thus, these two sets of abiotic variables each appear to correlate strongly with the potential of the land surface to support habitat favorable to fishers (e.g., large, dense forest) while also influencing the physical environment directly experienced by fishers (e.g., snow depth and microclimate). However, these abiotic factors cannot account for how disturbance history (e.g., fires, logging) affects actual vegetation. Notably, all of the highest ranking models also included one or two biotic variables that reflect aspects of vegetation composition and structure, such as total tree biomass or mean maximum tree age. The biotic variables included in top models all reflect previously established characteristics of fisher habitat quality, such as measures of tree size, age, and density. The results confirm that fishers are closely associated with dense, old, large forests in mid-elevation zones of the southern Sierra Nevada.

The highest ranking model (LAND8; Figure 2.1) combined elevation and annual precipitation with the total above-ground biomass of trees. It has very high AUC[24] values (0.941 using MAPE2 and 0.831 for MAPE) and explains 53.3% of deviance. No competing model was within 2.0 AIC units of LAND8, meaning that LAND8 was a clear "winner" with no need for model averaging[25]. All three variables (Figure 2.2) contributed significantly ($P < 0.05$ or better) to the predictive power of the model. Figure 2.3 evaluates the models' predictive accuracy. It shows frequency distributions for locations where fishers have been detected at least twice (unshaded bars) vs. never detected (shaded bars) relative to predicted habitat value (probability of fisher occupancy). Note that the vast majority of absence points are in sites with predicted probability of occupancy less than 0.3, and most fisher presence points are in sites with high predicted probability, but that there is overlap in the distributions. This is normal and expected: As mobile animals, and given the complexities of wildlife-habitat relationships, fishers may sometimes be detected in areas of lower predicted probability.

Figure 2.4 shows the partial response curves for model LAND8. These curves illustrate the relationship between each variable and modeled habitat value *given all variables in the model*. Each panel cannot be read as depicting the univariate relationship between the variable and fisher distribution, but must be evaluated in context with the other variables. Given this, it appears that habitat value increases with total forest biomass and is highest over intermediate elevations

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[24] An AUC of 0.5 indicates a model with no predictive power, an AUC of 1.0 indicates perfect predictive power, and an AUC > ~0.8 indicates a "good fit."

[25] If no model dominates in AIC weight (i.e., is at least 2.0 AIC units higher than any other model) then model averaging is generally recommended; otherwise the model with the lowest AIC weight should be selected as the strongest model (Burnham and Anderson 2002, W. Zielinski personal communication).

(~1300 – 2400 m). Finally, for those mid-elevation areas that have high forest biomass, fishers appear to favor those with less annual precipitation (PRISM). This may reflect that fishers appear to favor areas with less winter snow, as they are not highly adapted for traveling and foraging in deep snows.

Table 2.3. Statistical comparison of the top 10 fisher habitat GAM models, ranked in descending order by AIC weights. Metrics also include the % deviance explained; area under curve (AUC) for the receiver operating characteristic using the MAPE2 (conservative) and MAPE (liberal) interpretations of the fisher detection-nondetection data; AUC for the temporally independent fisher survey data test data set (Zielinski et al. 2000); and the mean and standard deviation of the five-fold cross-validation test using MAPE2. See Appendix C for variable definitions. The model shown in bold was selected for use in assessing current and future habitat conditions for reasons provided in text.

Model	Variable s	AIC_c Weights	% Deviance Explained	AUC MAPE2	AUC MAPE	AUC TEST SET	Mean 5-fold cross-validated AUC	SD 5-fold cross-validated AUC
LAND8	ADJELEV, PRISM, BIOMASS	0.68974	0.53260	0.94097	0.83145	0.63828	0.90499	0.07079
LAND7	ADJELEV, INSOL_INDEX, BIOMASS	0.15767	0.51768	0.93302	0.82264	0.61370	0.90332	0.05452
LAND13	ADJELEV, INSOL_INDEX, MAXAGE, BIOMASS	0.05790	0.51829	0.93372	0.82092	0.61370	0.89888	0.05642
LAND1	ADJELEV, INSOL_INDEX, MAXAGE	0.04921	0.50591	0.93001	0.82755	0.59438	0.87871	0.04170
LAND14	ADJELEV, INSOL_INDEX, MAXAGE, BIOM_NORF	0.03415	0.51296	0.93478	0.82685	0.59350	0.87069	0.04578
LAND3	ADJELEV, PRISM, MAXAGE	0.00732	0.48666	0.92294	0.82732	0.62511	0.88197	0.06829
LAND11	ADJELEV, INSOL_INDEX, BIOM_NORFBO, BIOM_BLKOAK	0.00205	0.48454	0.92153	0.81375	0.59087	0.89119	0.05815
LAND10	ADJELEV, INSOL_INDEX, BIOM_NORF	0.00091	0.46560	0.92082	0.82015	0.63652	0.87956	0.03619
LAND15	ADJELEV, INSOL_INDEX, BIOM_NORFBO, BIOM_BLKOAK, MAXAGE	0.00070	0.48454	0.92153	0.81375	0.59087	0.89119	0.05815
LAND12	ADJELEV, INSOL_INDEX, BIOM_BLKOAK	0.00011	0.44466	0.90597	0.80393	0.61896	0.88232	0.06142

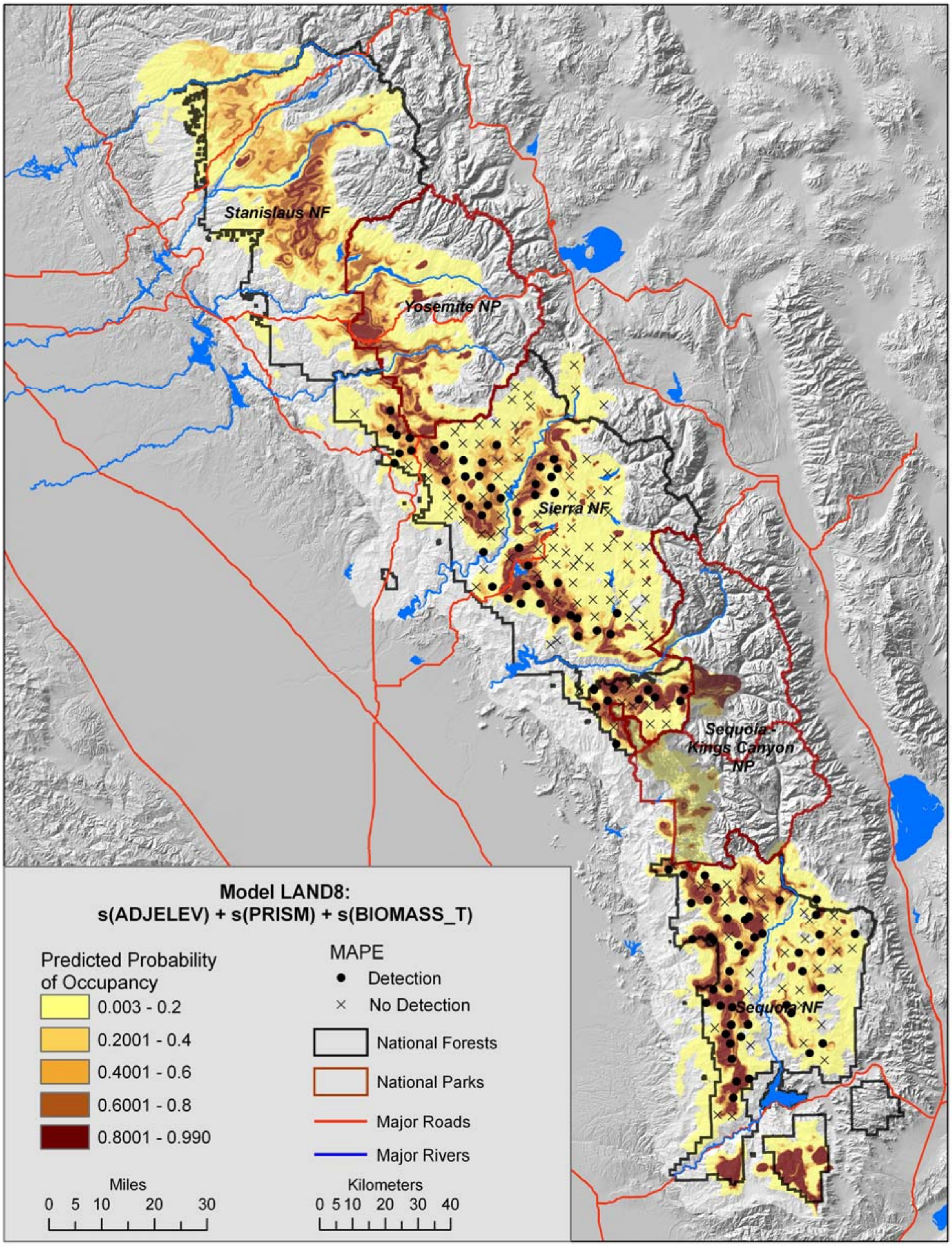

Figure 2.1. Predicted fisher habitat value (probability of occupancy) from model LAND8 using adjusted elevation, annual precipitation, and total tree biomass within a 5-km2 moving window and compared with fisher detection-nondetection monitoring data. (Data gaps for tree size result in under-prediction of habitat value in portions of Sequoia Kings Canyon NP.)

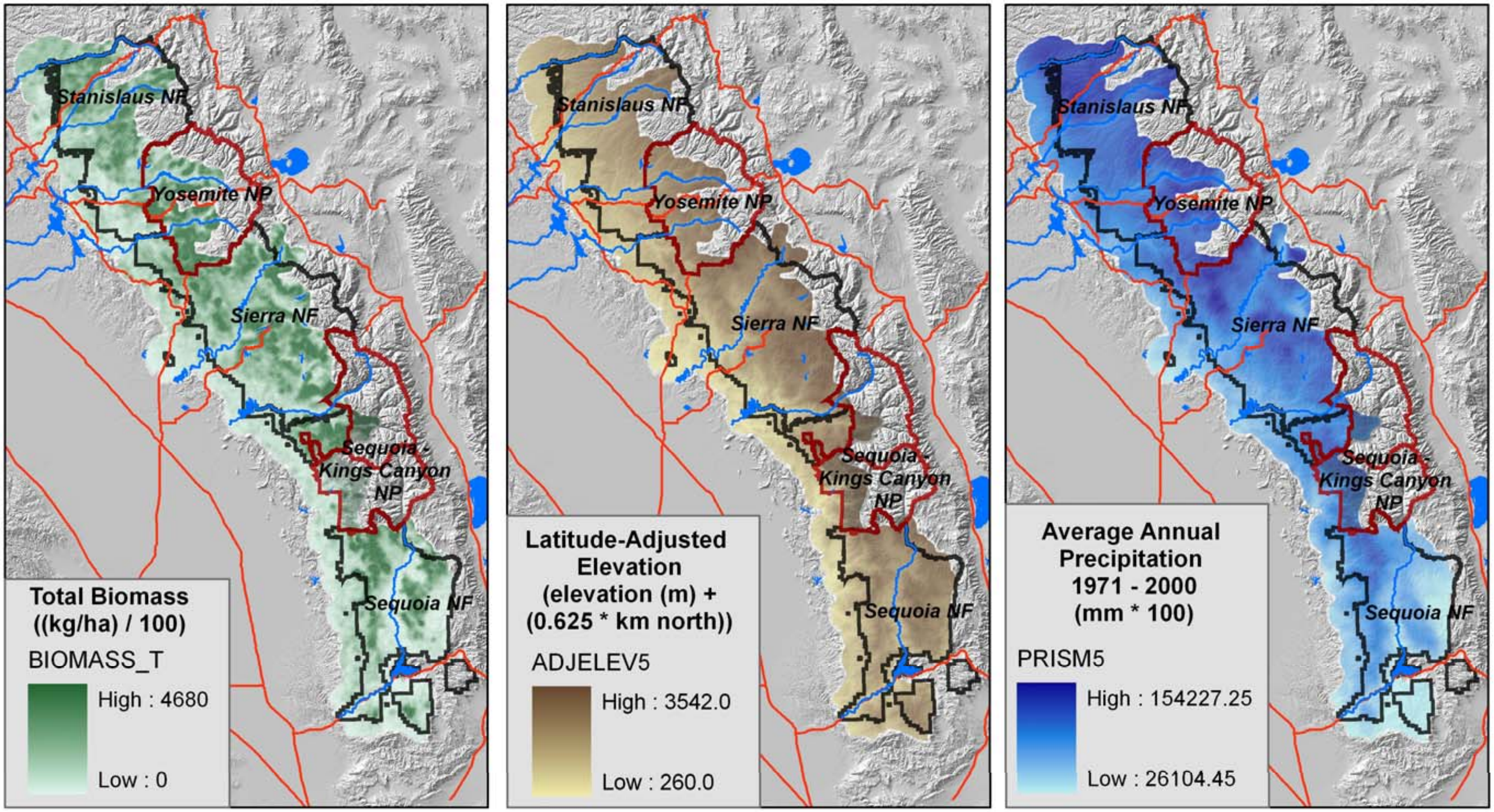

Figure 2.2. Distributions of the values for the three habitat variables included in model LAND8, averaged over a 5-km^2 moving window.

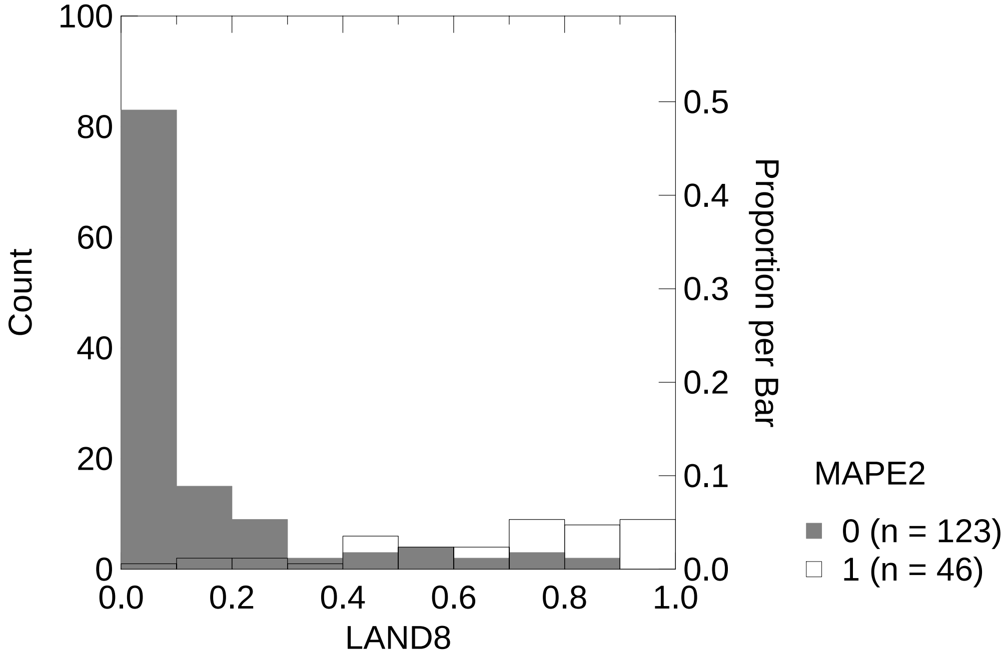

Figure 2.3. Accuracy of model LAND8 in predicting fisher presence and absence. Shaded bars indicate the proportion of fisher survey locations with no fisher detections and the open bars indicate those with detections in at least 2 years (MAPE2 dataset) as a function of predicted probability of occupancy using LAND8.

Amount of Predicted Suitable Habitat – There are over 384,000 ha total area of predicted suitable fisher habitat (defined as predicted probability of occupancy by fisher ≥0.5) based on LAND8 (Table 2.4) in the study area. Approximately two-thirds of the predicted habitat is on Forest Service land with most of the rest on the two national parks (Yosemite and Sequoia-Kings Canyon). It is important to note that on all figures, apparent gaps in habitat distribution on SEKI are at least partially artifacts of data gaps for tree sizes, resulting in under representation of habitat value there.

Table 2.4. Predicted area of currently suitable habitat by National Forest, defined using the 0.5 probability of occurrence cut-point and model LAND8).

Forest	**Hectares**
Stanislaus National Forest	60,361
Sequoia National Forest	90,031
Sierra National Forest	113,998
Subtotal, National Forests	264,390
Other	120,033
Grand Total	384,423

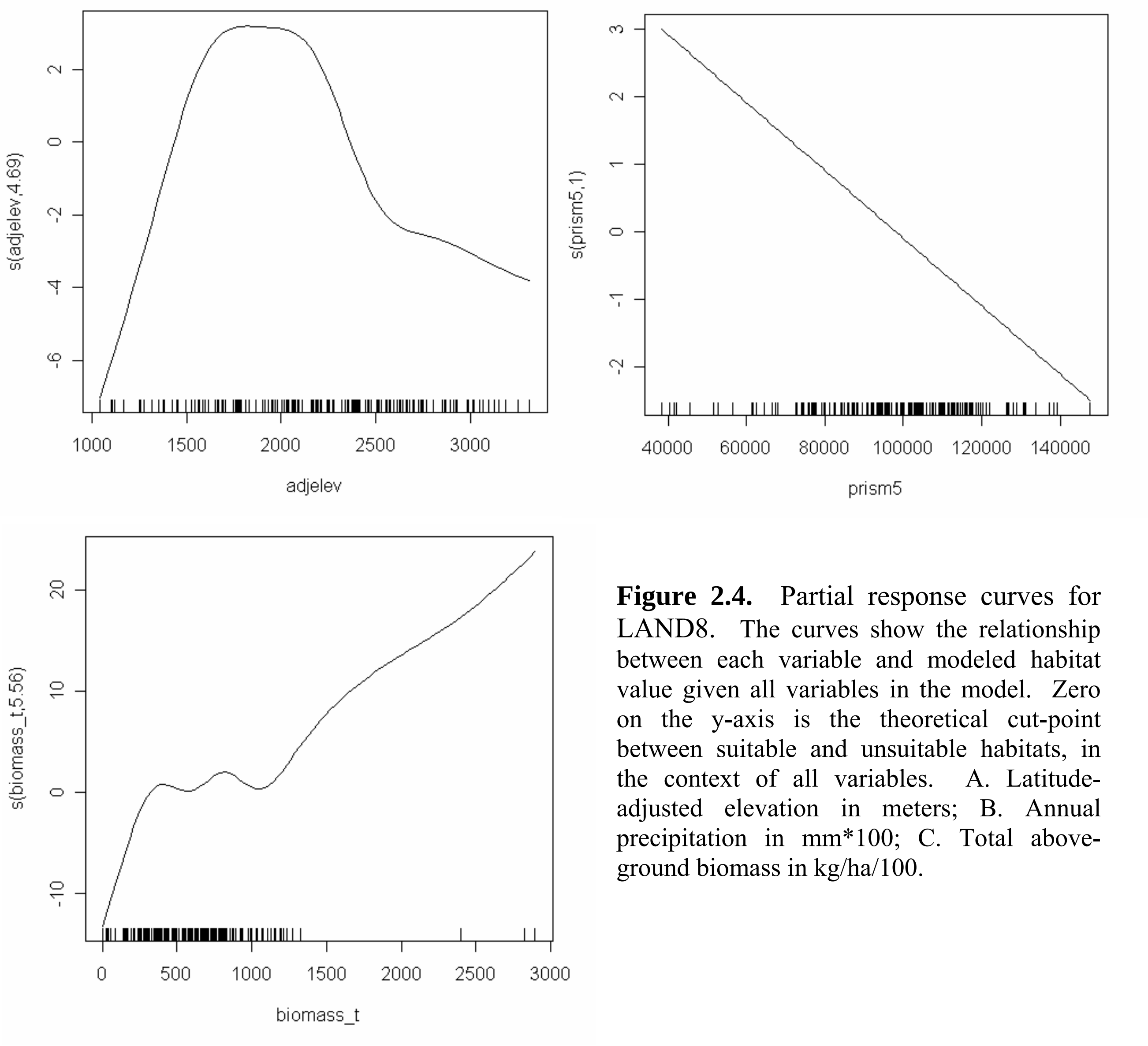

Figure 2.4. Partial response curves for LAND8. The curves show the relationship between each variable and modeled habitat value given all variables in the model. Zero on the y-axis is the theoretical cut-point between suitable and unsuitable habitats, in the context of all variables. A. Latitude-adjusted elevation in meters; B. Annual precipitation in mm*100; C. Total above-ground biomass in kg/ha/100.

2.2.2 Resting Microhabitat

Figure 2.5 illustrates resting microhabitat value, at approximate FIA point locations, overlaid on coarse scale predicted habitat suitability (using model LAND8). Predicted highly suitable resting microhabitats are rare in the study area, and the overall correlation between predicted resting microhabitat value and predicted coarse scale habitat value is moderate (0.290). Only 15 (5%) of the 250 FIA plots that were at least partially forested had predicted resting site suitability values greater than 0.5. Although the highest-rated resting microhabitat plots tend to cluster within areas of predicted suitable habitat at the coarse scale, many lower-value resting plots are also within high-value habitats at the coarse scale. Suitable coarse scale habitat must also contain sufficient resting microhabitat to support fishers (Zielinski et al. 2004a, 2004b, 2006), but how much is enough or how it must be distributed remains unknown.

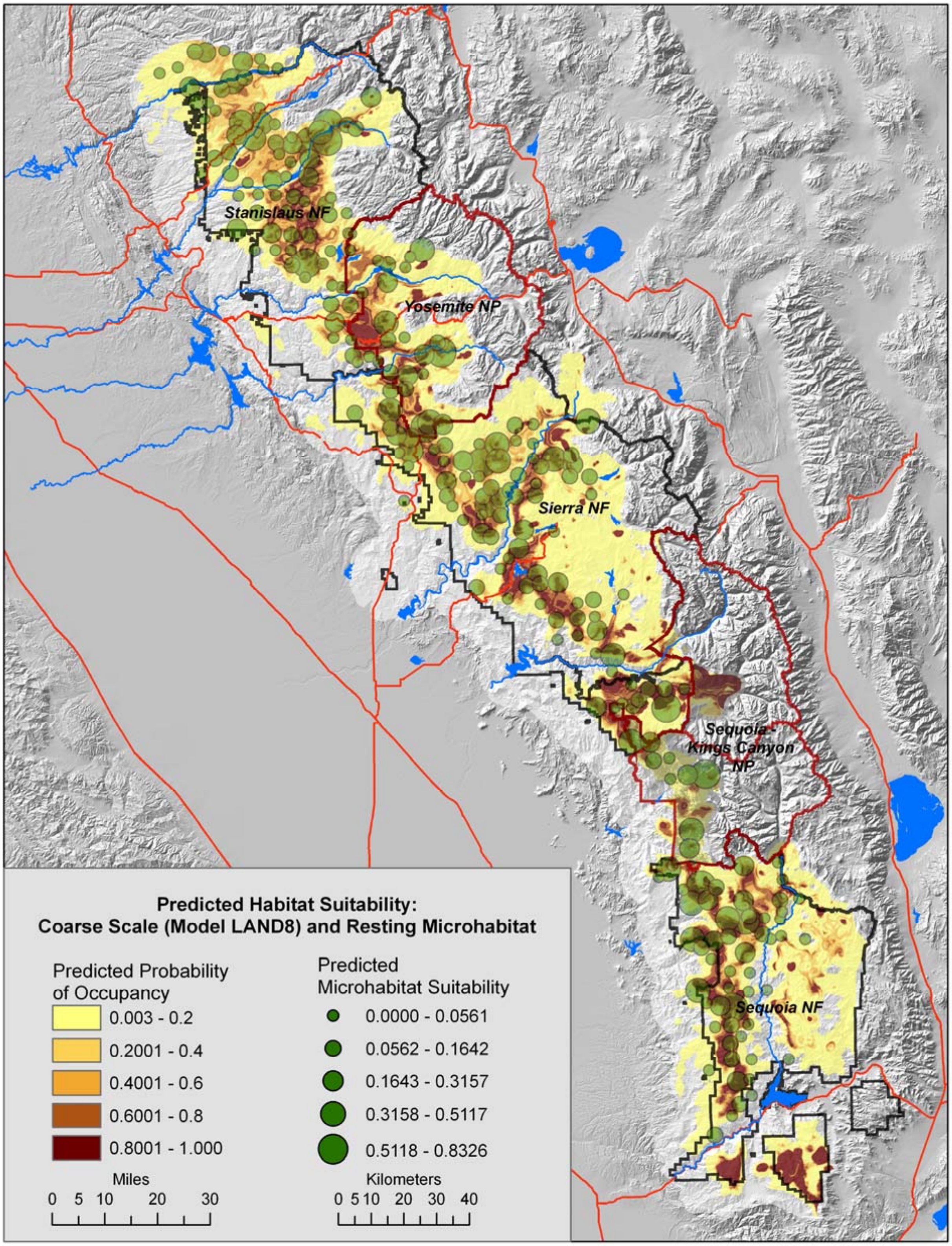

Figure 2.5. Predicted fisher resting microhabitat value at approximate FIA plot locations overlaid on coarse scale predicted habitat value (using model LAND8). (Data gaps for tree size result in under-prediction of coarse scale habitat value in portions of Sequoia Kings Canyon NP.)

2.2.3 Baseline Population Assessment

The equilibrium number of adult females by Year 40 was largely independent of starting population size (Figure 2.6 and Table 2.5), regardless of territory size and maximum dispersal distance, so we default to the intermediate value of 100 initial Stage-3 females in most of the results reported below. Likewise, population size and distribution are only very weakly affected by maximum dispersal distance (Table 2.6) so we default to the intermediate value of 50 km for most results reported below.

Table 2.5. Number of adult females at Year 40 as a function of initial population size. Populations of 50, 100, and 200 Stage-3 females started and run for 40 years, 20 replicates, using 860-ha territory size and 50-km maximum dispersal distance.

Initial Population	Median	Mean	Min	Max	SD
50	132.5	131.2	111	153	10.25
100	135.5	134.8	121	147	8.07
200	142.0	141.0	126	155	8.82

Table 2.6. Number of adult females at Year 40 as a function of maximum dispersal distance. **A**ll populations started with 100 Stage 3 females; 20 replicates, and 860-ha territory size.

Max. Dispersal Distance (km)	Median	Mean	Min	Max	SD
25	131.0	132.2	119	149	8.96
50	135.5	134.8	121	147	8.07
100	136.0	137.4	122	157	10.25

Table 2.7. Number of adult females at Year 40 as a function of territory size. All populations started with 100 Stage-3 females; 50-km maximum dispersal distance, 20 replicates.

Territory Size (ha)	Median	Mean	Min	Max	SD
500	255.5	255.2	241	273	7.64
860	135.5	134.8	121	147	8.07
1200	90.0	89.5	73	106	8.87

In contrast, equilibrium population size is, not surprisingly, strongly affected by territory size (Figure 2.6 and Table 2.7), because density and territory size will generally be inversely proportional, albeit with variance due to other parameter values and stochasticity. Assuming an average territory size of 500 ha, the currently occupied habitat area could support about 241 to 273 adult females (mean = 255.2); at 1,200 ha it could support about 73 to 106 adult females (mean = 89.5); and at the intermediate 860 ha, 121 to 147 adult females (mean = 134.8). For reasons expanded on in the Discussion, we believe that assuming a 500-ha territory size would grossly overestimate population size, and believe that numbers based on the higher territory estimates are more realistic (73 to 147 adult females). We default to the intermediate territory size of 860 ha in most results reported below.

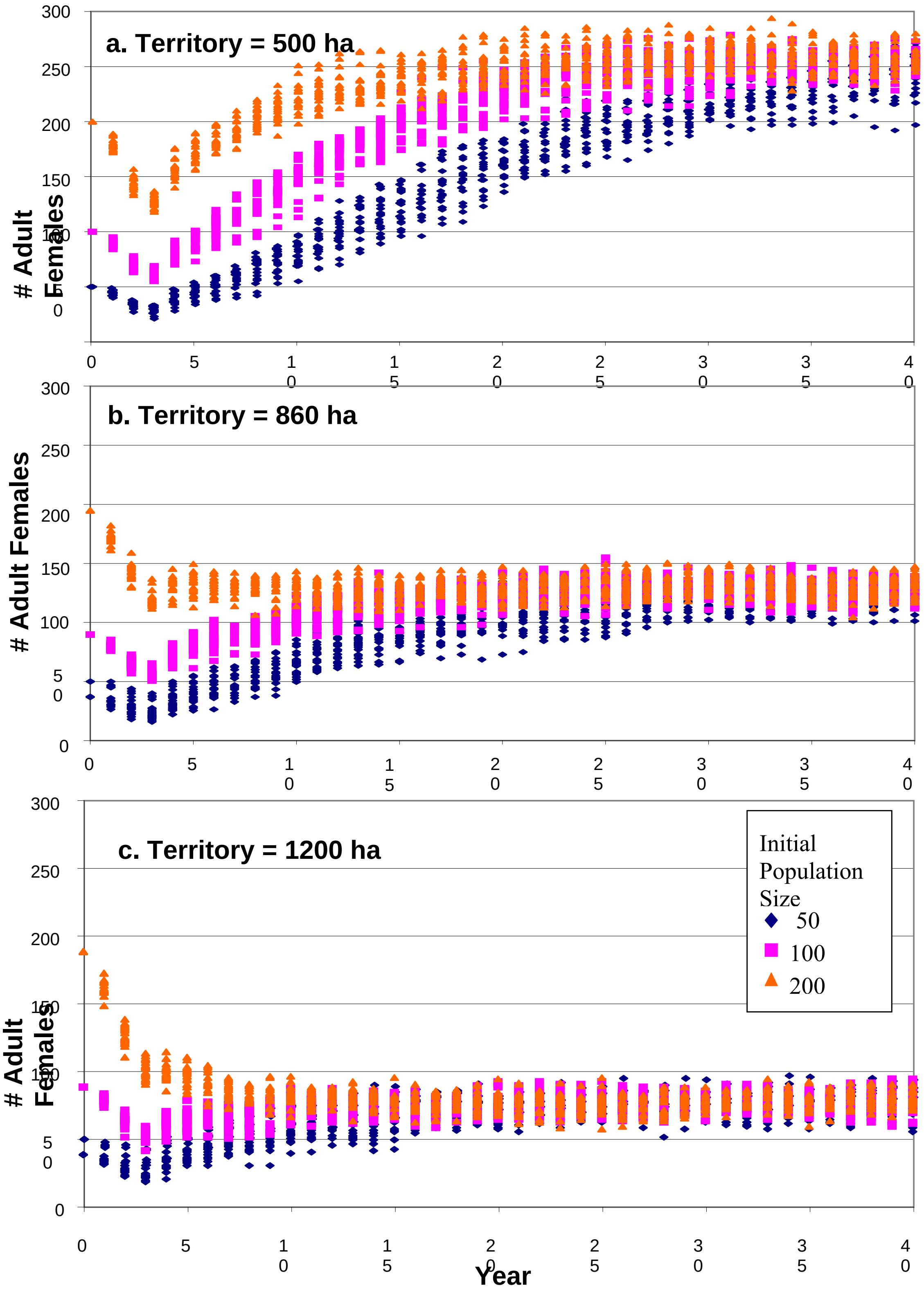

Figure 2.6. Effects of territory size and initial number of females on equilibrium population size south of the Merced River. Maximum dispersal set to 50 km; 50, 100 or 200 females started at Year 0 at each of three territory sizes (500, 860, and 1,200 ha). Years 0-20 are discarded from territory occupancy calculations because dynamics don't equilibrate until around Year 20.

Occupancy Pattern – Figure 2.7 shows overall occupancy of territories predicted by PATCH over Years 21-40, using intermediate values of all varied parameters (860-ha territories, 100 initial population, 50-km maximum dispersal distance). Occupancy is shown relative to results of model LAND8 (with probability of occurrence ≥ 0.1) and fisher detection points (MAPE = 1). Because any territory occupied at least once during 20 years and 20 replicates is included, this map approximates the maximum extent of occupiable habitats under the stated assumptions, including source as well as sink territories. Note that some small and isolated patches of potential habitat are never occupied. Note also there are few occupied territories on the Kern Plateau (Sequoia National Forest east of the Kern River) despite persistent fisher detections there.

Figure 2.8 illustrates average annual territory usage over the 20-year period, averaged over 20 replicates, using the same assumptions as in Figure 2.7. Because the census step in PATCH counts young-of-the-year within their mother's territory prior to dispersal, the highest quality territories can have annual usage > 1.0. The most-used territories clump within the largest contiguous blocks of highest value habitat predicted by LAND8. Sparsely used territories in the western portion of SEKI are at least partly due to missing environmental data, which results in under-representation of habitat value and hence territory value there. Again, territory usage on the Kern Plateau appears to be under-predicted by the model in the face of persistent fisher detections there.

Potential for Population Expansion – When we allow habitat value north of the Merced River to revert from zero to modeled (LAND8) value, the model fisher population gradually expands north into suitable habitat areas. Figure 2.9 illustrates annual territory usage over the 20-year expansion period using the intermediate parameter values (860-ha territory size, 100 starting population, 50-km dispersal distance). Note that within 20 years territories were colonized as far as about 20 km north of the Merced River, to the vicinity of the Tuolumne River, in areas of predicted moderate to high-value habitat. However, as expanded on in the Discussion, it is unclear whether this potential for rapid expansion is real, because the dispersal we modeled in PATCH may not adequately account for dispersal barriers or filters, nor for increased mortality risks that may be associated with crossing steep river canyons, suboptimal vegetation types, and roads.

Source-sink Dynamics – Figure 2.10 maps the observed net value of each hexagon (births – deaths) over the 20-year period (Year 21-40) using the same assumptions as Figure 2.9. Source territories (births > deaths) are shown in greens and sink territories (deaths > births) are shown in reds. Note that source territories tend to be clustered in association with the largest, most contiguous areas of predicted high-quality habitat, whereas most sink patches are peripheral to the best habitat areas. Note also that expansion areas north of the Merced River (approximated by magenta line) include some source territories. The dearth of source territories in SEKI may be an artifact of missing environmental data there that result in under-predicted habitat quality.

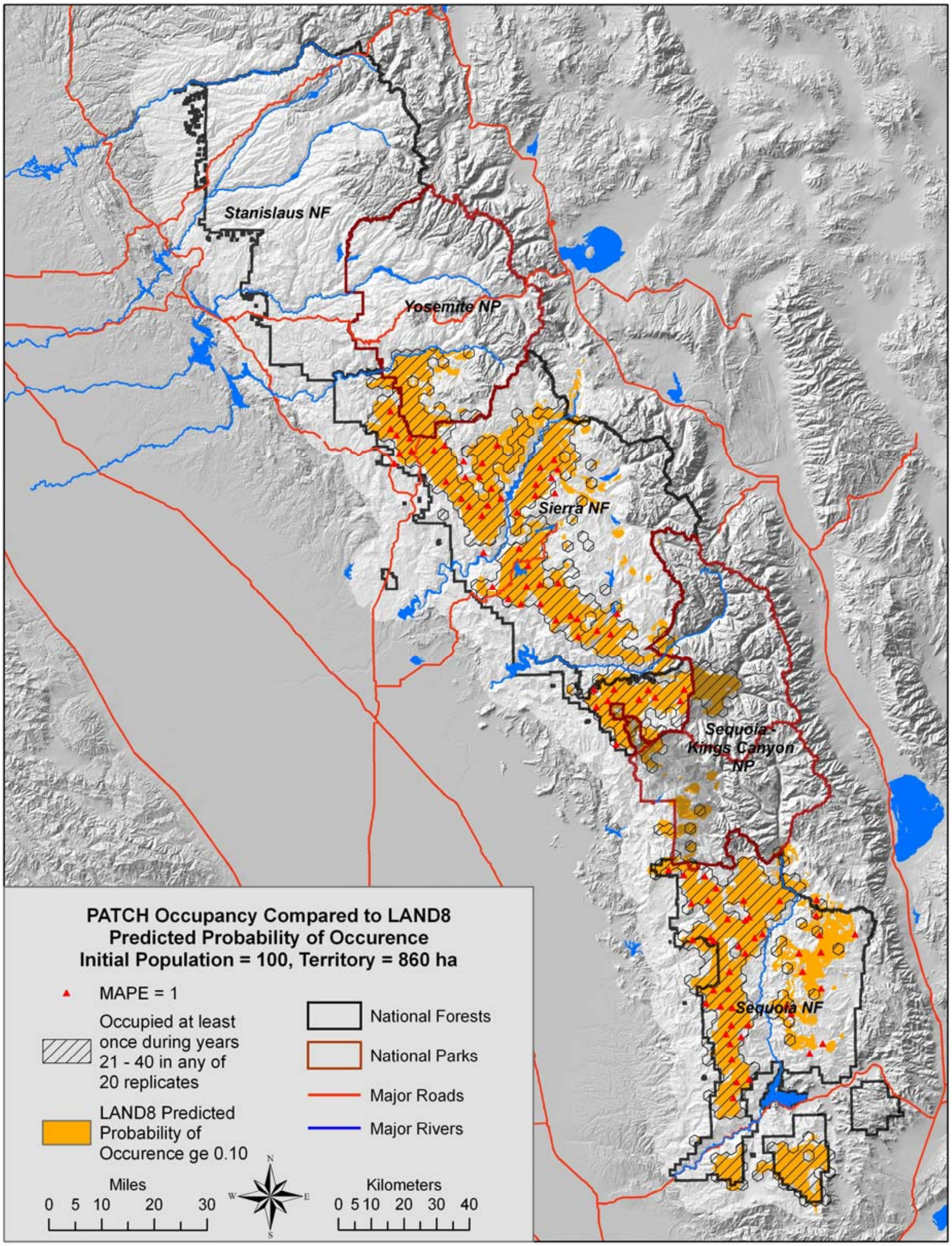

Figure 2.7. Extent of territory occupancy during Years 21-40 predicted by PATCH using intermediate parameter values (860-ha territory size, 100 initial females, 50-km dispersal distance) shown relative to probability of occurrence > 0.1 in LAND8. Habitat north of Merced River set to zero. Red triangles indicate fisher detections from monitoring data. Any territory occupied at least once during the 20-year period and 20 replicates is shown as occupied.

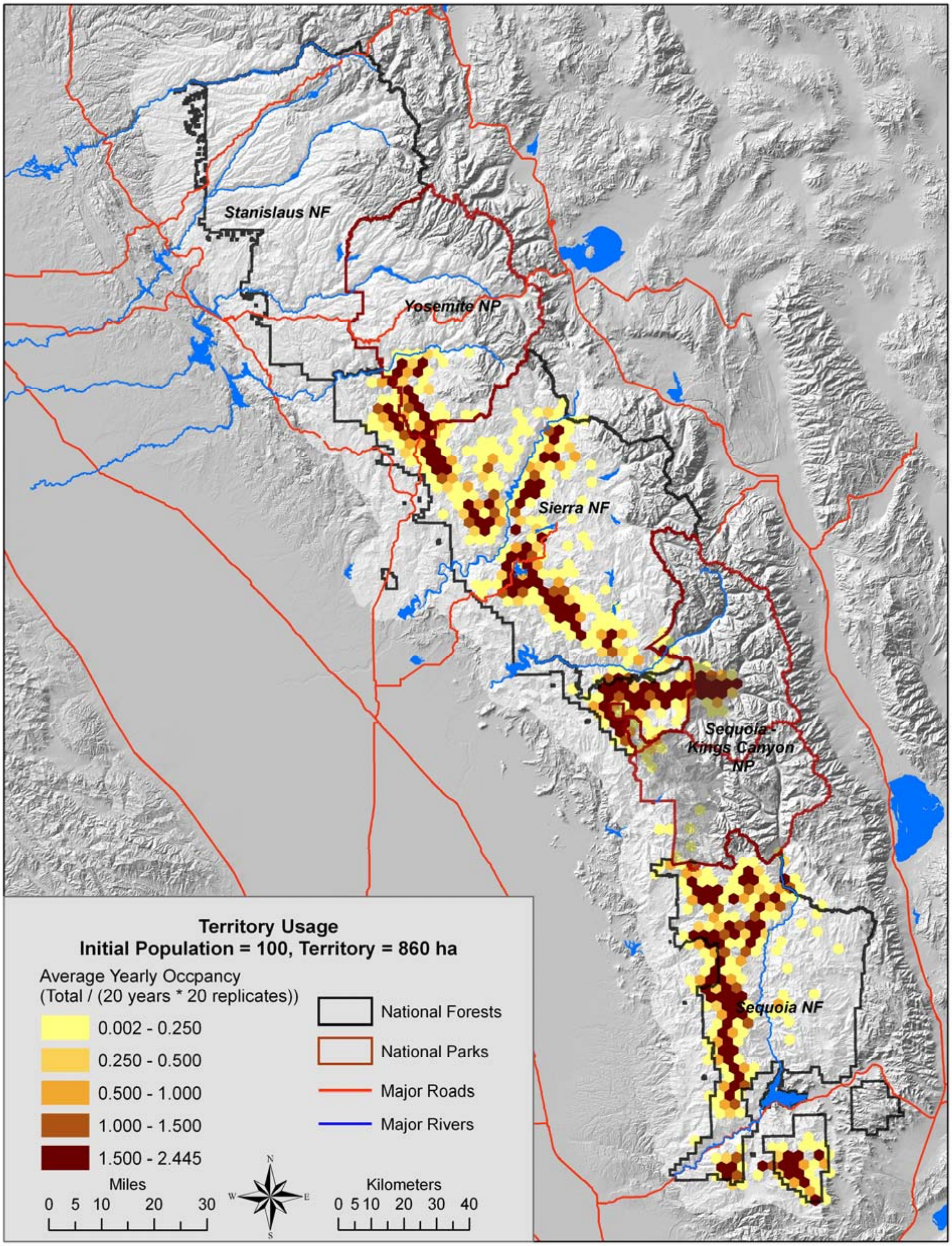

Figure 2.8. Average annual territory occupancy predicted by PATCH using the same assumptions as Figure 2.7. Territory usage includes juvenile females sharing a mother's territory, so highly productive territories can have average yearly occupancies > 1.0. Data gaps may result in under-representation of habitat value and hence territory value in Sequoia-Kings Canyon NP.

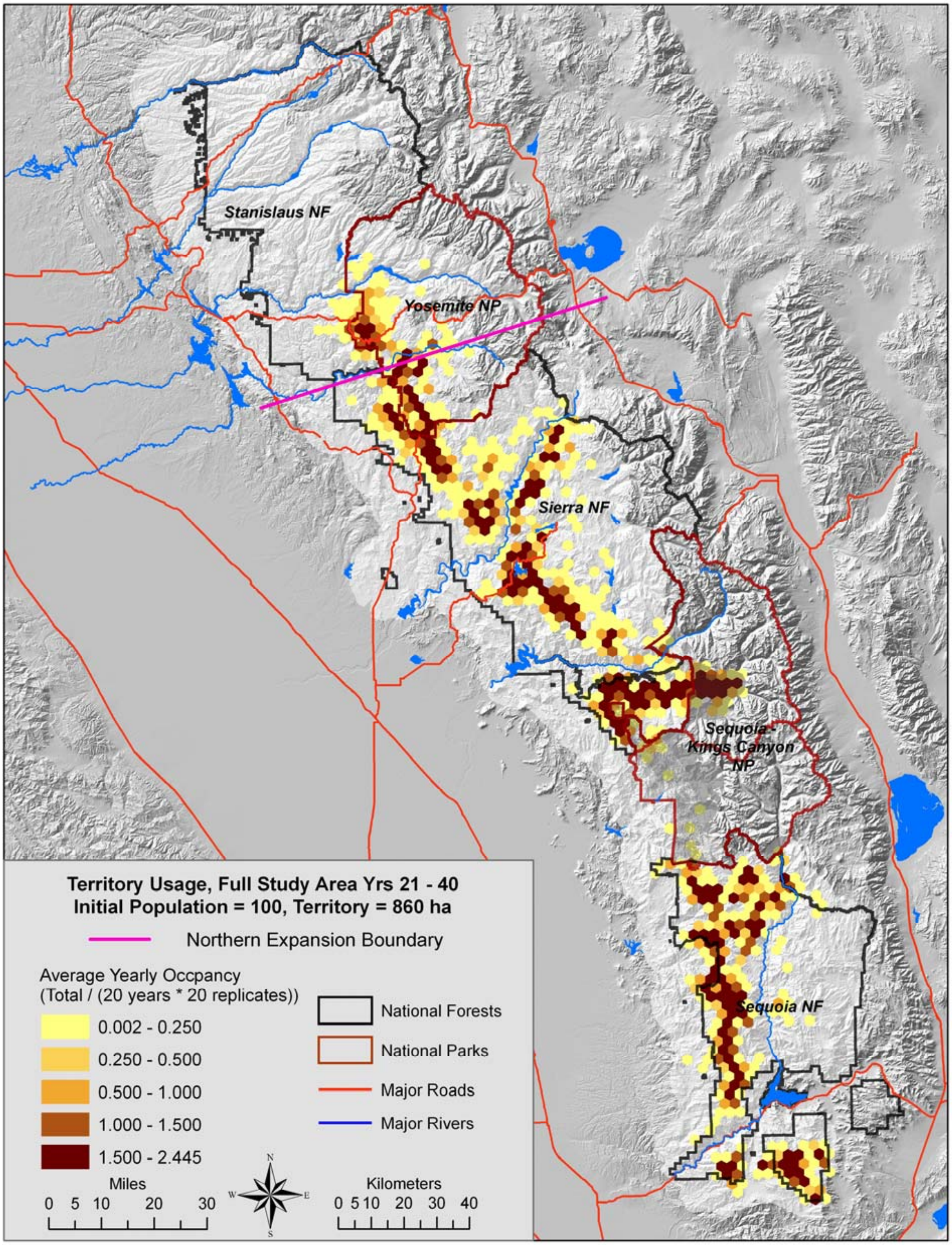

Figure 2.9. Average annual territory usage showing predicted northward expansion over 20 years. All assumptions the same as Figures 2.7 and 2.8 except that habitat value increased after Year 20 from zero to LAND8 value north of the Merced River (magenta line). Data gaps may result in under-representation of habitat value and hence territory value in Sequoia-Kings Canyon NP.

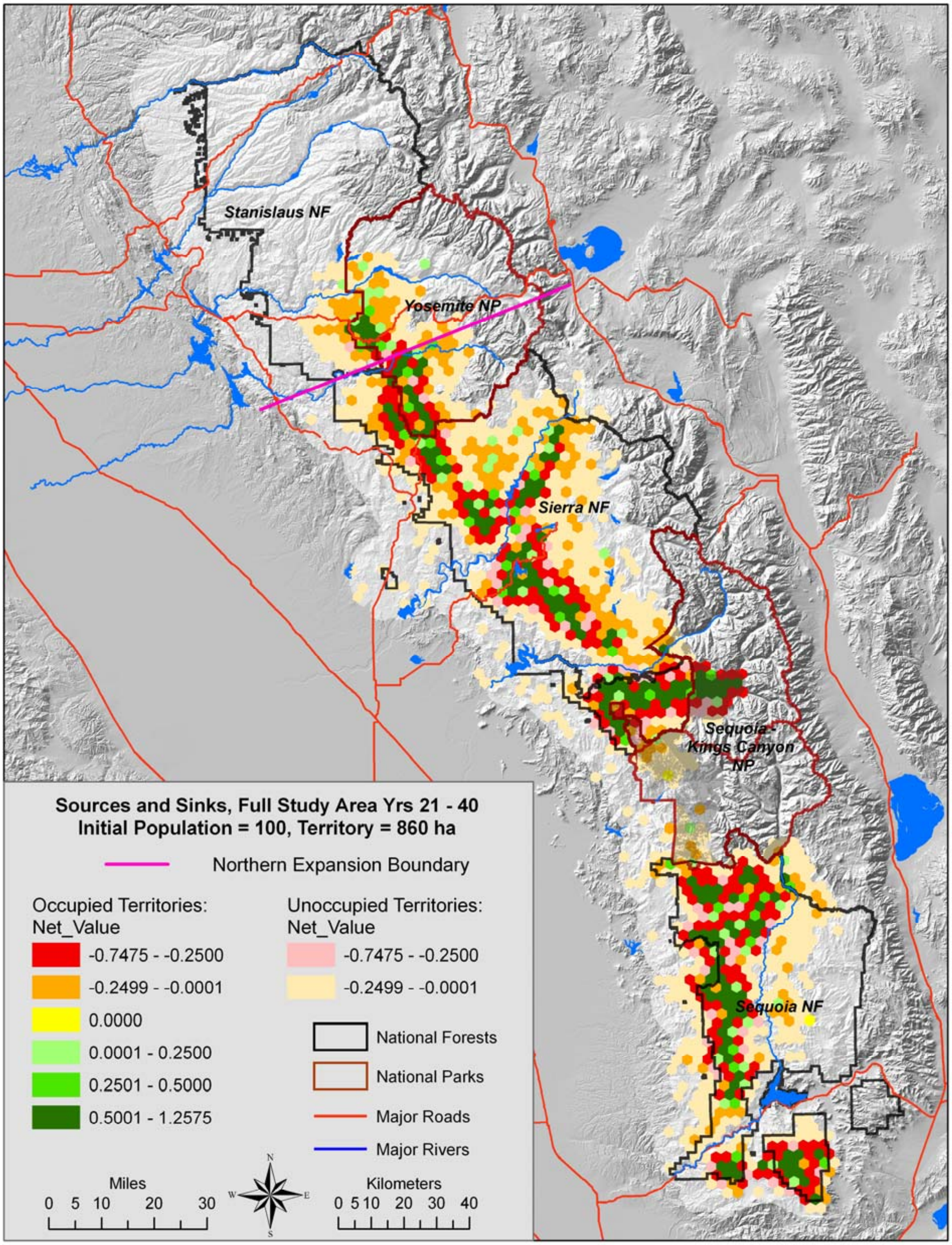

Figure 2.10. Net value of territories (annualized births – deaths) observed over Years 21-40 using the same assumptions as Figure 2.9. Source territories are in greens and sink territories in reds. Data gaps may result in under-representation of habitat value and hence territory value in Sequoia-Kings Canyon NP.

Sensitivity to Demographic Parameters – Similar to results of other fisher population modeling studies (Lamberson et al. 2000; Lewis and Hayes 2004) fisher population dynamics and viability were most sensitive to adult female survival. Decreasing adult female survival by 25% (from 0.9 to 0.675) resulted in a disproportionate 72.63% decline in average predicted female population size at Year 40 (Table 2.8). Although none of these runs resulted in population extinction, as few as 23 adult females remained in the population by Year 40 when adult survival was reduced to 0.675. In contrast, decreasing any other age-specific survival or fecundity rate by 25% resulted in a less than 20% decrease in equilibrium population estimate. If survival is simultaneously decreased by 25% across all age classes, the population crashes (8 of 20 runs resulted in extinction by Year 40, with no more than 2 adult females remaining in the other 12 runs). Simultaneously reducing fecundity across all ages in a similar manner had a much lesser effect, with only a 9.16% average decline in average population estimate and no extinctions.

Table 2.8. Sensitivity of the equilibrium population estimate (number of adult females at Year 40) to stage-specific demographic parameters. Each fecundity and survival rate was decreased by 25% while holding all others at default values (see Table 2.2 for default values). Mean and standard deviation calculated for 20 replicates with each combination of parameters. Percent change is relative to mean population estimate for the default model. Territory size = 860 ha; dispersal distance = 50 km; initial population = 100 adult females. Life stages: S_0 = kits (0-12 months); S_1 = subadults (13–24 mo); S_2 = subadults (25-36 months); S_3 = adults (>36 months).

Parameter Varied	**Mean**	**SD**	**% Change**
None - Default	134.80	8.07	
S_1 Fecundity	135.05	10.07	0.19
S_2 Fecundity	134.70	8.69	-0.07
S_3 Fecundity	123.85	9.16	-8.12
S_0 Survival	115.35	7.04	-14.43
S_1 Survival	112.15	8.43	-16.80
S_2 Survival	109.65	9.11	-18.66
S_3 Survival	36.90	7.58	-72.63
All Stages Fecundity	122.45	5.57	-9.16
All Stages Survival	0.60	0.68	-99.55

2.3 Discussion

2.3.1 Baseline Habitat Condition

Our analyses corroborate existing published descriptions of the current state of fisher habitat in the southern Sierra Nevada, and provide additional tools for assessing current as well as future conditions. Predicted fisher habitat in the southern Sierra Nevada is concentrated within a relatively narrow band of mid elevation forests (on the order of 5-15 km wide over most of its length), mostly on the western slope of the Sierra, in areas of relatively low annual precipitation, in mesic topographic positions (slopes experiencing relatively low annual exposure to solar radiation), and especially where older mixed-coniferous, sequoia, and ponderosa pine stands tend to abut or intermix with montane hardwoods, especially black oak. Predicted highly suitable resting microhabitats are even more restricted, being relatively rare even within areas of predicted suitable habitat at the coarse scale.

Coarse-scale Habitat Distribution – At the coarse scale, predicted suitable habitat is broken into somewhat discrete blocks or segments, generally corresponding with major river canyons, and exacerbated in some places by effects of previous disturbance, including large, severe fires. For example, extensive brush fields and young coniferous forests in the southern portion of Stanislaus National Forest may represent a gap in the distribution of predicted highly suitable habitat near Yosemite NP. We do not yet know enough about use of younger forests, brushfields, or other “suboptimal” habitats by fishers during foraging or dispersal, and it is unknown to what degree gaps in modeled fisher habitat represent barriers or filters to dispersal or to expansion of the fisher population into currently unoccupied habitat (W. Zielinski, R. Truex, R. Barrett, and K. Aubry, personal communications).

Much of the predicted suitable habitat on Sequoia and Sierra National Forests (roughly 204,000 ha total) appears to be occupied by fishers (except perhaps for the Piute Mountains and Breckinridge Mountain at the southernmost end of the study area), whereas large areas of predicted suitable habitat on the Stanislaus National Forest are not. Although the band of predicted habitat appears to be generally wider on the Stanislaus National Forest than on the steeper terrain to the south, it is mostly moderate value, with few locations having predicted very high probability of occurrence. There have been no fisher detections north of the Merced River, although all of the top-ranking habitat models, including LAND8 (see Figure 2.1), show varying degrees of moderate to high-value habitat there. Absence of fishers north of the Merced could be due to historical extirpation of fishers there and inadequate opportunity for fishers to recolonize (due to one or a combination of factors, from movement barriers or filters, to lack of sufficient reproduction and dispersal from the fisher population to the south, or simple lack of time).

Inaccuracies for the Kern Plateau – Most of the home-range-scale models we tested failed to predict as much suitable habitat on the Kern Plateau as would be suggested by the annual fisher detections in that area (R. Truex, personal communications). LAND8 performed better in this region than any other model, producing a reasonable fit to the detection-nondetection points by predicting a fair amount of moderate-value habitat and scattered patches of high value. However, most of this occupied habitat is not predicted to be highly productive by the fisher population dynamics model (see Figure 2.10). The apparent under-prediction of fisher habitat on the Kern Plateau could be due to one or a combination of factors, including limited fisher representation in the monitoring dataset (fisher typically account for <20% of the annual detections recorded by the monitoring program) or fundamental ecological differences between the west slope of the southern Sierra Nevada and the typically drier, higher elevation Kern Plateau (R. Truex, personal communication). The Kern Plateau is ecologically distinct from other portions of the study area, with less precipitation, gentler and less incised terrain, and perhaps different vegetation composition and structure (Miles and Goudey 1998). It is unclear whether these unique environmental conditions confound geographically broader habitat models (see also Davis et al., 2007). In addition, the 2002 McNally fire burned 60,985 ha (150,696 ac) on the Kern Plateau. It is unclear how this mixed-severity fire may have affected fisher occupancy patterns (R. Truex, personal communications). Alternative hypotheses for poor model predictions on the Kern Plateau could be that fishers detected there are surviving (perhaps temporarily) in suboptimal conditions or that they are animals that disperse from a nearby source population (e.g., from west of the Kern River). However, the persistent and perhaps expanding nature of the occupancy patterns in the monitoring data from Kern Plateau make these hypotheses seem unlikely (R. Truex personal communication). Thus, the potential that our models could be slightly under-predicting fisher occurrence and population viability on the Kern Plateau should be considered when interpreting the following results concerning population size and viability, and should be addressed by future research, perhaps including development of a separate habitat model for that area or use of higher resolution or updated environmental data layers.

Resting Microhabitat – Our analysis of the Zielinski et al. (2006) resting microhabitat model using FIA data indicates that high-value resting sites are rare and scattered, even within areas predicted to be high-value at the coarser scale. Unfortunately, we were unable to derive a way to couple the coarse- and fine-scale models, and the sample of FIA plots is too small to make definitive statements about availability of resting sites in different segments of the study area. Moreover, although characteristics of high-value resting sites (and presumably denning sites) are well described in the literature, we do not know how much resting microhabitat must be present, or how it should be distributed, relative to fisher home ranges or foraging habitat. For the simulations of potential future changes in fisher habitat value, we therefore tracked changes in individual variables that should correlate with abundance of potential resting structures, such as biomass of older cohorts of black oaks and firs (see Section 4).

2.3.2 Baseline Population Assessment

Our population modeling results provide insights concerning fisher population viability and potential expansion of the population in the future that are relevant to forest management. Here we discuss some uncertainties inherent to the results. We also compare the population estimates derived using the population dynamics simulations to estimates derived independently using different techniques.

Coupling of Models and Amplifying Uncertainties – Concerns are often raised that coupling multiple models—as we do here in using habitat values derived from LAND8 as inputs to PATCH–can compound errors and thereby amplify uncertainties in the predictions. This could be true, for example, if uncertainties resulting from assumptions in each model interact additively or multiplicatively. We do not see this as a huge concern in general for these models, although it may be contributing to inaccuracies in depicting occupancy and source-sink dynamics for certain locations, such as the Kern Plateau. Because LAND8 appears to undervalue habitat there relative to the monitoring data, and because we scaled demographic parameters with modeled habitat values, this likely compounds the LAND8 inaccuracies such that PATCH may be underestimating the potential contribution of Kern Plateau to the fisher population. A cursory visual inspection of maps suggests that correcting for this apparent under-representation of habitat on the Kern Plateau, as well as inaccuracies for SEKI due to missing data, might increase total population estimates on the order of 5 to 15%. On the other hand, high value habitat predicted in the Piute Mountains and the Breckenridge Mountain areas south of the Kern River that are outside the extent of fisher occurrence described by Grinnell et al (1937) and likely not occupied may slightly overestimate total population size by a small amount (perhaps up to 5%)

However, in general, the spatial patterns we are seeing with PATCH seem biologically reasonable and fit well with expectations of fisher experts (e.g., K. Aubry, R. Barrett, R. Truex, and W. Zielinski personal communications). Nevertheless, we reiterate that these results should be used primarily for their heuristic value and not for predicting absolute or site-specific conditions. Their utility is in looking at broader, landscape-level patterns, and especially for *relative* comparisons of how the fisher population may respond to different future management and fire scenarios.

Territory Size – Because territory size is inversely related to population density, it has strong effects on population estimates in PATCH and must be chosen carefully. Territory hexagons are contiguous, so the selected size should account for unused interstitial areas between territories, and hence may need to be larger than mean territory sizes measured by field studies for species that don’t use all portions of the landscape. Conversely, for species with extensive territory overlap, model hexagons may need to be somewhat smaller than mean territory sizes to accurately portray density. For fisher, we assume that these opposing possibilities somewhat cancel out, but that based on existing fisher space-use data in the study area (e.g., Zielinski et al. 2004b, Mazzoni 2002) and on the great heterogeneity of the study area, territory interstices are probably more important than overlap in affecting fisher densities.

For these reasons and other reasons, we have more confidence in population estimates using the larger territory sizes we tested (i.e., 860-1,200 ha). There have been only two field studies of fisher home range or territory size in the region, one in an area that appears to be in very high quality habitat on the Sequoia National Forest and resulted in the smallest estimate of female territory size of any study in North America (527 ha; Zielinski et al. 2004b) and one in an area predicted by our models to have more modest and variable habitat quality that yielded female territory sizes closer to those measured in other regions (1,192 ha; Mazzoni 2002). Given that the territory hexagons in PATCH should approximate the size of female territories averaged across the range of occupied habitat qualities in the study area, we feel that the territories recorded by Zielinski et al. (2004b) are too small to be representative, and that the larger territories recorded by Mazzoni (2002) may better represent the average for the region. This opinion was also shared by fisher experts (W. Zielinski and R. Barrett, personal communications).

Dispersal and Potential for Range Expansion – Although concerns have been raised about sensitivity of spatially explicit models to assumptions about maximum dispersal distances and mortality during dispersal (e.g., Ruckelshaus et al. 1997), others have addressed these concerns and found them largely unwarranted (Mooij and DeAngelis 1999, 2003; Carroll et al. 2004, Rustigian et al. 2003). Our sensitivity analysis on effects of maximum dispersal distance, from 25 km to 100 km, found no significant effect on population estimates. Consequently, we used the intermediate value of 50 km for most simulations. Lewis and Hayes (2004) also tested sensitivity over a range of maximum dispersal distances (25 to 75 km) and also settled on 50 km.

Of greater concern than sensitivity to dispersal distance is that the dispersal algorithms in PATCH may not adequately account for potential impediments to dispersal or increased mortality during dispersal. The model assumes that mortality is no higher during dispersal than during other periods, which may bias our assessment of the potential for a northward range expansion to be overly optimistic. Available dispersal data for fishers suggest that they are weak dispersers relative to other forest carnivores, may not cross broad expanses lacking dense overhead cover, and that long distance dispersal is rare (Arthur et al. 1993 Aubry and Raley 2002, Powell and Zielinski 1994). Carroll et al. (2001) hypothesized that this may limit rapid recolonization of areas from which fishers have been extirpated. Jordan (2007) and fisher experts R. Barrett, R. Truex, and W. Zielinski (personal communications) believe that the major river canyons of the southern Sierra Nevada represent filters to movement between areas of suitable forest habitat due to extremely steep terrain, large areas of non-forested or sparsely forested vegetation, large rivers, and heavily traveled highways. They suggest that the southern Sierra Nevada fisher population is effectively segmented into a series of subpopulations by these major river canyons. This hypothesis is supported by the phylogeographic analysis of Wisely et al. (2004) who found significant local genetic substructure within the southern Sierra population on the basis of microsatellite variability. R. Barrett (personal communication) also believes that mortality during dispersal may be very high, especially given traffic volumes and potential for roadkill for fishers crossing highways, such as Highway 140 along the Merced River Valley. Roadkill of fishers is not uncommon throughout the species' range (W. Zielinski, personal communication). Thus, although our PATCH results suggest that there is strong potential for northward expansion to areas north of the Merced River, this hypothesis may be overly optimistic. R. Barrett believes that expansion across the Merced River may take longer than predicted due to these unaccounted for effects. Continued monitoring of the fisher population should include special attention to its northern frontier, roadkill along Highway 140 and other roads, and dispersal movements of fishers. Moreover, we recommend conducting a habitat connectivity assessment in this potential expansion area, using an appropriate cost-weighted movement permeability model developed for the fisher, such as least cost corridor analysis (Singleton et al. 2002) or circuit theory (McRae and Beier 2007).

Demographic Parameters – Lamberson et al. (2000) believed that their high demographic parameter values (which we used here as maximum values) were "extremely optimistic and likely unrealistic." However, in their deterministic model, which ignored variation in habitat quality, using what Lamberson et al. considered the more biologically realistic medium and low values always resulted in rapid population declines and extinction within a few decades. This is counter to observations that (1) the southern Sierra Nevada fisher population has actually persisted, despite its small size and isolation, for many decades, and (2) with no apparent declines in occupancy, and some evidence of expansion, since systematic monitoring was initiated in the mid 1990s (R. Truex and W. Zielinski, personal communications).

The Lamberson et al. (2000) model, in not considering how variation in habitat quality affects life history rates, necessarily assumed that the selected parameter values represented averages across all occupied habitat. In contrast, we apply the high parameter values only to the highest-value territories (habitat score $\geq$ 7.5). The discounting of survival and fecundity rates with decreasing territory value ameliorates concerns that these parameters may be unrealistically high by establishing a source-sink balance. Figure 2.11 illustrates this by showing the relationship between territory value scores and the observed net value across all occupied territories for the Years 21-40 for our default model parameters (territory size = 860 ha, initial population = 100, max dispersal distance = 50 km). Net value is the observed, hexagon-specific average of births minus deaths across all time steps in which the hexagon was occupied. It is therefore a site-specific measure of site productivity[26]. Note that above the ceiling territory value of 7.5 (where the maximum demographic parameters from the Leslie matrix apply), net value varied from about 0.3 to greater than 1.0 (source habitat). Note that territories with scores below about 5.5 have net values ranging from 0 to about -0.75 (sink habitat), and territories with scores between 5.5 and 7.5 can represent either source or sink habitat. The scatter in net value at any given territory score is due to stochasticity as well as to effects of the territory's landscape context: Territories with lower than expected net value relative to their territory scores tend to be in isolated habitat patches, whereas territories with higher than expected net values tend to be clustered within larger, high-value habitat areas.

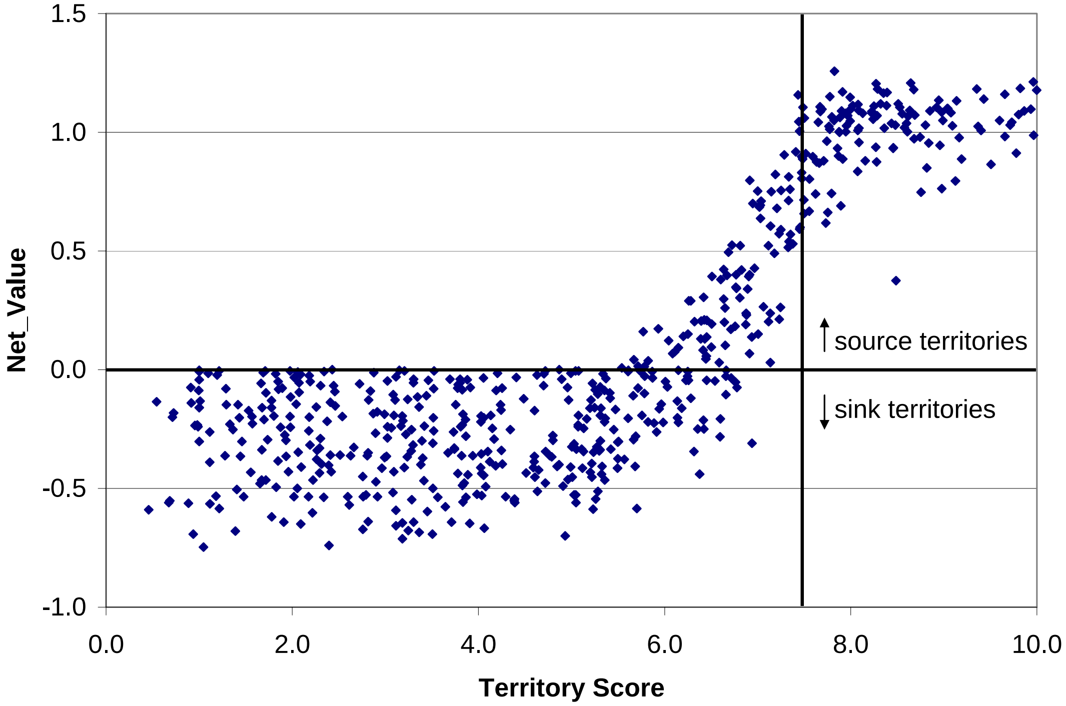

Figure 2.11. Scattergram showing relationship between territory hexagon score (assigned using Table 2.1) and observed net value over model Years 21-40, where net value = territory-specific

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[26] An average net value of 0 (births = deaths) corresponds to a population lambda of 1.0, or a stable population.

average of births minus deaths across all time steps in which a territory was occupied. (Territory size = 860 ha, initial population = 100, max dispersal distance = 50 km). Note that an average net value of 0 is equivalent to a population lambda of 1.0 (births = deaths).

Inaccuracies for Kern Plateau – None of our landscape-level habitat models, including LAND8, adequately fits the fisher detection-nondetection data on the Kern Plateau (east of the Kern River on Sequoia National Forest). Fishers have been consistently detected there in locations predicted to have low probability of occupancy, and the most recent (2007) survey results (not included in our model dataset) detected them in even more locations than previous surveys (R. Truex, personal communication). Because model LAND8 was used to establish territory scores in PATCH, the population simulations appear invariably to under-predict territory occupancy and usage on the Kern Plateau. This could be due to inaccurate environmental data used in the model, a poor model, or differences in how fishers use habitat on Kern Plateau relative to the rest of the study area. Because essentially all GAM models we tested under-represented fisher occurrence there, we suspect that unique environmental conditions on the Kern Plateau result in somewhat different selection of habitat by fishers there, so that our regional models extrapolate poorly over that portion of the study area. This may require modeling fisher occurrence separately for the Kern Plateau, an option that should be explored in more detail. A cursory visual inspection of maps suggests that adjusting to better account for under-represented fishers and fisher habitat on the Kern Plateau and SEKI could add roughly 5 to 15% to our assessments of the amount of habitat and fishers based on the current models.

Population Size Estimates – In general, despite uncertainties concerning the Kern Plateau, the overall spatial patterns emerging from these PATCH analyses seem realistic and informative, although precise quantification of population size, distribution, and change over time (e.g., potential range expansions) would require more careful parameterization and testing with additional data. Our sensitivity analysis suggests that the equilibrium population estimate within currently occupied areas is likely to vary modestly with realistic changes in fecundity and survival rates, although the potential for and rate of population expansions into currently unoccupied areas may vary more dramatically with such changes (not yet tested).

Based on these model results, we tentatively estimate the current carrying capacity of fisher habitat south of the Merced River (the "currently occupied habitat") at about 73 to 147 adult females. This range is based on the minimum number of females observed at equilibrium using a 1,200-ha territory size, up to the maximum number of females observed using the 860-ha territory size (see Table 2.7). As discussed above, we do not believe that estimates based on territory sizes <860 ha are realistic given the tremendous environmental heterogeneity of the study area and the wide range of habitat qualities that should be accounted for in setting average territory size.

Assuming a 1:1 adult sex ratio, the estimate of total adult population size would be 146 to 294, but this probably overestimates the number of males. Jordan's (2007) intensive study on the Kings River Project study area yielded slightly female-biased sex ratios (an average F:M ratio of about 6:4, with about 55-70% of the population consisting of females in 4 out of 5 years). Regarding the reproductive population, R. Barrett (personal communication) estimates that there are more likely 2-3 territorial females per territorial male, in part because male territories average roughly three times larger than females' (Powell and Zielinski 1994)[27]. Moreover, in polygynous mammals, males generally experience greater mortality than females, resulting in increasingly female-biased sex ratios with age (Clutton-Brock and Isvaran 2007). Consequently, although the quantitative data for fishers seem insufficient to confidently assume a ratio different from 1:1, fisher biology suggests that the 1:1 assumption probably results in overly optimistic (high) estimates of total adult population size (R. Barrett, W. Zielinski, R. Truex, personal communications). We therefore conclude that the equilibrium population size for fishers in the southern Sierra Nevada south of the Merced River is *no more* than about 294 adults and, all things considered, is more likely in the range of 125 to 250 individuals.

Accounting for juveniles and subadults is even more difficult–but also less important to understanding population viability. The timing of the census step in PATCH's calculations is such that it grossly overestimates the number of juveniles in the population because it counts all juveniles prior to mortality. Because juveniles do not contribute to reproduction, are highly variable over time due to phenology and mortality, and have little overall influence on a population's viability, accurately estimating their numbers is not critical. The number of subadults (Stages 1 and 2) in the population at year 40 averaged 90.55 (± 6.95 SD) individuals using our default model assumptions. Adding these ~84 to 98 subadults to our adult population estimate yields a total population estimate of about 230 to 392 individuals in the southern Sierra Nevada (ignoring juveniles).

We suspect that several assumptions may tend to inflate this estimate (e.g., assumed 1:1 sex ratio, average territory sizes of 860-1,200 ha, optimistic age-specific fecundity and survival rates, no accounting for increased mortality during dispersal, and no accounting for variation in other factors that may affect fisher populations, such as predators, prey, competitors, or human land uses). In addition, model LAND8 projects some high habitat value in the Piute Mountains and Breckenridge Mountain at the southern extreme of the study area, outside of the fisher's geographic range as delineated by Grinnell et al. (1937) and that probably don't support fisher. On the other hand, the under-representation of habitat value for Kern Plateau and Sequoia-Kings Canyon NP may tend to slightly underestimate the total population. Nevertheless, these estimates are consistent with various expert-opinion estimates, including those of Lamberson et al. (2000; 50-200 females) and R. Barrett (personal communication; no more than 300 or at most 350 total individuals).

These dynamic estimates also overlap with preliminary estimates by R. Truex (personal communication) of the total population size made by extrapolating the regional monitoring program's naive annual occupancy estimates to the entire population. Estimates vary depending on assumptions regarding occupancy rates within the population and the total number of fishers detected at each sample unit. Results from the monitoring data suggest occupancy rates vary somewhat among three segments of the southern Sierra population: (1)Yosemite NP and Sierra National Forest, (2) west-side Sequoia National Forest and Sequoia-Kings Canyon NP, and (3) the Kern Plateau (primarily on Sequoia National Forest, extending slightly into southernmost Inyo National Forest). Based on this assumption and by varying the number of fishers detected

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[27] Male vs. female territory size may be even more disproportionate in the southern Sierra Nevada, as Zielinski et al. (2004) found mean male territories to be 5.7 times larger than females' in their study area on the Sequoia National Forest.

at each sample unit in biologically defensible ways, R. Truex (personal communication) tentatively estimates the current size of the southern Sierra fisher population at 160 to 250 individuals (presumably including subadults and adults of both sexes but few if any detections of juveniles).

Yet another approach to estimating population size is to extrapolate measured fisher population densities from field studies over the area of occupiable fisher habitat in the study area. Unfortunately, there is only one reliable field estimate in the region, from the work of Jordan (2007) in the Kings River Project study area. Based on 3 years of mark-resight data using camera traps, Jordan estimated a total density of 10-13 fishers (presumably adults plus subadults), including 2-3 reproductive (adult) females, per 100 km^2. Extrapolating these density estimates over the area of predicted suitable and occupied habitat[28] we get total population estimates of 276 to 359 fishers (ignoring juveniles), of which 55 to 83 are adult females. Despite all the uncertainties in these static extrapolations, they also compare well with our estimates based on dynamic modeling.

Weighing the assumptions and uncertainties inherent to each of these three estimates of fisher population size along with the overlap in the ranges they provide, we conclude that total fisher population size in the southern Sierra Nevada is currently *160 and 360 total individuals, of which only about 55 to 120 are adult females — the most important class for sustaining a population*. Isolated wildlife populations this small are at risk of extirpation from a variety of causes, including stochastic events.

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[28] We applied the 0.5 cutpoint in probability of occupancy for model LAND8 south of the Merced River, yielding 276,077 total ha of predicted suitable, occupied habitat in the study area.

3.0 Simulation of Changing Fire and Landscape Conditions

Fire has played a crucial role in shaping the structure and function of forests throughout the Western United States, including the Sierra Nevada. Prior to widespread Euro-American settlement, fire was the predominant disturbance in the Sierra Nevada; and fire regimes were characterized by frequent, low-intensity surface fires that created a fine-scaled mosaic of fuel across the landscape (Kilgore and Taylor 1979, Collins and Stephens 2007, Beaty and Taylor 2008). Although the timing and extent of fires naturally varied over the last several millennia (Gavin et al. 2007, Beaty and Taylor 2008), the 20th century policy of fire suppression resulted in decades of exclusion of fire from these naturally fire-prone forests, to the point where the amount of fire on the landscape was far below historical estimates (Parsons and Landres 1998, Keeley and Stephenson 2000).

The unforeseen consequence of aggressive fire suppression was that the structure and composition of forests in the Sierra Nevada changed. The increased density and abundance of fuel that resulted from removing fire from the landscape now increases the likelihood that fires will become uncharacteristically large and intense, to the point that they are considered outside the historic range of variability (McKelvey et al. 1996, Stephens 1998, Keeley and Stephenson 2000, Sugihara et al. 2006, Westerling et al. 2006). Compounding the issue of fuel accumulation through fire suppression is that current and projected changes in climate favor increased incidence of fire (McKenzie et al. 2004, Flannigan et al. 2005). In addition, increased spring and summer temperatures, and earlier spring snowmelts, have resulted in more frequent, larger, longer-duration fires since the 1980s (Westerling et al. 2006).

While there is growing consensus that fire suppression has negatively impacted natural communities (Backer et al. 2004) and has substantially worsened the fire problem (Agee and Skinner 2005), the process of restoring natural fire regimes to many western forests has become a major challenge. With so much fuel accumulation, and the increased homogenization of fuels across the landscape (Beaty and Taylor 2008), the conditions in many parts of the forest are now so altered that it would be impossible to immediately restore fire regimes to those characteristic of pre-suppression conditions. In other words, many fires, especially under severe weather conditions, are likely to have uncharacteristically large patches of high-severity fire, unlike the low-severity fires characteristic of historic conditions Therefore, large patches of severe fires could result in dramatic reduction in large trees and aboveground biomass that could also have cascading ecological impacts (DellaSalla et al. 2004), particularly to fishers or other wildlife that require large old tress for their habitat. While current trends suggest that fire conditions are likely to become more extreme, if these large patches of severe fire did not occur, the increased biomass and fuel accumulation from fire suppression may actually favor fisher and increase their habitat quality.

Considering the trends of increasing size and severity of fire, it has become clear that management action will likely be necessary to reduce the risk of extreme fire behavior. Forest management designed to reduce the risk of large patches of high-severity fire typically involves a combination of fuels treatments and prescribed fire to reduce hazardous fuel loads. Those prescriptions that combine mechanical fuel manipulations (i.e., reducing vertical and horizontal continuity of canopy fuels) with prescribed fire are believed to be the most effective at reducing fire size and intensity (Agee et al. 2000, Schmidt et al. 2008). Nevertheless, some of the important variables to consider in developing a fuels treatment management plan include the rate of treatment (the area on the ground that is treated within a given time period); the intensity of the treatment (the amount and type of fuel removed), and the location of treatment.

Although the conceptual basis of fuels treatments is well-founded, there is uncertainty regarding their efficacy in modifying fire behavior. Understanding the potential efficacy of fuels treatments is complicated because natural fire regimes vary widely; and the effectiveness, or appropriateness, of a treatment may be influenced by factors such as treatment type, natural fire regime, weather conditions, and local topographical features. For example, high- elevation subalpine forests are naturally characterized by infrequent, high-severity fires; thus, mechanical fuel reduction to reduce fire threats may be considered inappropriate in these forests because dense trees do not represent abnormal fuel accumulation (Schoennagel et al. 2004).

Another factor complicating the evaluation of fuel treatment efficacy is understanding whether the treatments will have the same effect under more severe weather conditions as they do in moderate conditions. Fuel treatments were largely ineffective under severe weather conditions in the 2002 Hayman fire in Colorado, but this may have been due to the small size of the treatment area. On the other hand, fuel treatments effectively slowed and reduced the severity of the 2002 Rodeo-Chediski fire in Arizona under extreme weather conditions (Schoennagel et al. 2004).

To develop an understanding of how future landscape dynamics may affect fisher populations and habitat, particularly in light of concerns over increasing large, severe fires, it was important to first evaluate whether management actions designed to reduce the size and severity of fires would be effective. Furthermore, it was important to understand which types of treatment strategies would be the most efficient and effective. Therefore, the primary objectives for this section of the project were to answer these questions:

1. What are the long-term effects of fuels treatments on the fire regime (including fire rotation period, fire size, and fire severity) across large landscape in the Sierra Nevada?
2. What are the relative effects of treatment rate, intensity, and location on the fire regime?
3. Does the efficacy of treatment vary under more extreme weather conditions?

Because fire occurs sporadically over large areas, it is difficult to design experiments in the field to test the effectiveness of fuels treatments. A few studies have taken advantage of natural experiments, so there is some empirical evidence of how fires respond to individual fuel treatments. Nevertheless, field approaches, and even small-scale simulation modeling results, cannot easily account for multiple interacting factors that occur over long periods and broad spatial scales. Therefore, we used the LANDIS-II model to address these broad-scale questions.

3.1 Methods

3.1.1 Introduction to LANDIS-II

The choice of models often involves some level of trade-off or balance between the level of detail simulated and the scale of the simulation (Shugart 1998, Keane et al. 2004). More detailed models are typically better suited for finer space and time scales. To study large areas over long times, however, modelers often must generalize some fine-scale details that do not substantially affect outcomes at the coarser scale of interest, to meet the increased data demands and computational costs. For example, at the temporal scale of an individual fire, estimates of daily wind speeds are important to capture potential immediate threats to personnel and property. However, if multiple decades and 10s or 100s of fires are being simulated, daily changes in wind speed do not provide additional useful information relative to the increased effort required for their parameterization and computation. The differences between immediate fire modeling needs (tactical simulations) and longer term simulations designed to discern large-scale changes in system behavior (strategic simulations) distinguishes the various fire models available.

To simulate potential changes in forest and shrub vegetation in response to different fire and management combinations in the Sierra Nevada, we required a spatially explicit model that could simulate multiple and interactive disturbances (e.g., wildfires and fuel treatments) and successional processes (e.g., tree and shrub dispersal and growth), over large landscapes (millions of ha) at a biologically meaningful resolution. To meet these objectives, we chose LANDIS-II, which is a spatially dynamic and stochastic forest succession and disturbance simulation model (Scheller et al. 2007, Mladenoff et al. 1996, Mladenoff 2004) that has been extensively and successfully deployed in many forested and shrubland ecosystems throughout the world (e.g., Franklin et al. 2001, Gustafson et al. 2000, Schumacher et al. 2004, Ward et al. 2005, Xu et al. 2007, Syphard et al. 2006, Scheller et al. 2007). Whereas other models (e.g., FARSITE and FLAMMAP) are capable of simulating finer-scale fire behavior dynamics, LANDIS-II was able to integrate the interaction between vegetation succession and fire processes at broad spatial extents and long temporal scales. Nevertheless, LANDIS-II provided the necessary detail and complexity to simulate dynamic distributions of individual tree species, biomass, and habitat types (i.e. for the fisher). The model also had the ability to run many replicates to evaluate means and trends.

LANDIS-II simulates individual tree and shrub species. Each species is characterized by unique life history characteristics (Roberts 1996) that include longevity, age of maturity, fire tolerance, shade tolerance, seed dispersal distances, the ability to re-sprout, and reproduction following fire. These life history characteristics, in combination with the other spatial and non-spatial model inputs and parameters, both drive and respond to disturbance and successional processes. LANDIS-II does not represent individual trees; rather, trees are binned into species and age cohorts. Multiple species and multiple age cohorts may be present within a single cell, depending upon the initial conditions and establishment processes. Cohorts react individually to disturbances, depending on their age and tolerance to each disturbance type. Processes in LANDIS are highly stochastic, and probabilistic rules dictate the location and occurrence of fire and wind; so species distributions are a function of these multiple interacting processes and parameters. Therefore, LANDIS-II cannot be used to predict individual events at specific locations. Rather, the strength of LANDIS-II is in understanding how individual events and processes interact cumulatively and spatially to alter the entire landscape.

3.1.2 Model Initialization and Extensions

Within LANDIS-II, a landscape is divided into a grid of square cells. For the southern Sierra Nevada, we used a 100-m cell resolution (each cell represents 1 ha). Cells are aggregated into land types (or "ecoregions") that represent relatively homogeneous climatic and soil conditions. Land types need not be contiguously arrayed across the landscape. To stratify the study area into land types that captured differences in species' ability to establish and grow, we used an unsupervised clustering approach based on methods that were successful in another forested landscape in California (Franklin 2003). We standardized six GIS environmental data layers to range from 1 – 256: elevation, slope, solar radiation or insolation index (Gustafson et al. 2003,), minimum January temperature, maximum July temperature, and precipitation. The three climatic variables were estimated by PRISM (Daly et al. 2002). These layers were subsequently used to derive 10 relatively homogeneous layers using the ISOCLINE algorithm in ArcInfo. A supervised classification reduced the number of isoclusters to six. A satellite classification indicating permanent shrub fields was overlaid on top of these six land types to created a seventh land type. The resulting seven successional land types contained between 133,000 and 564,000 ha (Figure 3.1).

LANDIS-II requires life history characteristics for all tree or shrub species present. We compiled life history characteristics for 23 individual tree species as well as two chaparral species (*Adenostoma fasciculatum* and *Cercocarpus montanus*) and a riparian functional group comprised of willows (*Salix* spp.), black cottonwood (*Populus trichocarpa*), and alders (*Alnus* spp.) (Table 3.1).

LANDIS-II requires that initial communities are defined for the entire landscape. The initial communities are used by the Biomass Succession extension (described below) to generate initial biomass values for each mapped species and age cohort. The extension 'spins up' the initial above-ground live biomass for each cohort by growing the community beginning with the oldest cohort and successively adding cohorts from oldest to youngest. At the end of the spin-up phase, each cohort has an estimated biomass and the total above-ground live biomass can be calculated.

We used a combination of CWHR (California Wildlife Habitat Relationship) (Mayer and Laudenslayer 1988) attributes from the California EVEG GIS data layer and Forest Inventory and Analysis (FIA) data (Hansen et al. 1992) to estimate the initial community composition of the landscape. Mapped sequoia stands were extracted from the EVEG database and superimposed over the CWHR-type map layer. Recent clearcuts were identified either through the CWHR size type (type = 0) or from data provided by Region 5 identifying recent harvests on private lands near Stanislaus NF. The final vegetation polygons derived were then converted to a GRID with a 100m x 100m spatial resolution. This resulted in some modest generalization of the original data.

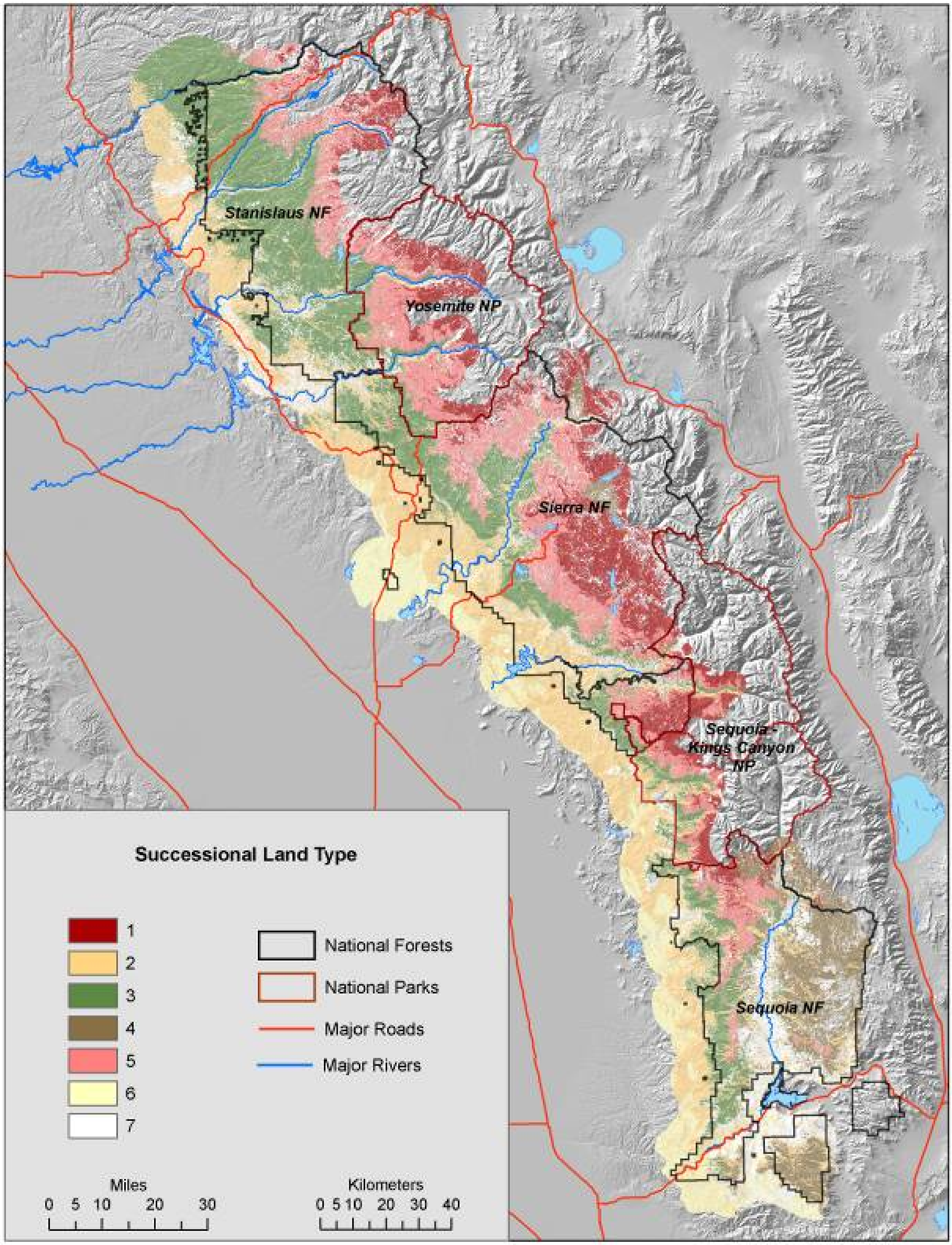


Figure 3.1. Successional land types used in the LANDIS-II modeling.

Table 3.1. Tree and shrub species life history parameters used in LANDIS-II vegetation dynamics model.

Common name	Species code	Longevity (yrs)	Sexual Maturity (yrs)	Shade tolerance[1]	Fire tolerance[2]	Seed dispersal effective distance (m)	Seed dispersal max. distance (m)	Vegetative reproductive probability	Sprout min. age (yrs)	Sprout max. (yrs)	Post-Fire Regeneration
Lodgepole pine	pinucont	150	7	1	1	20	60	0	0	0	none
Ponderosa pine	pinupond	350	20	2	4	35	150	0	0	0	none
Sugar pine	pinulamb	450	15	3	4	30	150	0	0	0	none
Jeffrey pine	pinujeff	450	18	2	4	50	350	0	0	0	none
Bull pine/Gray pine	pinusabi	200	18	1	3	30	1000	0	0	0	none
Western white pine	pinumont	400	12	1	4	30	400	0	0	0	none
Giant sequoia	sequgiga	3000	150	1	5	100	400	0	0	0	serotinous
White fir	abieconc	400	40	3	3	30	200	0	0	0	none
Limber pine	pinuflex	1000	20	2	1	30	5000	0	0	0	none
Red fir	abiemagn	400	40	3	3	30	200	0	0	0	none
Douglas-fir	pseumenz	300	15	2	3	30	1000	0	0	0	none
Mountain hemlock	tsugmert	600	20	5	1	30	250	0	0	0	none
Incense cedar	calodecu	500	40	4	3	30	3000	0	0	0	none
Sierra juniper	juniocci	1000	20	2	1	2	500	0	0	0	none
Quaking aspen	poputrem	175	15	1	2	30	7500	0.95	1	175	resprout
California black oak	querkelo	300	30	3	2	30	1000	0.8	1	300	resprout
Canyon live oak	querchry	250	20	3	1	30	1000	0.95	1	250	resprout
Blue oak	querdoug	250	20	3	1	30	1000	0.8	1	250	resprout
Interior live oak	querwisl	200	20	4	1	30	1000	0.8	1	200	resprout
Chamise	adenfasc	100	10	1	1	5	10	0.7	3	100	resprout
Mountain mahogany	cercmont	150	5	2	2	50	500	0.95	3	150	resprout
Annual grasses	anngrass	5	1	1	1	100	10000	0	0	0	none
Riparian areas[3]	riparian	150	5	1	1	50	3000	0.95	5	150	none

[1] Shade tolerance ranges from 1 – 5, with 1 requiring full sun to establish and 5 capable of establishment under very low sun.

[2] Fire tolerance ranges from 1 – 5, with 1 being least fire tolerant, 5 most fire tolerant

[3] Riparian areas represent deciduous species common along stream corridors, including willows (Salix spp.), black cottonwood (Populus trichocarpa), and alders (Alnus spp.).

FIA data represent a systematic survey of forested areas across the continental U.S. For each FIA plot within the study area (608 total) we derived estimates of age for each tree based on a linear regression of diameter against stand age for each species. Trees were subsequently binned into 5-year species and age cohorts. Each FIA plot was assigned a CWHR size, based on the 75th percentile diameter for all species. Because the majority of FIA plots belonged to CWHR size class 5, additional FIA communities were generated by de-ageing (subtracting a representative age from each cohort and then discarding cohorts aged zero or younger) each FIA plot to all younger CWHR size classes. The result was >6,000 communities derived from FIA plot data. Because very few FIA plots were located within old-growth sequoia groves, we created additional communities with an old (> 500 years) sequoia component. Each initial community generated was assigned a CWHR type based on characteristic species (Mayer and Laudenslayer 1988). Each community was also assigned to a successional land type dependent upon the location of the original FIA plot.

Finally, each one of more than 600,000 CWHR polygons (indicating areas with relatively homogeneous vegetation type) within the study area were randomly assigned one of the 6,000+ FIA communities, stratified by land type, CWHR size, and CWHR type. If there were less than 10 FIA communities with matching land type, size, and type, the pool of FIA communities with matching size and type was used for the random assignment; if there were less than 10 FIA communities with matching size and type, the pool of FIA communities with matching type was used.

Recent clearcuts were assigned a montane chaparral community with an age corresponding to the years since 2007. Locations with recent (<20 years) clearcuts of unknown age were assigned a montane chaparral community of age 5. However, due to seed dispersal, most recent clearcuts would transition to sapling and pole stages by the conclusion of our simulations.

3.1.3 Parameterizing and Calibrating Succession

The LANDIS-II Biomass Succession extension (version 2.0) (Scheller and Mladenoff 2004) requires estimates of the probability of establishment (P_{EST}) and aboveground net primary productivity (ANPP) by species and land type. These parameters were estimated through consultation with US Forest Service silviculturists and calibrated using Forest Vegetation Simulator (FVS) results for samples across the study area. In cooperation with the USFS silviculturist, Joe Sherlock, we iteratively calibrated and validated successional trajectories through comparison to the Forest Vegetation Simulator (FVS).

To begin the calibration process, the USFS generated maximum ANPP and AGB data directly from FVS, which allowed us to explicitly use those values and to compensate for the relative paucity and low quality of data available for the area. FVS was run for > 100 plots, and individual species ANPP and total AGB were compiled separately. Within these compiled data, the 95th percentile was used to provide our estimates of maximum ANPP and AGB. Maximum ANPP was generated (or estimated where necessary) for each of our six ecoregions (Figure 3.1); maximum AGB was generated only for the mid-elevation ecoregion (#3).

Next, we ran LANDIS-II for 24 FIA plots (a single cell simulation starting with initial communities derived as noted above). These results were compared to FVS simulations for identical FIA plots. Each simulation was run for 50 years and comparisons were made between total above ground biomass (AGB, Mg ha^{-1}). In general, this initial validation indicated that (a) our estimates of annual net primary productivity (ANPP) were originally too low, and (b) we needed to account for the much higher ratio of AGB to ANPP, as compared to eastern forests where the Biomass Succession extension was developed.

In general, the comparisons with FVS estimates indicate that LANDIS-II results follow the behavior of FVS to a relatively high degree (Figure 3.2). This fidelity to the FVS simulations occurs despite the large size of our ecoregions and the lack of discrimination for maximum AGB inputs among ecoregions. Within each ecoregion, under or over predictions could result from FIA plots that lie either at the maximum or minimum elevation within the ecoregion. In addition, when selecting FIA plots for validation, we did not eliminate plots that may have undergone recent thinning which would inflate our estimates relative to FVS as LANDIS-II does not consider stand density or any recent partial disturbances when spinning up to initial AGB.

3.1.4 *Fire and Fuels*

To simulate fire in the Sierra Nevada, we used the Dynamic Fire and Fuels System for LANDIS-II, which includes two extensions: the Dynamic Biomass Fuel Classification Extension and the Dynamic Fire Extension (Sturtevant et al. 2007). Both of these extensions interact together as well as with the LANDIS-II Base Harvest extension (which simulates fuels treatments).

The Dynamic Fire Extension has been under development by the USFS Northern Research Station for the past two years, and is currently being used by in Labrador (Northern Research Station), Siberia (Northern Research Station), and New Jersey (USFS Northern Global Change Program) to simulate fire effects and fire and fuels interactions across a variety of conifer-dominated systems. This extension is used for simulating the general characteristics of the fire regime, including fire frequency, fire size or duration, and fire severity. It also incorporates the effects of weather and topography on the fire regime. The fire regime can be parameterized to vary spatially across the landscape according to user-defined fire regions that are provided as spatial input to the model.

A new fuels classification extension, Dynamic Biomass Fuels, was written specifically for this project. The Dynamic Biomass Fuels extension was used to assign and characterize specific fuel types across the southern Sierra Nevada. The definition of multiple fuel types enables the user to qualitatively describe overall characteristics at an individual site (or cell) – such as stand structure, composition, surface fuels, ladder fuels, forest floor characteristics, and seasonal stand characteristics. Together, these fuel conditions influence characteristic fire behavior (e.g., rate of spread and severity) under defined burning conditions (Merrill and Alexander 1987). LANDIS-II fuel types capture variation in fuels characteristics across the landscape due to variations in species composition (weighted by biomass) and stand age. They do not capture fine-scale variation at the sub-cell (< 1 ha) scale, such as variation in duff layer depth, quantity of fine fuels, and the presence of grasses in the understory.

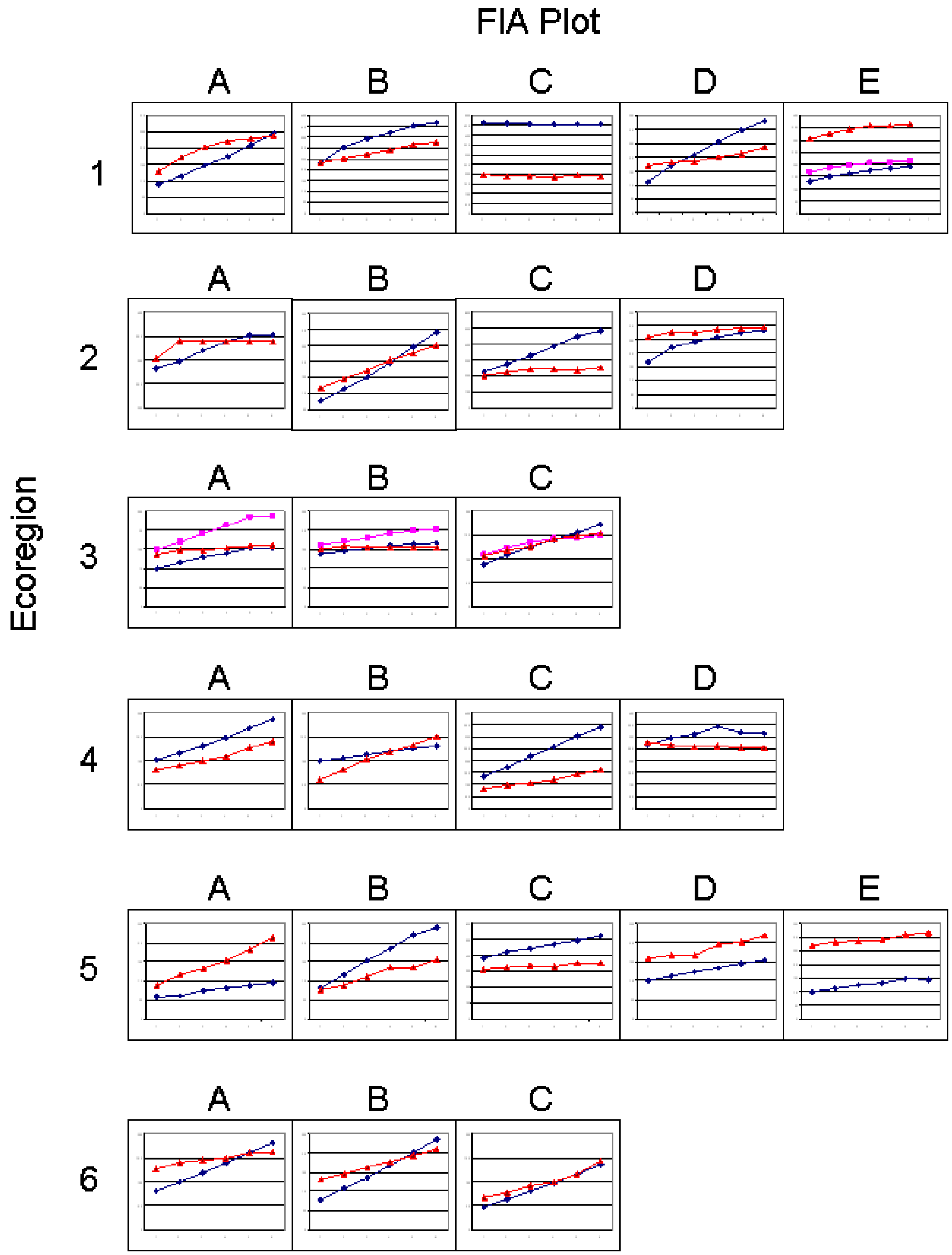


Figure 3.2. LANDIS-II simulations compared to FVS simulations for 24 FIA plots across 6 ecoregions. The x-axis indicates decade. The y-axis indicates above ground biomass in increments of 50 Mg ha^{-1} starting from 0. Red symbols and lines indicate LANDIS-II output; dark blue indicate FVS FFE (Fire and Fuels Extensions) output; dark pink indicate FVS with Jenkins biomass aboveground live biomass estimates.

3.1.5 Defining Fire Regimes

We developed parameters to simulate two different fire regimes in the southern Sierra Nevada. The "Baseline Fire Regime" is intended to represent fire patterns similar to those observed over about the last 20 years (1985-2006), which reflects the recent shift in trends toward increasing size and extent of fires (Westerling et al. 2006). We simulated this fire regime because we expected conditions in the near future to more closely represent recent fire history than to reflect historic fire regimes (frequent, low-intensity fires) or those that reflect complete fire exclusion.

Although no person or model can predict precisely how fire regimes in the southern Sierra Nevada may change in the future, current trends and projections suggest that wildfires are likely to become even larger and more intense, with a longer fire season. A longer fire season may also increase the potential for increased fire ignitions and fire frequency. Changes in the fire regime may be driven by climate warming, increased fuel accumulation due to succession and fire suppression, or a combination of both (Parsons 1991, McKelvey et al. 1996, Keeley and Stephenson 2000, Sugihara et al. 2006).

Considering this potential for fire to increase and become more intense, we also developed a "High Fire" regime to determine if management effectiveness or effects on fisher would vary under heightened fire conditions in the future. We did not try to attribute these changes to any particular cause (e.g., climate change, fuel accumulation due to regrowth and fire suppression, or a combination of both). We also did not try to project exactly how the fire regime will change. Instead, our goal was to determine the degree to which more severe fire weather conditions might influence the fire regime, and ultimately how a more extreme fire regime than that experienced in recent decades might influence fisher habitat and the effectiveness of fuels treatments. While the high fire regime may reflect one potential outcome of changing fire conditions, we emphasize that our definition of the high fire regime was not empirical, but hypothetical based on an assumption of more severe fire conditions in the future. Thus, the primary value of simulating landscape change under the Baseline Fire and High Fire assumptions is not intended to be a completely realistic portrayal of potential future fire regimes, but rather to estimate relative differences between fuels management combinations under a range of potential future fire conditions.

3.1.6 Delineating Fire Regions

We stratified the study area into three fire regions (Figure 3.3) that broadly reflect the effect of elevation on regional fire regimes (Agee 1993). The classes, created through the Jenks natural breaks classification method (that places class breaks based on clusters of data), included low (up to 1190 m), medium (~1190 – 2120 m), and high (above 2120 m) elevations. These zones roughly correspond to the foothill shrubland and woodland, lower-montane forest, and upper-montane forest ecological zones in the region (van Wagtendonk and Fites-Kaufman 2006). We next stratified these three regions into WUI and non-WUI subregions, to reflect the influences of humans and human development on fires. At both landscape and state-wide scales in California, significant spatial relationships have been observed between people and fires, with the majority of fires occurring in close proximity to housing development in the Wildland Urban Interface (WUI) and to roads (Syphard et al. 2007a, Syphard et al. in press). In the Sierra Nevada, lightning ignitions are also pervasive, and increase in density with elevation (from west to east) (van Wagtendonk and Fites-Kaufman 2006). These lightning ignitions were reflected in all of our fire regions. The WUI fire regions primarily captured the spatial influence of increased human ignitions in lower-elevation areas (where lightning is somewhat less frequent); and they reflected the smaller fire sizes typical of these human-dominated areas (due to more concentrated suppression efforts).

To delineate the WUI areas, we used data developed by a national WUI mapping project in the conterminous United States using housing density data from the U.S. Census and land cover data from the USGS National Land Cover Dataset (Radeloff et al. 2005). Whereas the WUI boundaries delineated by the California Fire Alliance focus on communities that are at risk from fire, we used the national WUI data because they have shown significant spatial relationships between humans and fire (and we were primarily trying to delineate areas where human influence on fire would be most prevalent, as opposed to where humans may be most at risk from fire). The maps were produced at the finest demographic spatial scale possible, using boundaries from the 2000 Census. Although other WUI maps and definitions exist; the maps we used were developed in accordance with the Federal Register definition (USDA and USDI 2001) and have been subject to statistical analyses demonstrating significant spatial relationships between fires and WUI (Syphard et al. 2007a).

Because many ignitions also occur along transportation corridors (Cardille et al. 2001, Yang et al. 2007, Syphard et al. in press), we incorporated roads (buffered by 100 m) into our WUI delineation. Thus, our final mapped fire regions contained six classes, with the three elevation bands each divided into WUI and non-WUI areas (Figure 3.3).

In LANDIS-II, the parameters that vary by fire region include fire size (or duration), daily weather conditions, fine fuel moisture levels, and number of fires. Fuel type and topographic data are assigned individually for each cell within the entire study area. For the three WUI fire regions, we assumed all parameters except number of fires would be the same as that of the non-WUI fire region in the same elevation band. Number of fires was higher in the WUI to account for higher ignition rates originating from those areas.

Fires can spread across ecoregion boundaries, and the model adjusts the fire size and spread rate to account for the ecoregional differences in parameters.

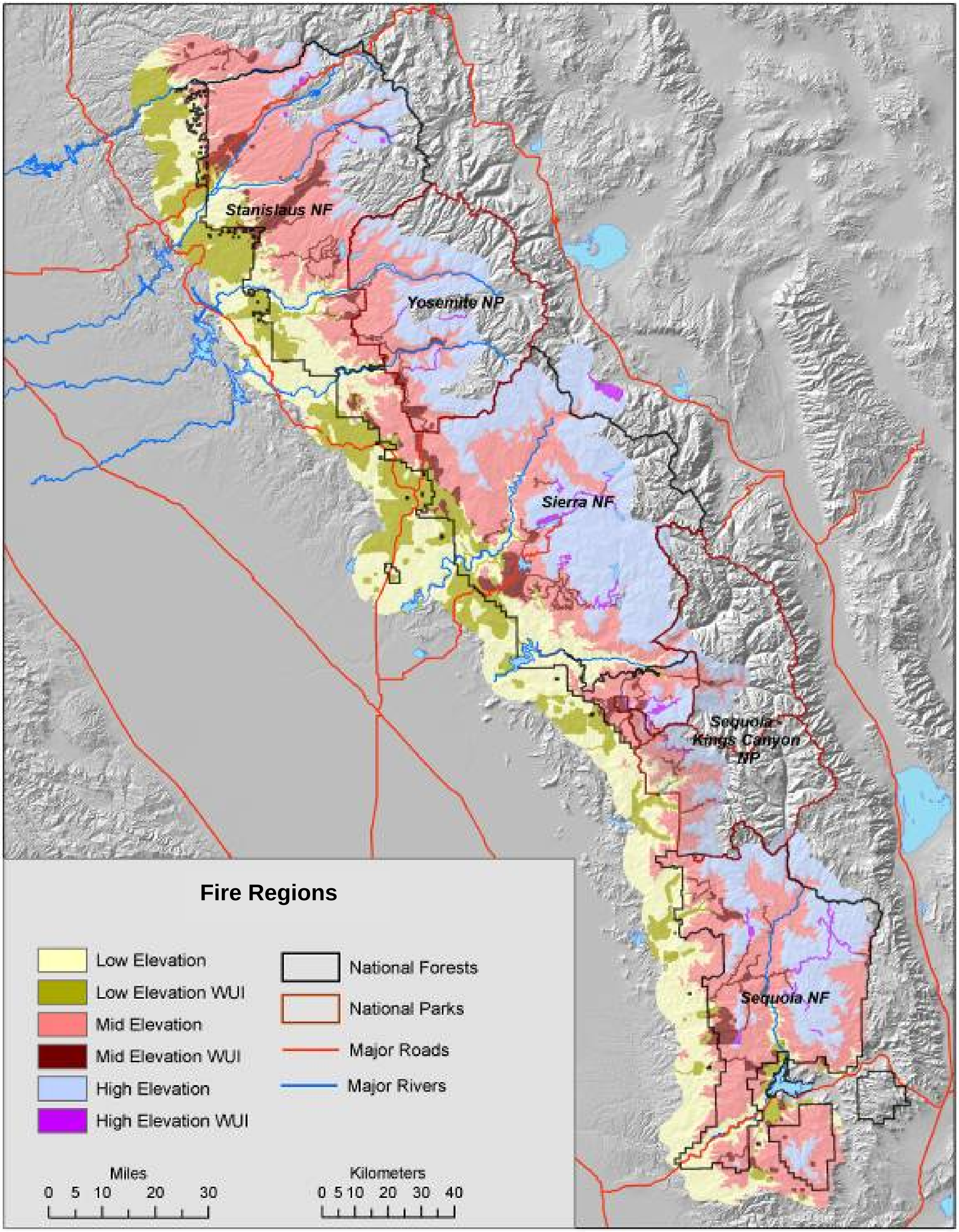


Figure 3.3. Fire regions developed for the modeling study area.

3.1.7 Calculating Fire Rotation Periods and Fire Sizes

To develop target fire regime parameters for our baseline fire simulations, we used historic fire perimeter data created as part of a cooperative effort between the California Department of Forestry and Fire (CDF), the United States Forest Service (USFS), the Bureau of Land Management (BLM), and the National Park Service (http://www.fs.fed.us/r5/rsl/projects/gis/data/calcovs/CA_R5_FireHistory06_1.html). The layer included fire perimeters from 1950 to 2006 with a minimum mapping unit of 10 acres (USFS land) or 300 acres (CDF jurisdiction). This minimum fire size restriction means that, while the majority of area burned occurs in larger fires, many small fires would be missed in this database.

We calculated mean historic fire rotation periods (FRPs) over the period 1985-2006 from the historic fire perimeter data by dividing the area of the fire regions by the mean area burned per year (WUI and non-WUI fire regions were grouped together). A FRP is defined as the time it takes to burn an area equivalent to the size of the area of analysis. Thus, it accounts not only for fire frequency but also for area burned. We used data from this ~20 year period to develop target fire regime parameters, because FRPs have generally been shorter in recent decades than they were 50 years ago (FRP = 90 – 140 years vs. FRP = 140 – 245 years, respectively; Table 3.2).

We did not calculate FRPs separately for the WUI fire regions because the WUI areas were mapped using Census boundaries from the year 2000. The number of houses in the WUI was historically much lower, and the extent of the WUI was smaller in the past. Therefore, we would not expect the influence of human-caused ignitions to be fully reflected in summary statistics of historic data.

The historic data show that size of fires in the study area is highly variable, and that variability has been increasing in recent years (Table 3.2). Fire size in LANDIS-II is simulated following a lognormal distribution function, where small fires are more likely to occur than large fires. We also calculated the mean, standard deviation, and maximum historic fire sizes from these data (Table 3.2).

Table 3.2. Historic fire regime in the Sierra Nevada, 1955-2006 and 1985-2006.

1985-2006

	Average size (ha)	Maximum size (ha)	Standard Deviation	Fire rotation
Low Elevation	401	19,460	4,788	90
Middle Elevation	513	32,060	2,822	140
High Elevation	577	60,490	4,789	120

1955-2006

	Average size (ha)	Maximum size (ha)	Standard Deviation	Fire rotation
Low Elevation	493	19,460	1,487	140
Middle Elevation	593	32,060	2,356	200
High Elevation	687	60,490	4,069	245

3.1.8 Dynamic Fire Extension Calibration

The Dynamic Fire extension provides two options for controlling fire size: size-based and duration-based. Duration time reflects the active burn periods during which there is significant fire growth. We used the duration-based approach (Pennanen and Kuuluvainen 2002) because it allows the landscape configuration of fuels to have a stronger influence on the fire regime. We also expected that under extreme weather conditions, fires using the same duration parameters would spread farther than they would under average conditions, which would more realistically simulate expected fire response if weather conditions became more severe over time.

To calibrate the fire and fuels extensions, we used a manual optimization process, which involved stepping through fire regime parameter ranges until the simulated FRPs and fire sizes approximated our targets. Because we calibrated based on duration, we stepped through ranges for three parameters: mean duration, mean variability of the duration, and number of ignitions attempted. These three parameters were uniquely varied for each fire region to achieve our calibration goals.

Our objective for calibrating the FRPs was to systematically adjust model parameters until the average FRP in each non-WUI fire region was within 15 years of the average observed over the last 20 years. We also refined parameters so that the baseline mean fire size for each non-WUI fire region was within 100 ha of the mean fire size as reflected in the historic data. In addition, we calibrated such that mean fire sizes in the WUI regions were substantially smaller than those in the non-WUI, to reflect the more effective fire suppression that occurs in the WUI.

Research shows that there is a highly significant spatial relationship between the distribution of people in the Wildland Urban Interface (WUI) and increased fire frequency in California (Syphard et al. 2007a, Syphard et al. 2007c). The actual degree of influence, however, may vary from region to region, depending on factors such as forest flammability, or the incidence of natural lightning ignitions. In California, we do know that a substantial proportion of housing growth in the WUI has occurred (and is continuing to occur) in the foothills of the Sierra Nevada (Hammer et al. 2007). Also, despite the fact that there are many more lightning ignitions in the Sierras than in the southern portion of the state, CDF fire records show that humans have historically caused more than 95% of the fires in the state (Syphard et al. 2007a).

Considering trends in human influence on California fire regimes, we parameterized the WUI fire regions differently than the non-WUI regions to reflect the spatial relationship between humans and fire. To simulate the combined effects of higher ignitions and better fire suppression capabilities in heavily populated areas, we calibrated the model to simulate smaller fire sizes in the WUI areas than the non-WUI regions, with FRPs up to 25% shorter (as in Syphard et al. 2007). The net effect was more frequent, smaller fires in the WUI, with most of the larger fires in the more remote forested areas.

High Fire Regime — To create the high fire regime, we selected a subset of the historic weather records for each fire region in all seasons that reflected the most severe weather conditions. Because the FWI represents a single integration of fire weather, we selected weather records with FWIs that were originally scaled as “Extreme” in the baseline calibrated regime. We rescaled these records into five new quintile classes and reran our simulations with only those weather records (Table 3.3). These simulations used the same parameter values as those in the baseline calibration, so the only difference was the weather records. Using only these records in the simulations allowed the model to naturally respond to weather conditions that were more "severe."

Table 3.3. Fire Weather Indices (FWI) broken into 5 classes for the baseline calibration and the high fire regime.

Percentile	Class	FWI Baseline Middle Elevation	FWI High Fire Middle Elevation
97 - 100	Extreme	35.14 – 37.17	36.03 – 37.17
90 - 96	Very High	34.58 – 35.13	35.53 – 35.99
75 - 89	High	33.27 – 34.57	35.21 – 35.52
50 - 74	Moderate	27.55 – 33.26	34.85 – 35.20
0 - 49	Low	10.01 – 27.54	12.34 – 34.84

Calculation of Fire Weather — Fire spread in the Dynamic Fire extension is a function of both weather and fuels. Weather data determine leaf status (leaf on, leaf off), wind speed velocity and direction, and percent curing of grass. Also, two of the required indices, the fine fuel moisture code (FFMC) and the Buildup Index (BUI)) (Van Wagner 1987) are derived from daily weather data (including daily temperature, wind speed, wind azimuth, relative humidity, and mean annual precipitation).

Our source of daily weather data was the California Climate Data Archive produced by the Western Regional Climate Center of Scripps Institution of Oceanography and the California Energy Commission (http://www.calclim.dri.edu/stationlist.html). For all of the weather stations located within the three elevation fire regions, we downloaded the full available history of daily weather data. Because some of the weather calculations (such as FFMC and BUI) require all days to be present within the fire season (because values depend on surrounding dates) we evaluated all the data to find the combination of stations that would provide the longest complete histories. For the low-elevation fire region, we used 1994, 1995, and 1997 from the Esperanza station and 2000-2006 from the Trimmer station, resulting in 10 years of data. For the middle-elevation fire region, we used 1991, 1992, 1994, and 1995 from Mariposa Grove; 1996, 1997, and 1998 from Shaver; and 1999-2006 from Johnsondale, resulting in 15 years of data. For the high-elevation fire region, we used 1995-2003 from Devils Post Pile and 2004-2006 from Sugar Loaf, for 14 years of data.

To simulate weather conditions, the fire season is separated into distinct seasons, which are defined by the user. The first day of spring, which is the beginning of the fire season, represents the first average snow-free day. The first day of summer represents the average leaf-out date for deciduous trees, and the first day of fall represents the average leaf-fall date for deciduous trees. The fire season ends on the first day of winter, which represents the average date of first snow accumulation. We defined our seasons, based on expert opinion of US Forest Service regional fire scientists, as follows: April 1 (spring), May 7 (summer), November 15 (fall), and December 15 (winter). The winter season was not used in the simulations, however, because no fires occur during this time.

During simulations, the model randomly selects daily weather records to individually regulate fire behavior for each simulated fire. To distribute the selection of weather records proportionally to their occurrence within each season, we calculated the proportion of all fires (across all fire regions) that occurred within each season for the full 56 years of fire perimeter database. Historically, 3% of fires occurred in spring; 89% occurred in summer; 5% occurred in fall; and 3% occurred in winter. Because the fire season only includes spring, summer, and fall, we allocated the proportion of fires within each season as 4% spring, 90% summer, and 6% fall. During simulations, the model selects weather records from tables with a frequency that is proportional to these percentages.

Fire Severity — In the Dynamic Fire extension, simulated fires reach varying levels of severity based on the fire's rate of spread and the critical surface fire intensity (the spread rate at which the crowns become involved in the fire). While the rate of spread is a function of weather, topography, and fuel conditions, surface fire intensity is a function of the fuel's crown base height (CBH: the height above ground that the live crown based begins) and the foliar moisture content (FMC: the moisture content of coniferous crown fuels) values of the season. Depending on variations in fuel types or weather conditions, simulated fires may be mixed-severity. A fuel type is an identifiable association of fuel elements of distinctive species, form, size, arrangement, and continuity that will exhibit characteristic fire behavior under defined burning conditions.

Foliar moisture content varies from 85% to 120%, depending on the season. Typically, the lowest FMC occurs in the early summer, resulting from a temporary increase in dry weight (Little 1970). Regional fire experts from the US Forest Service provided estimates of the conditions in the study area (Table 3.4). Due to the Mediterranean climate of California, summer drought conditions result in lower FMC in fall than any other season.

Table 3.4. Foliar Moisture Content parameters in the Sierra Nevada.

	Spring Season		Summer Season		Fall Season	
	Low FMC	High FMC	Low FMC	High FMC	Low FMC	High FMC
Low Elevation	120	120	85	100	85	90
Middle Elevation	120	120	85	100	85	90
High Elevation	120	120	85	100	85	90

We recalibrated fire severity to achieve a mean severity of ~3.5 (on a scale from 1 to 5) in the Baseline condition with a normal distribution. We specified a higher mean severity in the high fire regime (~4.5) due to the projections from regional fire experts that fires are likely to become more severe in the future under climate change or from fuel accumulation.

3.1.9 Defining and Parameterizing Fuel Types

Along with other parameters in the fire extension (e.g., wind speed, fuel moisture, or slope), fuel types determine fire behavior. We defined the different fuel types in our simulations based on characteristic species assemblages and age ranges that together exemplify relatively uniform fire behavior and rates of spread. These fuel types represent the *average conditions* across a broad landscape. The Dynamic Fire model does not require, nor does the Biomass Fuel extension provide, detailed estimates of actual fuel conditions. At fine scales (< 1 ha) there will be significant variation in fuel conditions (e.g., ladder fuels, duff layer depth, downed woody debris) due to gaps, wind throw, local mortality, etc., but these variations occur at a finer resolution than is possible or necessary to simulate at our regional scale of interest.

Working with US Forest Service, we derived seven basic fuel-type groups: Mixed Conifer, Red Fir, Pines and White Fir, Sequoia, Lodgepole and Hemlock, Chaparral and Scrub Oak (generically 'brush'), and Deciduous (predominately oaks). Within each group, fuel types were further broken into two or three age groups: young, mid-aged, and old. The resultant list of 20 fuel types (Table 3.5) is intended to capture significant variation of fuel characteristics due to species composition and successional status and not to represent more realistic fine-scale variation that may occur within stands.

Two additional fuel types were created to represent fuel conditions following a fuels treatment, depending on the intensity of the treatment (Table 3.5). These two fuel types represent the assumed efficacy of the simulated fuel treatments.

Assigning Fuel Types — During model simulations, the Dynamic Biomass Fuels extension assigns one fuel type to every 1-ha cell in the study area based on the species composition and age ranges present. Fuel type assignments are dynamic and can change with succession or harvest. For example, if an area burns, the fuel types following the burn should reflect the changed state of the forest. Likewise, as succession occurs and forests age, the fuel types will also change accordingly. Our fuel types are organized and assembled to follow a logical sequence of expected successional changes (Table 3.5).

Fuels classification in the Dynamic Biomass Fuels extension is determined by where a cohort falls in the fuel type age range (Table 3.5). The formula for classification is:

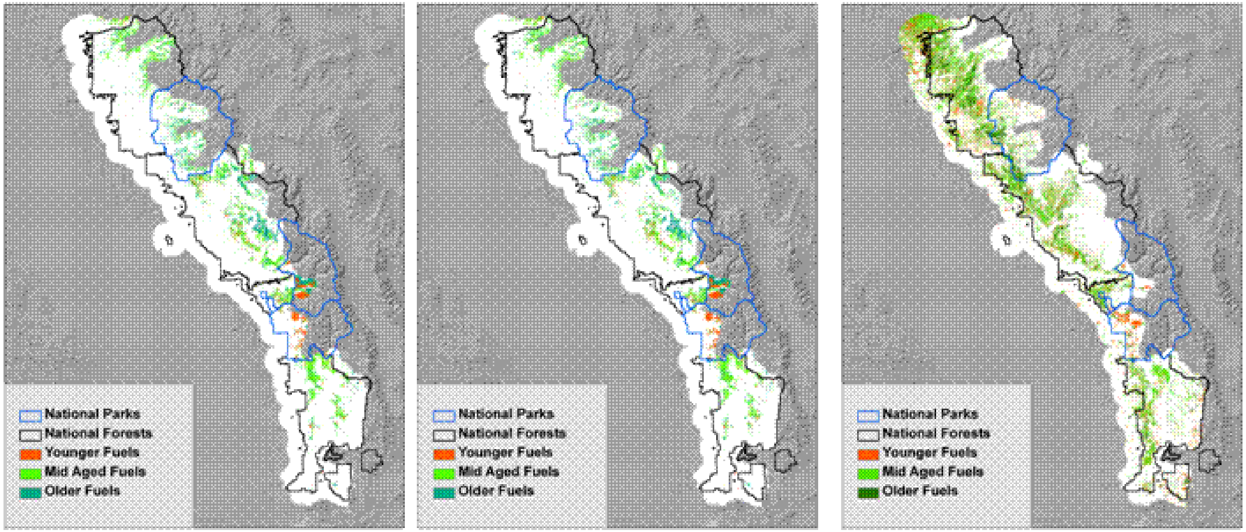
$$SpeciesValue = \sum_{0}^{n} Biomass \times SppCoefficient$$

where *Biomass* is the biomass (kg/ha) of all cohorts that falls within an age range (with a maximum and minimum) defined by the user; and *SppCoefficient* is a user specified weight (0 – 1.0) that can be assigned to each species (default = 0).

For each fuel type, these species values are summed (or subtracted if given a negative switch), depending upon which species are listed for the fuel type. The fuel type with the highest overall score is assigned to the cell. The resulting distribution of fuel types across the landscape is depicted in Figure 3.4.

Table 3.5. Fuel type parameters used in the Dynamic Fire extension for LANDIS-II. See Table 3.1 for species codes. The default grass fuel type is not shown.

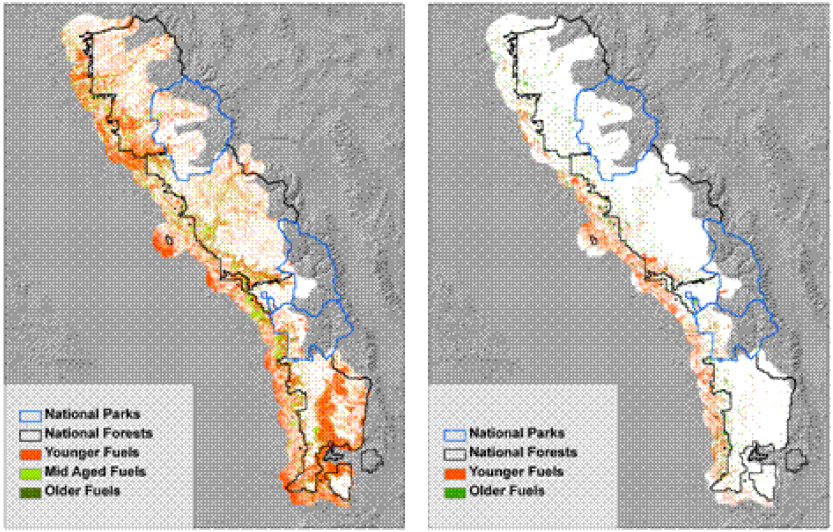
Description	Num	Ignition Probability	a	b	c	q	Mean BUI	Max BE	CBH (m)	Min Age (yrs)	Max Age (yrs)	Characteristic Species (See Table 3.1 for codes)
Young Mixed Conifer	FT1	0.01	110	0.0282	1.5	0.7	64	1.321	1	0	40	juniocci abieconc pseumenz pinupond pinulamb calodecu
Mid-aged mixed conifer	FT2	0.01	110	0.0282	1.5	0.7	64	1.321	2	41	80	juniocci abieconc pseumenz pinupond pinulamb calodecu
Old mixed conifer	FT3	0.01	110	0.0293	1.5	0.8	66	1.184	4	81	1000	juniocci abieconc pseumenz pinupond pinulamb calodecu
Young pine/white fir	FT4	0.01	110	0.0282	1.5	0.7	64	1.321	1	0	40	pinulamb pinujeff pinumont pinualbi abieconc pinupond
Mid-aged pine/white fir	FT5	0.01	110	0.0444	3	0.75	62	1.261	2	41	80	pinulamb pinujeff pinumont pinualbi abieconc pinupond
Old pine/white fir	FT6	0.01	110	0.0444	3	0.75	62	1.261	5	81	1000	pinulamb pinujeff pinumont pinualbi abieconc pinupond
Young red fir	FT7	0.01	110	0.0282	1.5	0.7	64	1.321	1	0	40	abiemagn
Mid-aged red fir	FT8	0.01	90	0.0649	4.5	0.9	72	1.076	2	41	80	abiemagn
Old red fir	FT9	0.01	90	0.0649	4.5	0.9	72	1.076	8	81	1000	abiemagn
Young sequoia	FT10	0.01	110	0.0282	1.5	0.7	64	1.321	1	0	40	sequgiga
Mid-aged sequoia	FT11	0.01	30	0.0800	3	0.8	62	1.197	3	41	80	sequgiga
Old sequoia	FT12	0.01	30	0.0800	3	0.8	62	1.197	10	81	3000	sequgiga
Young lodgepole/hemlock	FT13	0.01	110	0.0282	1.5	0.7	64	1.321	1	0	40	pinucont tsugmert
Mid-aged lodgepole/hemlock	FT14	0.01	30	0.0800	3	0.8	62	1.197	2	41	80	pinucont tsugmert
Old lodgepole/hemlock	FT15	0.01	30	0.0800	3	0.8	62	1.197	5	81	1000	pinucont tsugmert
Young brush	FT16	0.01	110	0.0282	1.5	0.7	64	1.321	1	0	40	cercmont querchry querwisl adenfasc
Mid-aged brush	FT17	0.02	110	0.0282	1.5	0.7	64	1.321	1	41	80	cercmont querchry querwisl adenfasc
Old brush	FT18	0.02	110	0.0282	1.5	0.7	64	1.321	1	81	1000	cercmont querchry querwisl adenfasc
Young deciduous	FT19	0.001	30	0.0232	1.6	0.9	32	1.179	1	0	40	querkelo querdoug poputrem riparian
Old deciduous	FT20	0.001	30	0.0232	1.6	0.9	32	1.179	2	41	1000	querkelo querdoug poputrem riparian
Intensity A	FT90	0.0001	30	0.0232	1.6	0.5	500	1.179	2			Effective for the 15 years following Prescribed Fire or Light Thinning
Intensity B	FT91	0.0001	90	0.0649	4.5	0.5	500	1.076	4			Effective for the 15 years following Medium Thinning



Red Fir

Pines and White Fir

Mixed Conifer



Chaparral and Scrub Oak

Deciduous

Figure 3.4. The spatial distribution of the five most common fuel types, divided by age class.

Fuel Type Parameters — The rate of fire spread is a function of weather, topography, and fuel type. When a fire is initiated on a site or spreads into a site, an Initial Rate of Spread (RSI) is calculated based on the fuels at the site and a randomly drawn weather record (stratified by season and fire region). From the weather record, fine fuel moisture conditions and the wind speed velocity are used to determine the Initial Spread Index (ISI). Fuel types were parameterized to exhibit characteristic rates of spread that vary across the ISI values (Figure 3.5).

Each fuel type includes four parameters (*a*, *b*, *c*, and *q*) that together determine RSI dependent upon ISI (Forestry Canada Fire Danger Group 1992, Sturtevant et al. 2008). We systematically adjusted these coefficients to reflect rates of spread characteristic of our fuels types based on expert opinion (B. Bahro, USFS, personal communication). For the deciduous fuel type, we derived parameters (from fuel class TL6; broad-leaf deciduous) from the Fire Behavior Fuel Models (FBFM) developed by Scott and Burgan (2005). The resulting spread rate curves are depicted in Figure 3.5. Other fuel type parameters (e.g., crown-to-base height) were also estimated from expert opinion (B. Bahro, USFS, personal communication).

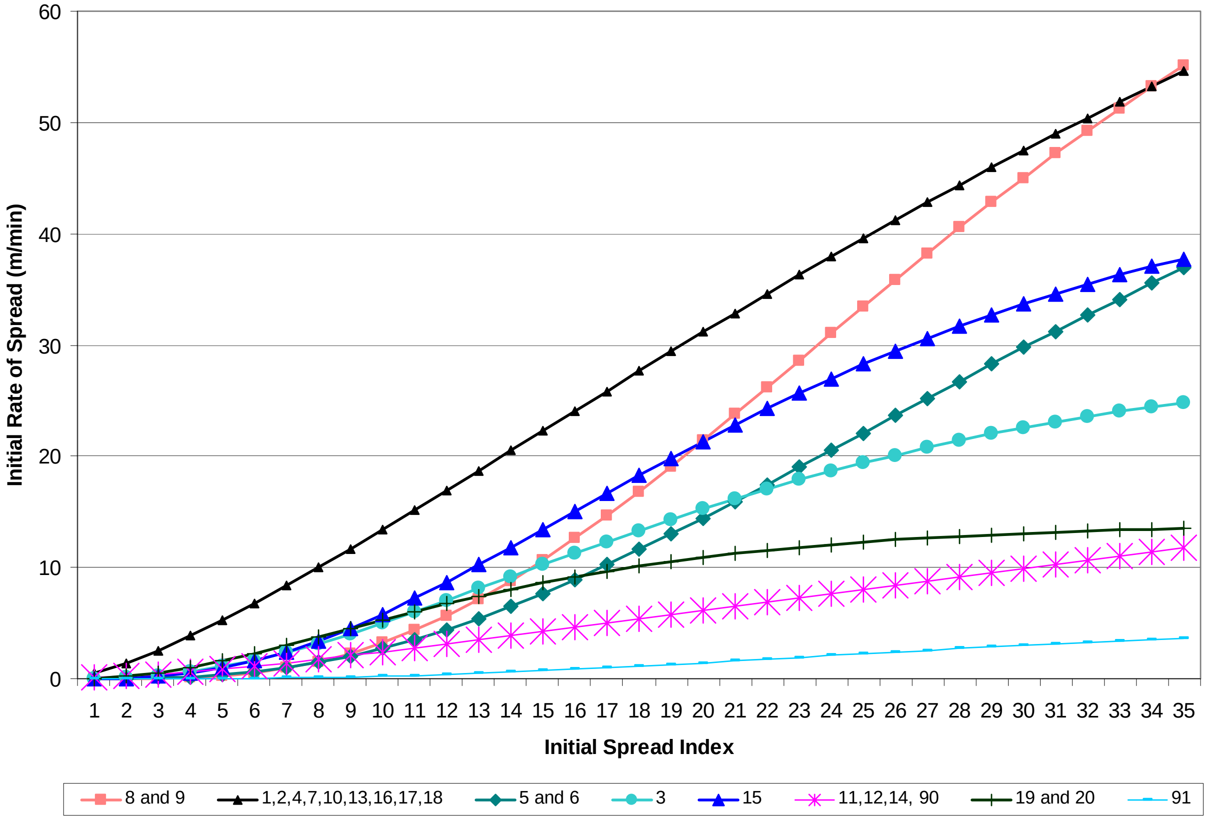


Figure 3.5. Initial Rate of Spread (RSI) as a function of the Initial Spread Index (ISI) for our 20 fuel types. See Table 3.5 for fuel type numbers and definitions.

After calculating the RSI, the rate of spread is multiplied by the build up index (BUI), which accounts for the influence of longer-term weather trends (10 days) on fuel-moisture. The BUI is calculated based on consecutive days in the weather database; however the influence of the BUI depends on the fuel type (which is referred to as the Buildup Effect) (Figure 3.6).

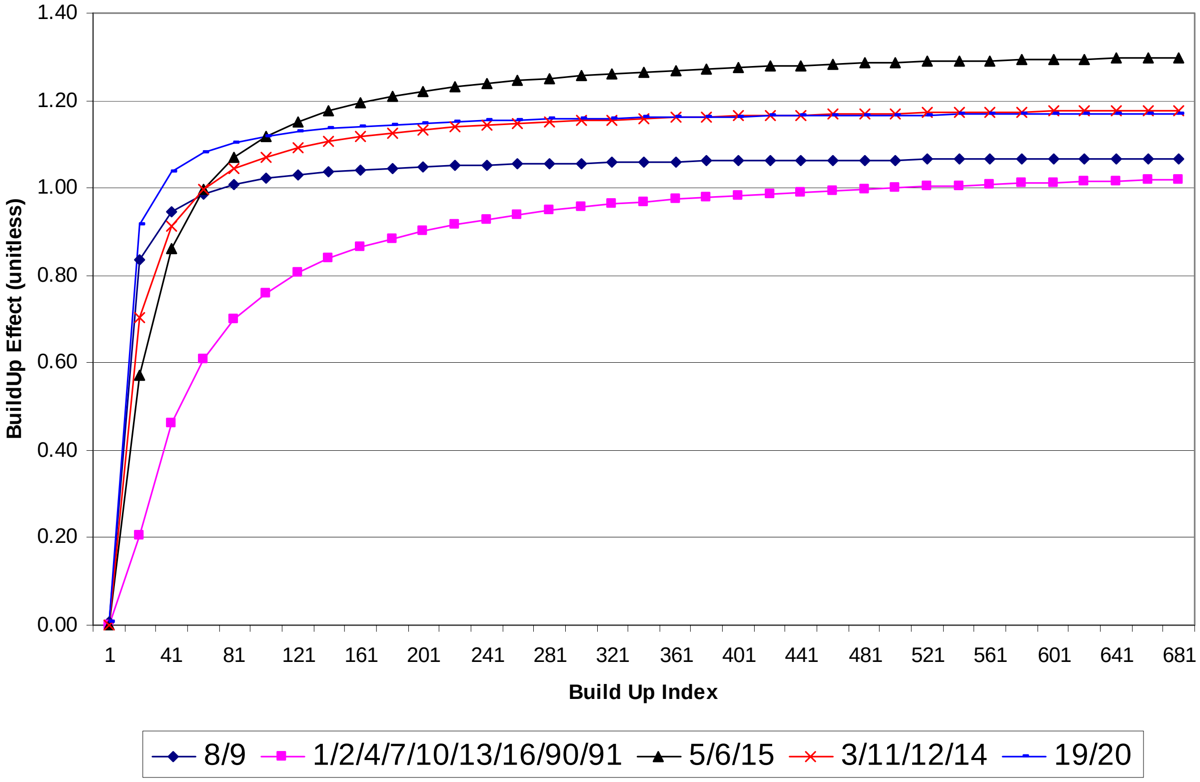


Figure 3.6. The relationship between Build Up Index (10 day weather) and the Build Up Effect for our 22 fuel types. See Table 3.5 for fuel type numbers and definitions.

3.1.10 Defining and Parameterizing Fuel Treatments

We simulated fuel treatments across the Sierra Nevada using a new extension, Biomass Harvest, created specifically for this project. The Biomass Harvest extension extends the functionality of the Base Harvest extension (Gustafson 2000, Scheller et al. 2007) by allowing the partial removal of cohort biomass whereas the Base Harvest only allowed total removal of cohorts. The partial removal of cohorts allowed us to more accurately reflect the 'thin from below' fuel treatments that are more commonly practiced in Sierra Nevada national forests.

The harvest extension simulates human modification of forest structure, including fuel treatments. These modifications can subsequently trigger changes in fuel types. The extension operates within a hierarchical framework. First, the total landscape is divided into management areas. Each management area is subsequently divided into stands, which are one or many adjacent sites that are treated as a unit. Within each stand are the multiple sites and within each site individual cohorts are either completely removed or their aboveground live biomass is reduced.

The harvest extension requires that specific prescriptions are defined. A prescription indicates how a stand is chosen for treatment (both minimum qualifications and a ranking algorithm is defined) and how cohorts are treated, specifically which species and which ages of cohorts are treated. Within each management area, a treatment is applied at a user determined rate. For illustration, a user could indicate that within a given management area, 10% of the area is treated every 5 years with a prescribed fire prescription.

Management Areas — All simulated fuel treatments were restricted to those areas that could *potentially* be treated by the US Forest Service (Figure 3.7). This potentially treatable area included lands inside of national forests but excluding non-treatable designations, such as existing and recommended Wilderness Areas, existing and recommended Wild and Scenic River areas (Wild and Scenic Rivers Act, 1968), Research Natural Areas, non-vegetated land, and spotted owl Protected Activity Centers (PACs) (Tanya Kohler, USFS, *personal communication*).

This potentially treatable area was subdivided into two slope categories (> 30% or ≤ 30% slope) because mechanical treatments would not be performed on slopes >30%. The landscape was further divided into groups of contiguous cells sharing the same initial conditions (henceforth, “stands”). Where stands were > 1000 ha, they were further subdivided into approximately equal areas to keep maximum stand size < 1000 ha. Average stand size was 7 ha. However, if stands were less than 20 ha, they were allowed to spread to neighboring stands that qualified for the individual prescriptions (defined below). Because stands were largely developed from their initial conditions and they were not shaped to contain fire or clustered or set into arrays, they are not equivalent to Strategically Placed Area Treatment (SPLAT), e.g., Schmidt et al. (2008).

Fuel Treatment Rate — Based on recommendations from the stakeholder group and science advisors, we initially tested the effects of treating 2% and 8% of the potentially treatable area per time step (every 5 years). The higher treatment rate (8% per 5 years) was suggested to simulate treating about 1/3 of the landscape over 20 years, or roughly the proportion of the landscape that fire-spread modeling suggests should be treated to significantly reduce fire incidence on a forested landscape (Finney et al. 2006). The lower treatment rate (2% per 5 years) was estimated as what the Forest Service has been treating recently, or planning to treat. However, pilot simulations performed on a subset of the total study area[29] indicated that treating only 2% of the landscape every 5 years had no discernable effect on fire rotation periods. We therefore dropped the 2% rate and replaced it with a 4% treatment area to compare with the 8% treatment area (Table 3.6).

Fuel Treatment Intensity — In close coordination with personnel from the US Forest Service Stewardship and Fireshed Assessment Cadre, we defined two different treatment approaches: Light and Medium Intensity. Both assumed a combination of mechanical treatment followed by prescribed fire on slopes < 30%, and prescribed fire only on slopes ≥ 30%. The treatment combinations differed in the prescriptions applied at sloped < 30%. In both treatment approaches the mechanical treatments were followed by prescribed fire, while steeper slopes were treated only with prescribed fires (the same prescription for both treatment combinations). Treatment combinations that indicate that they were treated with Low Intensity were simulated to receive the Light Thin prescription at slopes ≤ 30% and the Prescribed Fire prescription at slopes > 30% (prescription definitions below). Treatment combinations that indicate that they

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[29] We performed pilot tests of the fire and fuel treatment modeling on ~1/3 of the total study area, including all of the Sierra NF, a small section of the Stanislaus NF, and a small section of the Sequoia NF, before initiating simulations over the full study area.

were treated with Medium Intensity were simulated to receive the Medium Thin prescription at slopes ≤ 30% and the Prescribed Fire prescription at slopes > 30% (Table 3.6).

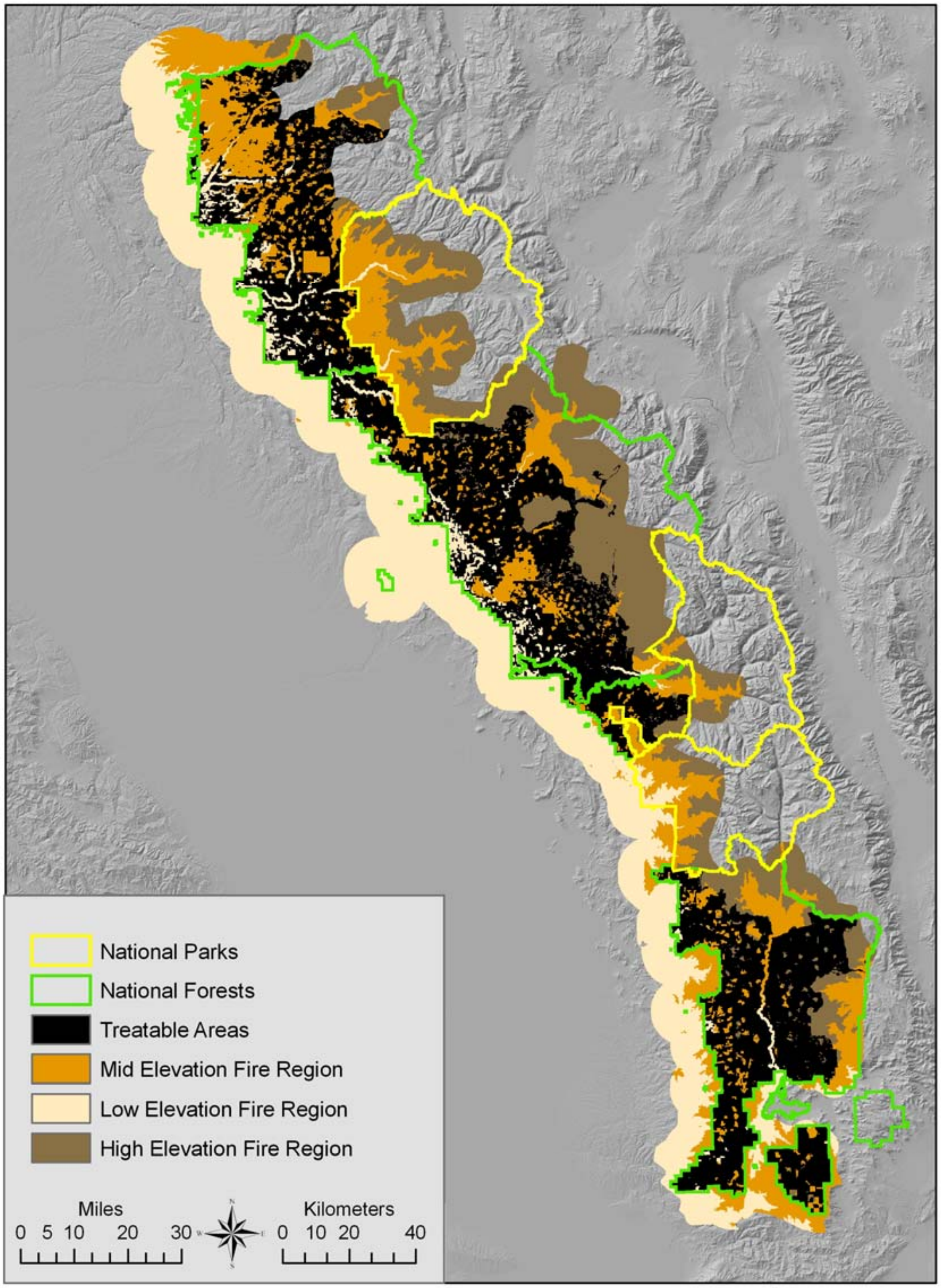

Figure 3.7. Potentially treatable area overlaid on top of three fire regions (the WUI fire regions were lumped with their respective elevations for display purposes).

Table 3.6. Rates and prescriptions used in the two alternate fuel treatment rates and fuel treatment intensities, respectively, as a function of slope.

Fuel Treatment Rate	Slope < 30%	Slope > 30%
Maximum Rate	8% every 5 years	8% every 5 years
Medium Rate	4% every 5 years	4% every 5 years
Fuel Treatment Intensity	Prescriptions applied when slope < 30%	Prescription applied when slope > 30%
Low Intensity	**Light Thin**: Mechanical thinning up to 12” DBH followed by prescribed fire with 2’ flame length	Prescribed fire with a 4’ flame length
Medium Intensity	**Medium Thin**: Mechanical thinning up to 30” DBH followed by prescribed fire with 2’ flame length	Prescribed fire with a 4’ flame length

Our prescriptions were designed to simulate effects of a relatively broad range but reasonable of management activities on stand structure and fire behavior. They were not specifically designed to mimic details of actual management actions as would be implemented by a forest manager. Rather, our prescriptions were designed to provide a consistent set of treatments across a broad area. Although the LANDIS-II Biomass Harvest extension allows an infinite combination of rules and thinning intensities, we opted to limit ourselves to three prescriptions that could be broadly applied: prescribed fire only (Prescribed Fire), light mechanical thin followed by prescribed fire (Light Thin), and moderate mechanical thin followed by prescribed fire (Medium Thin).

- *Prescribed Fire* - The Prescribed Fire prescription was applied as a stand-alone treatment only on slopes > 30% because this is considered too steep for mechanical thinning treatments. The Prescribed Fire prescription was designed to emulate effects of a 4-foot flame length (Klaus Barber, US Forest Service). The criteria for selecting stands for prescribed fire were slopes >_30% and having a mean maximum age > 50 years and must be dominated either by pines, firs, or Douglas fir (with ages between 40 and 200 years) or by oaks (with ages between 40 and 200 years). Prescribed fires could be re-applied to a stand only once at least 10 years had passed since the last treatment. Out of all the stands that met these criteria, the model randomly chose where to apply the Prescribed Fire fuel treatment. The effects of the treatment were simulated by removing biomass of tree cohorts according to a declining curve, with the greatest biomass removed for the youngest cohorts, and the least biomass removed for older cohorts (Figure 3.8). Following a Prescribed Fire, the fuel type of the stand was converted to Fuel Type 90 (see Fuel Type parameterization, Section 3.4). Although only applied once, we assumed that these treatments represented the full prescribed fire necessary to achieve the assumed fire spread rate.
- *Light thinning followed by prescribed fire* (*Light Thin*)– The Light Thin prescription was designed to emulate thinning from below with understory trees up to 12” in diameter being removed (Figure 3.9) followed by a prescribed fire with a 2-foot flame length (Klaus Barber, US Forest Service). Therefore, the Light Thin prescription would not leave any slash on the ground; and like the Prescribed Fire treatment, the fuel type was converted to FT90. Light Thin stand qualifications were: minimum age of the stand had

to be ≥ 50 years; treatment could only occur after 20 years since the last treatment; and slopes ≤ 30%. Stand that met these qualifications were ranked for treatment based on the species present on the stand and their ages: mature white fir (weight = 100), ponderosa pine (50), Douglas fir (50), incense cedar (40), lodge pole pine (25), Jeffrey's pine (25), and western white pine (25). The highest ranked stands were treated first.

- *Moderate thinning followed by prescribed fire* (*Medium Thin*) – The Medium Thin prescription was designed to emulate a more intense thinning from below (relative to Light Thin) up to 30" in diameter, again followed by a prescribed fire with a 2-foot flame length (Figure 3.9). The qualifications and stand ranking rules for choosing which stands to treat were identical to the rules for Light Thin.

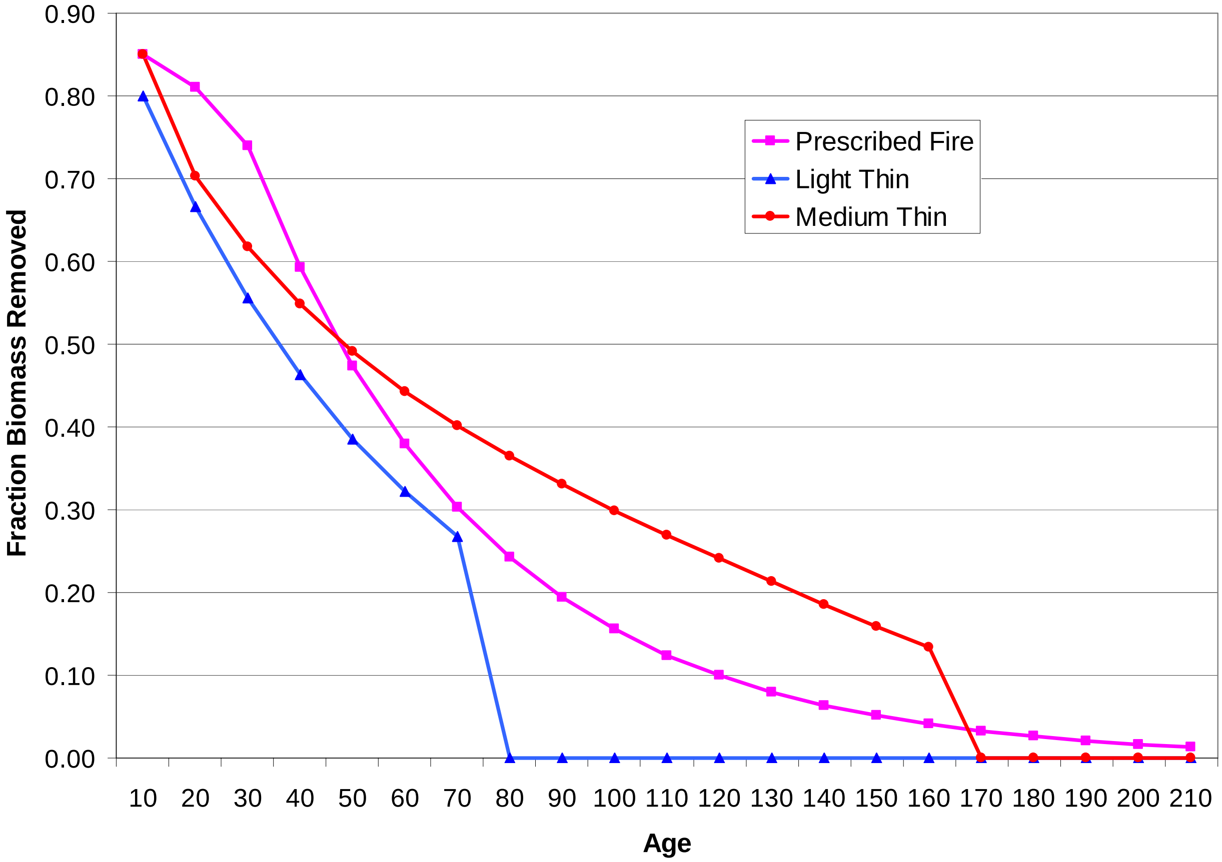

Figure 3.8. The proportion biomass removed as a function of age (years) for three prescriptions. The Light Thin and Medium Thin prescription include biomass removed from mechanical thinning and the follow-up prescribed fire. The lines represent the curves for mixed conifers (ponderosa pine, Douglas fir, Jeffrey pine, sugar pine) as a representative example.

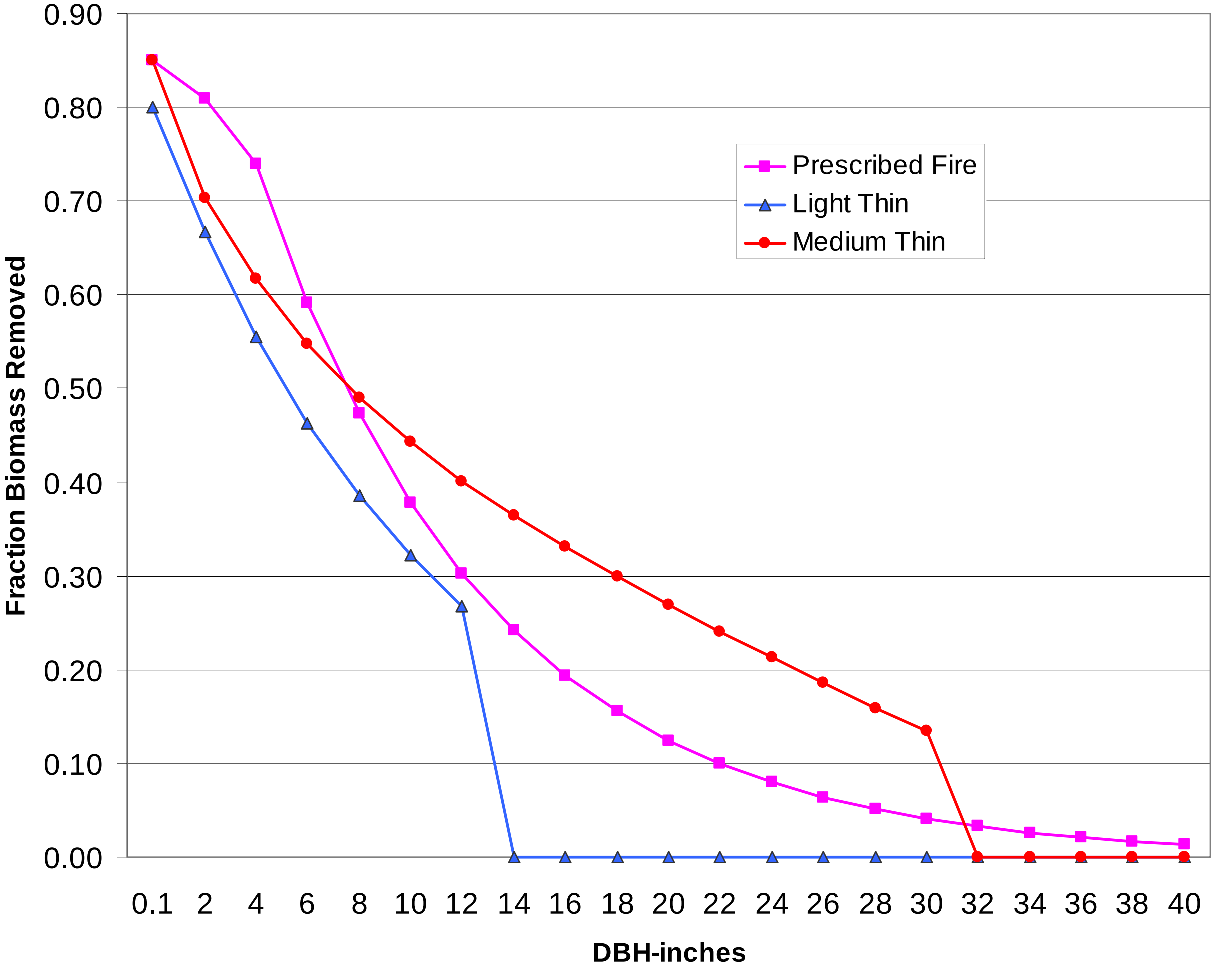

Figure 3.9. The proportion biomass removed as a function of diameter at breast height (DBH; inches) for three prescriptions. The Light Thin and Medium Thin prescription include biomass removed from mechanical thinning and the follow-up prescribed fire. The lines represent the curves for mixed conifers (ponderosa pine, Douglas fir, Jeffrey pine, sugar pine) as a representative example.

Location of Fuel Treatments — Ideally we would have dynamically assigned treatment locations based on where the GAM predicted the best fisher habitat would be at each time step within each replicate. However, there was no feasible way to allow the information from a GAM to feed back into the LANDIS-II Harvest extension.

As an alternative, fuel treatment locations were divided into three categories to test effectiveness of treating inside of fisher habitat, outside of fisher habitat, or both. For this purpose, we defined fisher habitat as those areas with a predicted habitat value (or probability of fisher occurrence) > 0.33 using the GAM model LAND8, averaged across all year-50 habitat replicates for both Baseline and High Fire regimes (with no fuel treatments, 20 maps averaged). Year 50 was chosen as most likely to represent the conditions that produced the final fisher population count. The fisher habitat value of 0.33 was used because it resulted in a 50:50 balance of within fisher habitat and outside fisher habitat, thereby removing the confounding effect of area (Figure 3.10). The identical prescriptions and rules for applying prescriptions were applied, the only difference being the spatial extent of where prescriptions could be applied.

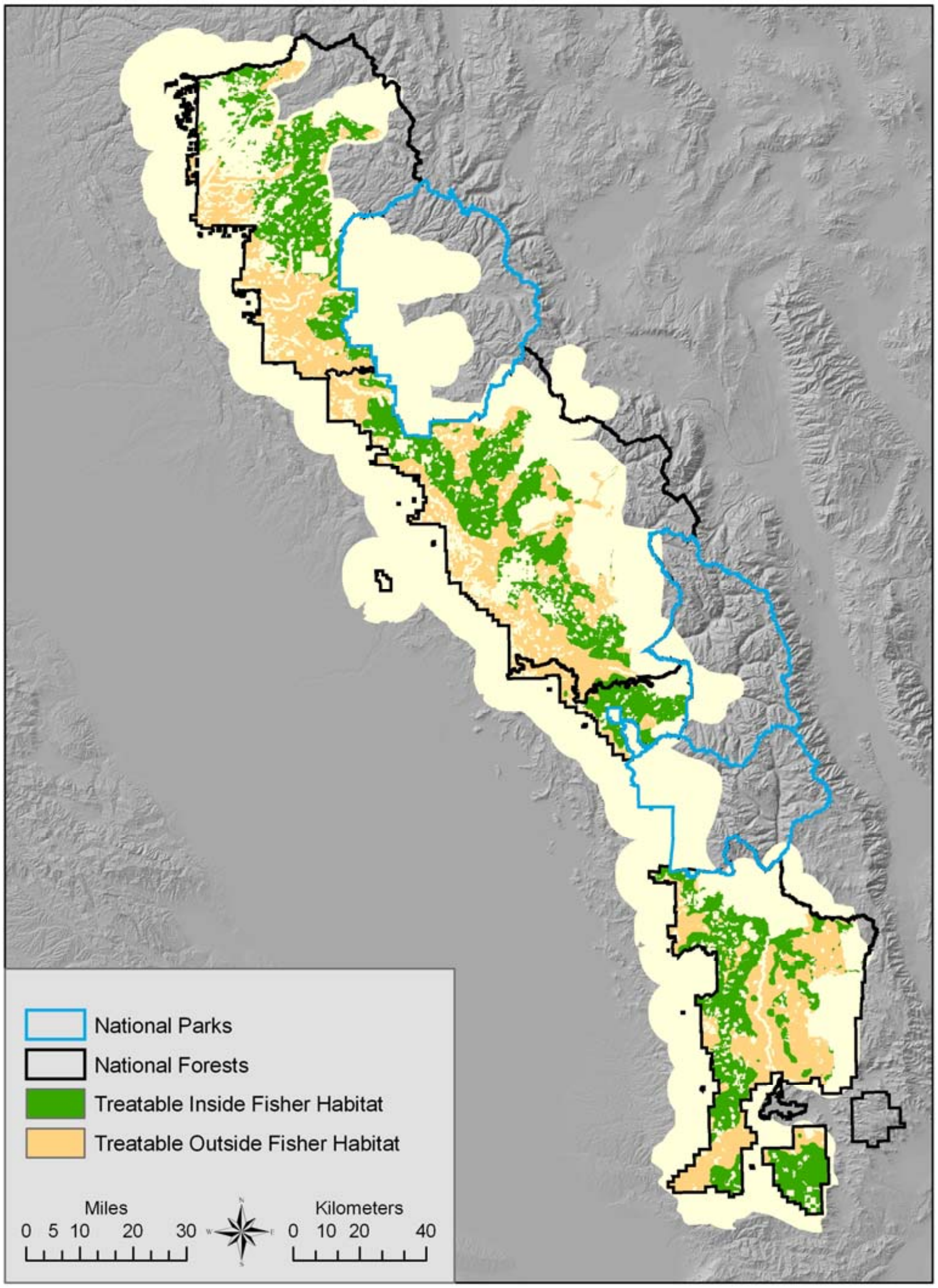

Figure 3.10. Potentially treatable areas inside and outside fisher habitat as estimated from 20 replicates at year 50.

3.1.11 Model Outputs

LANDIS-II outputs include both standard outputs for each extension chosen and additional outputs as indicated by separate output extensions. Standard extension outputs included: fire severity maps (every 5 years), a complete fire event data log file, fuel type maps (5 years), a fire summary log file for calculating fire rotation periods, fuel treatments applied maps (5 years), and the complete fuel treatment log file (5 years). We used the existing Biomass output extension to map total aboveground live biomass every 10 years.

We developed two new output extensions for this project. The Biomass-by-Age output extension summarized cohort biomass by species and by age class and mapped the results. This enabled us to calculate the area occupied by the different white fir, Ponderosa pine, Douglas fir, and black oak age classes. The Cohort Statistics output extension allowed us to map various age statistics (maximum and standard deviation of cohort ages) and species richness.

3.2 Results

3.2.1 Simulated Fire Rotation Periods

The fire rotation period (FRP) is defined as the time in years required to burn an area equal in size to area of interest. FRP therefore incorporates both fire size and fire frequency into one metric; and is thus useful for understanding relative differences in total amount of fire between different combinations of fuels treatments and fire. FRPs will be influenced by fuel treatments due to reduced spread rate and will be influenced by recent fires that also remove fuel and/or reduce spread rates. We specifically parameterized the fuel treatments as having a slower spread rate (i.e., they will be effective at local scales by definition); however, FRPs allowed us to examine the landscape-scale effect of fuel treatments on the fire regime.

In our calibrated baseline fire regime simulations, the mid-elevation (~1190 – 2120 m) fire region had longer fire rotation periods (FRPs) than the higher elevation (> 2120 m) or lower elevation (< 1190 m) fire regions; however, the mid elevation region also had the highest mean variability, which showed that there was high stochasticity in fire in this region. The low elevation fire region, on the other hand, had the most fire (shortest FRPs) and the least variability. These elevational trends in FRP were consistent for all of the treatment combinations we compared, including those with and without fuel treatments (Figure 3.11).

These elevational differences in FRP were consistent with the relative differences in the FRPs from the last 20 years in the fire history database as well as with expectations based on the literature. The low elevation region had the largest proportion of chaparral and area of Wildland Urban Interface, which explains why we calibrated the FRP to be shorter in this region than that of the other fire regions. Although the fire sizes were smaller in the low elevation, they were also more frequent, which reflects our intention to simulate the general trend of higher human-caused ignition frequency, but more effective suppression, in human-dominated areas. The higher variability of fire in the mid elevation region (and to a lesser degree in the high elevation) shows that, when there is somewhat less fire on the landscape overall, the variation from replicate to replicate is higher (Syphard et al. 2007c). Also, there was a greater shift in mixed conifer fuel type age classes in the mid elevation region (see below) than the other regions, which may also be related to the higher variability in FRP.

Under the high fire regime, fire rotation periods were shorter than the baseline by roughly 25 – 50 years (compare Figures 3.11 and 3.12), which indicated that substantially more area burned during the 50 years simulated in the high fire regime than the baseline, particularly in the high elevation. Although we did not intend to predict how weather may change when we restricted the weather database to reflect more extreme conditions, the response in FRP showed that fire was strongly influenced by weather in the simulations. While there is considerable uncertainty regarding how climate may change in the future, and how fire may respond, the shorter FRPs in the high fire regime were consistent with recent analyses that show increasing area burned in response to more severe weather (Calkin et al. 2005, Westerling et al. 2006).

.In general, we did not find large differences in the total area burned (fire rotation period) when we compared the combinations of treatment and fire, particularly in the low elevation fire region. This low effect in the low elevation region was likely due to the fact that much less area was treated in this region than the mid and high elevation regions. In the high elevation region in the baseline fire regime, FRPs were longer in fuel treatment combinations than they were in the high fire regime; and the longest FRP occurred with the maximum treatment rate with the medium treatment intensity. In the mid elevation, reduction in FRP in the fuel treatment combinations was similar in magnitude between the baseline and the high fire regimes; but the effects of different treatment combinations were different depending on the fire regime. Although the mean differences among the treatment combinations were not highly substantial, there was substantial variability among replicates within these combination of treatments and fires, particularly in the mid elevation fire region.

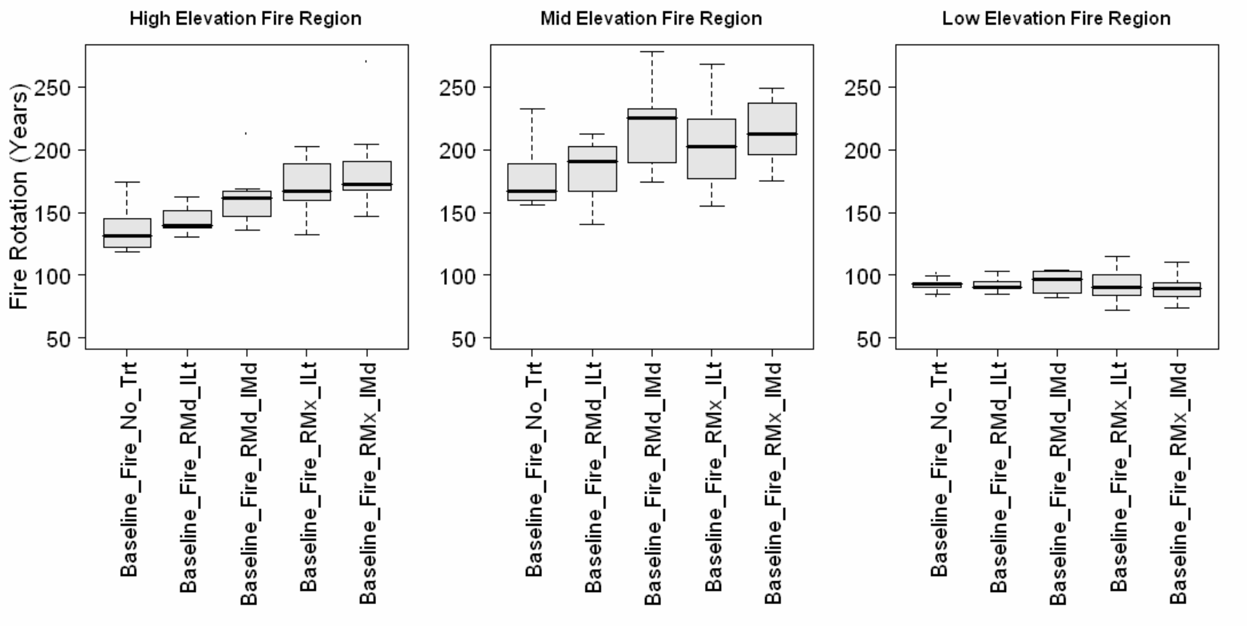

Figure 3.11. Simulated fire rotation periods (the time in years required to burn an area equal in size to the fire region) under the baseline fire regime for three fire regions (high, mid, and low elevation). WUI fire regions not shown. RMd indicates the medium rate of treatments (4%). RMx is the maximum rate of treatment (8%/5yr); ILt is the Light Intensity treatments; IMd is the Medium Intensity treatments.

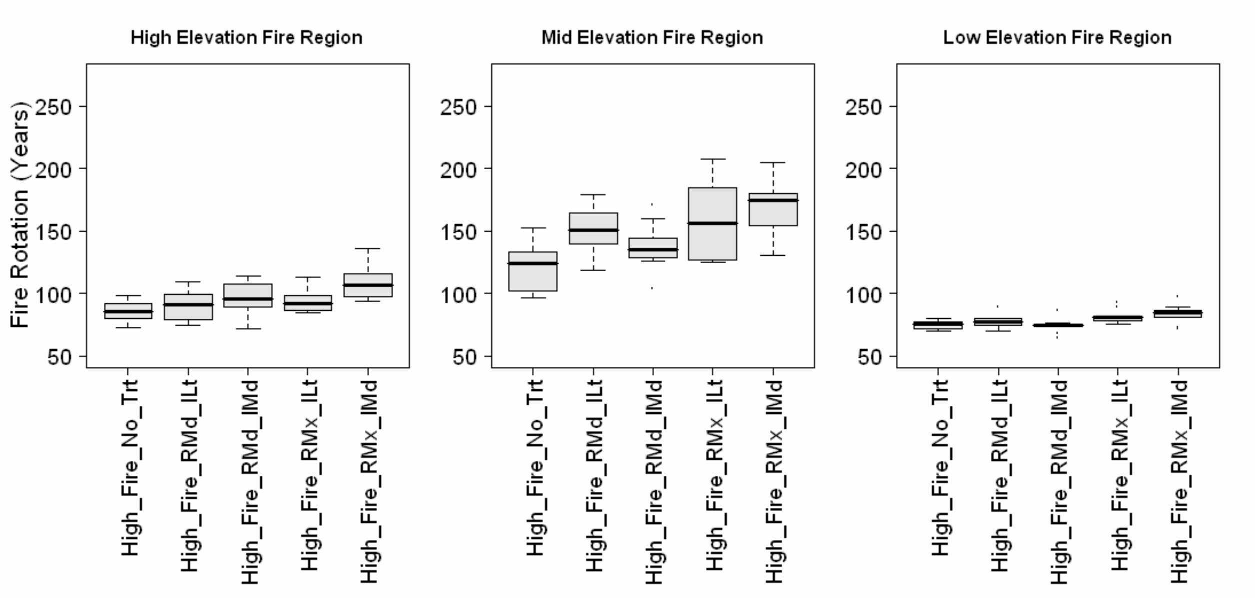

Figure 3.12. Simulated fire rotation periods (the time in years required to burn an area equal in size to the fire region) under the high fire regime for three fire regions (high, mid, and low elevation). WUI fire regions not shown. RMd indicates the medium rate of treatments (4%). RMx is the maximum rate of treatment (8%/5yr); ILt is the Light Intensity treatments; IMd is the Medium Intensity treatments.

3.2.2 Simulated Fire Severity

In LANDIS-II, fires vary in severity across an ordinal scale that ranges from 1 to 5. Although fire severity is a measure of fire effect on vegetation (i.e., high severity fires are more likely to kill vegetation than low severity fires), in LANDIS-II, it also indirectly accounts for fire intensity. Therefore, any effect of fuel treatment on lowering fire intensity would be reflected in the severity of the simulated fire. The model requires the specification of a fire severity coefficient in the parameters to adjust the distribution of mean severities throughout the simulations. Although individual fires ranged in severity from 1 to 5, and many fires were mixed severity, the mean severities in the baseline fire regime (i.e., without fuel treatment) ranged from approximately 3.27 to 3.83. Overall, fires were slightly less severe in the high elevation fire region and slightly more severe in the low elevation fire region (Figure 3.13). In the high fire regime, mean severities ranged from approximately 4.05 to 4.65. Again, fires were slightly less severe in the high elevation and slightly more severe in the low elevation (Figure 3.14).

There was no pronounced trend in the effect of fuel treatment on average landscape-scale fire severity, particularly in the baseline fire regime. In the high fire regime, mean fire severities for all treatment combinations were lower than those in the baseline in the low and high elevation fire regions; however, there was virtually no effect in the mid elevation fire region. Although there was enough variation in mean fire severities within the treatment combinations to create substantial overlap among the combinations of treatments and fire, the mean range of variation was relatively small compared to the full range of fire severities that occur in the simulations. Therefore, although individual fires in the simulations vary in severity as a function of local fuel conditions and foliar moisture content, there was generally not enough overall variation in mean severities across the landscape to statistically test for fire severity effects.

Because the fire severity coefficient regulates the mean level of severity within any fire regime, we cannot attribute the higher severity of fires in the high fire regime to the different weather conditions. Rather, we purposely adjusted the coefficient to attain those generally higher severities because we wanted to evaluate how fuel treatment efficacy and fisher population may vary under more severe fire conditions. Therefore, the reason that there was slightly more of an effect of treatment in lowering fire severity in the high fire regime may be that those fires were substantially more severe to begin with.

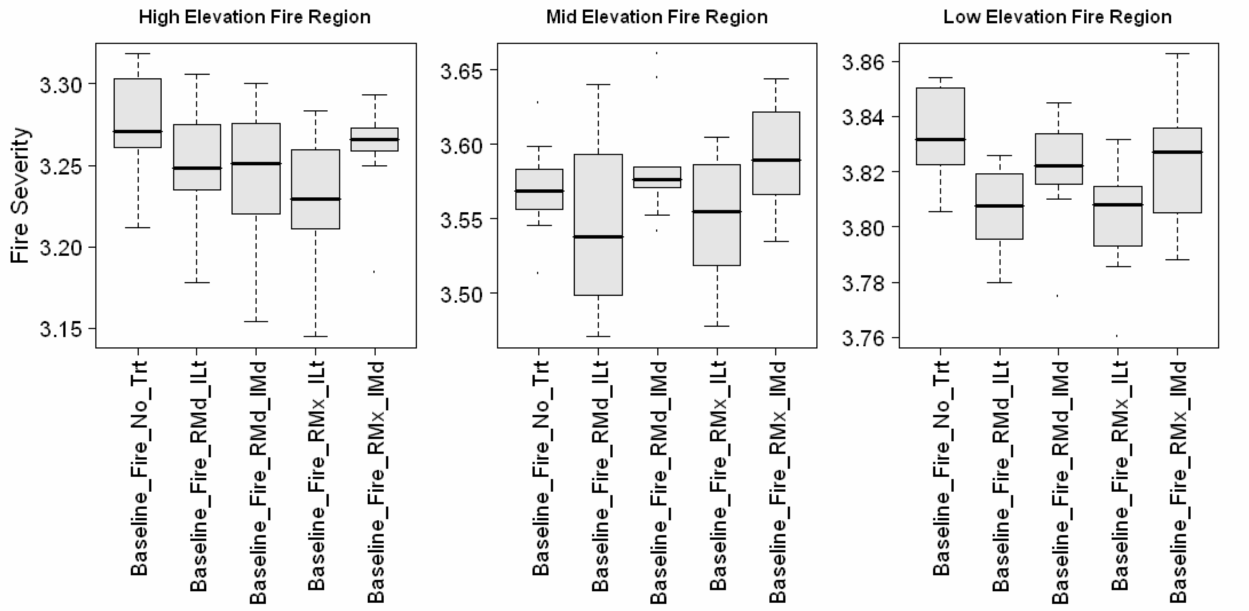

Figure 3.13. Simulated fire severities (scale 1 to 5) under the baseline fire regime for three fire regions (high, mid, and low elevation). WUI fire regions not shown. RMd indicates the medium rate of treatments (4%). RMx is the maximum rate of treatment (8%/5yr); ILt is the Light Intensity treatments; IMd is the Medium Intensity treatments.

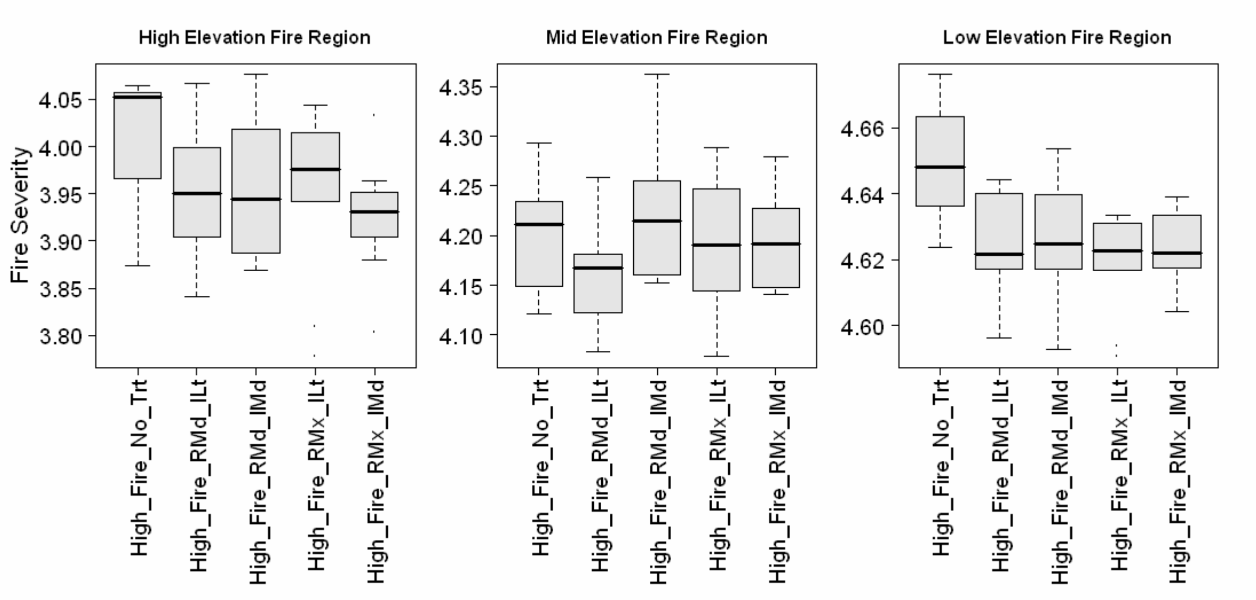

Figure 3.14. Simulated fire severities (scale 1 to 5) under the high fire regime for three fire regions (high, mid, and low elevation). WUI fire regions not shown. RMd indicates the medium rate of treatments (4%). RMx is the maximum rate of treatment (8%/5yr); ILt is the Light Intensity treatments; IMd is the Medium Intensity treatments.

3.2.3 Fuel Type Changes Through Time

We also assessed how fuel types changed through time to ascertain whether any combination of succession, fuel treatments, and wildfire were creating substantial changes in our defined fuel types. Although these results served primarily as a mean of model verification – ensuring that the model was not producing changes outside the bounds of general expectations – any large changes in fuel types may also help explain differences in the area burned or fire severity across our primary fire regions.

Under baseline fire, overall trends in fuel types were remarkably similar across the combination of treatments and fire (Appendices G – K), although variability was higher when there were no fuel treatments (Appendix A).

Within the high elevation (> 2120 m) fire region, there was a modest increase in young mixed conifer fuel type with a modest decrease in the mid aged and older red fir fuel type.

Within the mid elevations (~1190 – 2120 m) fire region, the area of young mixed conifer fuel type doubled from 15 to 30% of the area. Changes in the area of mid aged and older mixed conifers fuel types mirrored each other, indicating a demographic transition. The area chaparral and scrub oak declined modestly.

Within the low elevation (< 1190 m) fire region, there were only very modest changes in fuel types with a slight decline in chaparral and scrub oak and a slight increase in the area of young mixed conifer fuel type.

Under the high fire regime (Appendices L – P), the trends in fuel types were generally similar to those found under the baseline fire. At higher elevations, there was a somewhat larger increase of the young mixed conifer fuel type and a slightly larger decline of mid aged and older red fir fuel types.

Under all fuel treatment combinations (Appendices H – K and M – P), the area of treated fuels (fuel types 90 or 91, indicated as Light Thin or Medium Thin) was highest in the mid elevations and lowest in the low elevations. Under the medium fuel treatment rate (4%/5 yrs), the treated area occupies ~10% of the mid elevations. When the maximum fuel treatment rate (8%/5 yrs) was applied, the treated area increased to 15-20% of the mid elevations. The total treated area reached a plateau because, a) treated areas revert to native fuel types after 15 years, and b) some areas were treated repeatedly.

In conclusion, there were no unexpected or sudden changes in fuel types over time. This relative lack of change is likely due to a combination of factors: a) our fuel types were relatively broad and therefore relatively insensitive to minor changes induced by succession, b) any localized changes in fuel types were compensated for by equivalent changes in another part of the landscape, e.g., if young mixed conifer is lost due to fire in one location, an approximately equal area of young mixed conifer is being created through succession, and c) the lifespan of these trees is fairly long relative to the duration of the simulation. An assessment of potential changes in vegetation due to fire in the New Jersey pine barrens, also using LANDIS-II, demonstrated a similar landscape-scale stability over 50 years (Scheller et al. 2008). In conclusion, we found no large unexplainable changes in fuel types that would lead us to believe that the model was not behaving as expected.

3.2.4 Effect of Fuel Treatment on Fire

Although fuel treatments varied in rate, intensity, and location, evaluating total aboveground biomass removal allowed us to directly quantify treatment efficacy at a landscape scale.

The overall effect of simulated fuel treatments on fire varied by fire region and fire regime (Figures 3.15, 3.16, 3.17). Linear regression and ANOVA results demonstrated a significant positive relationship between fire rotation period and amount of biomass removed at mid and high elevations under both the baseline and high fire regimes. At low elevations, treatment effects were only significant under the high fire regime. These significant positive relationships show that, over the course of our 50-year simulations, the removal of biomass through fuel treatment effectively reduced the total amount of fire on the landscape. Aboveground biomass was greatest within the mid elevation fire region and lowest within the low elevation fire region.

The relative effectiveness of treatment is indicated by the slope of the estimated fit between FRP and biomass removed (Figures 3.15, 3.16, 3.17). In general, fuel treatments are effective under baseline and high fire regimes at both mid and high elevation fire regions (Figures 3.16 and 3.17).

Within the high elevation fire region, fuel treatment effectiveness declines under the high fire regime as compared to the baseline fire regime. At high elevations and under the baseline fire regime, areas with no fuel treatments have a lower FRP than treated areas (~160 and 190 years, respectively), indicating more overall fire. Under the high fire regime, the treated and untreated areas become more similar in FRP (~100 and 105 years respectively). This is because fires are larger under the high fire, fires burn from west to east, and therefore more fires are burning from the treated areas into the untreated areas.

Within the low elevation fire region, fuel treatments are not initially effective under the baseline fire regime and become effective under the high fire regime (Figure 3.17). This increase in effectiveness at low elevations may be related to the change in fire weather (fewer days with low FFMC and BUI, data not shown), chaparral fuel types that are highly sensitive to fuel moisture conditions (Figure 3.5), and/or the increase in fire-treatment intersections at low elevations (Figure 3.18).

However, we have potentially overestimated the effectiveness of fuel treatments in two cases. First, the Prescribed Fire treatments (on slope > 30%), when applied to chaparral, likely overestimated the duration of treatment effectiveness. Chaparral shrublands are quite flammable relative to other vegetation types due to low decomposition rates, high dead-to-live fuel ratios, dense community structure, and low fuel moisture (Christensen 1985). After fire, chaparral typically returns quickly to its pre-fire composition (Bond and van Wilgen 1995), and in some cases, may form a closed canopy as soon as five years. Although rates vary by species and site conditions, the relatively rapid increase in fuel and flammability with age in chaparral suggests that it would likely return to pre-fire conditions earlier than the 15-year duration specified in our treatments.

The other potential source of overestimation in fuel treatment effectiveness was a mis-parameterization of the mid-aged and old red fir fuel types. Although they are only locally dominant, their spread rates were uncharacteristically fast, which could potentially allow fires to grow to a larger size, and thus, lower the FRPs. Because the spread rates were parameterized to be relatively faster than they should have been, fuel treatment effectiveness in these regions may have been exaggerated. These red fir dominated areas only comprise ~8% of the treated area, however. Therefore, we believe that any exaggeration of fuel treatment effectiveness was likely small relative to the overall effect of treatment on the landscape.

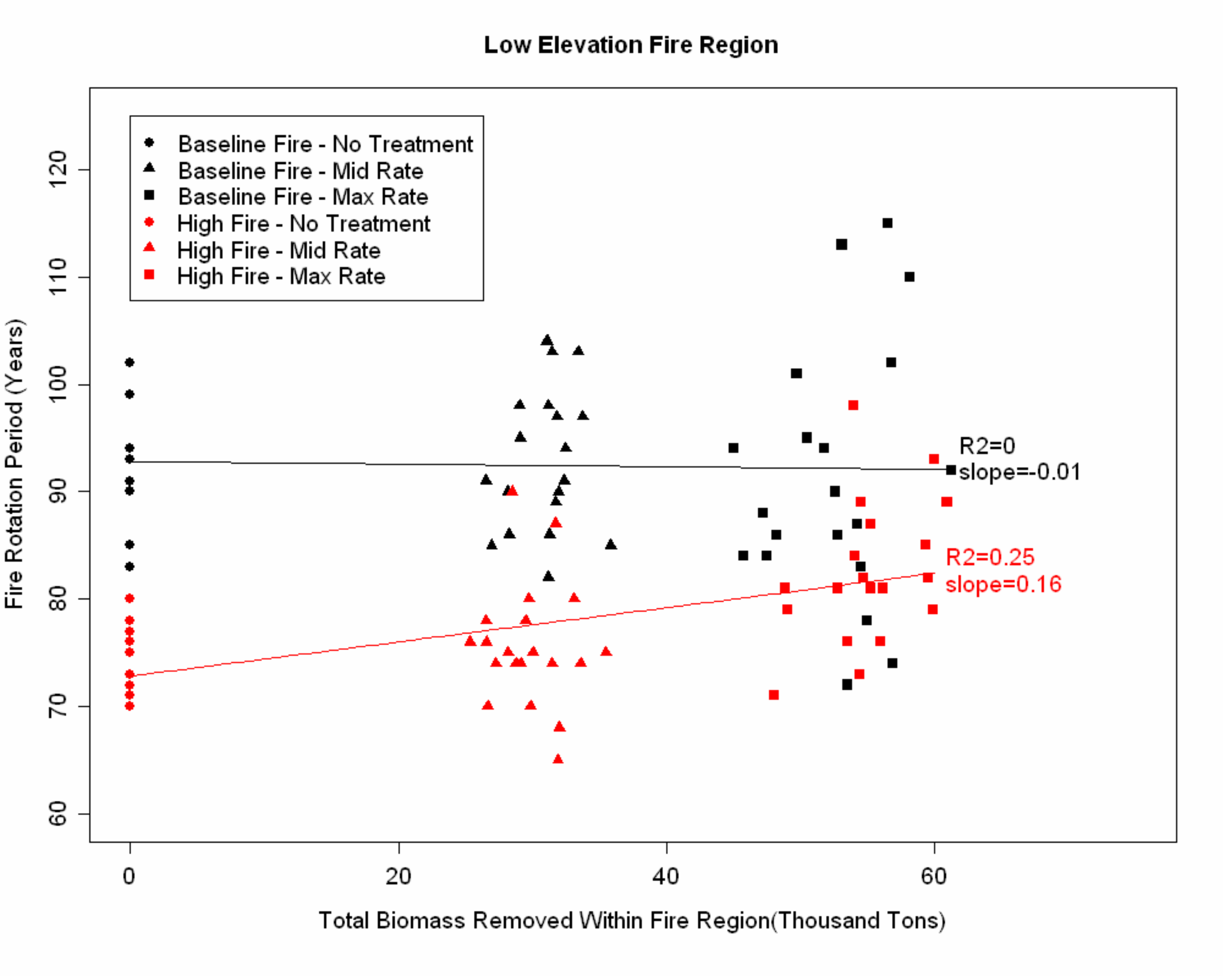

Figure 3.15. Simulated fire rotation periods within the low elevation fire region, under the baseline and high fire regimes, regressed against simulated biomass removal due to fuel treatments within the same fire region.

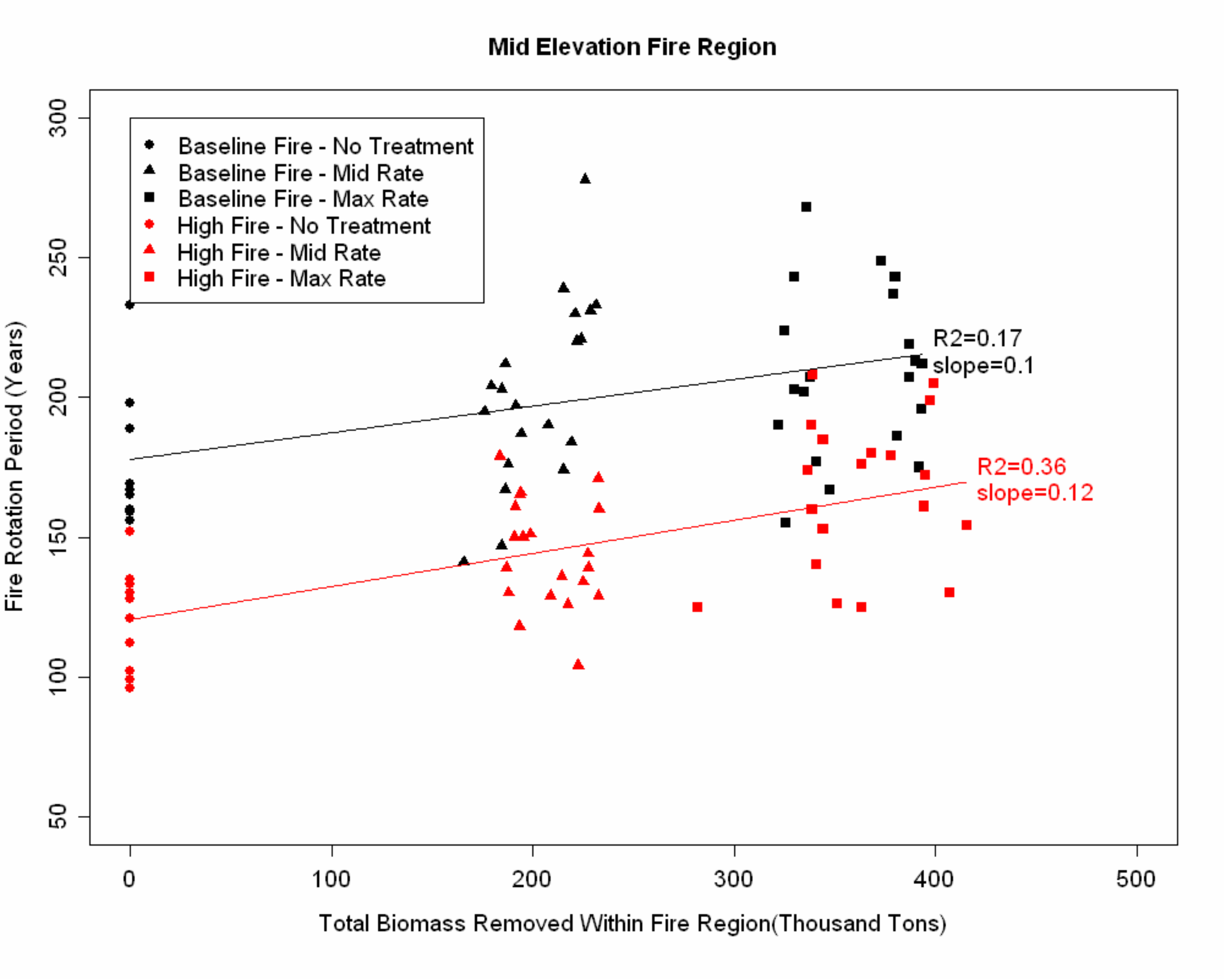

Figure 3.16. Simulated fire rotation periods within the mid elevation fire region, under the baseline and high fire regimes, regressed against simulated biomass removal due to fuel treatments within the same fire region.

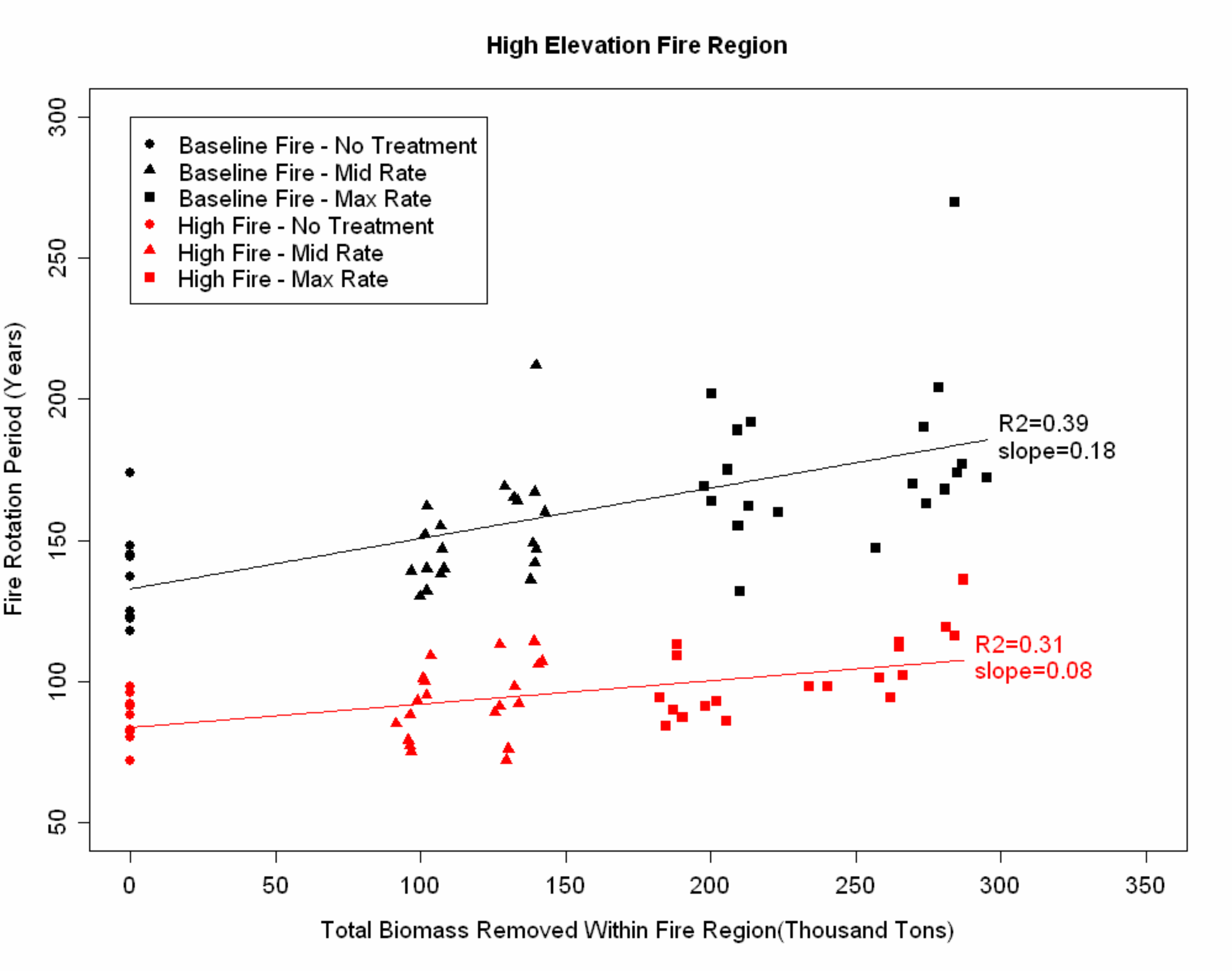

Figure 3.17. Simulated fire rotation periods within the high elevation fire region, under the baseline and high fire regimes, regressed against simulated biomass removal due to fuel treatments within the same fire region.

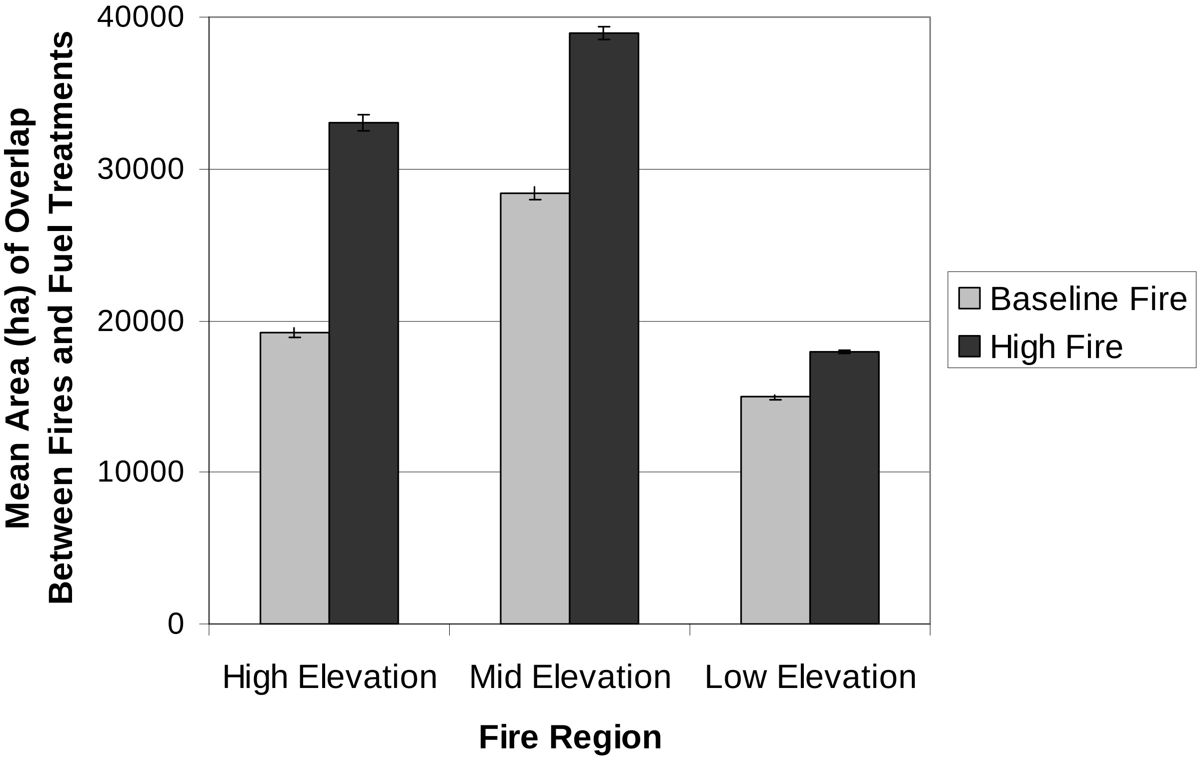

Figure 3.18. The mean area (in hectares) of active fuel treatments (within 15 years of the original treatment) that are burned for three fire regions and two fire regimes. Error bars represent standard errors.

Under the baseline fire regime, treatment intensity significantly and positively affected FRP (p = 0.015), but treatment area had no significant effect in the mid elevation region. In the high elevation region, however, both treatment area and treatment intensity were significant positive predictors of FRP (p = 0.002, p = 0.038; area and intensity respectively).

Under the high fire regime, treatment rate was a significant positive factor (p = 0.008) while treatment intensity was not in the mid elevation region. As in the baseline fire regime, both area and intensity significantly increased the FRP in the high elevation region (p = 0.030, p = 0.014; area and intensity respectively). Although we found rate and intensity to be significant predictors of fire rotation period under different fire regimes at mid elevations, we hesitate to attribute too much importance to these differences.

3.3 Discussion

There has been uncertainty in the literature about the efficacy of fuel treatments in reducing the size and severity of fires on the landscape, particularly in light of the potential negative ecological effects of fuel treatments (such as increasing invasive species or removing biomass that would impact wildlife, particularly the fisher). However, given the risk of uncharacteristically large patches of high-severity fire that expected to result from decades of unnatural fuel accumulation, climate change, and other factors, it is generally believed that, if treatments can effectively reduce the risk of unnaturally large, severe fires, they may provide benefits that offset their localized risks.

The objective of fuel treatments and prescribed fire is not to exclude fire from the landscape (which would further the paradigm of fire suppression), but to provide a means of slowing the spread of fires and reducing their intensity by restoring lower fuel amounts to the landscape (Agee and Skinner 2005). Fuel treatments also provide access points for firefighters to safely control fires. Therefore, although we used fire rotation period (FRP) as one of our primary metrics to evaluate fuel treatment efficacy, it is important to realize that FRP is just one facet of a fire regime; and the fire regime represented in both our baseline and high fire treatments represent conditions that are considered outside the natural range of variability for the Sierra Nevada (which would have experienced more frequent, but lower-intensity fires on a landscape with lower amounts and different distributions of fuel than current conditions).

Overall, our results showed that fuel treatments may effectively reduce the extent and severity of fire across a large, heterogeneous landscape during a 50-year time span. One of the main objectives for implementing fuel treatments is to reduce the incidence of large, high-severity fires. While our simulation results showed a slight reduction in fire severity with fuel treatment, it is likely that a significant broad-scale effect would only become apparent over a longer time than our 50-year simulation period. This time lag reflects that fact that the model was parameterized on recent conditions, which reflect high fuel accumulation due to decades of fire suppression. Therefore, any fire that occurs on the landscape that does not encounter a fuel treatment will likely burn at a relatively high severity. Although fires in areas that have been treated or have recently burned are expected to be less severe due to reduced understory and ladder fuels, it would take many decades for enough of the landscape to be affected by treatments and fires to substantially reduce the regional fuel conditions and thereby decrease average fire severity at the regional scale. In other words, the local effects of treatments and previous fires on fire severity may be lost in the landscape-scale average. Similarly, Scheller et al. (2005) found that fire severity did not substantially decline until after 40 years of simulated fire re-introduction in the Boundary Waters Canoe Area of Minnesota.

Both the area of treatment and the intensity of treatment were effective at reducing fire on the landscape. The effectiveness of fuel treatments, however, was greater at higher elevations and under the high fire regime. Although the chaparral and oak fuel types in the low elevation had fast fire spread rates, there were few treatments in those areas, and thus, we did not consider them in this particular analysis. Under the high fire regime, greater fuel treatment effectiveness likely reflected the greater probability of fires actually encountering fuel treatments (Figure 3.18). Therefore, at least in the forests represented in our simulations, and using the fuel treatment rates and intensities that we evaluated, fuel treatment may be more beneficial if weather conditions continue to become more severe. Although this was not the case in the Hayman fire, the low effectiveness under extreme weather may have been related to the small size of the treatments (Schoennagel et al. 2004).

The differences in treatment effectiveness between area and intensity in the mid elevation region, however, suggest that treatment is not universally effective under all conditions. The greatest area of intersection between treatments and wildfires was in the mid elevation for the high fire regime, which may be why treatment rate had a significant effect under the high fire regime but not under the baseline fire regime. Unlike the baseline fire regime, however, intensity was not important in the mid elevation. This could be because, with so much fire, there was already a much lower amount of total aboveground biomass on the landscape. Therefore, the differences in treatment intensity may not have been as big as those in the baseline fire regime.

One of the concerns over fuel treatment efficacy is that, for treatments to be effective, they must overlap with fires (which occur stochastically across space) (Rhodes and Baker 2008). Treating more area (i.e., 8% vs. 4% per 5-year time step) increases the probability that a spreading fire will encounter a treatment location. Treating only 2% of the potentially treatable landscape had no discernable effect on fire rotation period when this was evaluated on a subset of the study area. Had we performed simulations using the 2% treatment rate for the full study area, the strength of the statistical relationship between treatment rate and fire rotation period would have been even stronger. We cannot extrapolate beyond the 8% treatment rate to conclude that treatment effectiveness would continuously rise with treatment rate; however, the rate of 4-8% mirrors the SNFPA objectives of getting 20-30% of the landscape in an effective fuel condition in fire prone areas in the next 20 years.

4.0 Effects of Fire and Fuels Management on Fishers

Forest composition and structure are extremely dynamic in the southern Sierra Nevada due to interactions among fires, forest management, and a wide variety of other disturbances (e.g.,. insects, drought, disease, windthrow, climate change) as well as different pathways of succession following such disturbances. In addition to the immediate effects of these multiple factors on forest composition and structure, these effects are further influenced by temporal dynamics and cumulative impacts. In particular, the likelihood and nature of new disturbances occurring in an area will vary in response to the effects of previous disturbances. For example, one fire burning through an area can affect the behavior of future fires encountering the same area, thus potentially affecting future fires' spread rates, burn severity, and post-fire succession patterns. These dynamic, probabilistic, and interacting forces must be considered to make inferences about how, for example, fuels management and fires may cumulatively affect fisher habitat or populations over time. Inferences based on static habitat maps would be highly uncertain and inappropriate for estimating long-term changes in fisher habitat and populations.

To account for these interacting forces, we coupled the landscape dynamics model LANDIS-II (Section 3) with our landscape-level fisher habitat model (LAND8, Section 2), which we also coupled with the population dynamics model (PATCH, Section 2). This allowed us to explore how vegetation dynamics (due to fires, fuels management, and vegetation succession) may affect the distribution, abundance, and viability of fishers in the southern Sierra Nevada over the next 50 years.

This section presents the results of the simulations that show how the interacting forces of fuel treatments and wildfires may affect fishers across a range of potential conditions. We can visualize these interactions as three direct effects and one indirect effect (dashed line) that together influence fishers (Figure 4.1).

These four interactions can also be expressed as four hypotheses or expected interactions depicted on Figure 4.1:

H1 *Fuel treatments may reduce fire on the landscape.* This interaction was addressed previously in our discussion of the LANDIS-II modeling (Section 3.0), in which we found that simulated fuel treatments reduced the extent and severity of fires under select circumstances, and that both the rate and the intensity of treatment affected area burned.

H2 *Wildfire may directly and negatively affect fishers.* All wildfires reduce aboveground biomass (albeit with differential effects for different age-cohorts or fuel types). Because our landscape-level fisher habitat model depends upon total biomass as a predictor of habitat quality, fires will always result in some reduction in the simulated quality of fisher habitat. While the amount of biomass removed should be a function of area burned, fire severity is also important in determining the magnitude of this effect, for example, by determining whether biomass removal occurs primarily within younger, understory age cohorts or if fires are also consuming older, overstory trees.

H3 *Fuel treatments may have immediate, direct, negative effects on fishers.* Similar to H2, because all fuel treatments reduce aboveground live biomass, they will always result in some reduction in simulated habitat quality; however, the magnitude of this reduction in biomass and how it is distributed across different age cohorts is also important to determining effects on fishers as well as effects on future fire behavior and vegetation succession.

H4 F*uel treatments may have an indirect positive effect on fishers.* Because fuel treatments are intended to reduce fire spread ,and fire severity, and thereby reduce threats of huge, intense, or stand-replacing fires, they may also prevent loss and fragmentation of fisher habitat.

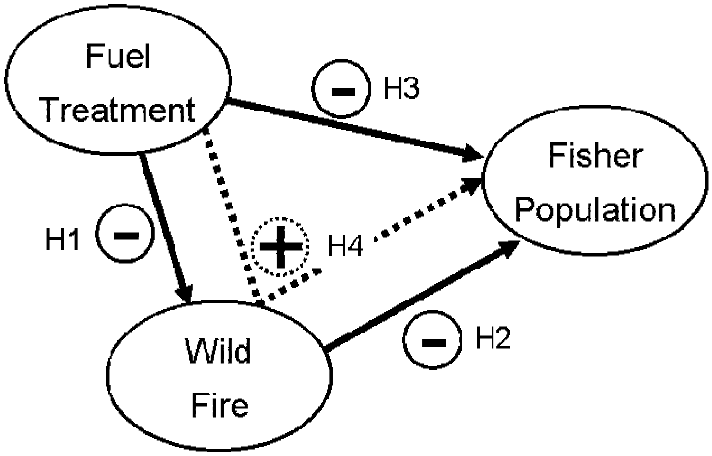

Figure 4.1. Expected interactions among fuel treatments, wildfire, and fisher populations in the southern Sierra Nevada. Solid lines represent direct effects; the dashed line represents an indirect effect.

Whether the indirect positive effect of fuel treatments on fishers (H4) outweighs the expected direct negative effect of fuel treatment (H3) is a critical question for determining whether and where fuel treatments should be deployed. The analyses in this section focus on understanding this potential trade-off between the positive and negative effects of fuel treatments on fishers and on how this balance is affected by changes in the fire regime and the area, intensity, and location of fuel treatments.

In addition to testing the effects of fuel treatments on fishers, we also assessed other variables that have been identified by fisher experts as potentially important to fisher habitat (R. Truex and W. Zielinski, personal communication). These included the amount of area occupied by two age classes of white fir (*Abies concolor*), Ponderosa pine (*Pinus ponderosa*), Douglas fir (*Pseudotsuga menziesii),* and black oak *(Quercus kelloggii).* We also evaluated the maximum age of forests, the median age of forests, and the standard deviation of the age of forests. Older age cohorts of these four species (white fire, Ponderosa pine, Douglas fir, black oak) are considered to provide potential resting and denning structures for fishers (Zielinski et al. 2004a, 2006). The maximum and median age of forests are also considered to be positively correlated with higher quality fisher habitat – older forests (as measured either by the maximum or median age) contain larger trees that provide critical resting habitat. Finally, the standard deviation of all the cohort ages at each site is indicative of whether a site is multi-aged and therefore has more structure, which should also be favored by fishers (W. Zielinski personal communications). A higher standard deviation is expected to in better fisher habitat.

Although all of these additional variables will be sensitive to wildfire and fuels management, none of them was retained in our fisher habitat model, LAND8, and none was therefore used to predict fisher population size. In other words, these variables were only assessed for broad changes over time that may indicate that important fisher habitat elements could increase or decrease under different simulation assumptions. *These are proposed indicators only and were not tested for their ability to predict fisher habitat.*

4.1 Methods

4.1.1 Coupling Fisher Population with LANDIS-II

As detailed in Section 3.0, we used the LANDIS-II estimates of total aboveground live biomass produced every 10 years to generate predicted probability of occurrence maps via the LAND8 probability of occurrence (or habitat value) model. For each combination of treatments and fire, and for each replicate, a new predicted probability of occurrence map was produced every decade by replacing the previous biomass map with the new biomass map and recalculating the habitat value model.

For each combination of fire and treatment factors we tested (14 total combinations), we ran 10 replicates using LANDIS-II for 50 years (a total of 140 LANDIS-II runs). After each simulation decade, the total above-ground live-tree biomass outputs from LANDIS-II were exported into LAND8 to change predicted habitat value in response to vegetation dynamics, for a total of 840 simulated habitat maps (including the year 0 base map for each run). These decadal habitat maps, in turn, served as inputs to PATCH, which assigned appropriate demographic parameter values based on the spatially explicit predicted habitat values. Because, like LANDIS-II, PATCH has stochastic components, we ran 10 PATCH replicates for each LANDIS-II replicate to assess variation in the predictions about population change, for a total of 1,400 population simulation runs.

For each run, PATCH simulations were initialized with the mean equilibrium number of adult females (135) that resulted using baseline habitat conditions (see Section 2.2.3) and using the initial predicted probability of occurrence maps. PATCH was first run for 10 years to allow the 4 fisher age classes to equilibrate (see Section 2.1.4). Results of this PATCH equilibration phase were discarded (i.e., PATCH year 10 was treated as simulation year 0 to avoid artifacts of the initial stage-class distribution from influencing results). The population dynamics were then run for the next 60 years, substituting in a new habitat map (reflecting biomass changes from LANDIS-II) every 10 years until year 50. Thus, population simulations extended 10 years beyond the last habitat change (to simulation year 60). We did not run PATCH any further into the future because simulations would not then be accounting for continuing changes in vegetation a due to disturbances, growth, and succession.

Adult female fisher population size at the end of the PATCH simulations (year 60) were used as our final estimate of fisher populations for statistical analyses of fire and treatment effects (Section 4.1.3). Note that the absolute accuracy of these estimates is not critical, because their utility is in comparing relative effects among different sets of factors.

Figure 4.2 shows one example of 10 PATCH replicates conducted for one LANDIS-II replicate (using the High Fire regime with no fuel treatments). Notice that (1) for all 10 replicates, the fisher population gradually increased over the baseline size (presumably due to increasing habitat value with vegetation succession), and (2) a large fire (or fires) reduced fisher habitat at year 40, causing a substantial reduction in the simulated fisher population in all replicates. Note that due to stochastic events the other 9 LANDIS-II replicates, for this same set of factors, would all be different, with associated differences in the average population response demonstrated by PATCH.

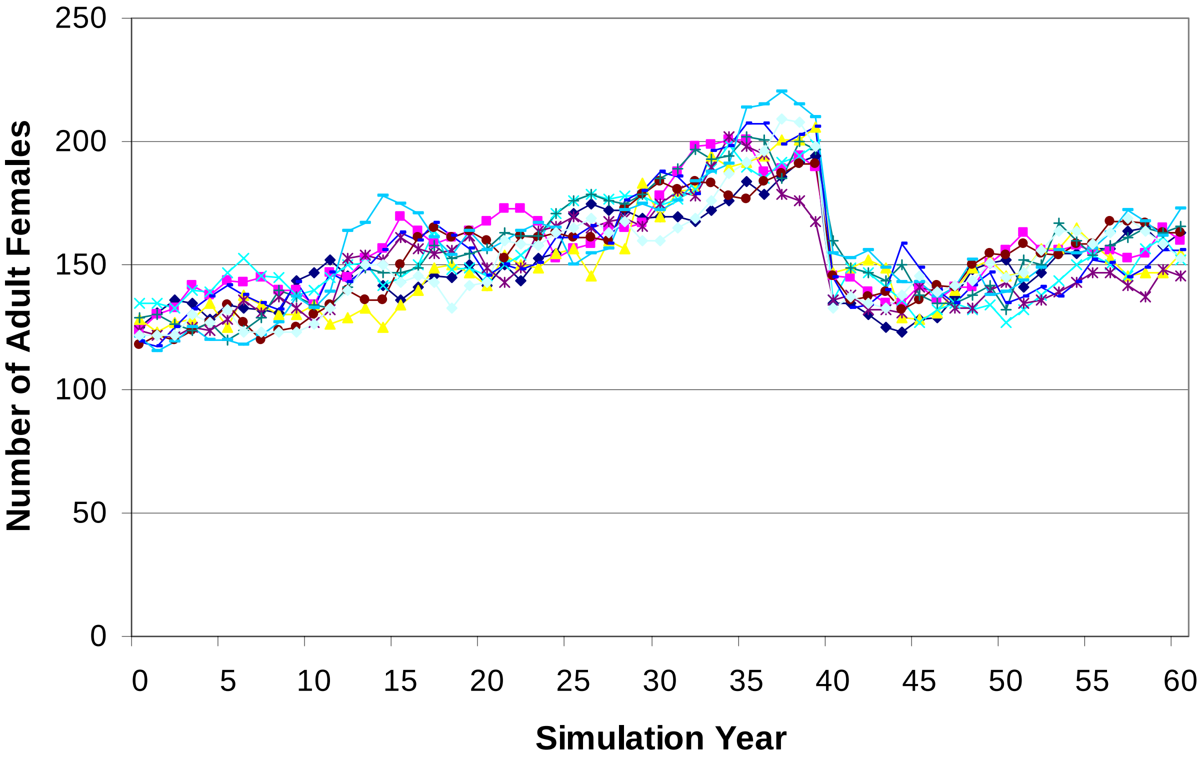

Figure 4.2. Example fisher population dynamics from 10 PATCH replicates run for one LANDIS-II replicate (replicate 7 from the High Fire regime with no fuel treatments). LANDIS-II aboveground biomass maps are used to generate habitat probability maps, via the GAM model LAND8, every 10 years. These maps are input into PATCH at 10 year increments, excluding the PATCH initialization period (not shown). In this particular example, all replicate populations steadily increased for about 40 years (presumably due to vegetation growth) followed by a drop after major fire(s) in simulation year 40**.**

4.1.2 Experimental Design

To evaluate the relationships among fire, fuel treatments, and fisher populations, we developed a factorial experimental design that allowed us to systematically explore the effects of fuel treatment rate, fuel treatment intensity, fuel treatment location, and fire regime on fisher populations, individually and in combination. This systematic, factorial approach (with 10 replicates for each combination of factors) allowed us to use standard statistical tests to determine the absolute and relative strength of the relationships among these potential determinants of fisher populations, as well as their interactions. This approach was not constrained to testing predefined fuels management plans or by considering economic factors. Instead, the objective of this systematic, factorial approach was to structure understanding of how the complex interactions between the various fuel treatment factors and fires may cumulatively affect fisher habitat and fire behavior. Armed with this understanding, we can work toward management approaches that best reduce the potential for uncharacteristically large patches of severe wildfire and preserve fisher habitat.

We separated our experimental design into two systematic analyses. By doing so we reduced the number of combinations simulated and thereby were able to simulate a larger number of replicates per combination (10). In the first analysis, we simulated and analyzed all combinations of fire regime (2 categories); fuel treatment rate (2 categories), and fuel treatment intensity (2 categories). In addition, we simulated fire without treatment to serve as a baseline for comparison with the treatment combinations. This first analysis allowed us to examine the relative effects and interactions among fire regime, treatment rate, and treatment intensity. For the second analysis, our objective was to estimate whether treatment placement could potentially reduce wildfire while treating a smaller total area of forest. For this analysis, we simulated all combinations of fire regime and fuel treatment location (3 categories), while simulating the highest treatment rate and the highest treatment intensity, again with 10 replicates.

4.1.3 Statistical analyses

For our systematic factorial analysis, we evaluated the absolute and relative strength of the relationships between fire, fuels treatments, and fisher populations using several types of statistical analyses. Multiple regression and Analysis of Variance (ANOVA) conducted with the R statistical package were used to establish correlations among these factors and with fisher population size. Because our simulations were designed as landscape-scale experiments, we were able to assign causation to the trends and correlations revealed. We also used Structural Equation Modeling (SEM) to test hypotheses about factors that cause the simulated fisher population to increase or decrease. SEM is derived from regression and ANOVA but is specifically designed to test hypotheses about causation and to test for indirect interactions (e.g., A changes B, and B in turn changes C, therefore A has an indirect effect on C). SEM is therefore regarded as a more robust tool for testing causation than are correlation analyses. SEM was conducted using the Stata statistical package.

Dependent variables. To test the direct effects of biomass reduction and fire rotation period, and the indirect effect of fuel treatment on fisher populations, we used the predicted number of adult females at simulation year 60 as our dependent variable.

Independent variables. Although we used fire rotation periods for the different fire regions as dependent variables (to evaluate fuel treatment efficacy), we also used them as independent variables to evaluate how fire affected fisher populations. We did not include the WUI fire regions in this calculation. Although WUI is an important source of ignitions, area burned was highly stochastic, and due to the small size of the WUI, results there are not representative of the broader landscape and could be misleading at the regional level.

As a way of analyzing the overall effects of fuel treatment on fire (directly) and fisher populations (directly and indirectly), we used total biomass removed as an independent variable. Due to the relationship between biomass and fisher habitat, we tested the effect of biomass removed on fisher populations; and we also evaluated its effect on fire rotation period to answer questions about treatment efficacy. Aboveground biomass removed is an integrating variable that combines the effects of treatment area and treatment intensity.

In addition to biomass removed via fuel treatments, we also systematically evaluated the effect of treatment location (three categories) on fisher population size. Recall, however, that we only evaluated the effect of treatment location using the highest treatment rate (8%/5 yrs) and the highest treatment intensity (Medium Thin). For all analyses, we evaluated the effects of the independent variables under both fire regimes: Baseline and High Fire.

4.2 Results

4.2.1 Interactions Among Fuel Treatments, Fires, and Fishers

As expected, fuel treatment rate and treatment intensity were positively correlated with the amount of biomass removed by fuel treatments, under both the baseline fire and high fire regimes (Table 4.1). In both cases, fuel treatment rate was a larger determinant of biomass removed via fuel treatments than fuel treatment intensity (Table 4.1).

Table 4.1. Coefficients from total biomass removed by treatment regressed on fuel treatment rate and fuel treatment intensity. The multiple regression was drawn from the scenarios with high/medium fuel treatment rate, and medium and low fuel treatment intensity. Each scenario had a sample size of 10 for a total of 40 observations per fire regime. The coefficients indicate the relative strength and direction of the effect.

	Fuel Treatment Rate Coefficient	Fuel Treatment Intensity Coefficient	Adjusted R^2
Baseline Fire	466	169	0.99
High Fire	466	165	0.99

Based on simulations conducted over 1/3 of the total study area, we confirmed that, *without fire*, fuel treatments would directly reduce potential fisher population size due to the immediate reduction in habitat quality (Figure 4.3). The results of this unrealistic analysis (because fire cannot and should not be totally removed from the landscape) showed a significant negative linear effect of biomass reduction on fisher populations. This negative correlation between biomass reduction by treatments and fisher populations serves as a default expectation (a null hypothesis) against which we can compare the same relationship when fire is simulated. Again, *this result should not be used out of context to imply that fuels treatments are unilaterally bad for fishers, because it unrealistically assumes that no fires will occur and does not balance this direct negative effect with potential indirect positive effects.*

When fires were simulated along with the fuel treatments, the direct negative effect of fuel treatment on fishers, as measured by total biomass removed, was offset by the positive indirect effects of treatments on fishers. In other words, the reduction of fire due to treatments compensated for the direct negative effects of treatments, such that *there was no significant effect of fuels treatments on fishers under the baseline fire regime considering the entire landscape as a whole* (Figure 4.4). Moreover, *under the high fire regime, there was a positive trend toward increased fisher population size with increasing biomass removed by treatment* (Figure 4.5), although this correlation was not statistically significant (p = 0.16). Both rate and intensity of treatment were positively correlated with biomass removed (Table 4.1), but the differences between them are not statistically significant with regards to fisher populations. Figures 4.4 and 4.5 represent the *total* effect (combining the direct negative and indirect positive effects) of fuel treatments on fisher, again at the whole-landscape scale.

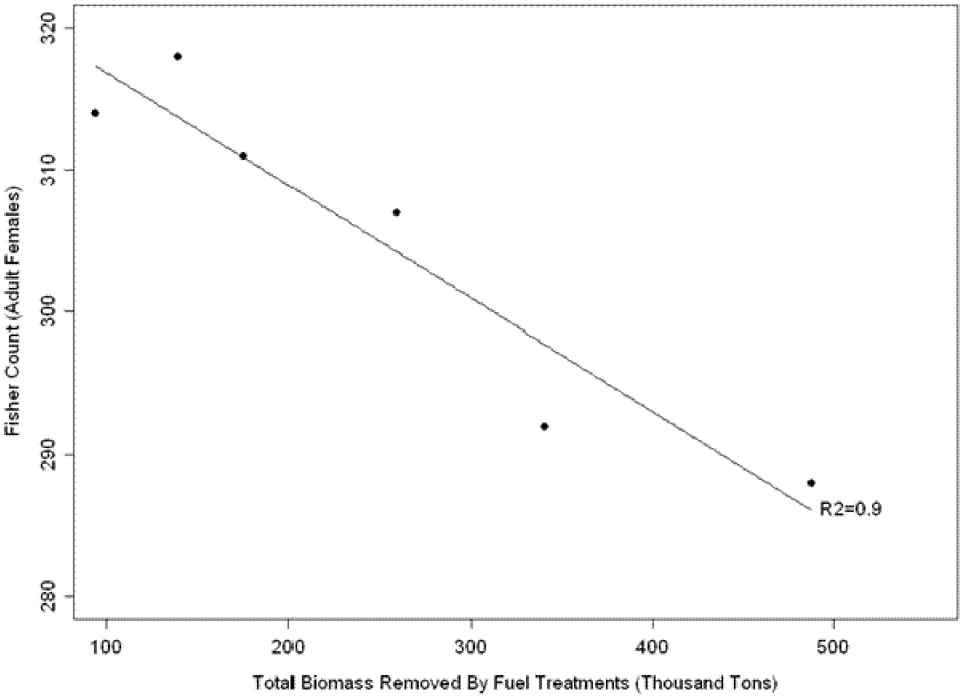

Figure 4.3. Estimated adult female fisher population size regressed on total biomass removed for a subset of the total landscape assuming no fires would occur over 50 years.

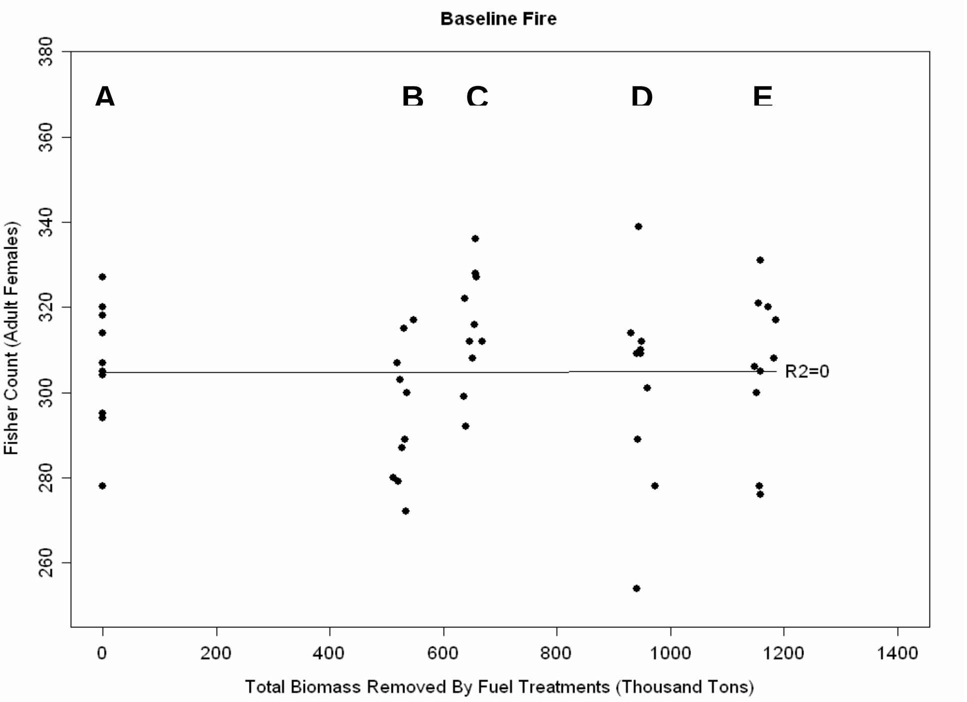

Figure 4.4. Estimated adult female fisher population size regressed against total biomass removed via fuel treatments for all replicates excluding the location tests for the simulated baseline fire regime. (A= no treatment, B= 4% rate at low intensity, C= 4% rate at medium intensity, D= 8% at low intensity, E = 8% at medium intensity).

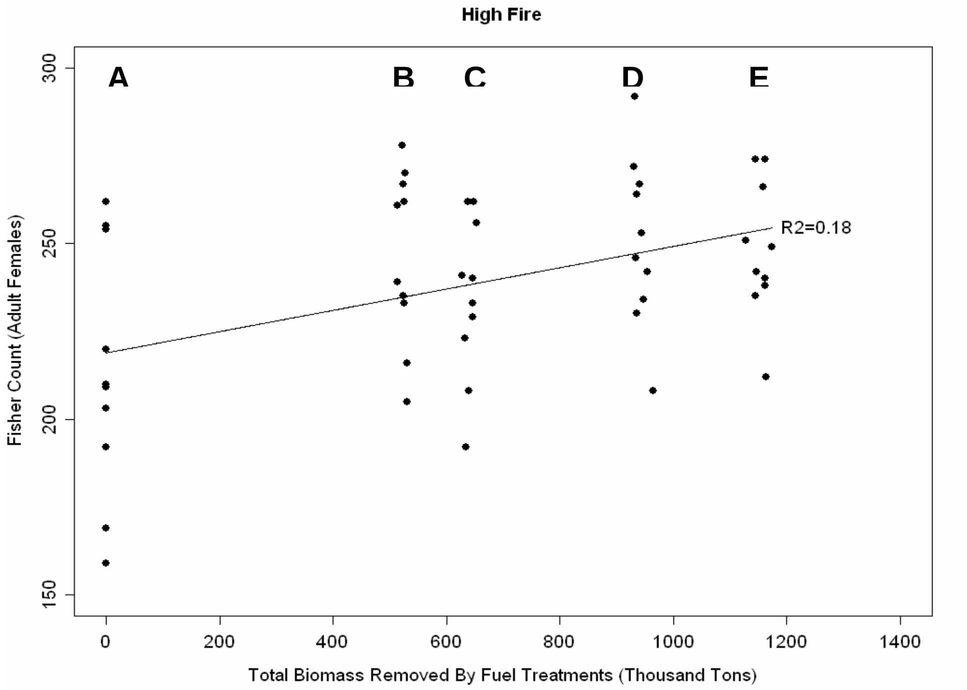

Figure 4.5. Estimated adult female fisher population size regressed against total biomass removed via fuel treatments for all replicates excluding the location tests for the simulated high fire regime. (A= no treatment, B= 4% rate at low intensity, C= 4% rate at medium intensity, D= 8% at low intensity, E = 8% at medium intensity).

We next evaluated the separate, independent influences of the positive indirect effect and the negative direct effects of fuel treatments on fisher populations using structural equation modeling and stratifying the analysis into mid elevation and high elevation fire regions. None of our structural equation models included the "no treatment" scenarios as this created an unbalanced dataset.

The positive indirect effect of fuel treatment on fisher was statistically significant in each case, except for the mid-elevation region under the baseline fire regime (Table 4.2; Figure 4.6). The negative direct effects of treatments were never significant (Table 4.2), and the relative strength of the negative direct effects of fuel treatments on fishers was always smaller than the indirect positive effects of limiting fire on the landscape (Figure 4.6). These results are consistent with our previous conclusions that fuel treatments are more effective at reducing the extent and severity of fire at higher elevations (or more generally, areas with faster fuels) and are more effective under the high fire regime, due to the greater probability of fires encountering treatments (Figure 4.7).

Table 4.2. Coefficients (and standard errors) from a structural equation model that measured the negative direct and positive indirect effects of fuel treatments on fisher population size as a function of the fire region and fire regime. $^*p < 0.01$, $^{**}p < 0.005$.

	Positive Indirect Effect	Negative Direct Effect
Baseline Fire – Mid Elevation	0.034 (0.025)	-0.017 (0.027)
Baseline Fire – High Elevation	0.084 (0.032)*	-0.061 (0.045)
High Fire – Mid Elevation	0.085 (0.033)*	-0.053 (0.033)
High Fire – High Elevation	0.132 (0.043)**	-0.077 (0.050)

The structural equation model also allowed us to compare the circumstances (i.e., fire regime and fire region) providing the best trade-off between the positive and negative effects of fuel treatment on fisher. Under the baseline fire regime, the negative direct effect was proportionately greatest at high elevations (only 26% smaller than the indirect effect). The negative direct effect was only half as large (54%) as the positive indirect effect at mid elevation under the baseline fire regime. Although it is tempting to conclude that the trade-off favors treating the mid elevations in the baseline fire regime, the results were not statistically significant in the mid elevation fire region and should thus be interpreted with caution. Under the high fire regime, the differences between mid and high fire regions vanished, and the negative direct effects were roughly 60% as large as the indirect effects at both elevations. Therefore, for this analysis we cannot make any strong conclusion about which circumstances (baseline or high fire, mid or high elevations) provide the best trade-off in regards to treatment effects on fisher.

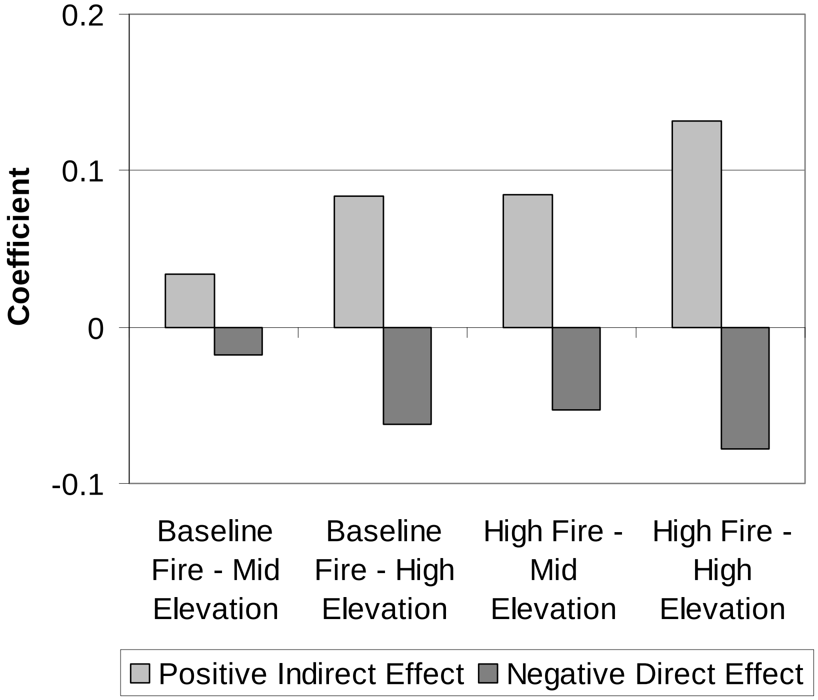

Figure 4.6. Coefficients from a structural equation model illustrating the relative strength of the direct and indirect effects of simulated fuel treatments on estimated fisher population sizes for baseline and high fire regimes at mid and high elevations. The model used pooled results from management combinations using the moderate and high treatment rates and the low and medium treatment intensities (a total of 40 replicates per fire regime). The coefficients indicate the relative strength and direction of the effect.

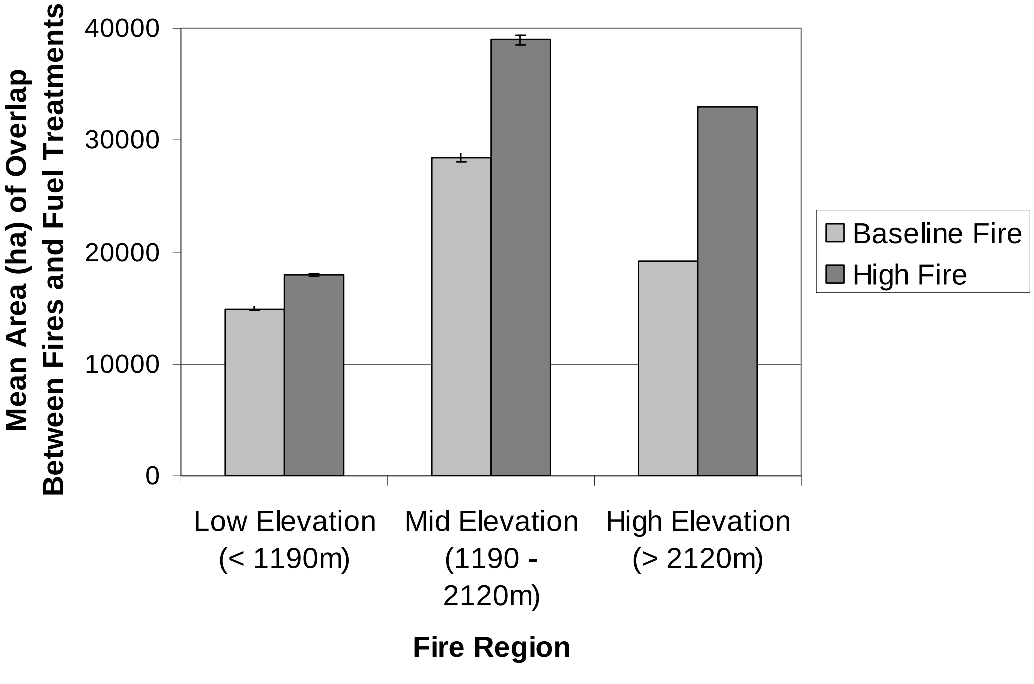

Figure 4.7. The mean area (in hectares) of active fuel treatments (within 15 years of the original treatment) that are burned for three fire regions and two fire regimes. Error bars represent standard errors.

4.2.2 Effect of Treatment Location

Our initial analysis of fisher populations as a function of fuel treatment location (Figure 4.8) appeared to demonstrate little effect of location. When we examined the amount of total biomass removed for the six treatment combinations (2 fire regimes x 3 fuel treatment locations) (Figure 4.9) there was less biomass removed when treatments were performed exclusively outside the highest probability fisher habitat than there was for the treatments inside fisher habitat or when there were treatments in both locations. The reason there was less biomass removed in the treatments outside of fisher habitat is that the best fisher habitat has higher levels of biomass to begin with.

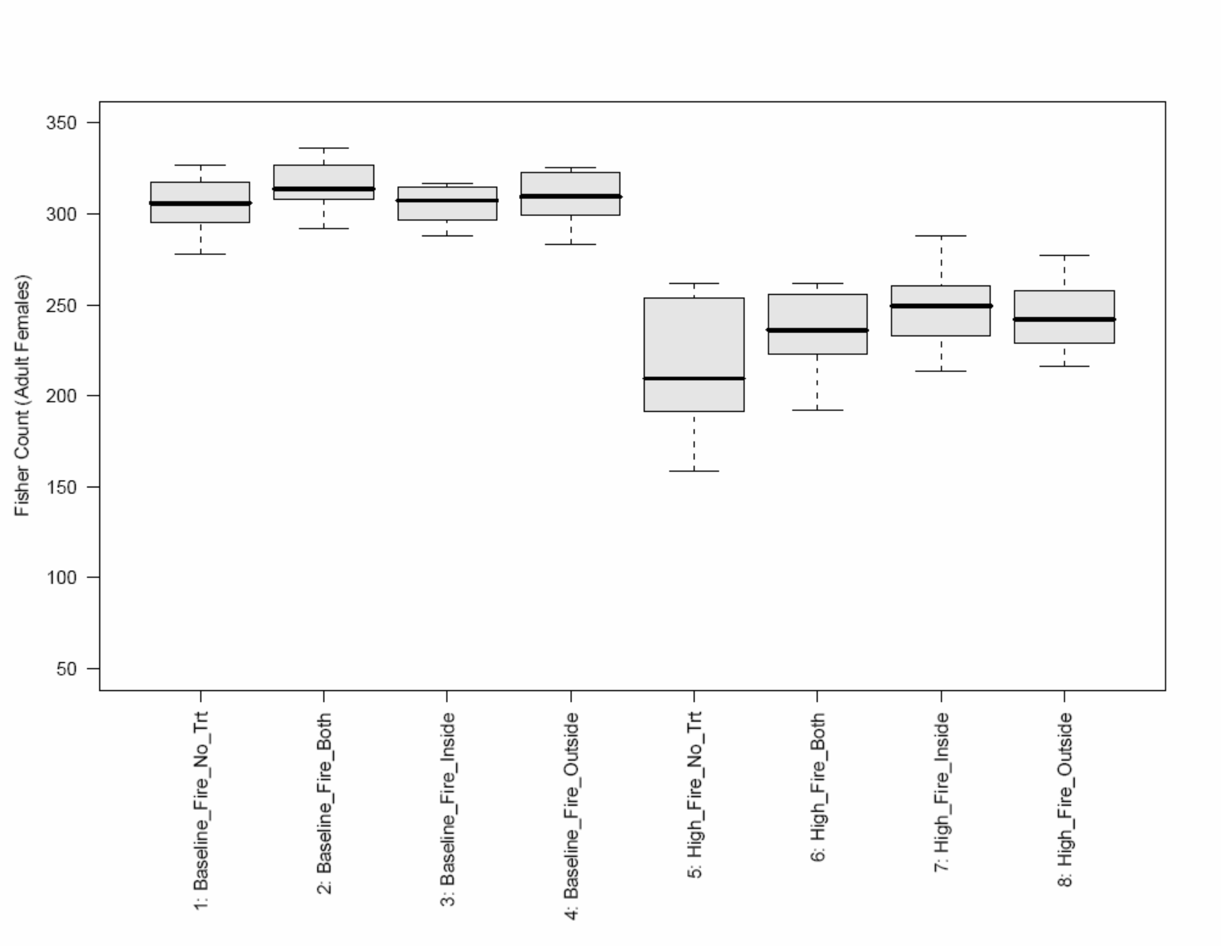

Figure 4.8. Estimated adult female population without fuel treatments (No_Trt) and under 3 fuel treatment locations and 2 fire regimes (Baseline and High) (N=10 replicates per combination). *Both* indicates fuels treated throughout the treatable area; *Inside* indicates treatments only inside fisher habitat (> 0.33 mean probability cutpoint defined across all replicates with no treatments at year 50); *Outside* indicates treatments occurred only outside fisher habitat (< 0.33 cutpoint). To balance the total area treated, values for *Both* used the medium treatment rate (4%/5 yrs) and the values for *Inside* and *Outside* used the maximum treatment rate (8%/5 yrs).

Based on this initial analysis, we could not conclude that fuel treatment location does or does not have a strong effect. However, the reason that these treatment combinations failed to demonstrate an effect of treatment location was that we used a static map of fisher habitat: the map used to define fisher habitat was based on the average of 20 maps at the end of the PATCH simulations and represented the best habitat *on average*. However, fisher habitat is spatially dynamic and the LANDIS-II harvest extension cannot accommodate these dynamic changes. As a result, the efficacy of our intended design became highly variable: treatments that were meant to be outside fisher habitat may have been inside and vice versa. In addition, this analysis also suffered from a low sample size that prevented analysis via structural equation modeling.

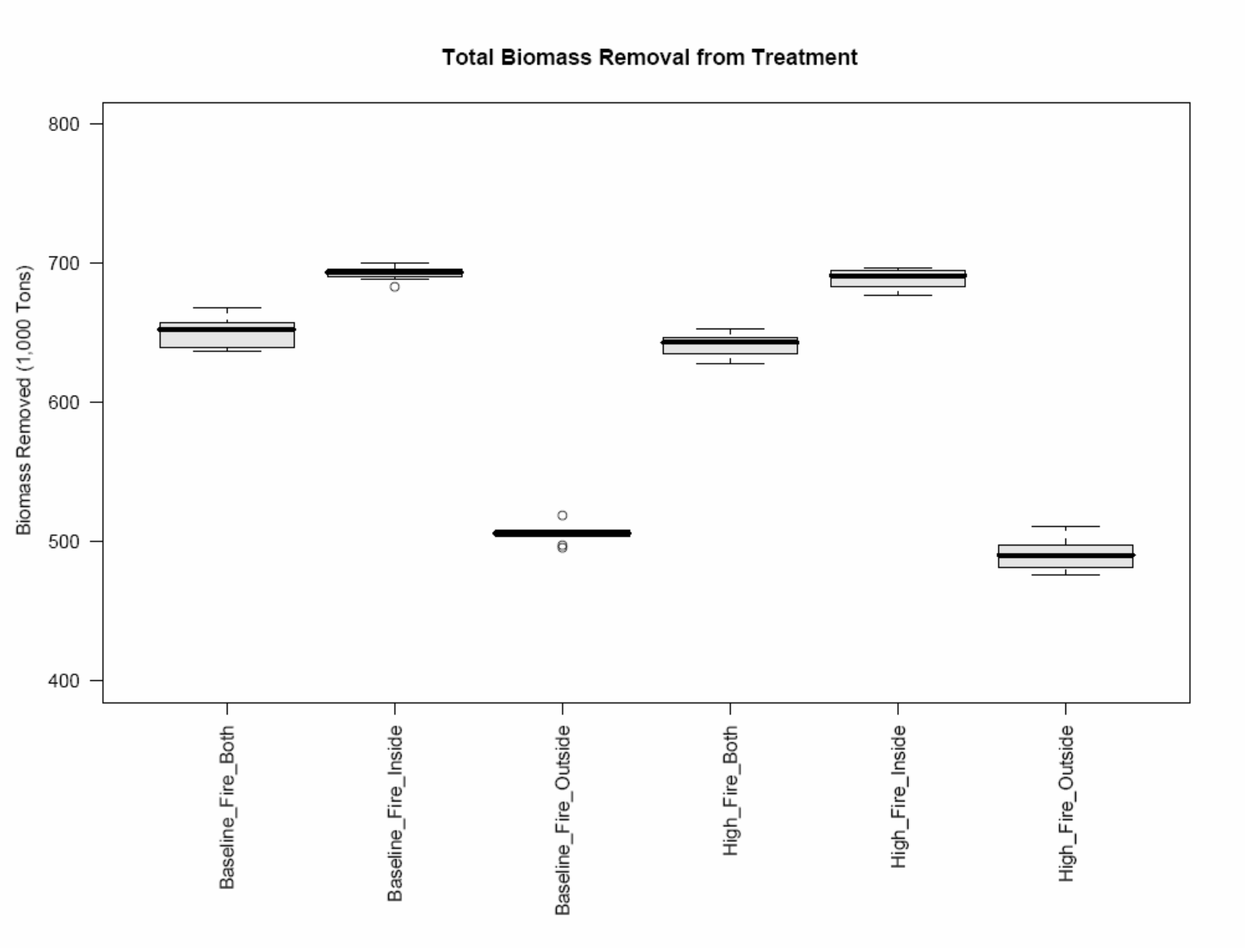

Figure 4.9. Biomass removed by simulated fuel treatments under 3 fuel treatment locations and 2 fire regimes (Baseline and High) (N=10 replicate per combination). *Both* indicates fuels treated throughout the treatable area; *Inside* indicates treatments only inside fisher habitat (> 0.33 mean probability cutpoint defined across all replicates with no treatments at year 50); *Outside* indicates treatments occurred only outside fisher habitat (< 0.33 cutpoint). To balance the total area treated, values for *Both* used the medium treatment rate (4%/5 yrs) and the values for *Inside* and *Outside* used the maximum treatment rate (8%/5 yrs).

Because these results were inconclusive, we did a retrospective analysis of treatment location based on the initial 80 simulations (2 fire regimes x 2 treatment rates x 2 treatment intensities x

10 replicates). For this analysis, we calculated the amount of above-ground live biomass removed by treatment based on the dynamic fisher habitat maps estimated by LAND8 at 10-year time steps (using the 0.5 fisher occurrence cutpoint to define inside vs. outside fisher habitat). Therefore, our assignment of a treatment to either inside or outside fisher habitat changed over simulation time. We also calculated FRP for each of the 80 simulations based on whether fires were inside or outside fisher habitat.

Next, we used structural equation modeling to determine whether fuel treatments were most effective and/or most detrimental based on their location as determined by the retrospective analysis and on the fire regime (Table 4.1 and Figure 4.10). Under the baseline fire regime and outside of fisher habitat, neither the direct effect of fuel treatments on fisher (due to biomass reduction) nor the indirect effects (due to wildfire reduction) were significant (Table 4.3). However, under the baseline fire regime and treatments inside fisher habitat, both effects were significant, and the positive indirect effect was larger than the negative direct effect. Consequently, *this analysis demonstrates that under the baseline fire regime, the simulated treatments inside fisher habitat provided a net benefit to fishers, because their effects on reducing fire damage to fisher habitat were greater than the direct negative effects of the treatments themselves on fisher habitat.*

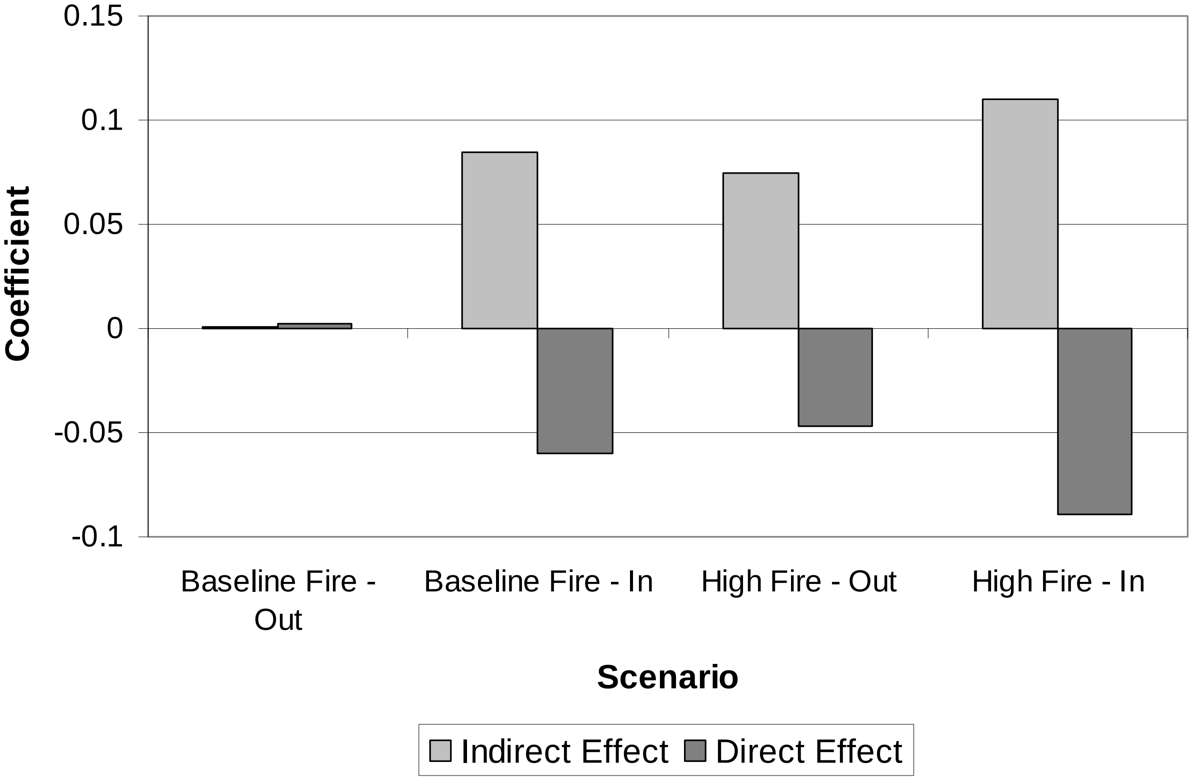

Figure 4.10. Coefficients from a structural equation model of the positive indirect effects and the negative direct effects of simulated fuel treatments on estimated fisher population sizes for baseline and high fire regimes, pooling mid and high elevation fire regions. The coefficients indicate the relative strength and direction of the effect. The structural equation model was drawn from treatment combinations using high and medium fuel treatment rates and medium and

low fuel treatment intensities (10 replicates each for 4 combination of treatments and fires = 40 total replicates per fire regime). Fuel treatments were assigned to being either inside or outside of fisher habitat based on probabilities of fisher occupancy (using the 0.5 probability cutpoint) estimated at 10 year time steps.

Under the high fire regime, fuel treatments outside fisher habitat had a significant indirect positive effect while the direct negative effect of treatments was marginal (p = 0.18). The indirect effect outside fisher habitat is due to larger fires under the high fire regime, which would have more readily spread from outside fisher habitat into fisher habitat. If there was any direct effect, it was because fishers could occupy habitat outside of the 0.5 probability of occupancy area, albeit at a reduced probability. Under high fire and within fisher habitat, both the direct and indirect effects were significant, and again, the indirect positive effects on fisher were larger. Therefore, *fuel treatments within fisher habitat are slightly more effective under the high fire regime than the baseline regime.* However, the negative direct effect is a smaller proportion of the total effect (positive indirect + negative direct) outside of fisher habitat. Treatments outside fisher habitat may provide some protection for fisher habitat if large fires can encroach on fisher habitat from adjacent areas. From the perspective of weighing the trade-offs between the positive and negative effects of treatment, *strategic placement of treatments outside of fisher habitat appears most beneficial under the high fire regime, where we observed the higher ratio of positive to negative effects*.

Table 4.3. Coefficients (and standard errors) from a structural equation model that measured the direct and indirect effects of fuel treatments on fisher population size as a function of fuel treatment location and fire regim**e**. *p = 0.18, **p<0.01, ***p< 0.005.

	Indirect Effect	Direct Effect
Baseline Fire - Out	0.001 (0.004)	0.002 (0.028)
Baseline Fire - In	0.085 (0.026)***	-0.060 (0.023)**
High Fire - Out	0.075 (0.026)***	-0.047 (0.035)*
High Fire - In	0.110 (0.035)***	-0.089 (0.024)***

4.2.3 Changes in Old Tree Habitat Elements

Under the baseline fire regime, the area of the two age classes defined for white fir and Ponderosa pine remained fairly stable over time (Appendix G-K). Both age classes of Douglas fir declined by about a third over the course of the simulations. The area of black oak aged 50 – 100 years increased by a small amount, while the area of black oak > 100 years of age nearly doubled.

Under the high fire regime, the area occupied by white fir, Ponderosa pine, Douglas fir, and black oak remained similar to those observed under the baseline fire regime (Appendix L-P). Changes in age classes over time reflect demographic transitions and/or losses due to fire.

None of these patterns was strongly influenced either by fire regime or by treatment combinations (compare graphs in Appendices G through P). This lack of sensitivity in trends in the abundance of tree species by age class is not overly surprising given (1) the short duration of the simulations (50 years) relative to the life spans and rates of succession in these species and (2) the relatively coarse age cohorts we were able to track. However, these results may also suggest that changes in the species-age composition of Sierra Nevada forests over the next 50 years may be influenced more strongly by processes already set in place by the current condition (which reflects many decades of altered disturbance patterns due to logging, fire suppression, etc.) than by management actions from this point forward.

In conclusion, with the exception of Douglas fir, these indicators of fisher habitat generally increase or remain stable through time in all simulations. No one treatment combination or group of treatment combinations indicate any precipitous declines or dramatic increases in these indicators of potential fisher habitat. However, it is notable that under all treatment combinations the models predict that the distribution of old black oaks is likely to increase in the study area, which may provide increased rest-site and prey availability for fishers.

Our analyses of changes in landscape age structure (maximum age, median age, standard deviation of age) all indicate a modest increase for these indicators of fisher habitat quality over time (Appendix Q). These trends held true under both the baseline and high fire regimes and with little apparent influence of management. This may again suggest that, at this very coarse scale and considering these coarse age classes, successional patterns for these species are already established by the current condition and not likely to respond dramatically different to different management approaches.

4.3 Discussion

Above-ground live biomass was a useful, if coarse, currency for understanding the interactions among fuel treatments, wildfire, and fisher populations at the very broad scales we examined because it was a single metric that could be related to all of our analyses and variables of interest. There is no doubt that fisher distribution at the regional scale is highly correlated with areas of high biomass, and that both fires and fuels treatments reduce biomass. What is critical to keep in mind, however, is that the distribution of that biomass on the ground, and how it is differentially affected by treatments and fires, is also hugely important. Large patches of severe wildfires (that are most likely to occur under extreme fuel and weather conditions) can remove entire stands of trees, including large, old canopy trees that are required by fishers. The fuels treatments that we simulated were designed to remove biomass in the form of smaller trees and shrubs: surface fuels and ladder fuels in the understory. However, in relying on total biomass as a proxy variable in determining fisher habitat value, the analyses we've presented do not fully discriminate between the size and age of trees that were removed and thus do not account for this critical difference in the form of biomass removal. Consequently, we strongly suspect that our models may be over-representing the direct negative effects of fuels treatments on fishers, and perhaps under-representing both the direct negative effects of fires and the indirect positive effects of fuels treatments. Thus, although total biomass enabled us to draw conclusions about the factors controlling fisher populations at very large scales, it also limits our ability to make predictions at finer (< 100 ha) scales, where a different set variables are likely to serve as better predictors of fisher habitat (Zielinski et al.2004a, 2004b, 2006). *Management of Sierra Nevada forests must consider all scales of interest, from site-specific effects of fuel treatments on fine-scale habitat elements, to the broad, landscape-level patterns we've investigated here.*

Under select conditions, we can conclude that the longer-term positive effects of fuel treatments (due to the reduction of fire hazard) outweigh the short-term negative effects of fuel treatments (due to immediate loss of biomass). This was born out by our statistical analyses at high elevations (under both baseline and high fire regimes) and at mid elevations if the simulated fire regime was more severe. As stated above, we suspect that our analyses may be overstating the negative effects of fuels treatments on fishers and understating the beneficial effects, such that treatments in the mid-elevation region, even under the baseline fire regime, may actually also have indirect, positive effects on fishers. We hesitate to suggest that the differences in effects were entirely due to elevation. The objective of delineating fire regions into elevation bands was to capture broad-scale differences in fire regimes; there are also general differences in fuel structure and fisher habitat quality among the elevations. However, elevation is a continuously varying phenomenon, and our results cannot indicate exactly where within any one broad fire region the effects may have occurred. Also, there is substantial spatial variation and heterogeneity (in many environmental variables, including elevation) within each region. Therefore, while analyzing the results by fire regions provides a solid framework for analysis, differences between the regions may a function of multiple interacting factors.

Our statistical analyses also demonstrated that the positive indirect effects of treatments on fisher outweighed the negative direct effects when the treatments were conducted inside fisher habitat, regardless of fire regime, or were conducted outside fisher habitat under the high fire regime. The situation that provided the highest ratio of positive to negative effects of fuel treatment on fisher (the best trade-off) was treating outside of fisher habitat under the high fire regime. These results suggest that the negative effects of fire on fisher (i.e., removing biomass) are local (at least in terms of the scale of our analysis), and it will be important to consider the location of fisher habitat when determining where to place treatments. In other words, if treatments are placed too far from fisher habitat, they may only affect the fires that occur in those locations and not provide the indirect positive benefits that would occur from reducing the spread of severe fire near fishers. We suggest that if the anticipated increase in severe fire conditions is accurate, strategic placement of treatments both inside and outside of fisher habitat may be beneficial.

Again, these conclusion are predicated on our simulation assumptions and the relatively narrow range of treatment effects we modeled: If actual fuel treatments remove substantially more biomass than what we simulated (for example, removing more large trees) or if actual treatments do not reduce fire spread rates and severity as much as was modeled, then we would expect to see much more parity between the positive and negative effects of fuel treatments. Conversely, if a greater reduction in fire size and fire severity could be achieved with the removal of less biomass, the positive effects of fuel treatment would be greater than our results indicated. Strategically reducing understory biomass by proportionally targeting more of the younger cohorts (ladder fuels) and less of the older cohorts should have less adverse effect on fisher habitat, if this would sufficiently reduce fire spread rates and severities. However, we did not test this alternative, nor can we assert with any confidence that the desired reduction in fire spread rate and severity could be achieved with this approach.

Although the amount of biomass removed by fuel treatments was largely controlled by the rate of fuel treatment (proportion of treatable landscape treated per 5-year time step), this is largely because the range of rates we tested was greater than the range of intensities we tested. The medium intensity treatments removed only ~20% more biomass than the low intensity treatments, whereas the high treatment rate (8%/5 yrs.) was twice as large as the medium treatment rate (4%/5 yrs.). In addition, where slopes were > 30%, the same fuel treatments (simulated prescribed fire with 4-foot flame lengths) with identical intensities were applied across all fuel treatment combinations. If a broader range of treatment intensities were tested, we may see a greater range of effects. Nevertheless, our analysis in Section 3 indicated that both treatment rate and intensity are important in determining area burned, dependent upon the local conditions. Whether the treatment rates we simulated could actually be achieved, or whether treatments could be made more efficient (i.e., remove less biomass while achieving equivalent fire spread rates and severities) are management questions beyond the scope of our analyses.

If we measure the efficiency of a fuel treatment as the total treatment area required to achieve the same fisher population size, then the strategic placement of fuel treatments should increase the fuel treatment efficiency. Our results showed that, where fuel treatments are most effective varies by elevation, by proximity to fisher habitat, and by the assumed fire regime. In general, the most efficient strategy would to minimize biomass removed by fuel treatments while maximizing the ability of fuel treatments to reduce fire size and severity. Under the baseline fire regime, our results suggest that placing such treatments inside of fisher habitat provides the greatest benefit to fishers. It is also apparent that the benefits of fuel treatments rise if the risk of large patches of high- severity fire increases, as simulated. Under a heightened fire regime, our results suggest that strategic placement of treatments outside of fisher habitat may also be beneficial, or that a combination of treatments both inside and outside of habitat is best. It could also be the case that fires will become more frequent and/or larger in the future with no attendant increase in fire severity. However, most of our collaborators and science advisors deemed this situation to be unlikely.

Another future consideration with regards to treatment location is that, to identify the best locations for strategically placed fuels treatments, it is first necessary to determine how and why fire patterns vary across the landscape (DellaSala et al. 2004). If treatments are placed in areas where there is a greater risk of fire, they will likely be more effective (as opposed to treating areas where fires are unlikely to occur anyway). There have been a number of approaches developed for mapping fire risk and probability using biophysical and climate variables (e.g., Preisler et al. 2004). Human influence on fire (i.e.., ignitions and suppression patterns) may also be incorporated into probability maps (Syphard et al. in press). Therefore, our results support the differential effectiveness of treatment at different elevations, under different fire regions, and in relation to fisher habitat; we also suggest that future management consider how treatment efficiency could be further maximized by focusing on areas that have a relatively high probability of experience large patches of severe fire.

Finally, our analysis of potential changes in the abundance of other fisher habitat indicators (e.g., large old oaks, firs, or pines) seem to indicate that these are relatively insensitive, at this spatial resolution and over the next 50 years, to the alternative fire and management combinations we tested. The area occupied by the four important tree species, including of older cohorts, may remain roughly constant or increase over 50 years. None of these potential indicators of fisher resting structures declined precipitously under any treatment combination, and the abundance of oaks older than 100 years increased in all combination of treatments and fires. Again, however, management actions must consider all spatial scales. Site-specific analysis of fuel treatment projects should strive to retain fisher habitat elements while changing fuel profiles in a manner that reduces fire spread rates and severity.

5.0 Management Recommendations

The results of our analyses should be considered just one form of input to forest management and planning decisions, with all their assumptions and uncertainties kept firmly in mind. Management of Sierra Nevada forests must consider all scales of interest, from site-specific effects of fuel treatments on fine-scale habitat elements, to the broad, landscape-level patterns we've investigated here. While the interacting system of models we have used here provide a landscape context for management decisions concerning fuels, forests, and fishers, local, on-ground information is just as essential for evaluating the risks of habitat loss due to fires as well as how best to influence future fire outcomes and sustain forest conditions that support fishers and other at-risk species. It is critical that any strategy for managing fire risks and fisher habitat constantly ground itself by ensuring it considers both the fine scale as well as the landscape scale.

The analytical process and results described in this report also emphasize the need to carefully consider the temporal scales of management effects on fires and resources, from immediate, often short-lived, direct effects, to longer-term cumulative effects that may ripple through vegetation succession patterns and species population dynamics over many years or decades. Due to the complexity of nature and the uncertainty it brings to any forecasting exercise, it is essential that we continually adapt to new findings about fisher habitat use and biology as well as for changing landscape conditions through management actions and environmental changes.

Based on the results from our broad-scale analyses, we offer the following recommendations, which should be refined in consultation with fuels management experts and fisher biologists.

5.1 Baseline Status of Fishers and Fisher Management Goals

Maintaining Habitat Quality and Continuity – Our analysis of the current status of fisher habitat and the fisher population in the southern Sierra Nevada confirms that the population is small (160 to 360 individuals) and at risk of extirpation by stochastic events, including uncharacteristically large or severe wildfires that could fragment habitat and isolate fishers in smaller areas. *Management should strive to sustain and increase the area and continuity of fisher habitat at the landscape scale. Fuels management efforts should be prioritized in areas at highest risk of large, severe wildfire that could move through the relatively narrow band of fisher habitat or widen gaps between current habitat segments (e.g., corresponding with major river canyons).*

Maintaining Resting Site Microhabitat Value – Highly suitable resting microhabitats appear to be relatively rare even within areas of predicted suitable habitat at the coarse scale. *Management should strive to maintain and if possible increase the distribution and abundance of forest stands that provide high-value resting sites* (Zielinski et al. 2004a, 2006, Aubry and Raley 2006), especially within areas predicted to be high value at the coarse-scale. Doing so will also presumably increase availability of suitable natal and maternal denning habitat. However, in light of concerns that continuous forest canopies may be at risk of carrying large, severe crown fires, it may not be necessary to maintain dense canopies over large contiguous areas. In consultation with fisher habitat experts, *we recommend deriving and analyzing spatial rules for the size and continuity of dense forest patches that may best retain fisher resting habitat value while minimizing risks of crown fire.*

Maintaining or Improving Potential for Population Expansion – Other than potential northward expansion, there appears to be little suitable habitat south of the Merced River that is not already occupied by fishers. Increasing the amount or value of fisher habitat south of the Merced may result in modest increases in fisher population size and viability and should not be precluded by management decisions. In addition, it appears that northward expansion of the population onto the Stanislaus National Forest has strong potential to significantly increase population size and extent, and hence viability. However, there is uncertainty about the likelihood of such expansion occurring naturally, due to potential dispersal impediments (e.g., steep canyon slopes, open habitats, the Merced River, and heavily traveled roads). *We therefore recommend a focused analysis of habitat connectivity and the potential for fisher movement between habitat areas on either side of the Merced River Canyon using an appropriate cost-weighted movement permeability model developed for the fisher, such as least cost corridor analysis (Singleton et al. 2002) or circuit theory analysis (McRae and Beier 2007).*

5.2 Fuel Treatments

Fuel Treatment Rates – Our simulations suggest that treatment rates on the order of 4% to 8% of treatable area every 5 years can significantly reduce fire sizes and fire severity and thereby benefit fishers. More strategic placement of treatments than we were able to simulate may decrease the total amount of area that needs to be treated. Even though our simulations at the 2% rate were inconclusive or not very effective, reduced treatment rates may be effective at reducing fire size and severity if carefully designed and implemented by fuel managers to treat areas with high fire risk and arranged to strategically slow fire spread (Finney et al. 2006). Existing analysis processes, such as those of the Stewardship and Fireshed Assessment, should continue to be used to identify and evaluate areas of higher fire risk that overlap with high value fisher habitat to inform strategic placement of treatments.

Fuel Treatment Intensity – The moderate intensity treatment we simulated tended to decrease the spread and amount of fire on the landscape more than the light intensity treatment under certain conditions and in certain fire regimes (e.g., at mid and high elevations for the baseline regime and high elevations for the high fire regime). Moreover, the moderate intensity treatment appeared to reduce fire severity under certain conditions (e.g., across all elevations under the baseline fire regime and for mid elevations under the high fire regime). Although there was a general trend, over the range of treatment combinations we simulated, for simulated fisher populations to be positively correlated with total biomass removal, no clear thresholds can be deduced, and the relative merits of light versus moderate treatment intensities need to be evaluated on a finer scale. We recommend that fuel treatments be designed to reduce fire spread rates and severity based on site-specific analysis that also considers fisher habitat value in and near the treatment. In areas of very high fire hazard and outside of fisher habitat, a higher intensity treatment may be warranted. *Within fisher habitat, treatments should be designed to balance desired fuel conditions with maintaining sufficient overstory and habitat elements to* *sustain or encourage occupancy by fishers. Removing larger trees and other essential habitat elements should generally be avoided within fisher habitat, to the degree feasible while meeting fuel reduction and landscape vegetation management goals.* There may be benefits to fisher of removing, for example, some larger firs or cedars to stimulate growth of black oaks, which provide important habitat elements for fishers and their prey.

Fuel Treatment Location **–** Our simulation results suggest that placing treatments inside fisher habitat is not necessarily detrimental to fisher (at least for the limited range of treatment types and at the scale we simulated). The positive indirect effect of treatments in reducing fire size and severity can help protect fisher habitat value despite potential short-term, localized, negative effects on fisher. Because treatment effects on fire spread are relatively local, treatments inside landscape-level fisher habitat (areas of large tree biomass) may better protect fisher habitat than those placed outside fisher habitat (at least under the baseline fire regime). However, treatments in high biomass areas should still strive to maintain sufficient overstory canopy and avoid removing fisher habitat elements, such as large old trees that provide resting structures.

If the fire regime is truly expected to become more severe in the near future, than some treatments outside of fisher habitat may also benefit fishers by helping reduce the spread of large fires from outside into fisher habitat. Again, strategic placement and design of treatments must be done at the local scale through close collaboration of experts in fuels management and fisher biology.

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