

**Southern Sierra Nevada
Fisher Baseline Assessment**

Final Report:

**Baseline Evaluation of
Fisher Habitat and Population Status
in the Southern Sierra Nevada**

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Executive Summary

This report assesses the current status of fisher habitat and fishers (*Martes pennanti*) in the southern Sierra Nevada, California. These results will serve as a baseline for additional analyses of how fisher habitat and the fisher population may respond in the future to potential forest management practices and natural disturbances like wildfire.

The habitat and population models presented in this report are being integrated with a vegetation dynamics model, LANDIS-II, which simulates how forest vegetation is expected to change over time in response to disturbances (e.g. fires), management actions (e.g. fuels reduction projects), climate change, and ecological succession. The ultimate goals of the study are to help the three southern Sierran Forests (Sierra, Sequoia, and Stanislaus) improve vegetation and fuels management intended to reduce risks of catastrophic wildfires, restore more natural fire regimes to Sierran forests, and conserve and enhance habitat value for fishers to ensure their continued persistence, and perhaps expansion, in the Sierra Nevada.

This Final report is based on our Draft report of August 2007, but it excludes sections concerning the vegetation dynamics modeling being performed with LANDIS-II. Those efforts are summarized in a separate progress report being issued simultaneously with this one. The contents of this Final report focus on the current baseline assessment (our Task 1) and reflect changes made based on comments received from science advisors and other interested parties on appropriate sections of the August draft. It also includes new sections concerning fisher population size and dynamics, which were not covered in the August draft. The Science Advisors are listed in Appendix A, and comments received on the August draft in Appendix B.

Methods

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. 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-km²; 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.

At the coarse or home-range scale¹, 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 environmental predictor variables derived at 1-ha resolution from GIS data. Potential predictor variables included a large array of abiotic variables (e.g., derived from digital elevation and climate data) and biotic variables (e.g., derived from digital landcover maps). They included both continuous (e.g., elevation, insolation index, distance measures) and categorical variables (e.g., from vegetation type, density, and size polygons). During the habitat modeling process, we also derived some new continuous variables using the vegetation dynamics model LANDIS-II (e.g., above-ground biomass of trees). These continuous LANDIS-derived variables proved to be strong predictors of fisher habitat value, and have the added advantage of being more directly applied to predicting future habitat changes using LANDIS-II for the future predictions phase of the project (not included here).

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-km² circle centered on points where fishers have been detected (presence) or not detected (presumed absence). We built these multivariate statistical models using a fairly 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 weights², AUC values³, percent deviance explained, and mean and standard error in AUC based on five-fold cross validation⁴.

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

¹ 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 km² as opposed to the regional (population) scale (1,000-10,000 km²) and the microscale (0.1-1 ha).

² 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.

³ 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.”

⁴ 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.

and 2,300 meters elevation⁵ 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 static approach (supplied by R. Truex, personal communication) was to apply sampling theory to the fisher monitoring data to calculate annual fisher occupancy rates, adjusting for detectability and characteristics of the sample population, to derive a total 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, 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. By coupling population dynamics with habitat models in a spatially explicit way, this approach is useful 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 set all habitat north of the Merced River to zero, so that model fishers would not disperse there, 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 habitat value north of the Merced River to revert to modeled values after the population reached equilibrium south of there (Year 20), and ran simulations 20 more years into the future.

Results and Discussion

Home-Range-Scale Habitat Models – A number of the coarse-scale habitat models provided very good fits to the fisher data, but no single model ranked highest on all statistical tests or dominated in AIC weight⁶. There was considerable spatial overlap in predictions, and minor differences in statistical metrics, among the top 10 models⁷. In selecting a model to use for assessing baseline habitat conditions and potential future conditions, we also considered which

⁵ 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.

⁶ Although model averaging is generally recommended for such situations, at least for parametric models, it is unclear how to do model averaging for non-parametric GAM models (W. Zielinski, personal communications).

⁷ For example, AUC for the top 10 models using MAPE2 ranged from 0.909 to 0.928, and AUC using the more liberal MAPE data varied from 0.808 to 0.828. 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.”

of the top-ranking models relied on variables that would be useful indicators of habitat change due to fires or fuels management. For example, mean maximum tree age and mean tree biomass are correlated—and models using one or the other as predictor variables were highly concordant—but biomass would be more responsive to modeled fuels manipulations than maximum tree age, because thinning (and low-severity fires) are not expected to affect the oldest trees in a stand, whereas removing numerous small trees can measurably affect stand biomass⁸.

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) along with one or more biotic variables that reflect vegetation characteristics favorable to fishers. The biotic variables included in top models all reflect previously established characteristics of fisher habitat associations, including measures of tree size, age, and density. 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 al. 2004a, 2004b, 2005, Davis et al. 2007). The results also suggest that potential habitat is significantly more extensive than occupied habitat, particularly on the Stanislaus NF.

Figure ES-1 illustrates results for the model we selected: LAND14. Although this model ranked second to LAND1 in AIC weight, it scored better on some other statistical metrics and better met project objectives because it was the highest-ranking model that used at least one variable responsive to fuels manipulations: mean biomass of trees (excluding red fir). The other variables included were the same as those used in model LAND1: adjusted elevation, insolation index, and mean maximum tree age. LAND14 was only 1.12 AIC units⁹ from LAND1 and was nearly indistinguishable from LAND1 in statistical comparisons and map results, with 99% spatial agreement between the two using the 0.5-probability cut-point in probability of occupancy. It also has a very good fit to the fisher monitoring data (AUC = 0.928).

Resting Microhabitat Model – Figure ES-2 illustrates resting microhabitat value at approximate FIA point locations overlaid on the predicted coarse-scale habitat map using model LAND14. 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 modest (0.342). 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.

⁸ In addition, F. Davis (personal communication) points out that maximum tree age has a highly skewed distribution, which is not ideal for a predictor variable.

⁹ 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).

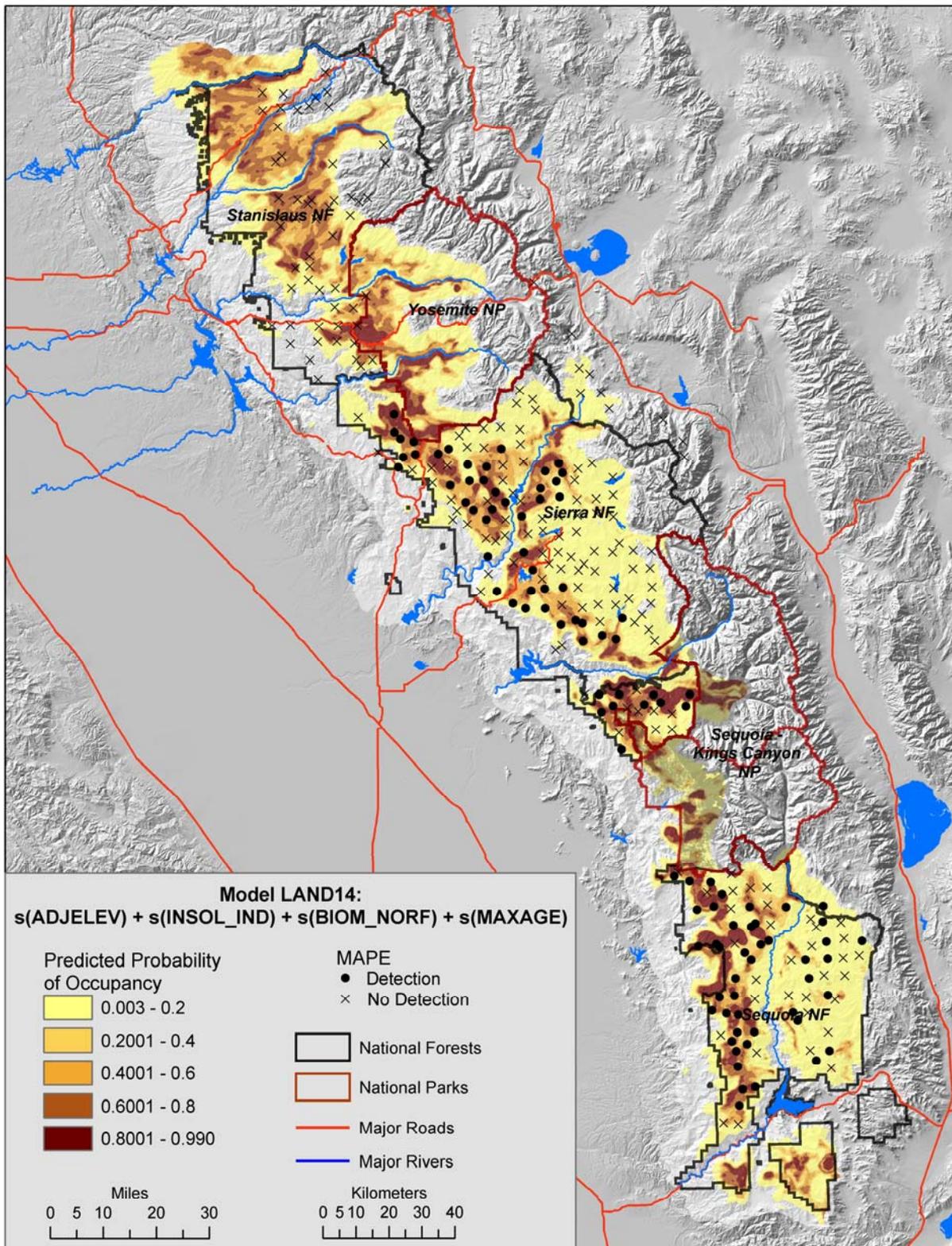


Figure ES-1. Predicted fisher habitat value (probability of occupancy) from model LAND14 using adjusted elevation, insolation index, maximum tree age, and total tree biomass (excluding red fir) within a 5-km² moving window. (Data gaps for tree size result in under-prediction of habitat value in portions of Sequoia Kings Canyon NP.)

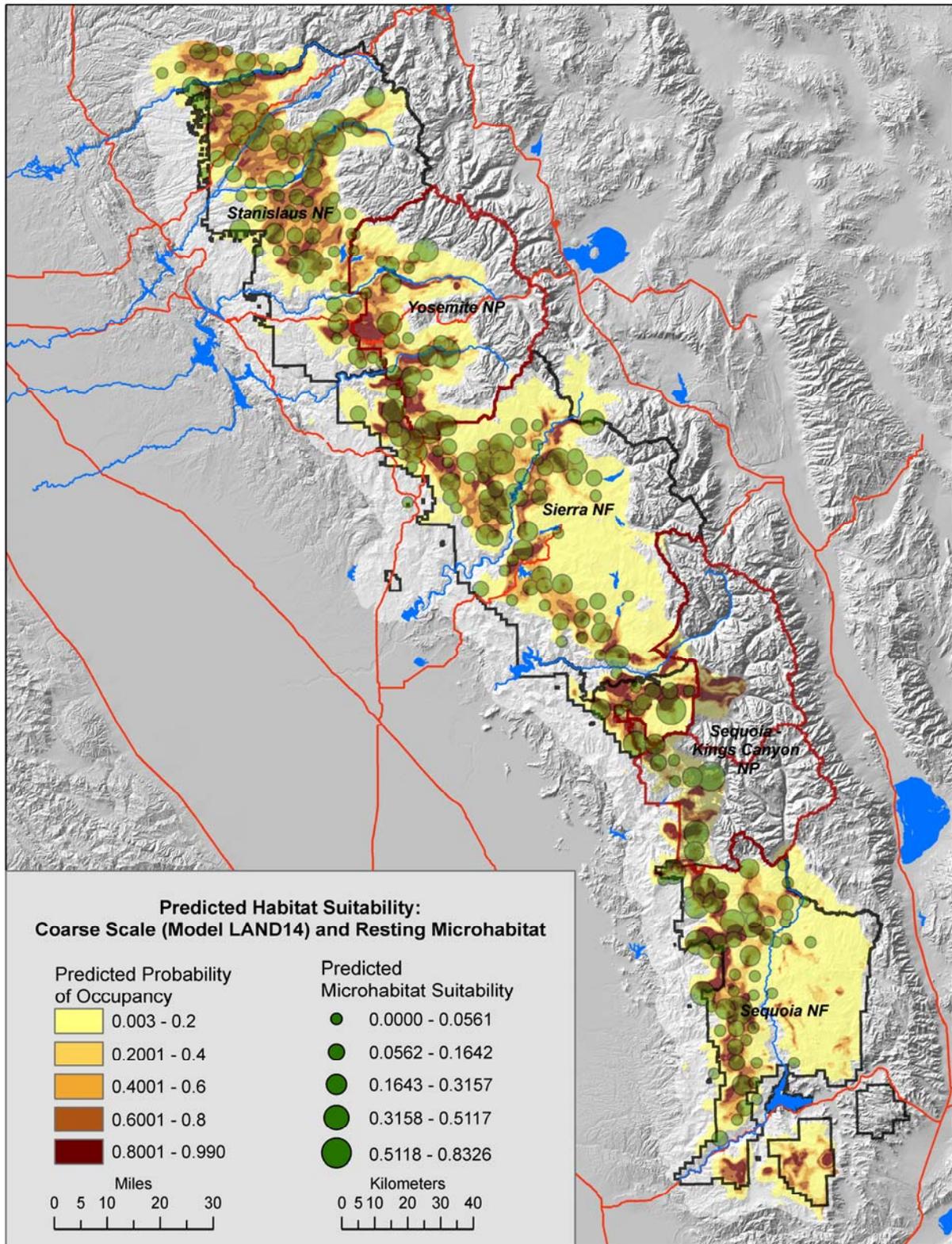


Figure ES-2. Predicted fisher resting microhabitat value at approximate FIA plot locations overlaid on coarse scale predicted habitat value (LAND14). (Data gaps for tree size result in under-prediction of coarse scale habitat value in portions of Sequoia Kings Canyon NP.)

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 in the southern Sierra Nevada is restricted to a relatively narrow band of mid-elevation forests, mostly on the western slope of the range, particularly on mesic topographic positions (i.e., slopes experiencing relatively low annual exposure to solar radiation), and especially where older stands of mixed conifers, sequoia, or ponderosa pine tend to abut or intermix with montane hardwoods, especially black oak. The zone of predicted suitable habitat broadens in more northerly portions of the study area (Stanislaus NF), although this area is not currently occupied by fishers.

Predicted coarse-scale fisher habitat is broken into somewhat discrete blocks or 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. Among the occupied segments, the one comprising parts of the Kings River Administrative Study Area (between the Kings and San Joaquin Rivers) appears to be particularly vulnerable to disruption, as it is a relatively long and narrow stretch of variable but mostly moderate quality habitat according to our models. None of the FIA plots in that area have very high resting microhabitat value, although the sample size is small (~22). 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 NF (roughly 192,000 ha total) appears to be occupied by fishers, whereas large areas of predicted suitable habitat on the Stanislaus NF (about 84,000 ha) are not.

None of the home-range-scale models we tested predicted much suitable habitat on the Kern Plateau, despite numerous and persistent fisher detections there (R. Truex, personal communications). This could be due to one or a combination of factors, such as 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). These apparent inaccuracies in predicting fisher occurrence 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.

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 71-147 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 142-294 adults fishers. Accounting for subadult animals provides a rough estimate of 220-360 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 and should be viewed with caution. 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, and no accounting for increased mortality during dispersal). 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 (using the 0.5 cutpoint in probability of occupancy for model LAND14 south of the Merced River) yielded population estimates of 57-86 adult females and 285-370 total fishers. Although this estimate accords fairly well with those from PATCH and other methods, it is a very 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 matches fairly closely with our estimate using PATCH.

PATCH results suggest that potential source habitat (where births exceed deaths) are fairly well distributed in association with the larger, more contiguous areas of predicted high-value habitat, with sink habitats (deaths exceed births) in more peripheral and isolated habitat areas (Figure ES-3). The Kings River study area, between the San Joaquin and Kings Rivers, appears to be a fairly narrow and tenuous set of territories relative to more contiguous blocks of high-value habitat north and south of there. This area should be managed to prevent loss or further fragmentation of habitat value and to increase the contiguity of dense, older forest stands. Overall, forest management should strive to increase the size and connectivity of source habitat areas.

Our model test for the potential of northward expansion onto the Stanislaus NF from occupied habitat south of the Merced River suggests that there is a strong potential (Figure ES-3). 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.

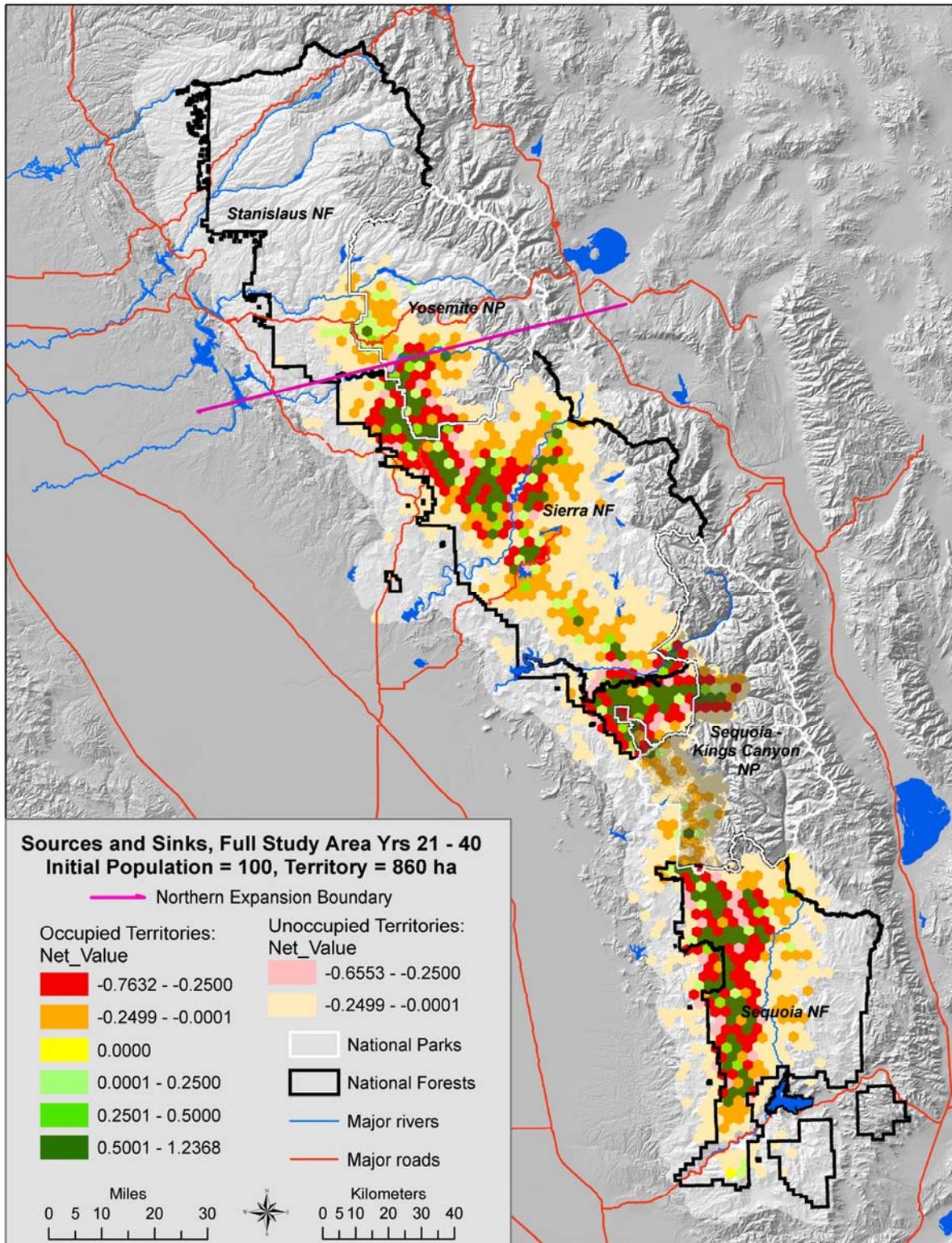


Figure ES-3. Net value of territories (annualized births – deaths) observed over 20 years in program PATCH (territory size = 860 ha; maximum dispersal = 50 km; 100 initial females). Source territories are in greens and sink territories in reds. Data gaps may result in under-representation of habitat value and hence territory value in Sequoia-Kings Canyon NP. Note predicted population expansion north of the Merced River (magenta line) and tenuous nature of territories on central Sierra NF.

Conclusions and Recommendations

We emphasize that the results of the models documented in this report should be viewed more as functional hypotheses subject to further testing than as black-and-white conclusions. However, given the extensive and rigorous approach to modeling, our testing of at least some assumptions, and the close scientific input and review by our Science Advisors, we can confidently offer some general conclusions and recommendations based on these results:

- Utilizing the remarkable systematic fisher monitoring dataset compiled by the USDA Forest Service (Region 5), and the powerful multivariate tool of Generalized Additive Modeling (GAM), we were able to derive a highly predictive habitat suitability map at the home-range scale for fishers in the southern Sierra Nevada. The model uses a combination of two abiotic variables (adjusted elevation and insolation index) and two LANDIS-II biotic outputs (total tree biomass minus red fir and maximum tree age). This is encouraging, because with LANDIS-II providing strong predictive variables, it should now be possible to reliably evaluate the interaction of fuels management and wildfires on coarse-scale fisher habitat into the future. Further, coupling habitat change modeling with spatially explicit population models (such as PATCH or an alternative, RAMAS) will also allow us to investigate potential changes in population viability under alternative scenarios.
- However, given the importance of fine-scale variation in habitat features to fishers, future predictions about changes in fisher habitat or populations should not rely exclusively on home-range-scale models. Unfortunately, we were not able to derive a reliable method of integrating fine-scale habitat models (i.e., the fisher resting microhabitat model using FIA data; Zielinski et al. 2006) with landscape-level models. Consequently, we recommend using LANDIS-II to track changes in the abundance and distribution of individual variables known to be important to fishers at the finer scale, such as the presence of large black oaks or other structures used for denning and resting (see recommendations of W. Zielinski, Appendix B). In the next phase of work, CBI should work with Science Advisors to *identify key variables whose fates can and should be tracked using LANDIS-II in future scenarios (for example, biomass and age of black oaks, and biomass of dead wood for various tree species)*.
- Based on both static and dynamic estimates of population size, and weighing the various uncertainties in all their assumptions, we conclude that *the fisher population in the southern Sierra Nevada (ignoring juveniles) is between 160 and 360 total individuals (and probably less than 300). Of these, about 57 to 147 (and probably less than 120) are adult females, which comprise the most important class for sustaining a population.* Given that this small population does not experience immigration from other regions, it is at risk of extirpation by a variety of stochastic influences. Consequently, *management should strive to increase the extent, value, and contiguity of fisher habitat* throughout the study area (and perhaps beyond, to the north) to maximize population persistence and enable population growth.
- Population persistence is less dependent on the total number of individuals than on the *effective population size* (N_e ; Wright 1931). Effective population size is a measure of the rate at which genetic variation changes over time. It is inversely proportional to a population's probability of extinction, and is smaller than actual population size due to

effects of population fluctuations, uneven sex ratios, age-structuring of populations, and other factors. Genetic work should therefore be used to determine N_e of the southern Sierra Nevada fisher population and its potential for future population growth. Recent genetic work on fishers suggests that this population is genetically depauperate (Wisely et al. 2004), meaning that N_e will probably be quite low (W. Zielinski, personal communication). *If effective population size is below about 50, extinction is probable over a relatively short term, and population augmentation may be warranted.*

- Highly suitable resting microhabitats are relatively rare even within areas of predicted suitable habitat at the coarse scale. Management should strive to *increase the distribution and abundance of forest stands that provide high-value resting sites and hopefully denning sites* (Zielinski et al. 2004a, 2006, Aubry and Raley 2006) especially within areas predicted to be high value at the coarse-scale.
- The Kern Plateau supports more habitat and fishers than our models predict. This should be addressed by further research and perhaps by developing separate models there using updated and/or higher resolution environmental data layers.
- The Kings River Administrative Project Area should be a focal area for increasing habitat value and contiguity. Fisher habitat between the Kings and San Joaquin Rivers is a relatively long and narrow stretch of variable but mostly moderate-quality habitat, with rather tenuous potential as a source population area under current conditions, according to our models. *Management should strive to increase the value, extent, and connectivity of fisher habitat between the Kings and San Joaquin Rivers.*
- Habitat on the Stanislaus NF, although not currently occupied by fishers, has potential to become occupied in the future, barring disturbances that may reduce the probability of fishers dispersing to and successfully colonizing from the south. Vegetation succession may increase the probability of northward expansion over the current condition, but a finer-resolution assessment would be required to identify specific locations where management intervention may or may not be beneficial.
- Field studies and continued monitoring should also address dispersal characteristics of fishers in the study area and whether or to what degree steep canyons, sparse forest vegetation, roads, or other factors serve as barriers or filters to movement. Again, coupling genetic studies with field and modeling studies may prove helpful in better understanding dispersal dynamics and other important aspects of fisher biology (Aubry et al. 2004). *If natural expansion to areas north of the Merced River is deemed unlikely, active translocation of fishers to portions of the Stanislaus NF may be prudent.*

Acknowledgements

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Our technical work is 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, 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 objective advice and insights about fisher habitat and populations and for his important insight about the role of incident solar radiation in defining fisher habitat; 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; and Peter Stine for advice on data sources, uncertainties, and assumptions in modeling.

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stakeholder groups have also added considerably to the thought process, especially Sue Britting and Craig Thomas of the Sierra Forest Legacy; Steve Brink of the California Forestry Association, Steve Self of Sierra Pacific Industries, Bill Corcoran of the Sierra Club, Cynthia Wilkerson and Mike Skuja of Defenders of Wildlife, John Buckley of CSERC, Linda Blum of Quincy Library Group, and John Robinson of On My Mountain Environmental Consulting. Finally, Chad Hanson submitted thoughtful comments on the role of fires in maintaining fisher habitat for consideration in this report.

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 the Sierra Nevada, fishers currently occupy less than half their historic range, with a small population persisting in the southern Sierra Nevada, south from Yosemite National Park through the Greenhorn Mountains in southern Tulare County (Zielinski et al. 2000, 2005). In 2004, the U.S. Fish and Wildlife Service (2004) found that the west coast distinct population segment of the fisher was warranted protection under the Endangered Species Act, but that listing the species was precluded by higher priorities.

Forces that may reduce habitat value for fishers range from stand-replacing wildfires to management actions intended to preclude such fires by reducing the amount and continuity of forest fuels. The complex and probabilistic interplay between such habitat threats, as well as incomplete information on fisher biology, creates great uncertainty about the current health of the southern Sierran fisher population and how it is likely to change in the future. The Conservation Biology Institute (CBI) was therefore commissioned by Region 5 of the U.S. Forest Service to compile available data on fisher populations and habitat in the southern Sierra Nevada, assess their current status, and predict how they are likely to respond to various alternative forest management actions, as well as unplanned events like wildfire.

The project involves extensive collaboration with other scientists, stakeholders, and agencies. A group of independent science advisors (Appendix A) oversees technical aspects of the analyses and advises 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 are 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.

Objectives of the Study and This Report

CBI was tasked with assessing the current status of fisher (*Martes pennanti*) habitat in the southern Sierra Nevada, assessing risks to the southern Sierra fisher population, and forecasting likely cumulative effects of management actions and other disturbances (especially wildfire) on fisher habitat and populations. More specifically, CBI was asked to address three main objectives:

1. Create a scientifically credible set of habitat maps and fisher population estimates for the southern Sierra Nevada based on current conditions (contemporary baseline).

2. Using spatial modeling, assess potential risks to fisher habitat due to wildfire and proposed forest management activities aimed primarily to reduce fire hazards and return fire regimes to more natural, pre-suppression conditions.
3. To the degree possible, predict fisher distribution and population viability based on probable habitat changes due to wildfire and proposed forest management actions over a reasonable range of alternative scenarios.

This report presents our findings for Objective 1 – a baseline assessment of the current condition of fisher habitat and the fisher population occupying the southern Sierra Nevada. These results establish the baseline against which to assess potential future changes that may occur due to fires and management actions (Objectives 2 and 3), which are to be addressed during 2008¹⁰. The overarching goal of this report is to summarize everything we've learned to date and focus those lessons on the next phase of analyses, whose goal is to objectively assess how various aspects of potential fuels management scenarios may affect fire regimes and fisher habitat, and ultimately to guide us to a fuels management approach that best balances the conservation of fishers and other management objectives.

¹⁰ Progress on Objectives 2 and 3 using the vegetation dynamics model LANDIS-II are addressed in a separate progress report (Syphard et al. 2007) being issued simultaneously with this Final report on the fisher baseline assessment (Objective 1).

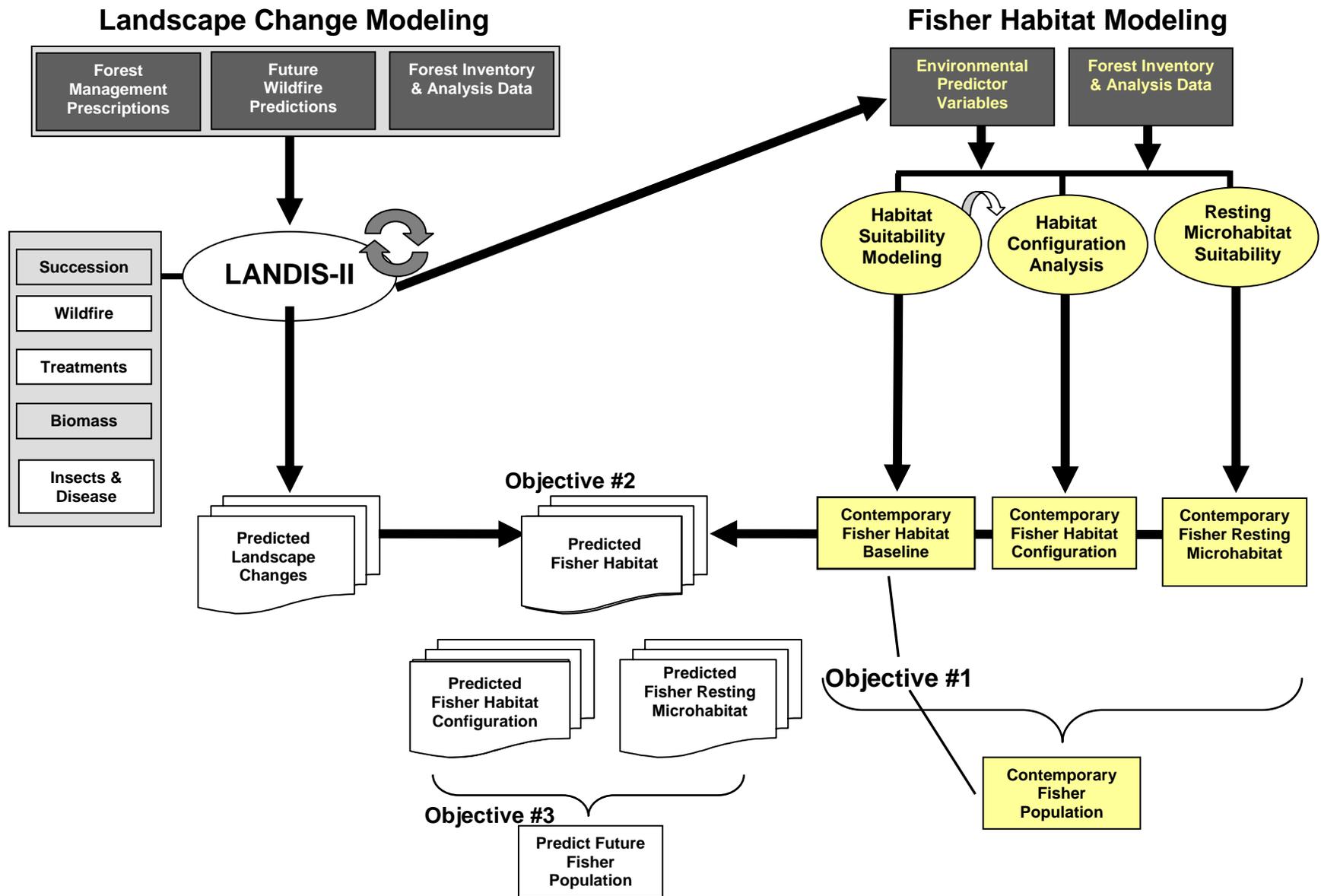


Figure 1. Flow diagram illustrating the major components of the project and the context for this report. It shows the relationships and interactions between data, analyses, and objectives. Boxes illustrated in yellow are addressed in this report.

Study Limitations and Constraints

Throughout this report, we strive to be clear and transparent in stating assumptions that may lead to uncertainties in our methods and interpretations of results. However, 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 methods, results, and recommendations in 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.

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 NF (south of the Merced River) although historically fishers occurred throughout the length of the Sierra Nevada (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; Zielinski et al. 2005) and insufficient time, dispersal ability, or source of potential colonizers for the population to expand back into that area. 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 NF. 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.*

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 (Appendix B) 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 poor resolution or inaccurate environmental data for that area 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). The Kern Plateau appears to be ecologically unique (Miles and Goudey 1998) and fishers may select habitat based on different features there than in other areas. At this time, we do not know the relative contributions of such factors to the inaccuracies in model predictions for the Kern Plateau (and possibly other areas), and this concern should be kept in mind during further research and monitoring.

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 km²). Because the models average variables over a 5-km² 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.

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. 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.

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, Appendix B). 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 and Appendix B).

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, we are not currently considering the other side of climate change – growth and survival response by tree species – nor the effects of insects and disease. The degree to which these factors may influence our vegetation change and fisher habitat models is unknown.

Methods

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 2). 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 NF.

Based on early analyses of the distribution of fishers, fisher habitat, and wildfires, we modified the study area somewhat for modeling purposes (Figure 2). 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 risk of wildfires within the area of greatest interest.

Data Sources and Data Gaps

Existing data on environmental factors and management actions were compiled into a Geographic Information System (GIS, Appendix C). These data layers were used to derive potential predictor variables for the creation of multivariate fisher habitat models (Appendix D). 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 NF and locations of giant sequoia groves.

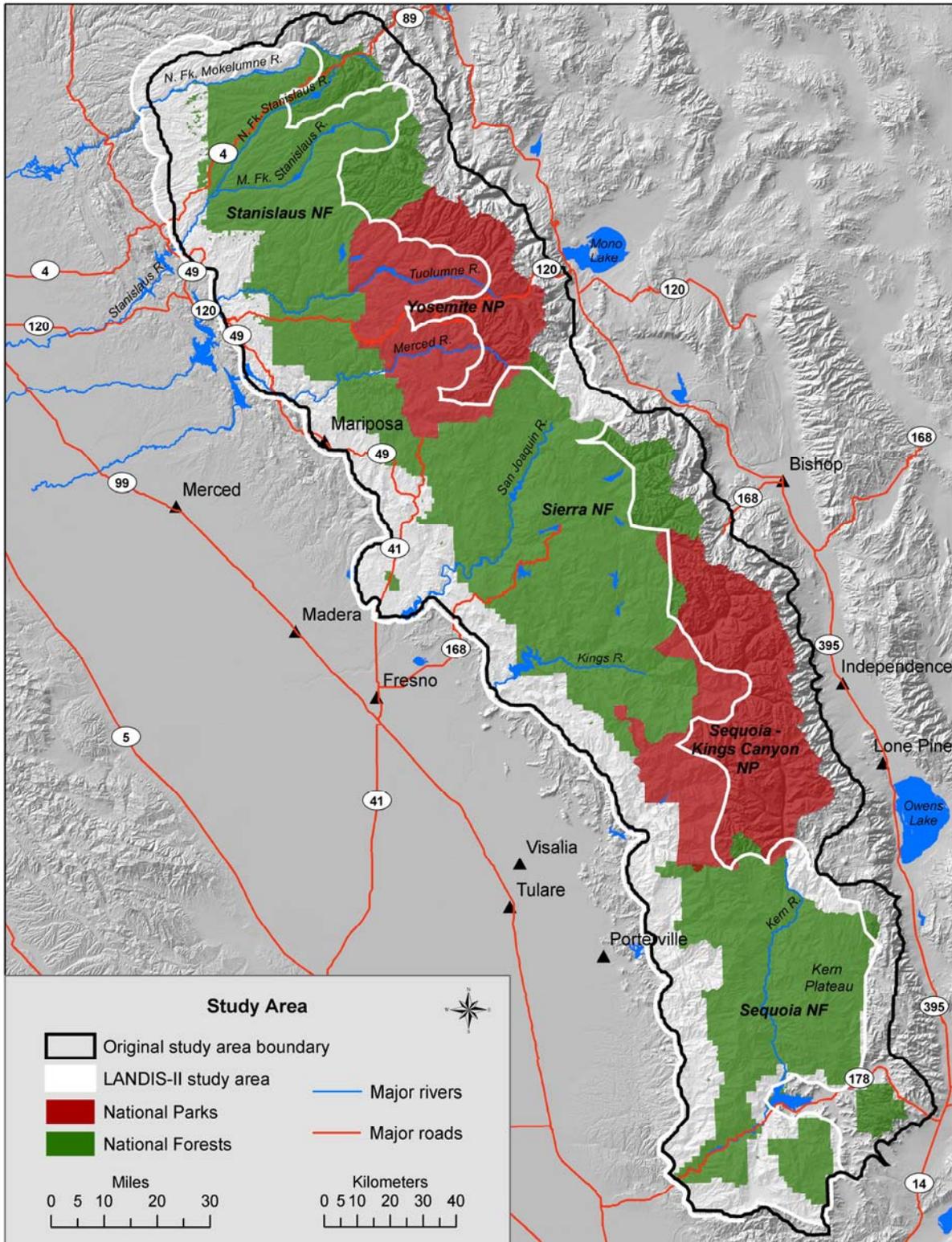


Figure 2. Original study area boundary (black outline) and revised modeling study area (white outline).

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-km² moving windows. 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.

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 E) given the nature of the available data, exploratory tests, and project objectives. 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 km² 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 decisions¹¹, 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

¹¹ 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.

stronger statistical associations at 3 km² than 10 km² (they did not test 5 km²) in the southern Sierra Nevada. They believed that their 10-km² 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 Syphard et al. 2007) 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 C lists data layers used in modeling and Appendix D defines potential predictor variables that were derived from these layers

Response Variables – 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 NFs, most monitoring sites have been sampled between two and four times since 2001, while on Stanislaus NF sites have generally only been sampled once or twice¹². 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 *MA*rtes *PE*nnanti): This includes all survey points south of the Merced River¹³ 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.

¹² The database we used included results through 2006, since which time additional data have been collected.

¹³ 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).

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 communications; see Appendix B) 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 (see also comments from K. Aubry, Appendix B, questioning whether the monitoring data can truly be used to distinguish transient vs. resident habitat use). 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 2002)¹⁴. 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.¹⁵ 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

¹⁴ 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.

¹⁵ 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).

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 F 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)¹⁶, AIC weights, percent deviance explained, residual deviance, and Area Under Curve (AUC) of the Receiver-Operating Characteristic (ROC)¹⁷.

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 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.

It is important to emphasize that statistical fit to the empirical data was not the only consideration in selecting which coarse-scale habitat model(s) should be used for predicting future changes in habitat value. It is also important that any model used for predicting future conditions use at least one variable that can be reliably derived from the landscape dynamics model (LANDIS-II); and it would be highly desirable that included variables are responsive to modeled fire and management actions (e.g., thinning prescriptions). We realize that the process we ultimately used to select, refine, and apply GAM models strayed from the strict “information-theoretic” philosophy of modeling (Burnham and Anderson 2002) due to the particular objectives and practicalities of this project, schedule constraints, and the way new variables became available over the course of the project.

¹⁶ 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.

¹⁷ 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.”

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 Northwest Research Station. We applied the model to FIA plot locations between 1100 and 2300 meters elevation¹⁸ 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.

Baseline Population Assessment¹⁹

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 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 directly from the fisher monitoring data using sampling theory and 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 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).

¹⁸ 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.

¹⁹ 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).

Territory Size – We used three estimates of fisher home range²⁰ 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 NF, 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 NF.
- 860 ha: This is the median of 527 and 1,192 ha.

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 (LAND14) (Table A). Each 1-ha pixel within a territory hexagon was provided an integer score from 0 to 10 based on Table A, and the mean habitat score across all pixels was used as the territory score.

Table A. Cross walk of mean probability of occurrence (habitat value) calculated using habitat model LAND14 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 |

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 B). 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 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).

²⁰ 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).

Table B. 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 (λ) 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, \geq 37 mo) |
|-----------|----------------------|----------------------------|----------------------------|-----------------------------|
| Fecundity | 0 | 0.315 | 0.49 | 1.62 |
| Survival | 0.5 | 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 (Zielinski and Stauffer 1996). 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.

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 about 20 years to approach an equilibrium in population age structure and size (see Results), so the first 20 years are discarded, and results are 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)²¹, 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).

²¹ Tallies include territorial females plus non-territorial floaters and any juveniles residing in their mother's territory.

Results

Home-Range-Scale Habitat Models

Model Characteristics and Selection of Preferred Model – Table C summarizes characteristics of the top 10 performing habitat models (and Appendix G lists all candidate models we compared) in order of AIC weights, from highest to lowest. No single GAM model ranked highest on all statistical tests or dominated in AIC weight, and the mean cross-validated AUC values are all extremely close, with widely overlapping standard deviations. Although model averaging is generally recommended for such situations, at least for parametric models (Burnham and Anderson 2002), it is unclear how to do model averaging for non-parametric GAM models (W. Zielinski, personal communications).

All of the top models included a combination of two abiotic variables--adjusted elevation and either insolation index or 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 (LAND1; Figure 3) combined elevation and insolation index with mean maximum tree age. It has high AUC²² values (0.926 using MAPE2 and 0.824 for MAPE) and explains 49% of deviance. Unfortunately, the variable maximum tree age would not be responsive to fuels manipulations--which rarely if ever would be expected to remove the oldest trees from a stand--making this model unsuitable for evaluating future changes in habitat value that may occur due to forest management²³. The second-highest ranking model (LAND14; Figure 4) also scored very high, and has better utility for predicting future changes, due to the addition of a LANDIS-derived variable that is responsive to fuels manipulations and fires: total tree biomass excluding red fir. Selecting this model is within recommended statistical

²² 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.”

²³ F. Davis (personal communication) also points out that maximum tree age has a highly skewed distribution, which is not ideal for a predictor variable.

Table C. 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 D 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 |
|---------------|--|-----------------------------|----------------------------|--------------|-------------|--------------------|--|---|
| LAND1 | ADJELEV, INSOL_INDEX, MAXAGE | 0.40479 | 0.4904 | 0.92577 | 0.82389 | 0.62248 | 0.89635 | 0.05495 |
| LAND14 | ADJELEV, INSOL_INDEX, MAXAGE, BIOM_NORF | 0.23173 | 0.4954 | 0.92824 | 0.82662 | 0.61984 | 0.88226 | 0.04824 |
| LAND3 | ADJELEV, PRISM, MAXAGE | 0.17973 | 0.4821 | 0.92029 | 0.82646 | 0.64179 | 0.89260 | 0.05517 |
| LAND8 | ADJELEV, PRISM, BIOMASS | 0.11306 | 0.4775 | 0.92400 | 0.82771 | 0.69359 | 0.88558 | 0.05812 |
| LAND7 | ADJELEV, INSOL_INDEX, BIOMASS | 0.03961 | 0.4669 | 0.92206 | 0.82482 | 0.64442 | 0.88574 | 0.05064 |
| LAND13 | ADJELEV, INSOL_INDEX, MAXAGE, BIOMASS | 0.02074 | 0.4711 | 0.92400 | 0.82763 | 0.63301 | 0.89724 | 0.04439 |
| 231 | ADJELEV, INSOL_INDEX, LRGHDWD, CWHR2 | 0.00240 | 0.4493 | 0.91534 | 0.81718 | 0.65935 | 0.88139 | 0.06293 |
| 239 | ADJELEV, INSOL_INDEX, PHDWD, CWHR2 | 0.00150 | 0.4445 | 0.91622 | 0.81781 | 0.67428 | 0.89990 | 0.04666 |
| 237 | ADJELEV, INSOL_INDEX, LRGHDWD, LRGFOR | 0.00125 | 0.4427 | 0.91181 | 0.81134 | 0.67515 | 0.87509 | 0.06640 |
| LAND11 | ADJELEV, INSOL_INDEX, BIOM_NORFBO, BIOM_BLKOKAK | 0.00112 | 0.4416 | 0.90898 | 0.80806 | 0.63213 | 0.87167 | 0.04695 |

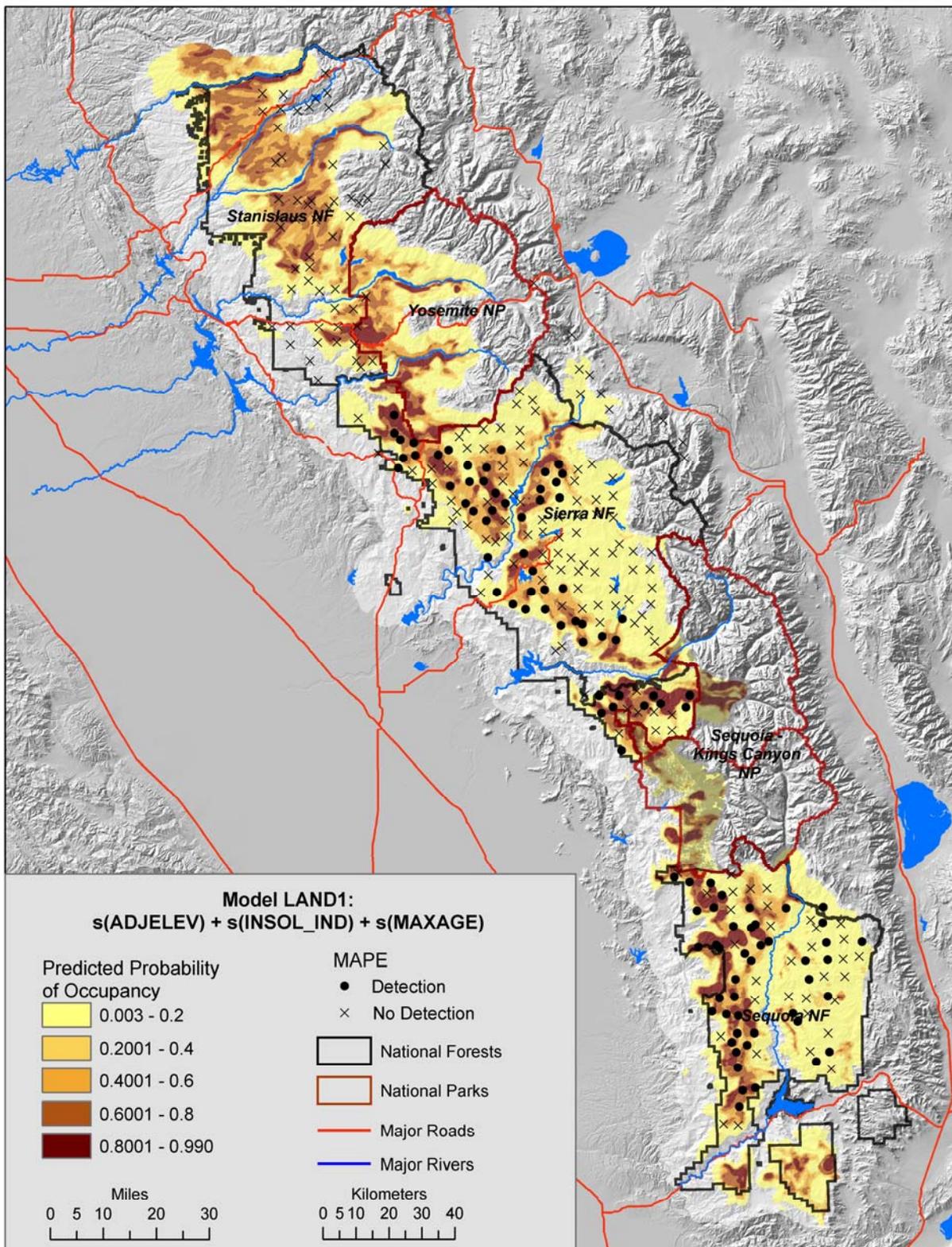


Figure 3. Predicted fisher habitat value (probability of occupancy) from the highest ranking model (LAND1) using adjusted elevation, insolation index, and maximum tree age within a 5-km² moving window and compared with fisher detection-nondetection monitoring data. (Data gaps may result in under-prediction of habitat value in portions of Sequoia Kings Canyon NP.)

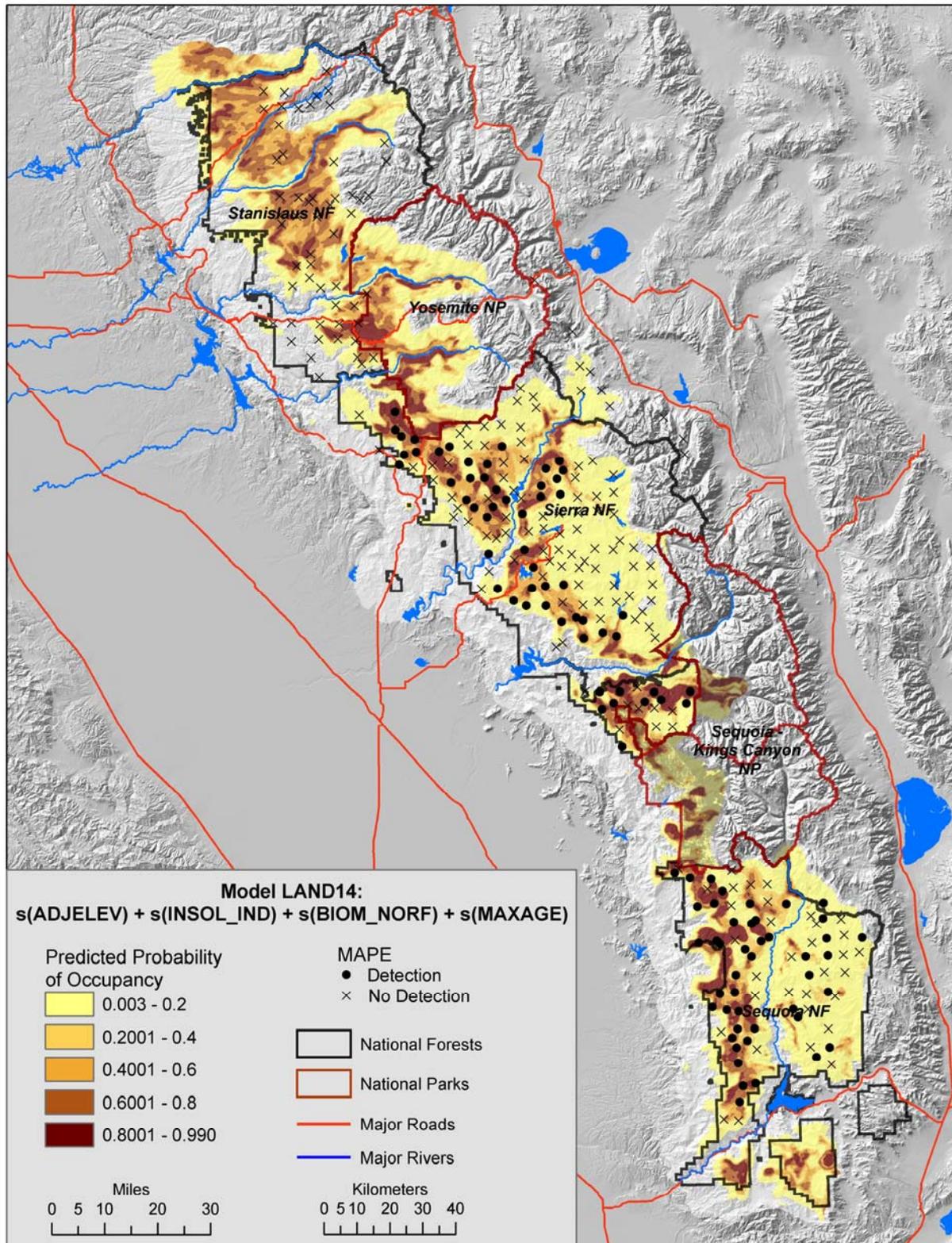


Figure 4. Predicted fisher habitat value (probability of occupancy) from the second highest ranking model (LAND14) using adjusted elevation, insolation index, mean tree biomass (excluding red fir), and maximum tree age within a 5-km² moving window and compared with fisher detection-nondetection monitoring data. (Data gaps for tree size may result in under-prediction of Sequoia Kings Canyon NP.)

guidelines²⁴. Additionally, LAND14 tested well in other diagnostics, with AUC values nearly identical to those of the top model using MAPE, MAPE2, and an independent fisher survey dataset; as well as similar cross-validated mean AUC values (Table C). Furthermore, there is nearly 99% spatial agreement between LAND1 and LAND14 using the 0.5 probability cut-point.

Another reason to consider further analyses using the second highest ranking model, LAND14, is to investigate how the different measures of forest condition included may interact to influence habitat quality. Total tree biomass and mean maximum tree age are correlated variables (correlation coefficient = 0.8823), but there are locations where biomass is high despite lower average tree age, and vice-versa. Figure 5 shows mapped residuals from the correlation between the variables maxage and total biomass (excluding red fir). Green areas represent locations where mean maximum tree age is greater than expected based on total tree biomass, and purple where total tree biomass is greater than expected based on stand age. Note that many mid-higher elevation areas appear to have lower than expected biomass, whereas lower elevation areas appear to have greater than expected biomass (due perhaps to high site productivity, or an abundance of dense, young forests). These differences deserve additional consideration for modeling as well as management reasons.

To summarize, we selected the 2nd highest ranking model (LAND14) to use in assessing current and future fisher habitat, for the following reasons:

- The top two models both combine adjusted elevation and insolation index, which together appear to robustly capture the abiotic universe of conditions favorable to fishers at this scale.²⁵
- LAND14 also contains a variable (tree biomass) that is responsive to management manipulations and fire, which is lacking for LAND1.
- Statistical comparisons among the top two models are extremely close. They rank inconsistently among one another across the various tests of model fit (Table C) and have extensive overlap in mean \pm standard deviation for the cross-validation test.
- There is an extremely high degree of spatial overlap between the two (98.6% agreement using 0.5-probability cut-point).

Finally, further investigating model LAND14 may have scientific and management value, because this may allow us to evaluate the contributions of both forest biomass and forest age to fisher habitat condition. This may allow us to better understand and perhaps monitor how fishers respond to forest succession in, for example, highly productive versus less productive forest areas, or to different types of fuels treatments.

²⁴ LAND14 is less than 2.0 AIC units from the top model (the recommended rule-of-thumb for selecting models other than the top model; Burnham and Anderson 2002, W. Zielinski personal communication). LAND14 is 1.12 units away.

²⁵ Note that 8 of 10 top models used this combination of variables (the other two substituted precipitation for insolation index). The strength of this relationship in setting the abiotic “envelope” within which fisher habitat is likely to be expressed was suggested by one of our science advisors (R. Barrett personal communication), which appears to have been supported by the analysis.

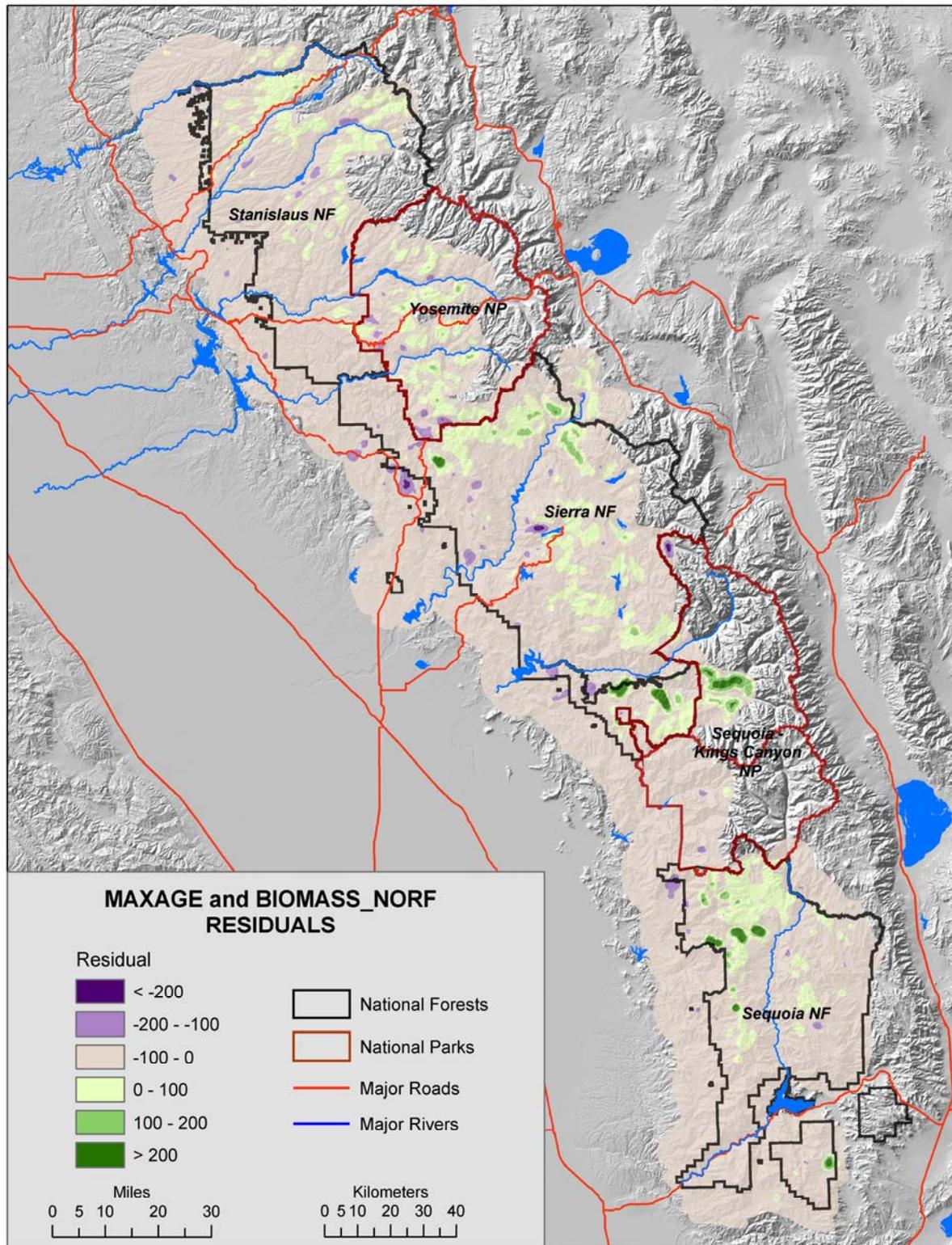


Figure 5. Mapped residuals from the correlation between mean maximum tree age and tree biomass within the 5-km² moving window. Purple areas represent stands where biomass is higher than expected based on maxage; green is where biomass is less than expected based on maxage.

Figure 6 shows the partial response curves for model LAND14. (Curves are similar for other models, which are available upon request.) 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 forest age, increases with total forest biomass (excluding red fir), is highest over intermediate elevations (~1500 – 2300 m); and decreases with increasing annual solar radiation.

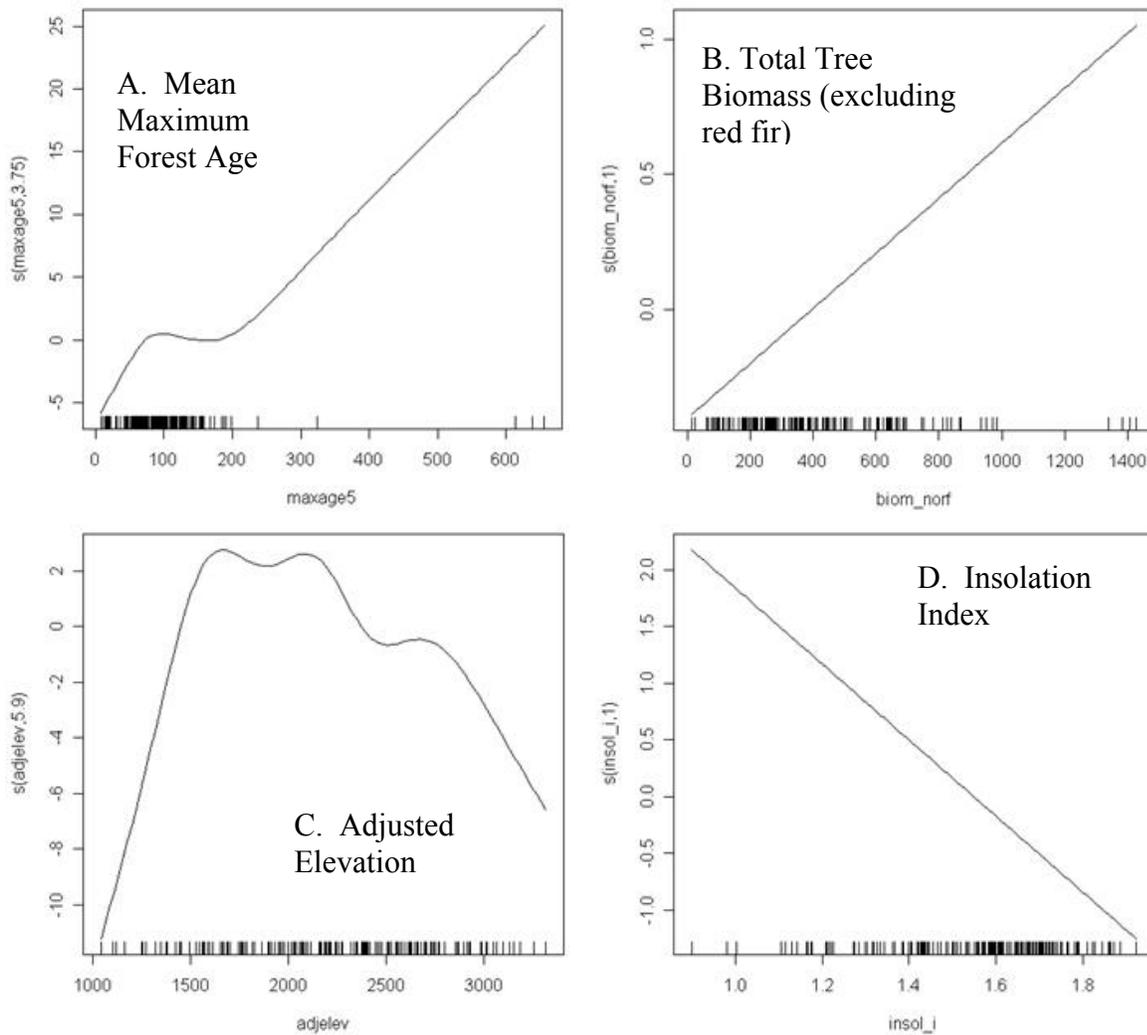


Figure 6. Partial response curves for LAND14. The curves show the relationship between each variable and modeled habitat value given all variables in the model. Zero on the y-axis is the theoretical cut-point between suitable and unsuitable habitats, in the context of all variables.

Amount of Predicted Suitable Habitat – There are over 400,000 ha total area of predicted suitable fisher habitat (defined as predicted probability of occupancy by fisher ≥ 0.5) based on LAND14 (Table D). 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. In addition, the amount of suitable habitat predicted on Kern Plateau appears to be under-represented relative to fisher monitoring results there.

Table D. Predicted area of currently suitable habitat by National Forest, defined using the 0.5 probability of occurrence cut-point and model LAND14).

| Forest | Hectares |
|----------------------------|-----------------|
| Stanislaus NF | 84,471 |
| Sequoia NF | 97,564 |
| Sierra NF | <u>94,103</u> |
| Subtotal, National Forests | 276,138 |
| Other | <u>161,045</u> |
| Grand Total | 437,183 |

Resting Microhabitat

Figure 7 illustrates resting microhabitat value, at approximate FIA point locations, overlaid on coarse scale predicted habitat suitability (using model LAND14). 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 (0.342). 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.

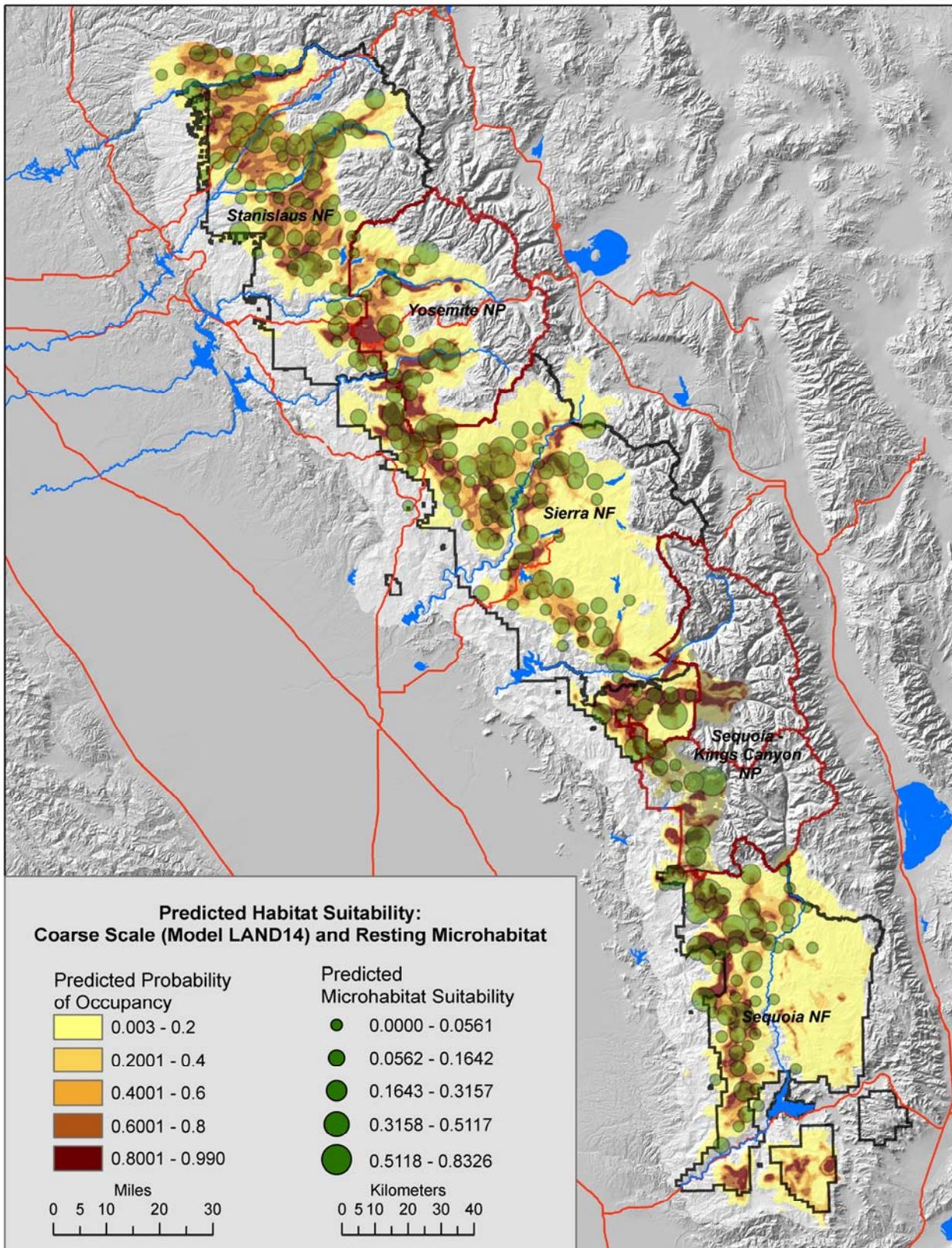


Figure 7. Predicted fisher resting microhabitat value at approximate FIA plot locations overlaid on coarse scale predicted habitat value (using model LAND14). (Data gaps for tree size result in under-prediction of coarse scale habitat value in portions of Sequoia Kings Canyon NP.)

Baseline Population Assessment

The equilibrium number of adult females by Year 40 was largely independent of starting population size (Figure 8 and Table E), 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 F) so we default to the intermediate value of 50 km for most results reported below.

In contrast, equilibrium population size is, not surprisingly, strongly affected by territory size (Figure 8 and Table G), 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 215 to 259 adult females (mean = 233.8); at 1,200 ha it could support about 71 to 98 adult females (mean = 85.1); and at the intermediate 860 ha, 113 to 147 adult females (mean = 126.3). 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 (71 to 147 adult females). We default to the intermediate territory size of 860 ha in most results reported below.

Table E. 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 | 122.5 | 124.0 | 107 | 143 | 9.71 |
| 100 | 125.5 | 126.3 | 113 | 147 | 7.29 |
| 200 | 133.0 | 130.55 | 114 | 145 | 9.05 |

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

| Max. Dispersal | | | | | |
|-----------------------|---------------|-------------|------------|------------|-----------|
| Distance (km) | Median | Mean | Min | Max | SD |
| 25 | 127.5 | 127.25 | 110 | 136 | 6.52 |
| 50 | 125.5 | 126.3 | 113 | 147 | 7.29 |
| 100 | 122.0 | 122.7 | 112 | 146 | 8.75 |

Table G. 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 | 235.0 | 233.8 | 215 | 259 | 11.47 |
| 860 | 125.5 | 126.3 | 113 | 147 | 7.29 |
| 1200 | 84.5 | 85.05 | 71 | 98 | 7.17 |

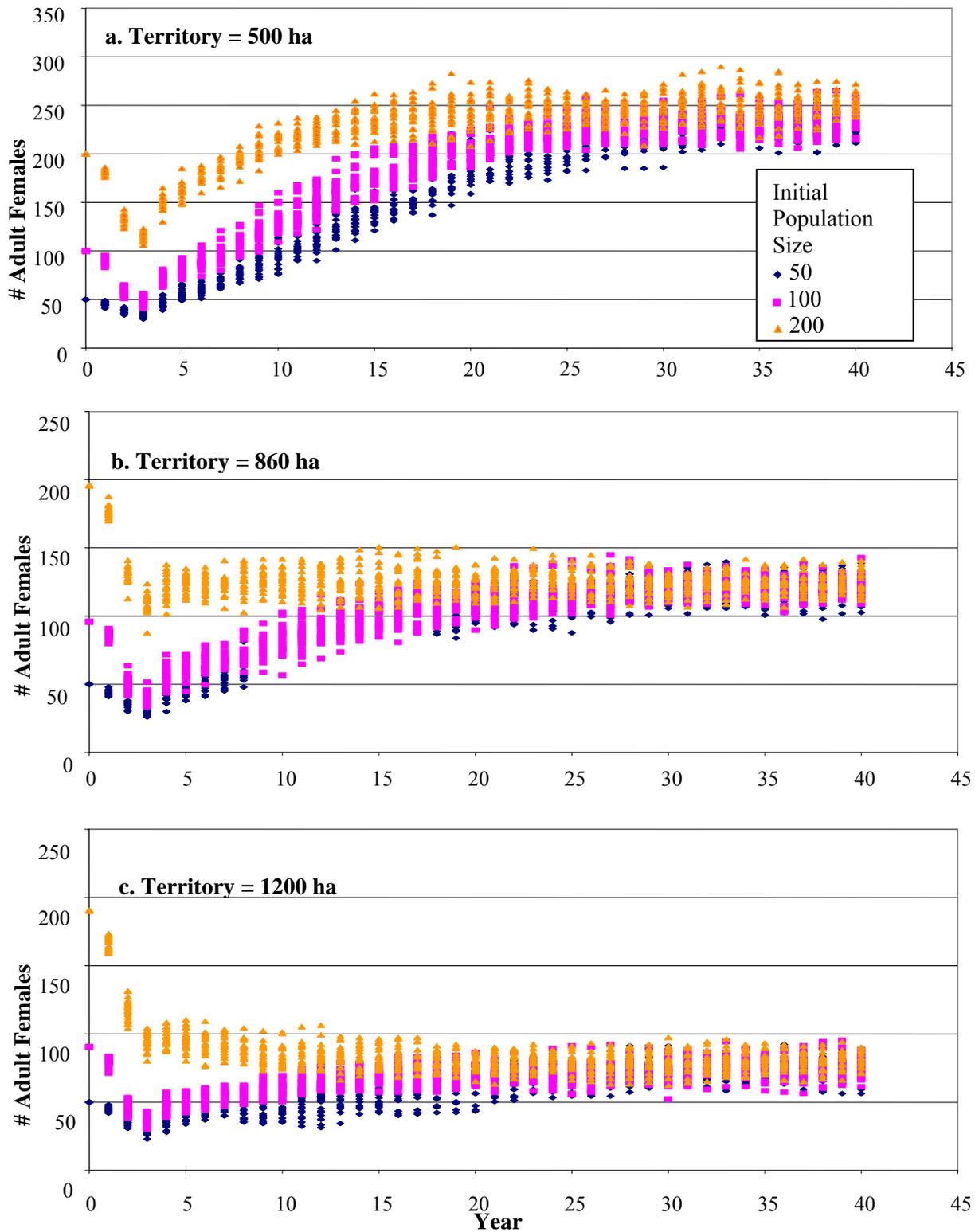


Figure 8. Effects of territory size and initial number of females on equilibrium population size in habitats south of the Merced River. Maximum dispersal set to 50 km; 50, 100 or 200 females started at Year 0 at each of three territory sizes (500, 860, and 1,200 ha). Years 0-20 are discarded from territory occupancy calculations because dynamics don't equilibrate until around Year 20.

Occupancy Pattern – Figure 9 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 LAND14 (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 NF east of the Kern River) despite persistent fisher detections there.

Figure 10 illustrates average annual territory usage over the 20-year period, averaged over 20 replicates, using the same assumptions as in Figure 9. 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 LAND14. 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. Note that fisher habitat in the Kings River Project Study Area (south-central portion of Sierra NF, between the Kings and San Joaquin Rivers) represents a fairly narrow and tenuous set of territories relative to more contiguous blocks of high-value habitat north and south of there. Again, territory usage on the Kern Plateau appears to be poorly 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 (LAND14) value, the model fisher population gradually expands north into suitable habitat areas. Figure 11 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 30 km north of existing fisher detections, near the Tuolumne River, where high-value habitat is predicted by LAND14. However, as expanded on in the Discussion, it is unclear whether this potential for rapid expansion is real, because dispersal 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 12 maps the observed net value of each hexagon (births – deaths) over the 20-year period (Year 21-40) using the same assumptions as Figure 11. 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.

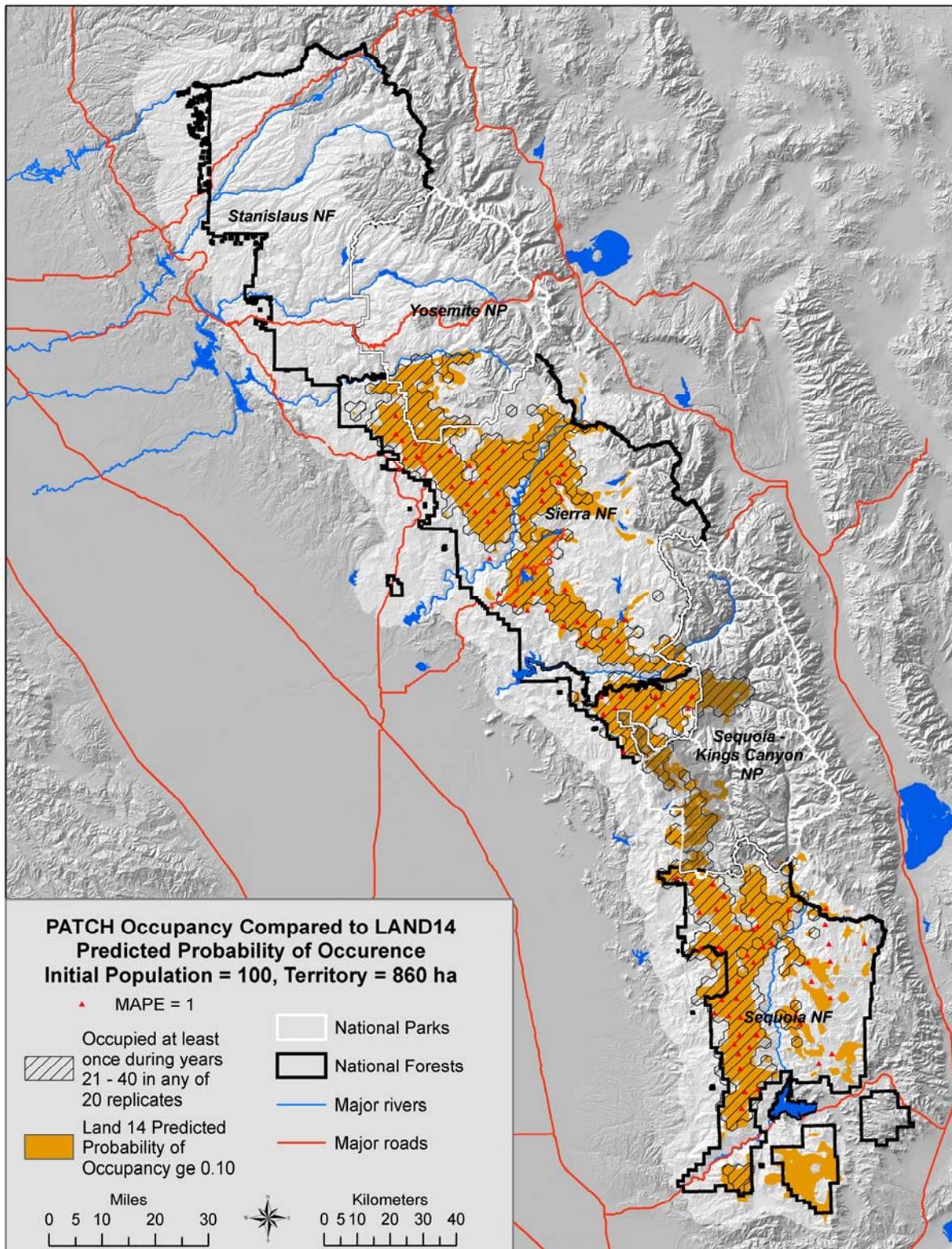


Figure 9. Extent of territory occupancy during Years 21-40 predicted by PATCH using intermediate parameter values (860-ha territory size, 100 initial females, 50-km dispersal distance) shown relative to probability of occurrence ≥ 0.1 in LAND14. Habitat north of Merced River set to zero. Red triangles indicate fisher detections from monitoring data. Any territory occupied at least once during the 20-year period and 20 replicates is shown as occupied.

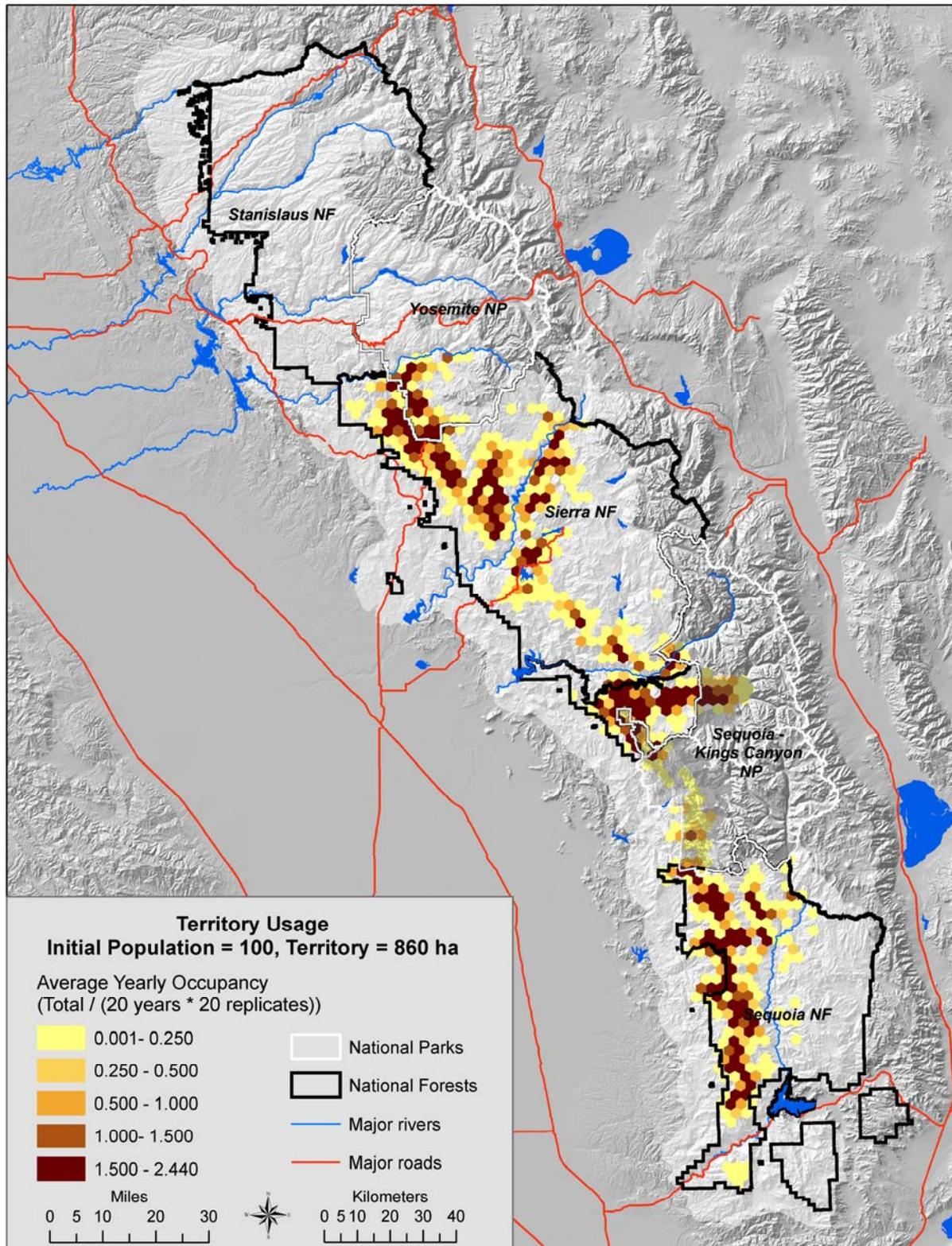


Figure 10. Average annual territory occupancy predicted by PATCH using the same assumptions as Figure 9. Territory usage includes juvenile females sharing a mother’s territory, so highly productive territories can have average yearly occupancies > 1.0. Data gaps may result in under-representation of habitat value and hence territory value in Sequoia-Kings Canyon NP.

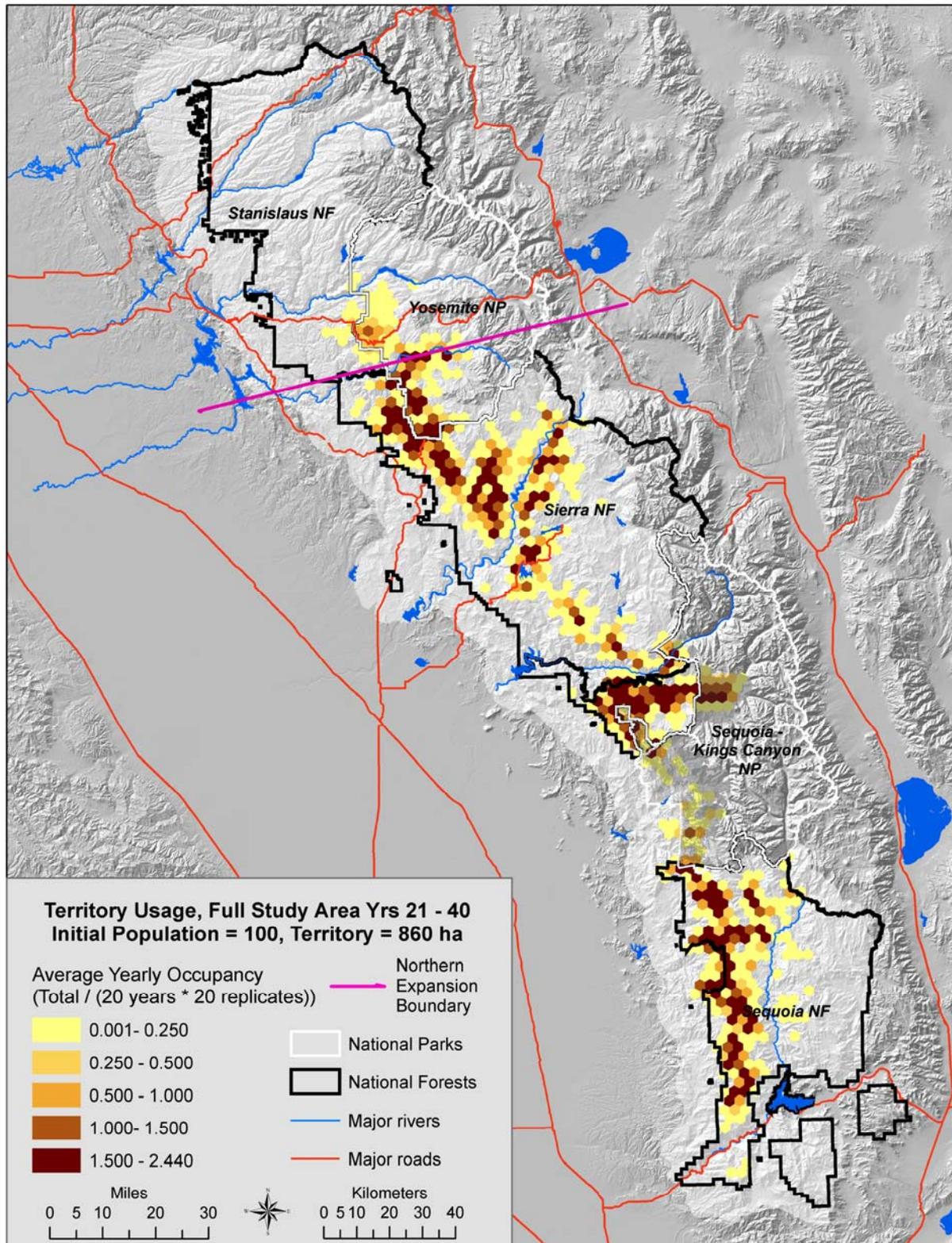


Figure 11. Average annual territory usage showing predicted northward expansion over 20 years. All assumptions the same as Figures 9 and 10 except that habitat value increased after Year 20 from zero to LAND14 value north of the Merced River (magenta line). Data gaps may result in under-representation of habitat value and hence territory value in Sequoia-Kings Canyon NP.

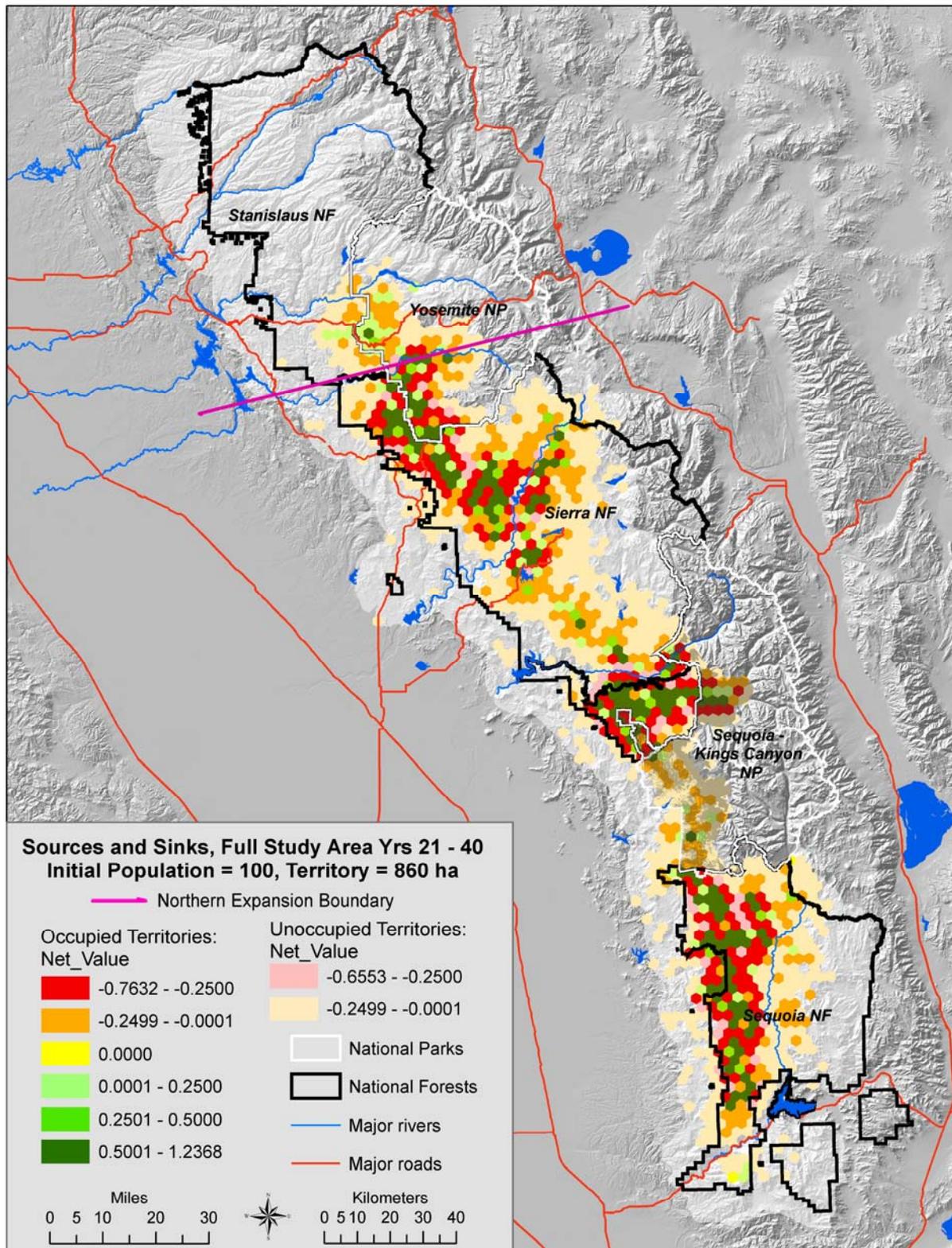


Figure 12. Net value of territories (annualized births – deaths) observed over Years 21-40 using the same assumptions as Figure 11. Source territories are in greens and sink territories in reds. Data gaps may result in under-representation of habitat value and hence territory value in Sequoia-Kings Canyon 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 survival by 25% (from 0.9 to 0.675) resulted in a disproportionate 73% decline in average predicted female population size at Year 40 (Table H). Although none of these runs resulted in population extinction, as few as 6 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 25% decrease in equilibrium population estimate. If survival is simultaneously decreased by 25% across all age classes, the population crashes (15 of 20 runs resulted in extinction by Year 40, with no more than 4 adult females remaining in the other 5 runs). Simultaneously reducing fecundity across all ages in a similar manner had a much lesser effect, with only a 16.75% average decline in average population estimate and no extinctions.

Table H. Sensitivity of the equilibrium population estimate (number of adult females at Year 40) to demographic parameters. Each fecundity and survival rate was decreased by 25% while holding all others at default values (see Table B 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.

| Parameter Varied | Mean | SD | % Change |
|--------------------------|-------------|-----------|-----------------|
| None - Default | 126.30 | 7.29 | |
| S ₁ Fecundity | 122.50 | 7.74 | -3.01 |
| S ₂ Fecundity | 122.60 | 8.49 | -2.93 |
| S ₃ Fecundity | 110.45 | 10.70 | -12.55 |
| S ₀ Survival | 108.70 | 10.75 | -13.94 |
| S ₁ Survival | 102.75 | 7.38 | -18.65 |
| S ₂ Survival | 98.15 | 6.43 | -22.29 |
| S ₃ Survival | 34.25 | 9.96 | -72.88 |
| All Stages Fecundity | 105.15 | 9.86 | -16.75 |
| All Stages Survival | 0.45 | 1.00 | -99.64 |

Discussion

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 2-6 km wide south of the Merced River, and perhaps 6-15 km wide north of the Merced), mostly on the western slope of the Sierra, 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 and perhaps post-fire management (C. Hanson, Appendix B). For example, extensive brush fields and young coniferous forests in the southern portion of Stanislaus NF 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 NF (roughly 190,000 ha total) appears to be occupied by fishers, whereas large areas of predicted suitable habitat on the Stanislaus NF are not. There have been no fisher detections north of the Merced River, although all models show varying degrees of moderate to high-value habitat there (see Figures 3 and 4). This could be due to poor model performance (perhaps due to use of coarse or inaccurate predictor variables that don’t reflect actual habitat value well in that area) or 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 – None of the home-range-scale models we tested predicted much suitable habitat on the Kern Plateau, despite numerous and persistent fisher detections there (R. Truex, personal communications). This could be due to one or a combination of factors, such as 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). 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 (and according to C. Hanson, Appendix B., was not as heavily salvage-logged as fires in other areas). 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 western 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 apparent inaccuracies in predicting fisher occurrence 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. Nevertheless, we can identify particular microhabitat variables that may be useful to track in future vegetation dynamics modeling, to investigate how available resting microhabitat features (e.g., large black oaks or dead-wood biomass) are likely to change over time and space under various scenarios.

Baseline Population Assessment

Our population modeling results should be used with caution, primarily as hypotheses in need of further testing and refinement. Their utility lies mainly in informing future analyses and field studies, although they do provide some preliminary 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 and some suggestions for addressing them before we conclude with some management and research recommendations.

Coupling of Models and Amplifying Uncertainties – Concerns are often raised that coupling multiple models--as we do here in using habitat values derived from LAND14 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 LAND14 appears to undervalue habitat there relative to the monitoring data, and because we scaled demographic parameters with modeled habitat values, this likely compounds the LAND14 inaccuracies such that PATCH may be substantially 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%. We believe this issue

requires further investigation and that model results for the Kern Plateau and SEKI be viewed skeptically for the time being.

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. Aubrey, 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 NF 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 advisors (W. Zielinski and R. Barrett, personal communications). Large home range sizes also result in more conservative population estimates, a precautionary approach that is justified for a species of urgent conservation concern.

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 and also settled on 50 km.

Nevertheless, the dispersal algorithms in PATCH may not adequately account for important influences on dispersal behavior in fishers, and they assume 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 Aubrey 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 dramatic river canyons of the southern Sierra Nevada represent strong 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. 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.

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 ≥ 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 13 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 productivity²⁶. Note that above the ceiling territory value of 7.5 (where the maximum demographic parameters from the Leslie matrix apply), net value varied from 0 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 areas, whereas territories with higher than expected net values tend to be clustered within larger, high-value habitat areas.

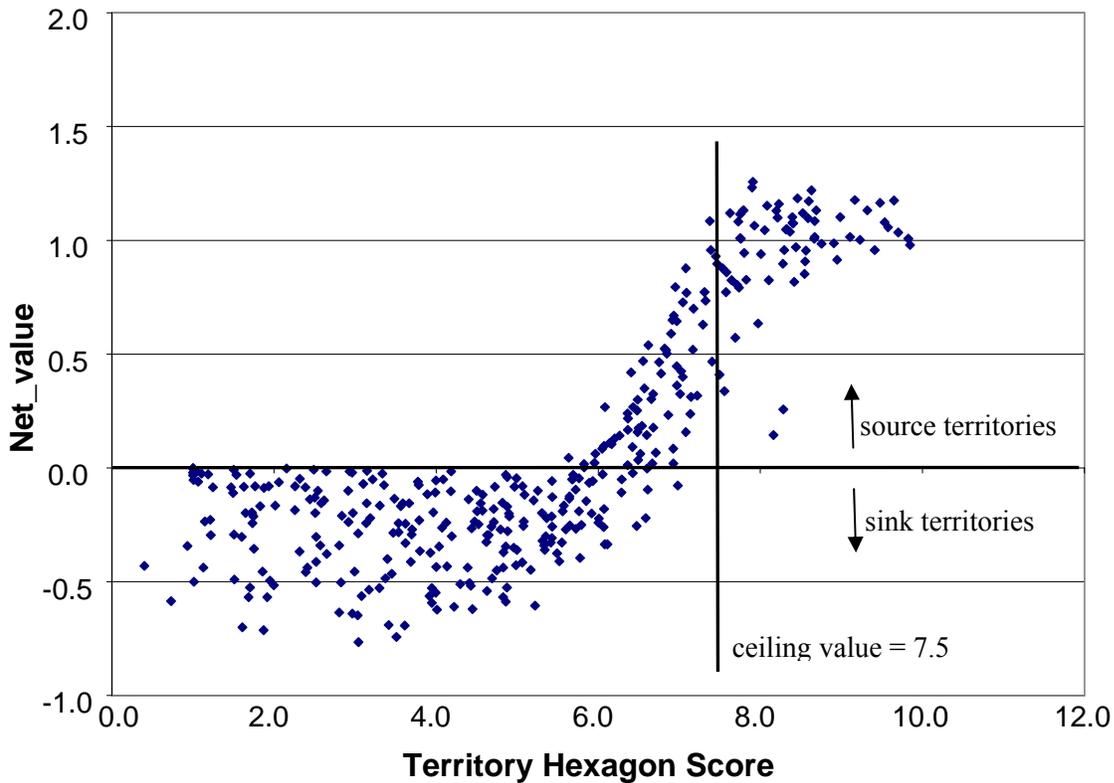


Figure 13. Scattergram showing relationship between territory hexagon score (assigned using Table A) and observed net value over model Years 21-40, where net value = territory-specific average of births minus deaths across all time steps in which a territory was occupied. (Territory size = 860 ha, initial 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 LAND14, adequately fits the fisher detection-nondetection data on the Kern Plateau (east of the Kern River on Sequoia NF). Fishers have been repeatedly 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,

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

personal communication). Because model LAND14 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 71 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 G). 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 142 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

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 NF.

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 150 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 averaged about 90 ($\pm \sim 20$) individuals per year for our default model assumptions. Adding these subadults to our estimate of adult population size yields a total population estimate of about 220 to 360 individuals in the southern Sierra Nevada (ignoring juveniles).

These estimates are fairly 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 NF, (2) west-side Sequoia NF and Sequoia-Kings Canyon NP, and (3) the Kern Plateau (primarily on Sequoia NF, extending slightly into southernmost Inyo NF). Based on this assumption and by varying the number of fishers detected 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 km². Extrapolating these density estimates over the area of predicted suitable and occupied habitat²⁸ we get total population estimates of 285 to 370 fishers (ignoring juveniles), of which 57 to 86 are adult females. Despite all the uncertainties in these static extrapolations, they also compare well with our estimates based on dynamic modeling.

²⁸ We applied the 0.5 cutpoint in probability of occupancy for model LAND14 south of the Merced River, yielding 285,163 total ha of predicted suitable, occupied habitat in the study area.

Future Work: Coupling Population Dynamics Models with LANDIS-II – We did not run PATCH further into the future than 20 years for these analyses²⁹, because they assume that the landscape and thus the habitat value map are static, whereas in reality vegetation and habitat value will be changing over time due to disturbances, growth, and succession. Vegetation dynamics are likely to have increasing effects on fisher population processes and patterns of occupancy with increasing duration, making inferences based on static habitat maps increasingly uncertain. In future analyses, we plan to couple PATCH (or potentially another spatially explicit population model, such as RAMAS GIS) with the landscape dynamics model LANDIS-II to explore how vegetation dynamics (due for example to fires, management, and vegetation growth) may affect the distribution, abundance, and viability of the fisher population. Note that as additional data accrue from the monitoring program, and especially from field studies of fisher population demography, genetics, dispersal, etc., we can apply these models with greater confidence to answer a wider variety of essential questions. The utility of these spatially explicit models will continue to expand as we learn more and can apply them to more refined management questions.

²⁹ Recall that although we actually ran the model for 40 years, the first 20 years are simply to equilibrate the dynamics, so that model Year 20 represents the current time, and model Year 40 represents 20 years into the future.

Management and Research Implications

We emphasize that the results of the models documented in this report should be viewed more as functional hypotheses subject to further testing than as black-and-white conclusions. However, given the extensive and rigorous approach to modeling, our testing of at least some assumptions, and the close scientific input and review by our Science Advisors, we offer some general conclusions and recommendations based on these results with some confidence:

Modeling Fisher Habitat at the Home-Range Scale – Utilizing the remarkable systematic fisher monitoring dataset compiled by the USDA Forest Service (Region 5), and the powerful multivariate tool of Generalized Additive Modeling (GAM), we were able to derive a highly predictive habitat suitability map at the home-range scale for fishers in the southern Sierra Nevada. The model uses a combination of two abiotic variables (adjusted elevation and insolation index) and two LANDIS-II biotic outputs (total tree biomass minus red fir and maximum tree age). This is encouraging, because with LANDIS-II providing strong predictive variables, it should now be possible to reliably evaluate the interaction of fuels management and wildfires on coarse-scale fisher habitat into the future. Further, coupling habitat change modeling with spatially explicit population models (such as PATCH or an alternative like RAMAS) will also allow us to investigate potential changes in population viability under alternative scenarios.

Fine-scale Habitat Value and Future Scenario Analyses – Future predictions about changes in fisher habitat or populations should not rely exclusively on home-range-scale models, because of the importance of fine-scale variation in habitat features to fishers. Unfortunately, we were not able to derive a reliable method of integrating fine-scale habitat models (i.e., the fisher resting microhabitat model using FIA data; Zielinski et al. 2006) with landscape-level models. Consequently, we recommend using LANDIS-II to track changes in the abundance and distribution of individual variables known to be important to fishers at the finer scale, such as the presence of large black oaks or other structures used for denning and resting (see recommendations of W. Zielinski, Appendix B). In the next phase of work, CBI should work with Science Advisors to identify key variables whose fates can and should be tracked using LANDIS-II in future scenarios (for example, biomass and age of black oaks, and biomass of dead wood by various tree species).

Population Size and Viability – Although there is uncertainty in the precision of any population estimate, the precision is not as important as the magnitude, and all reasonable estimates of the southern Sierra Nevada fisher population agree that the population is very small by the standards of population biology. Given that this population does not experience immigration from other regions, it is at risk of extirpation by a variety of stochastic influences, and likely needs to grow to avoid extinction.

Based on the various approaches used to estimate population size, and weighing the various uncertainties in their assumptions, we conclude that *the fisher population in the southern Sierra Nevada (ignoring juveniles) is between 160 and 360 total individuals (and probably less than*

300). *Of these, about 57 to 147 (and probably less than 120) are adult females, which comprise the most important class for sustaining a population.*

Importantly, however, we have not addressed the *effective population size* (N_e) of the population (Wright 1931) which requires genetic analyses. N_e is a measure of the rate at which genetic variation changes over time and is inversely proportional to a population's probability of extinction. Effective population size is smaller than actual population size, or even the size of the breeding population, as it is reduced by such influences on genetic variation as population fluctuations, uneven sex ratios, unequal contributions of gametes by individuals to the next generation, and age-structuring of populations. Genetic work should be used to determine N_e of the southern Sierra Nevada fisher population and its potential for future population growth. Combining genetic work with monitoring studies and other field research has proved a powerful tool for understanding important aspects of fisher biology (Aubry et al. 2004). Recent genetic work on fishers suggests that the Sierra Nevada population is genetically depauperate (Wisely et al. 2004), meaning that N_e will probably be quite low (W. Zielinski, personal communication). If effective population size is below about 50, extinction is probable over a relatively short term, and population augmentation may be warranted.

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 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.

Inaccurate Model Results for Kern Plateau – The Kern Plateau apparently has more suitable habitat and supports more fishers than our models predict. This should be addressed by further investigation and perhaps by developing separate models there with updated and/or higher resolution environmental data layers (W. Zielinski, personal communications).

Role of Kings River Administrative Project Area – The Kings River Administrative Project Area should be a focal area for increasing habitat value and contiguity. Fisher habitat between the Kings and San Joaquin Rivers is a relatively long and narrow stretch of variable but mostly moderate-quality habitat, with rather tenuous potential as a source population area under current conditions, according to our models. Further reductions or fragmentation of fisher habitat in this population segment may disrupt metapopulation dynamics sufficiently to increase extinction risks to the north and south (i.e., both north of the San Joaquin River and south of the Kings River) and consequently of the population as a whole.

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. Although increasing the amount or value of fisher habitat south of the Merced (e.g., in the Kings River Project Area) may result in modest increases in fisher population size and viability, it appears that northward expansion onto the Stanislaus NF has the greatest potential to significantly increase population size and hence viability. Given climate change, southward

expansion (e.g., into the Paiute Mountains) seems unlikely to contribute to population size or viability.

Our spatially explicit population models suggest that habitat on the Stanislaus NF, although not currently occupied by fishers, has potential to become occupied in the future, barring disturbances that may reduce the probability of fishers dispersing to and successfully colonizing from the south. Vegetation succession may increase the probability of northward expansion over the current condition, but a finer-resolution assessment would be required to identify specific locations where management intervention may or may not be beneficial.

How best to protect and enhance these connections is a focus of the next phase of analysis. The future dynamics modeling (using LANDIS-II) will explore tradeoffs between fuels treatments and wildfire on the amount and continuity of fisher habitat over space and time. If fuels treatments result in a net benefit to fishers and their habitat, these treatments may also decrease the risk of extirpation. In the meantime, it seems prudent to avoid or minimize logging or other activities, including salvage logging following fires, anywhere it may reduce fisher habitat value—especially in critical locations, such as the current northern frontier of the fisher population near the Merced River and on currently unoccupied areas of potential habitat north of the Merced.

Field studies and continued monitoring should also address dispersal characteristics of fishers in the study area and whether or to what degree steep canyons, sparse forest vegetation, roads, or other factors serve as barriers or filters to movement. If natural expansion to areas north of the Merced River is deemed unlikely, active translocation of fishers to portions of the Stanislaus NF may be prudent.

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