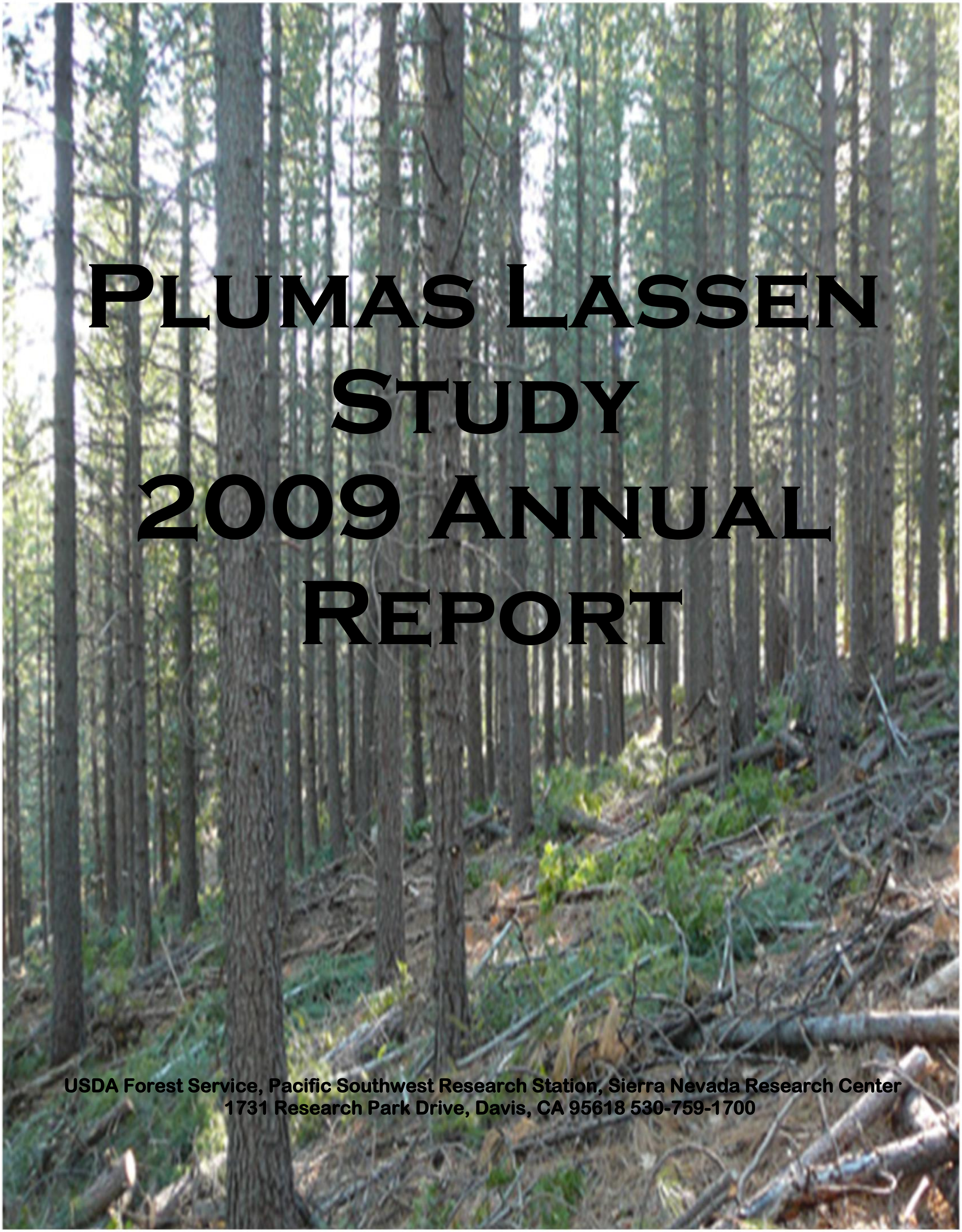

PLUMAS LASSEN STUDY 2009 ANNUAL REPORT

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Introduction

The Pacific Southwest Region and the Pacific Southwest Research Station agreed in 2002 to jointly develop and fund an administrative study to fill management information needs concerning the relationship between management-caused changes in vegetation and their effects on spotted owl habitat and population dynamics. The detailed discussions explaining how this program was started is provided in previous Annual Reports. Copies of previous Annual Reports for this program are available on the Sierra Nevada Research Center web site (www.fs.fed.us/psw/programs/snrc/forest_health) or upon request.

This is the eighth such Annual Report that we have compiled. The primary purpose of this is to provide a periodic synopsis of what we have been learning so all interested parties can remain abreast of the progress. Research products resulting from this effort will be disseminated as they are ready and this will vary from module to module, project to project, and from year to year. We expect that there will be a continuous flow of findings documented primarily with publications in both refereed journals and other publication outlets. The cadre of scientists, support staff, students, and others contributing to this effort will also be making oral presentations and providing other kinds of outreach materials to help inform interested parties and our peers on the results of this work.

We provide some review information here to reinforce the intent of our work. This background information provides a general overview on the purpose of this research program and helps set the context for the report. We have emerged from the initiation phase and we have collected an impressive amount of information. Many publications have been completed and some key ones are in development. We expect to provide useful information, particularly from the spotted owl module, in the immediate future. Of course much of our research purpose depends on forest management treatments to be put in place and then observe short and even long term response to those treatments. Such treatments are now being executed in some locations and thus some of our potentially most significant work has only recently begun. Observations of response after treatments will logically take place in the ensuing years. We anticipate that the main body of field research will conclude by the end of Fiscal Year 2012. New studies are being contemplated and we may collectively, in collaboration with land managers, choose to extend facets of this work beyond this date.

We recognize that response of different elements of the forest can occur immediately after treatments however it is also possible that response can occur slowly and not be recognized for some period of time depending on the response variable of interest. Alternatively it is also possible that some response variables exhibit a notable initial response and then return to a state similar to that of before the treatments. Thus we believe it is prudent to look at a fairly long period of post treatment response if possible, even if funding limitations require scheduling follow-up work in stages over time with periods of inactivity.

Purpose of the Study

This study is interdisciplinary by design, examining at least five groups of response variables (spotted owls, small mammals, terrestrial birds, vegetation, and fuels conditions) through collaboration between researchers of the USDA Forest Service Pacific Southwest Research Station (PSW) and cooperators from the Universities of California, Berkeley and Davis, and the PRBO Conservation Science. The study addresses some of the most significant uncertainties that confound management decisions in the Sierra Nevada today, including in the HFQLG Pilot Project Area. How do old-forest-dependent species respond to vegetation management over space and time? Do fuels management approaches effectively address fuels loadings without negatively affecting species viability? How effective are landscape level fuels management strategies in modifying fire behavior and reducing the extent and severity of wildland fire? These and related questions are the focus of the work being done in this study.

Objectives of Study

The original overarching objective of this proposed research was to address an array of related ecological questions in a coordinated, integrated effort, thereby providing empirical data to inform future management decisions. The landscape scale of this design was both the driving force addressing the key questions as well as the largest impediment to successful construction of a scientifically credible experimental design and implementation in the field. Our research team believes that assessing many of the key elements of forest ecosystems should be done over larger spatial and temporal scales than has typically been investigated in past research. The important difference we are investigating is the response to changes in forest structure and composition over space and time rather than simply site specific and immediate response. We believe this difference is especially relevant to forest management practices that are designed for large landscapes, executed over relatively long time frames, such as landscape level fuels treatment strategies.

This research program is designed to address the three principal issues described below. These issues are specifically addressed through research questions and attending investigational approaches tailored for five different research components of this research program. These specific questions are detailed in the individual study plans for each module. Here we simply highlight the main objectives of the integrated research program and summarize the primary research questions that we plan to pursue.

- ***Wildland Fire Behavior and Protection.*** How do landscape level fuels and silvicultural treatments affect potential fire behavior and effects? Are specific combinations of defensible fuel profile zones (DFPZs) and subsequent individual tree selection or area treatments to thin the matrix effective in reducing the extent and severity of wildland fires? Are realized fire management benefits consistent with hypothesized results in reducing fire risk and altering fire behavior?

- ***Landscape Dynamics.*** How do combinations of DFPZs, subsequent individual tree selection or area treatments to thin the matrix, group selection, riparian protection standards, and species-specific protection measures affect landscape dynamics such as forest structure, composition, and succession at multiple scales of space and time?

- ***Species Viability.*** Induced by a forest management regime, how will old-forest dependent species, particularly the California spotted owl and its prey base comprised of various species of small mammals, respond to changes in vegetation composition, structure, and distribution over space and time? How is response to treatments manifested at the individual and population levels of biological organization?

In 2009 the PLAS program made some adjustments as a result of both the normal course correction that a research program experiences as well as a conscious shift to be more responsive to the Storrie Fire restoration efforts. A good portion of the funding that supports the PLAS efforts is now coming from the Storrie Fire settlement funds thus we have shifted some of our priorities to specifically address restoration considerations with respect to a) forest management approaches to prevent large, high severity fires and b) management/restoration requirements after fires of varying severities. Below we provide brief summary statements that capture the essence of the questions we are currently pursuing under this revised research agenda. The specific reports under each of the modules will reflect this shift, to the extent that each module has changed in 2009.

Fuels and Fire Module

1) Quantitatively characterize the pattern of fire severity and erosion following the 2000 Storrie fire.

2) (a) Assess change in protected wildlife habitat resulting from recent large wildland fires in the northern Sierra Nevada/southern Cascades
(b) Explore the modification of both potential fire behavior and owl habitat suitability in areas treated for fuels reduction.

3) Retrospectively analyze effects of proposed fuel treatment strategy on Storrie fire and assess longevity of fuel treatment effectiveness.

4) Assess the potential change in fire regime of the northern Sierra Nevada and southern Cascades under changing climatic conditions.

Vegetation Module

1) Seed dispersal and seedling survival models for reforestation. Foresters need to know distances tree seeds will travel from a parent tree, and probability of survival of established seedlings, so they can predict natural regeneration after

canopy–opening disturbance such as high-intensity fire or group-selection silviculture.

2) Follow-up on Experimental Thinning and Group Selection Stands. Understory vegetation in experimentally treated stands in the first phase of the Veg Module will be remeasured 5 years after treatment

3) Large Tree Survival and Growth. What tree and site conditions are associated with higher survival rates and rapid growth after high-intensity burns?

Small Mammal Module

1) Habitat Associations of Small Mammals. Complete a spatially extensive survey of small mammals at a large number of point count transects established by PRBO biologists.
2) Long-term Small Mammal Monitoring at Focal Trapping Grids. Continue with our annual surveys at 12 sites that were established to assess biotic responses to two levels of thinning.
3) Comprehensive Demographic Research on Key Prey Species. Complete efforts to assess the habitat selection preferences of northern flying squirrels in the northern Sierra.

Bird Community Module

1) What are the short-term and predicted longer term local effects of DFPZ treatments on a suite of landbird species in west-side Sierran forest?

2) What are the short and medium term effects of wildfire (of varying intensity) and post-fire management on avian species abundance and distribution?

3) What are the most desired future conditions to ensure an ecologically stable system that supports the range of habitat types and attributes upon which wildlife depend?

California Spotted Owl Module

1) We are continuing to monitor the distribution, abundance, and demographics of California spotted owls across the baseline PLS study area, including our first year of monitoring owl post-treatment response in the seminal Meadow Valley Project Area and baseline monitoring in the highly controversial Creeks Project Area on the Lassen National Forest.

2) A second year of post-fire monitoring was conducted in the Moonlight-Antelope Complex Fire Areas, approximately 88,000 acres (35,612 ha) that burned at mostly high severity in 2007. The fire reduced the amount of suitable owl habitat from 65-70% of the landscape to about 6% following the fires.

3) Continued demographic monitoring on the Lassen Demographic Study Area provides an irreplaceable source of empirical data on CSO survival, reproduction, recruitment, and population trends that serves to address continuing interests and legal challenges regarding the status of the CSO in the HFQLG Project Area and the Sierra Nevada.

4) Measure the associations and responses of CSOs to the full range of landscape conditions comparing treatment versus wildfire risk. Expand our CSOs research into the Cub-Onion Fire Area (Lassen) and into the Scotts John Creek (Lassen) and Empire (Plumas) Project Areas. This work expands our base PLS work so that we have empirical data on CSO distribution and abundance across landscape conditions ranging from untreated, treated (current or projected to be treated soon), burned at low-moderate severity (Cub-Onion Fire Area) and burned at high severity (Moonlight-Antelope Complex Fire Area). Results from this work will allow us to provide real empirical data on how CSOs are associated with these landscape conditions and advance the discussion regarding treatment versus fire risk.

5) In conjunction with biologists on the Plumas and Lassen NFs we are considering field surveys to assess the distribution and abundance of CSOs in some portions of the Storrie Fire Area. This information will provide insight into the association of CSOs with this burned landscape 8-10 years following wildfire.

Progress to Date

Given that we have completed an eighth year of work many findings have taken shape and others are well along in their development. Some results, based on primarily pretreatment data, are crystallizing and findings are being reported. Some of the work described here includes activities from other locations but are potentially relevant to the Plumas and Lassen National Forest landscape, thus they are included in this summary. Each module will report on their respective findings and publications.

Summary

This work represents some significant scientific study that has occurred over the last eight years. Our original expectation was to continue for up to another three years within the HFQLG Pilot Project area to capture adequate post-treatment data. When we began this study the pilot project was scheduled to end in 2005 and since then it has been extended twice, now to 2012 to enable the complete pilot project to be implemented. If funding support persists we will continue to pursue field work through 2012. Upon completion of the field work the remainder of the effort will be devoted to data analysis and reporting. Any future work that is deemed mutually desirable may continue (or start anew) as this phase of work winds down. That will depend on the interest of forest managers and the availability of funding. As alluded to above we are planning to initiate a new study, beginning with site selection in 2010, to examine response of riparian areas to various approaches of restoration. We expect this study will continue for approximately five years. If other questions arise we will consider addressing those as well.

We understand there is some uncertainty and sometimes controversy over how various forest elements will respond to planned forest management practices. This is likely to be the case under any chosen management regime. The objective of PSW was to tackle the difficult scientific challenges derived from the salient management questions. PSW, as a research organization, remains wholly objective in executing this charge. We have assembled an excellent team of scientists with the appropriate areas of expertise and we have done the best we can to design our work to address the important questions. Many of these questions present significant challenges to experimental design of field ecology experiments and management constraints further constrain our ability to test questions with traditional hypothesis testing approaches. We expect to make the most of these opportunities in advancing our scientific understanding of forest ecosystem response to management practices.

Plumas-Lassen Administrative Study Terrestrial Bird Module:

2009 Monitoring Report

February 2010

Ryan D. Burnett, Paul Taillie, and Nathaniel Seavy
PRBO Conservation Science

Contribution Number 1726

Executive Summary

In 2009 the avian module of the Plumas-Lassen Administrative Study (PLAS) expanded to address important questions related to post-fire habitat and its management. The primary objective of this new part of the study is to assess the influence of post-fire conditions on spatial and temporal variation in bird abundance, and to use this information to inform forest management practices that can maintain avian diversity across multiple spatial scales. We began sampling three areas affected by fire within the boundaries of the original PLAS study: the Storrie, Moonlight, and Cub fires.

Avian species richness and total bird abundance were higher in green forest than all fire areas combined though a similar number of species were detected in fire areas as in green forest. The highest total bird abundance recorded in 2009 in our study area and highest species richness in any fire area was found in high severity areas of the Storrie fire – nine years after it burned.

The abundance of avian species in post-fire habitat was considerably different than green forest in the study area. Eight species were significantly more abundant in post-fire habitat whereas eleven were in green forest of the PLAS. Hairy Woodpecker - a Management Indicator Species in the Sierra Nevada – and White-headed Woodpecker were significantly more abundant in post-fire habitat compared to green forest. Black-backed Woodpecker were only detected within fifty meters of observes in the Moonlight fire though we have observed them in both the Cub and Storrie fire and rarely in the PLAS green forest.

A number of cavity nesting species showed a preference for certain tree species and large diameter trees. All cavity nesting species we studied showed a preference for decayed snags which were not readily available in both of the younger fires (Cub and Moonlight) regardless of the severity at which a plot burned. Nesting densities of cavity nesting species increased as the area of the plot with greater than 50% overstory tree mortality increased.

In the Sierra Nevada considerable debate surrounds the management of post-fire habitat. As the area affected by wildfire appears to be increasing after nearly a century of suppressed fire activity, a greater understanding of the value of these habitats and the critical habitat elements required by the unique and relatively diverse avian community will be important to sustaining biological diversity in the Sierra Nevada. Post-fire habitats are necessary components of the Sierra Nevada ecosystem that support a unique, diverse, and abundant avian community that should be considered when managing these areas.

Post-Fire Habitat Management Recommendations

- Restrict all activities that may disturb breeding birds to the non-breeding season (August - April)
- Consider post-fire habitat as important component of the ecosystem necessary for maintaining biological diversity in the Sierra Nevada
- Consider the area of a fire that burned in high severity, as opposed to the area of the entire fire, when determining what percentage of the fire area to salvage log
- Consider the landscape context (watershed, ranger district, forest, ecosystem) and availability of different habitat types when planning post-fire management actions
- Manage a portion of post-fire areas for large patches (minimum of 50 acres or 20 ha) of high severity habitat
- Retain high severity areas in locations with higher densities of larger diameter trees and existing snags with relatively high levels of decay
- Manage a portion of post-fire areas for early successional shrub and herbaceous dominated habitats and natural regeneration of conifers
- Retain snags in salvaged areas greater than green forest standards of 4 snags per acre
- Retain some patches of high severity fire adjacent to intact green forest patches as the juxtaposition of unlike habitats is positively correlated with a number of avian species including declining species such as Olive-sided Flycatcher, Western Wood-Pewee, and Chipping Sparrow
- Manage post-fire areas for diverse and abundant understory plant community including shrubs, grasses, and forbs. Understory plant communities provide a unique and important resource for a number of species in a conifer dominated ecosystem
- Retain snags (especially large pine trees that decay slowly) in areas being replanted as they can provide the only source of snags in those forest patch for decades to come and we have found that scattered large snags in plantations surrounded by forest are readily used by cavity nesting birds

Introduction

The primary objective of the landbird module of the Plumas-Lassen Administrative Study is to assess the impact of forest management practices in sustaining a long-term ecologically stable forest ecosystem at the local and landscape scales. We know the avian community in the Sierra Nevada is comprised of species that are associated with a wide range of forest seral stages, vegetative composition, and structures. This vegetation, and hence avian diversity, is constantly changing as a result of natural disturbances (primarily fire) that create a dynamic and diverse ecosystem. Therefore, it is imperative for managers to consider how natural disturbance events interact temporally and spatially with management actions, and how ecological integrity can be achieved in an inherently dynamic system.

In the Sierra Nevada, there is a pressing need to understand the nexus of silvicultural practices, wildfire, and fuels treatments in order to maintain forested ecosystems that are ecologically diverse and resilient. In the context of a century of fire suppression, at the core of the debate over how to manage Sierra forests is the debate over how to most appropriately manage areas where natural disturbances have been disrupted. Forest Service managers need a better understanding of the suitability of habitat created through fire suppression, fuel treatments (DFPZ, groups, mastication), and wildfire and post wildfire management.

The challenge of integrating wildfire and forest management into wildlife conservation is not unique to the Sierra Nevada. Because large, infrequent disturbances are responsible for long-lasting changes in forest structure and composition (Foster et al. 1998), they are recognized as a critical element of bird community dynamics (Brawn et al. 2001). In many regions of western North America, fires burn with considerable spatial and temporal variability (Agee 1993), creating complex mosaics of vegetation patches. In these systems, changes in bird abundance are often linked to post-fire vegetation characteristics and landscape composition (Saab et al. 2002, Huff et al. 2005, Smucker et al. 2005).

In addition to fire suppression, there are a number of management activities that influence post-fire vegetation characteristics and landscape composition in working forests. These activities include salvage-logging, the mechanical mastication and herbicidal treatments to reduce broadleaf shrubs, and planting of conifer species that are favored by forestry. As a result, management activities may have profound influences on post-fire conditions- locally and across the landscape.

Beginning in 2009 the avian module of the Plumas-Lassen Administrative Study (PLAS) expanded to address important questions related to post-fire habitat and its management. The primary objective of this new part of the study is to assess the influence of post-fire conditions on spatial and temporal variation in bird abundance, and to use this information to inform forest management practices that can maintain avian diversity across multiple spatial scales. We began sampling three areas affected by fire within the boundaries of the original PLAS study: the Storrie Fire that burned in the Fall of 2000, the Moonlight Fire that burned in the Fall of 2007, and the Cub Fire that burned in the Summer of 2008. Each of these fires burned at similar elevations and through primarily mixed conifer and true fir vegetation communities but with varying intensity patterns. This report provides results from the first year of fire monitoring and uses ongoing monitoring of unburned or “green” forest in the study area to provide context.

Methods

Study Location

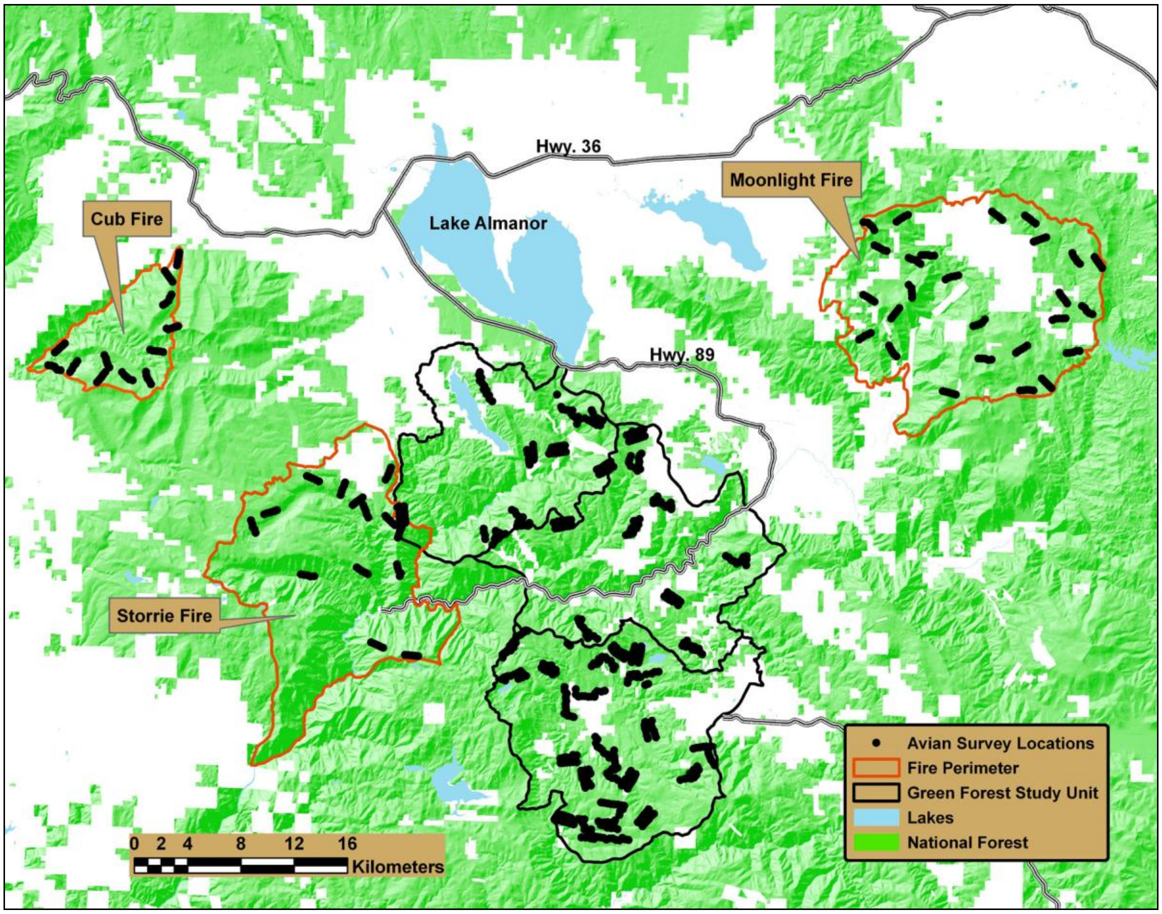
The Plumas-Lassen Area Study avian module study encompasses portion of the Mount Hough Ranger District of Plumas National Forest and the Almanor Ranger District of Lassen National Forest in the Sierra Nevada Mountains of Northeastern California (Figure 1). In 2009 we added three separate burned areas to our study within this same area. The elevations of sites surveyed ranged from 1126 – 1998m (0.7-1.2 mi) with a mean of 1658 in the Cub fire, 1199 – 2190m (0.7-1.4 mi) with a mean of 1779 in the Moonlight Fire, 1107 – 2011m (0.7-1.2 mi) with a mean of 1528 in the Storrie fire, and 1094 – 1902m (0.7-1.2 mi) with a mean of 1483 at the existing PLAS green forest sites.

Site Selection

A total of 52 transects (260 stations) were established across the three fires. Twenty-six transect were surveyed in the Moonlight fire, 12 in the Cub Fire, and 13 in the Storrie Fire. Site selection for PLAS green forest study sites followed a similar random selection protocol except each transect contained 12 points instead of five and approximately 25% of transects were systematically established in areas where treatments were planned (many now implemented). The PLAS site selection protocol for the unburned “green forest” sample is described in detail in the study plan and previous annual reports (Stine et al. 2005, Burnett et al. 2009).

Random starting points for each fire transect were generated in ArcGIS 9.2 within the boundaries of each fire. The sampling area was limited to forest service land and sites with a slope of less than 40 degrees to allow access and safe navigation on foot in a timely manner. We maintained a minimum distance between transect starting points of 1500m (0.9 mi) to ensure transects would not overlap and maximize a spatial balance within the sampling frame of each fire. Four more points were added to the starting point on a random compass bearing at 250m (273 yd) spacing resulting in a 1km (0.6 mi) long five point transect.

Figure 1. The location of PRBO study sites in the Plumas-Lassen study area in 2009.



As we attempted to implement our sampling protocol logistical constraints necessitated minor deviations from the methods described above in order to establish the minimum number of transects desired for each fire. In the Storrie Fire, two transects that were in remote areas (ST8 and ST7) and accessible only by the Pacific Crest Trail were moved from the random location to the trail or adjacent to the trail due to rugged un-navigable terrain. In three cases of extreme topography (two in Cub Fire and one in Storrie Fire; CB13, CB14, and ST12) we moved the transect to a more navigable location in the same general area. Five originally selected transects (ST1, ST2, ST3, ST5, and ST11) were dropped after field crews were unable to safely access or navigate them and replaced with new transects that were in locations of known accessibility. Finally there were five transects (ML8, CB4, CB6, CB15, and ST14) that were dropped completely as we selected a few extra transects in each fire assuming some would have to be dropped. As a result of these terrain limitations, fairly large sections of the Cub and especially the Storrie fire were not surveyed (see Figures 1 & 2).

Figure 2. Location of PRBO point count locations overlaid on composite burn index fire severity maps for each of three fires in the study area. Red = high intensity, Orange = moderate, Lime = low intensity, and green is low to unburned.

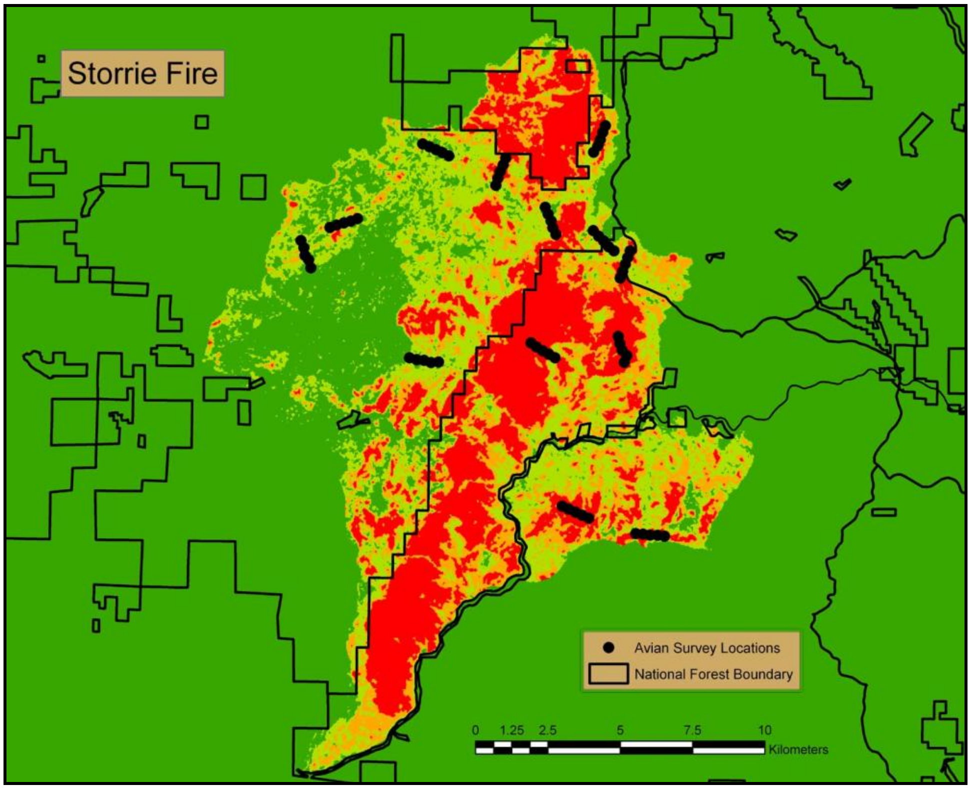
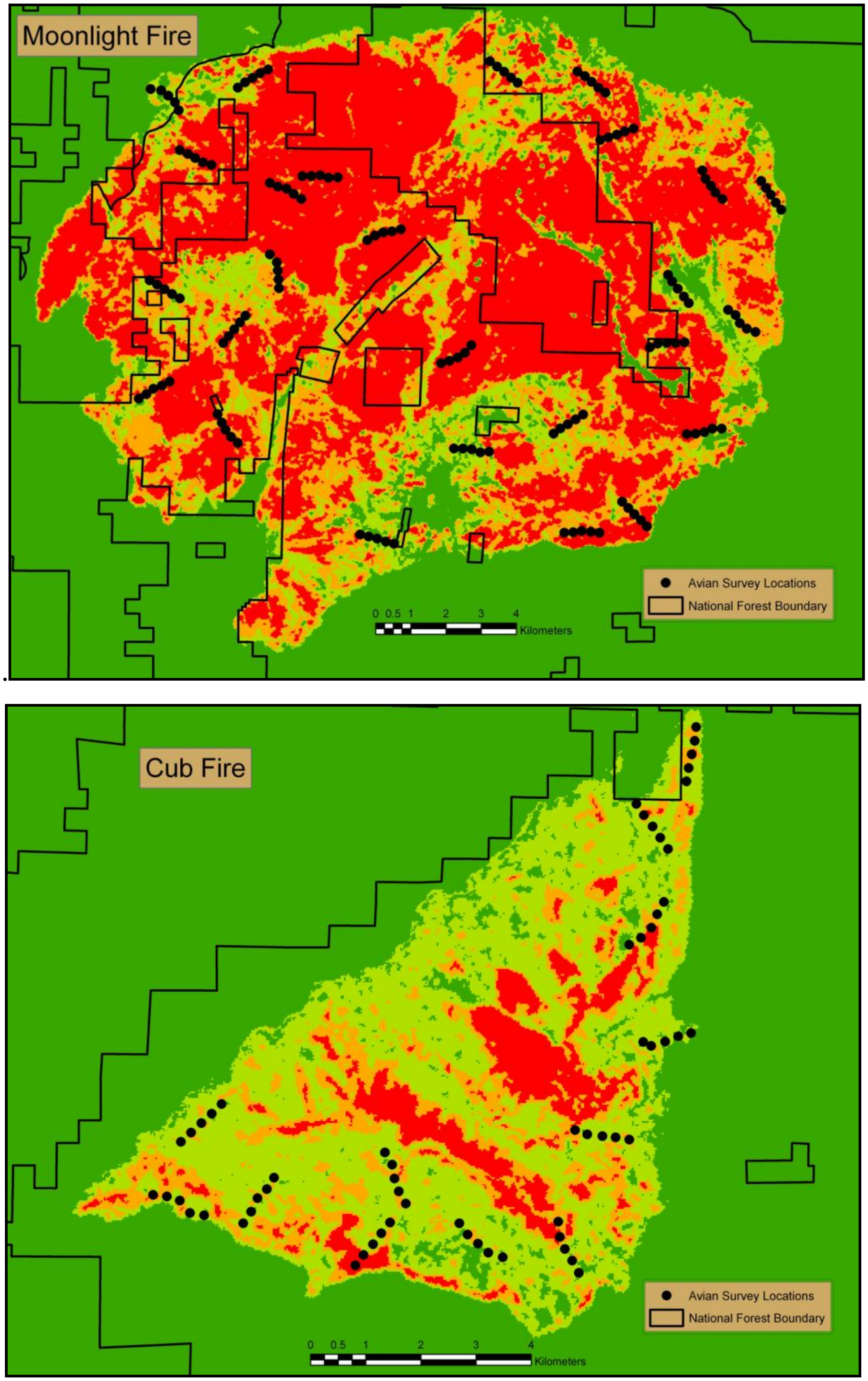


Figure 2 continued.



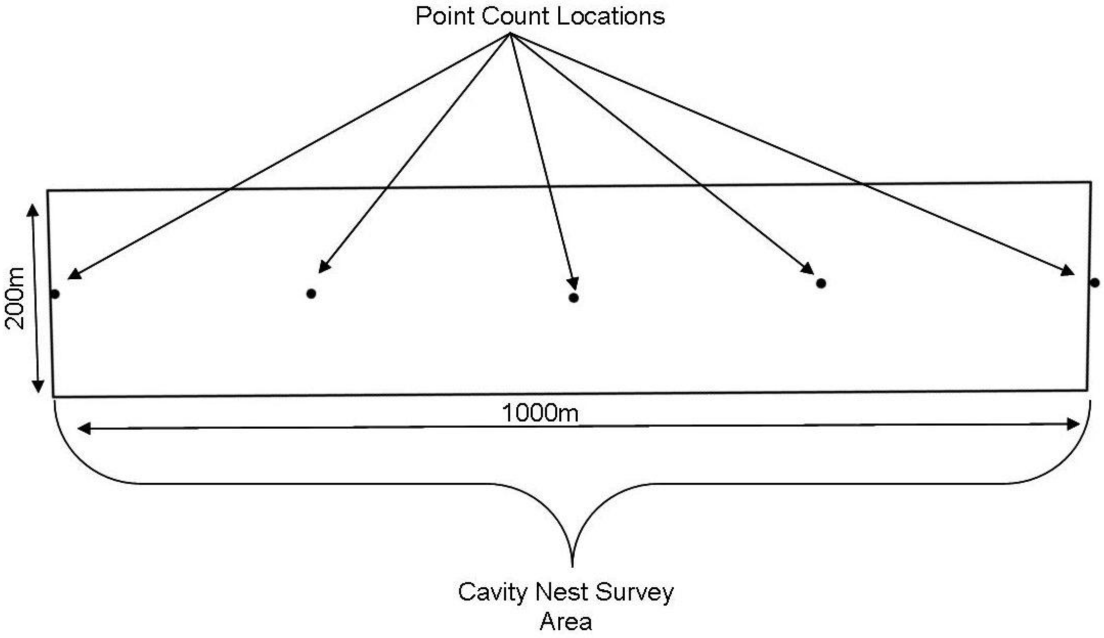
Bird community surveys

The avian community was sampled using a five minute exact distance point count census (Reynolds et al. 1980, Ralph et al. 2005). In this method points are clustered in transects, but data are only collected at the individual stations. All birds detected at each station during the five-minute survey were recorded according to their initial distance from the observer. The method of initial detection (song, visual, or call) for each individual was also recorded. All observers underwent an intensive three week training period focused on bird identification and distance estimation prior to conducting surveys. Laser rangefinders were used to assist in distance estimation at every survey point. Counts began around local sunrise, were completed within four hours, and did not occur in inclement weather. Each transect was visited twice during the peak of the breeding season from mid May through the first week of July (Appendix C).

Cavity nest surveys

In addition to the point count census, at each fire transect a 20ha area (200m x 1000m or 0.1mi x 0.6mi rectangle) was surveyed for nests of cavity-nesting birds following the protocol outlined in “A field protocol to monitor cavity-nesting birds” (Dudley and Saab 2003). After the point count census was complete, the nest survey was conducted for between two and four hours depending on the habitat and terrain and time spent waiting to confirm a cavities status. All nest surveys were completed by noon. The primary search method for finding nests was bird behavior though once suspicious birds were located observers often conducted a systematic search of snags in the vicinity. Once a potential nest was found it was observed from a distance for a maximum of 20 minutes to confirm the cavity was an active nest. If that cavity was confirmed active a variety of characteristics of both the nest tree and the cavity were recorded. These characteristics included diameter at breast height (DBH), tree height, cavity height, tree decay class, and the orientation of the cavity opening. For tree decay, we used a qualitative scale of decay ranging from one to eight one being a live, intact tree and eight a severely decayed stump (see Figure 7). If the observer was unable to confirm the cavity was active, its location was recorded to aid nest searchers during the second visit. Only confirmed active nests were used in analysis presented herein.

Figure 3. PRBO Northern Sierra post-fire habitat survey plots.



Vegetation surveys

Vegetation data was collected in all post-fire plots (with the exception of two in Moonlight) in 2009. We measured vegetation characteristics within a 50 m (164 ft) radius plot centered at each point count station following the relevé protocol outlined in the original PLAS bird module study plan (Stine et al. 2005). On these plots we measured shrub cover, live tree cover, and herbaceous cover as well as the relative cover of each species in the shrub and tree layers through ocular estimation. We also collected basal area of live trees and snags using a 10-factor basal area key. To estimate the density of snags across the plot, we recorded data (e.g. DBH, species, height, decay) on every snag within 11.3m (37 ft) of the center of the point count. In addition to the point count stations, we selected five random points in every plot, and collected the same snag data within 11.3m (37 ft). At the center of this random plot, a tree was chosen at random to serve as the control for the nest tree. The same descriptive characteristics that were collected for nest trees were also collected for these random trees at the center of the random snag plots.

Analysis

A per point index of abundance (detections within 50m or 164ft of observer summed across two visits) at all 260 point count station was calculated for 28 species that occurred in at least one of the study areas. The species selected are comprised of all of the California Partner's in Flight Coniferous Forest Focal species, (CALPIF 2002), for which we had adequate detections to conduct meaningful analysis, as well as three woodpecker species, and a suite of other species that were among the most abundant in the study area or showed some large differences between post-fire habitat and green forest.

In order to quantify the overall songbird community in the study areas we used three different metrics, the Shannon Index of species diversity, species richness, and total bird abundance. The Shannon index used a transformation of Shannon's diversity index (or H', Krebs 1989) denoted N_1 (MacArthur 1965). The transformation expresses the data in terms of number of species and thus is more easily interpreted. Expressed mathematically:

$$N_1 = e^{H'} \text{ and } H' = \sum_{i=1}^{i=S} (p_i)(\ln p_i)(-1)$$

Where S = total species richness and p_i is the proportion of the total numbers of individuals for each species (Nur et al. 1999). High Shannon index scores indicate both high species richness and more equal distribution of individuals among species. Species richness is defined simply as the number of species detected within 50m (164ft) of each point summed across the two visits and total bird abundance is the sum of all species detected per visit within 50m (164 ft) of observers. All species that do not breed or naturally occur in the study area and those that are not adequately sampled using the point count method including waterfowl, shorebirds, and raptors were excluded from each calculation. In addition, we calculated these metrics for each fire according to fire severity- as determined from on the ground vegetation surveys-and defined by percent overstory tree mortality. High severity was classified as sites with greater than 50% tree mortality and moderate to low severity being less than 50% tree mortality. These metrics were investigated for each fire and the Plumas Lassen Administrative Study green forest sample (Table 2).

Results

Bird community composition

A total of 97 species were recorded during point count censuses in the study area in 2009: 59 species on the 60 stations in the Cub fire, 72 species on the 130 Moonlight stations, 61 species on the 65 Storrie stations, and 78 species on the 468 PLAS green forest stations. Once species not adequately sampled using the point count method were removed (see paragraph above), there were 55 species in the Cub, 64 in the Moonlight, 55 in Storrie, and 61 in PLAS green forest. No species were unique to the Cub fire, seven species (Ash-throated Flycatcher, Canyon Wren, Cedar Waxwing, Clark's Nutcracker, Gray Flycatcher, Pygmy Nuthatch, and White-crowned Sparrow) were unique to the Moonlight fire, four species (Blue-gray Gnatcatcher, Chestnut-backed Chickadee, Vesper Sparrow, and Western Bluebird) were unique to the Storrie fire, and four species (Brewer's Blackbird, Red-winged Blackbird, Swainson's Thrush, and Western Meadowlark) were unique to the PLAS green forest sample.

Of the 28 species investigated, the index of abundance was highest in one of the fires for 21 of the species and higher in green forest for seven of the species. When all fires were combined, 16 species (8 significant) were more abundant in fire areas compared to 12 (11 significant) in green forest. The species with the highest index of abundance in the Cub fire were Mountain Chickadee, Red-breasted Nuthatch, Oregon Junco, and Audubon's Warbler and Hermit Warbler. In the Moonlight fire the most abundant were Lazuli Bunting, Oregon Junco, Dusky Flycatcher, Fox Sparrow, and Western Tanager. In the Storrie fire they were Lazuli Bunting, Oregon Junco, Fox Sparrow, Spotted Towhee, and MacGillivray's Warbler. In the PLAS green forest study area they were Hermit Warbler, Nashville Warbler, Audubon's Warbler, Dusky Flycatcher, and Oregon Junco.

Hairy Woodpecker - a Management Indicator Species in the Sierra Nevada – and White-headed Woodpecker were significantly more abundant in post-fire habitat compared to green forest. Black-backed Woodpecker were only detected within fifty meters of observes in the Moonlight fire though we have observed them in both the Cub and Storrie fire and rarely in the PLAS green forest. Lewis' Woodpecker, which we have never detected in the PLAS green forest study area, was fairly common in both the Moonlight and Storrie Fire areas.

Western Wood-Pewee, Mountain Bluebird, American Robin, Lazuli Bunting, Chipping Sparrow, and Cassin's Finch were all significantly more abundant in post-fire areas than PLAS green forest sites. Hammond's Flycatcher, Dusky Flycatcher, Cassin's Vireo, Mountain Chickadee, Red-breasted Nuthatch, Nashville Warbler, Audubon's Warbler, Golden-crowned Kinglet, Black-headed Grosbeak, Western Tanager, and Hermit Warbler were all significantly more abundant in PLAS green forest than all fire areas combined. For many of these species there abundance varied between each of the fires and many of these effects were significant (Table 1).

Table 1. An index of the abundance (detections within 50m of observers per station summed across 2 visits) for 28 species in each of three burned areas and the adjacent unburned Plumas-Lassen Area Study (PLAS) in 2009. Species are listed in taxonomic order and means are presented with standard errors.

Species	CUB	MOONLIGHT	STORRIE	PLAS
Calliope Hummingbird	0.00	0.11 (.03)	0.08 (.03)	0.05 (.01)
Hairy Woodpecker	0.15 (.06)	0.35 (.07)	0.12 (.05)	0.05 (.01)
White-headed Woodpecker	0.28 (.09)	0.11 (.04)	0.11 (.04)	0.04 (.01)
Black-backed Woodpecker	0.00	0.05 (.02)	0.00	0.00
Olive-sided Flycatcher	0.05 (.03)	0.05 (.02)	0.03 (.02)	0.03 (.01)
Western Wood-Pewee	0.05 (.03)	0.14 (.04)	0.20 (.08)	0.05 (.01)
Hammond's Flycatcher	0.27 (.07)	0.22 (.05)	0.03 (.02)	0.41 (.03)
Dusky Flycatcher	0.30 (.09)	0.72 (.09)	0.23 (.06)	0.69 (.05)
Cassin's Vireo	0.03 (.02)	0.03 (.02)	0.08 (.04)	0.36 (.03)
Warbling Vireo	0.23 (.09)	0.18 (.06)	0.09 (.06)	0.14 (.02)
Mountain Chickadee	0.78 (.14)	0.41 (.08)	0.29 (.08)	0.66 (.05)
Red-breasted Nuthatch	0.68 (.10)	0.16 (.04)	0.34 (.08)	0.56 (.04)
Brown Creeper	0.35 (.08)	0.17 (.04)	0.32 (.11)	0.19 (.02)
Golden-crowned Kinglet	0.27 (.09)	0.12 (.04)	0.17 (.06)	0.59 (.04)
Mountain Bluebird	0.02 (.02)	0.17 (.05)	0.02 (.02)	0
American Robin	0.08 (.04)	0.17 (.04)	0.34 (.09)	0.10 (.02)
Nashville Warbler	0.12 (.06)	0.15 (.04)	0.34 (.10)	0.78 (.05)
Audubon's Warbler	0.58 (.11)	0.37 (.06)	0.29 (.08)	0.69 (.05)
Hermit Warbler	0.37 (.09)	0.30 (.07)	0.15 (.05)	1.26 (.06)
MacGillivray's Warbler	0.17 (.05)	0.21 (.06)	0.40 (.09)	0.33 (.03)
Western Tanager	0.22 (.07)	0.46 (.07)	0.14 (.06)	0.56 (.04)
Black-headed Grosbeak	0.08 (.04)	0.04 (.02)	0.20 (.07)	0.19 (.03)
Lazuli Bunting	0.02 (.02)	0.79 (.11)	0.83 (.21)	0.04 (.01)
Spotted Towhee	0.15 (.07)	0.07 (.03)	0.52 (.14)	0.13 (.02)
Chipping Sparrow	0.00	0.25 (.05)	0.23 (.08)	0.06 (.01)
Fox Sparrow	0.17 (.07)	0.46 (.10)	0.69 (.19)	0.43 (.06)
Oregon Junco	0.60 (.11)	0.77 (.10)	0.72 (.13)	0.69 (.04)
Cassin's Finch	0.02 (.02)	0.09 (.03)	0.06 (.06)	0.02 (.01)

The Shannon index of diversity ranged from 5.77 in PLAS to 4.56 in the Cub Fire (Table 2). Similarly, species richness ranged from 6.37 in PLAS to 4.92 in the Cub Fire. Total bird abundance ranged from 5.08 in PLAS to 3.55 in the Cub Fire. Comparing these metrics by fire severity between each fire all three indices were higher in low severity areas of the Cub and Moonlight fires and higher in high severity areas of the Storrie fire (Figure 4). Species richness was highest in green forest of the PLAS and high severity areas of the Storrie fire, Shannon diversity was highest PLAS green forest, and total bird abundance was highest in the high severity areas of the Storrie fire.

Table 2. The mean per point Shannon diversity and species richness indices for the three fires and the Plumas Lassen Administrative Study in 2009 with standard error.

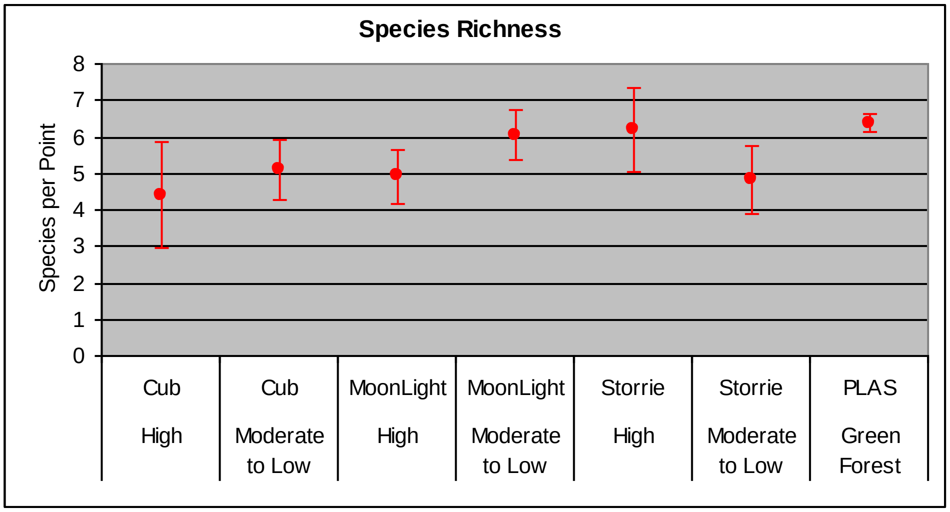
Metric	CUB	MOONLIGHT	STORRIE	PLAS
Diversity	4.56 (.31)	4.94 (.22)	4.77 (.30)	5.77 (.11)
Richness	4.92 (.34)	5.39 (.25)	5.38 (.35)	6.37 (.12)
Total Bird Abundance	3.55 (.28)	4.12 (.35)	4.58 (.35)	5.08 (.12)

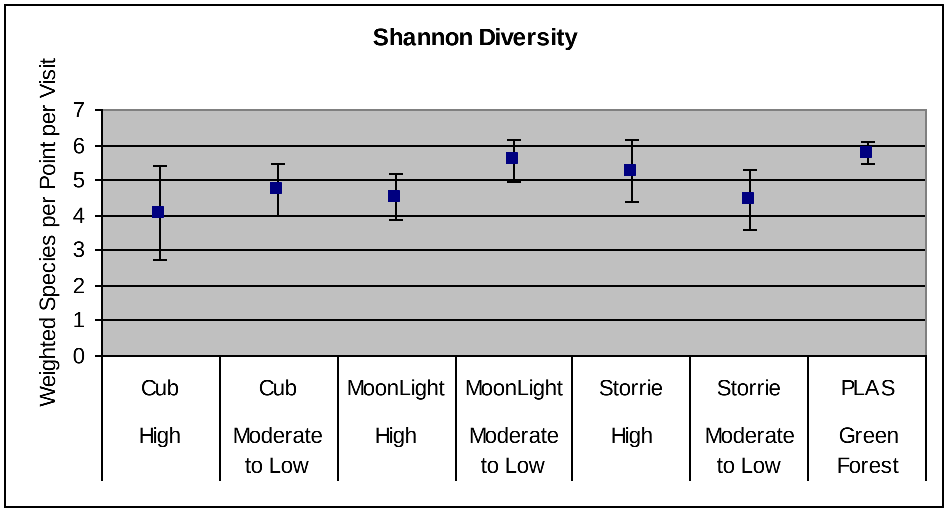
Cavity nest characteristics

A total of 64 active cavity nests were confirmed in 2009. The nest trees selected were quite variable in both size and species. The diameter at breast height (DBH) ranged from 12cm (4.7 inches) to 130cm (51.2 inches) and the height off the ground ranged from 2 to 31 meters (7-102 ft). Nests were located in at least nine different tree species and we compared use of nest trees versus their availability across each of the three fires for all bird species combined (Figure 5). 38% of nests were found in true fir (primarily white fir) while it comprised 51% of the available trees, 17% were in Douglas Fir compared to 6% of available, 14% were in yellow pine compared to 13% of available, 6% were in aspen compared to 1% of available, 5% were in black oak compared to 4% of available, 2% were in incense cedar compared to 7% of available, and no nests were found in sugar pine though it comprised 4% of the available. For 16% of the nests and 13% of the random "nests" we were unable to identify the nest tree to species.

Some bird species showed a preference for tree species selected. White-headed Woodpecker (4 out of 10) and Mountain Bluebird (3 out of 10), nested in Yellow Pine (ponderosa or Jeffrey) more than its availability with 35% of their nests in these species while it comprised only 13% of the available nest trees. Red-breasted Sapsucker nests were only found in true Fir, Douglas fir, or Quaking Aspen. Hairy Woodpeckers nested primarily in Fir (both Douglas and true) with 8 out 10 nests in fir.

Figure 4. Avian community indices by burn severity at three fires in 2009 and green forest in the adjacent Plumas-Lassen Administrative Study area with 95% confidence intervals High severity = >50% overstory tree mortality and Moderate to Low severity = <50%.





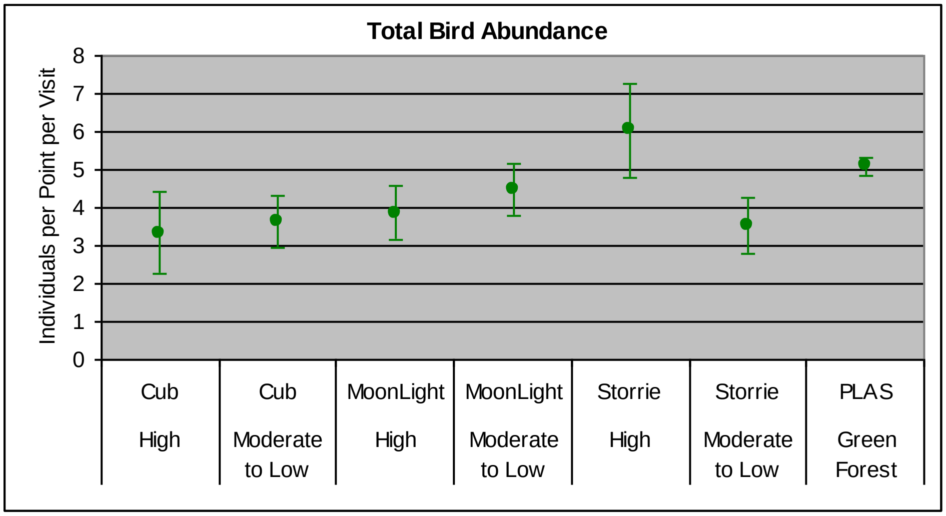
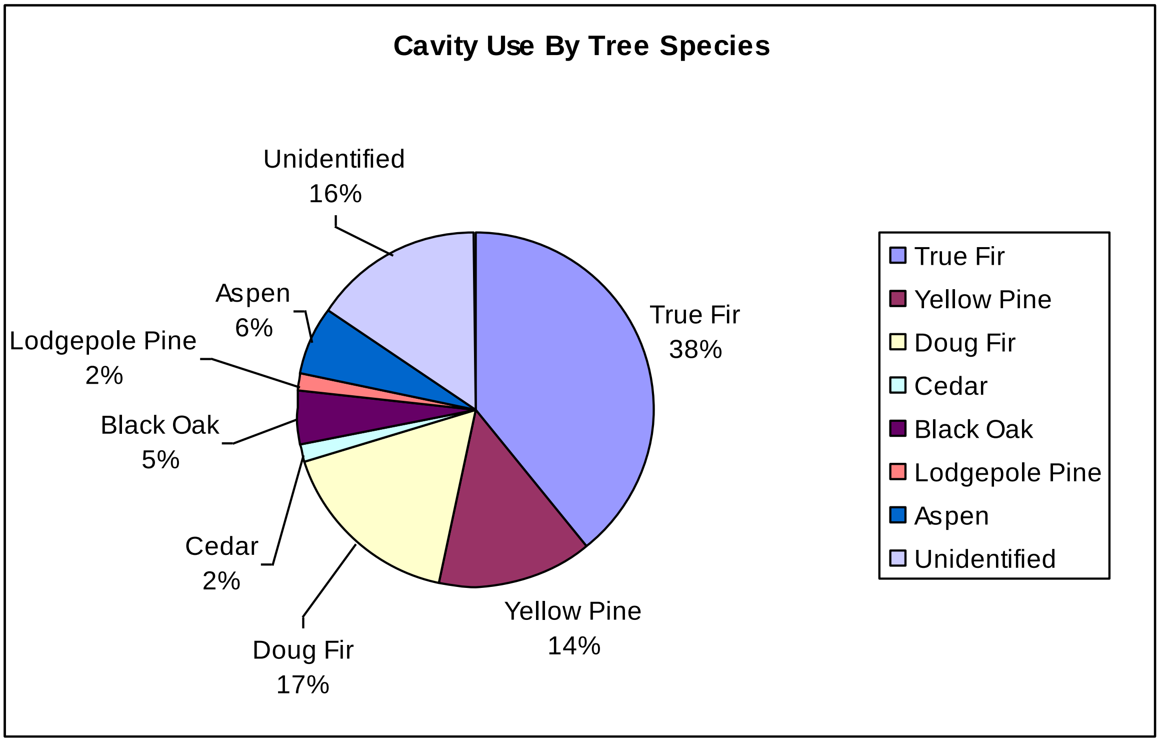
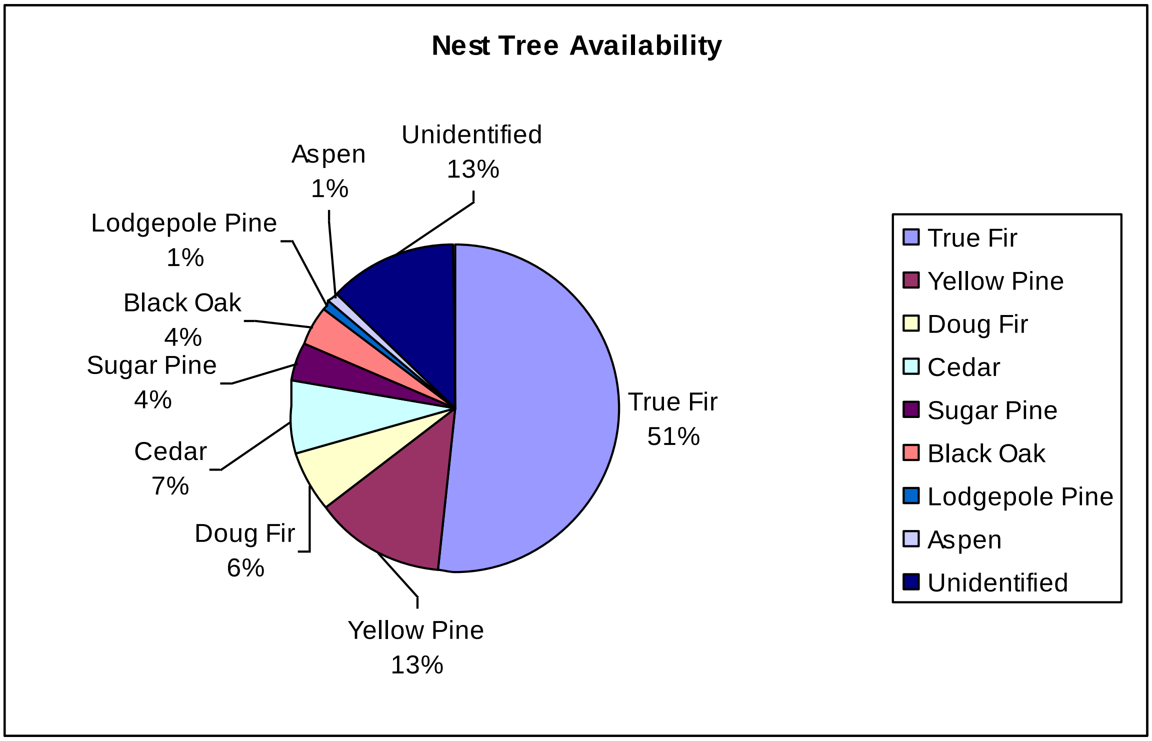


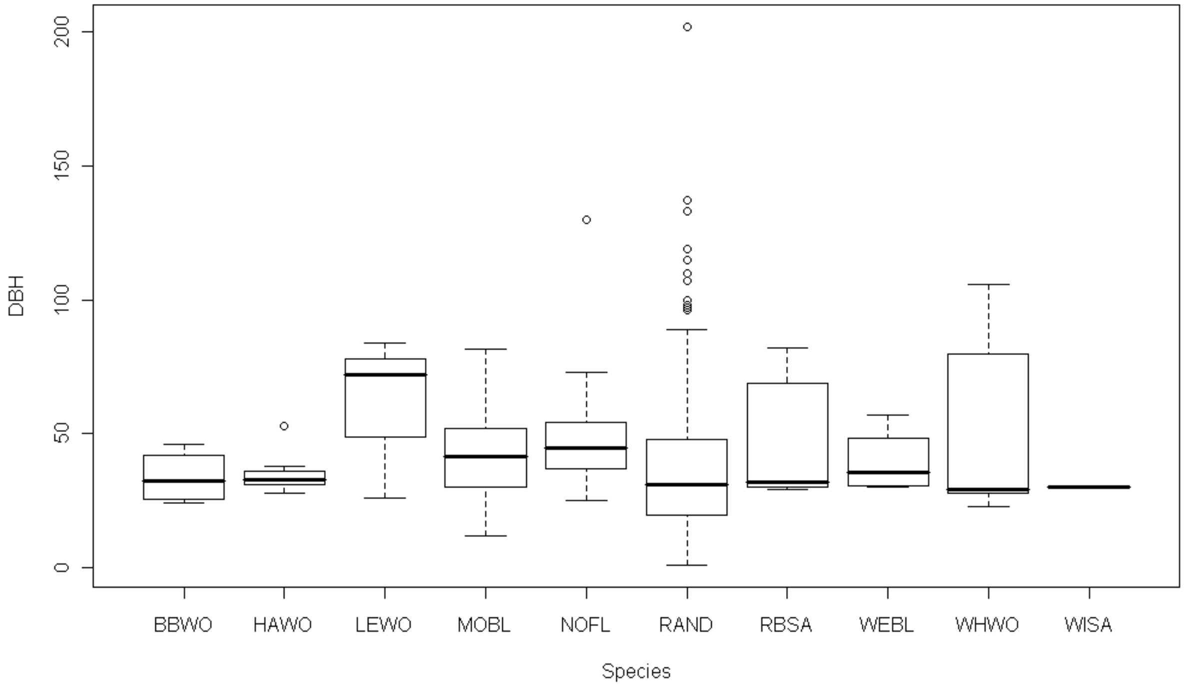
Figure 5. Avian cavity nest tree use compared to availability in the Cub, Moonlight, and Storrie Fires in 2009. Ponderosa and Jeffrey pine are combined under Yellow Pine and Red and White fir are combined under True Fir as snags of these species were difficult to determine to species.





Almost all nests were in snags larger than 25cm (10 in) DBH and typically in snags larger than the average snag size available. The size of trees used was quite variable and patterns were evident by species (Figure 6). Lewis' Woodpecker, Mountain Bluebird and Northern Flicker tended to select for considerably larger DBH nest trees compared to what was available while the remaining species used trees slightly larger than the average available. We found nests in trees with a DBH as small as 12cm or 5 in (Mountain Bluebird) - and as large as 230cm or 91 in (Northern Flicker).

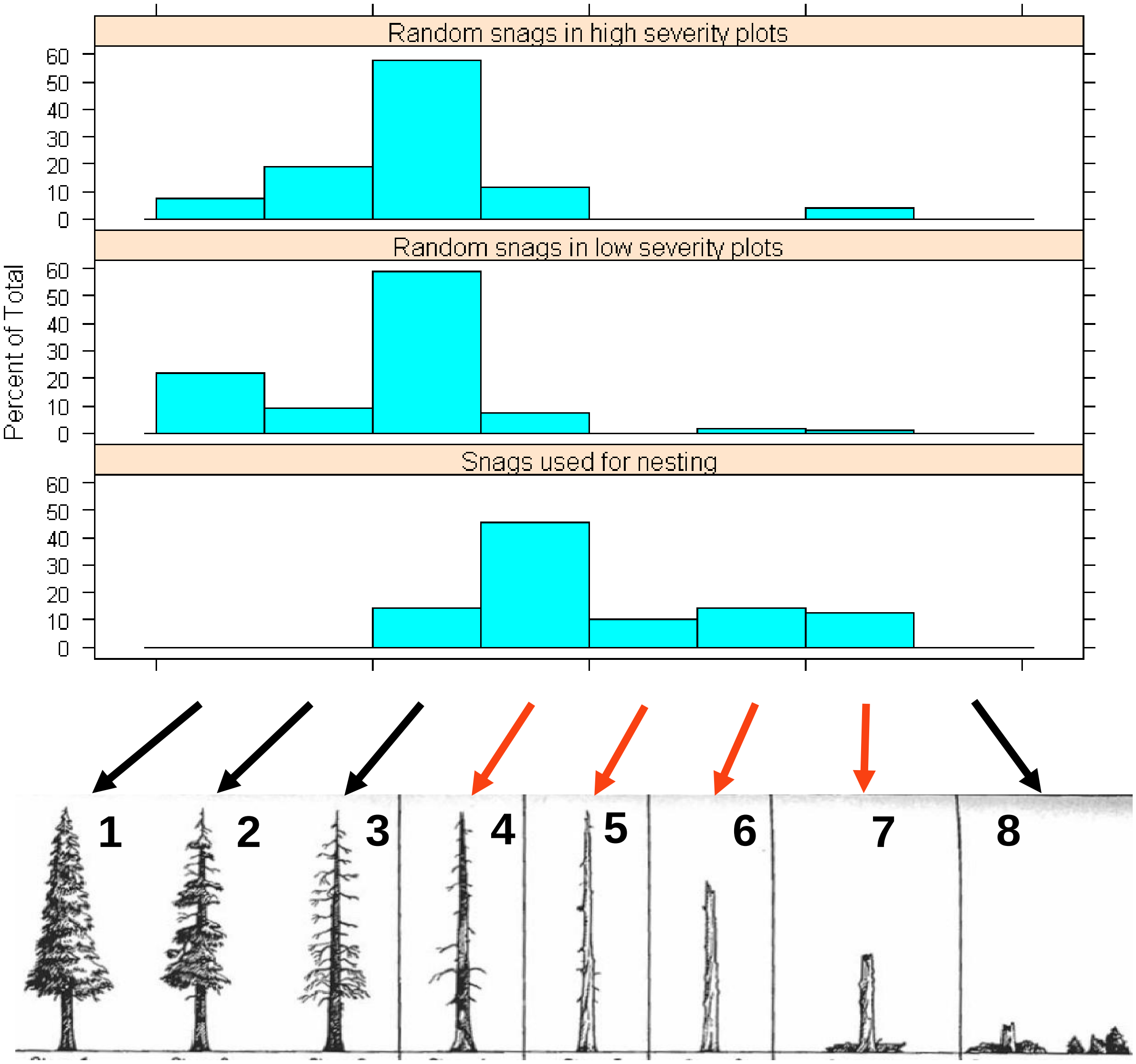
Figure 6. The DBH of nest trees according to bird species with the random "nest" trees shown as species RAND. (BBWO – Black-backed Woodpecker, HAWO – Hairy Woodpecker, LEWO – Lewis's Woodpecker, MOBL – Mountain Bluebird, NOFL – Northern Flicker, RBSA- Red-breasted Sapsucker, WEBL - Western Bluebird, WHWO – White-headed Woodpecker, WISA – Williamson's Sapsucker). The diameter or snags used by cavity-nesting birds was almost always larger than 25 cm in diameter and typically larger than random snags.



There were two different patterns that emerged in terms of decay class, those bird species that preferred slightly decayed trees and those that preferred very decayed trees. However, all species selected for trees that were a higher decay class than was available (Figure 7). Only five nests were in trees with intact tops, four in trees with forked tops, one in a pile of logs at a landing, the other 54 were in trees with broken tops. The average decay class of nest trees for Black-backed Woodpecker, Hairy Woodpecker, Northern Flicker, Red-breasted Sapsucker and Williamson's Sapsucker were all between four and five. Whereas the average decay class for

Lewis' Woodpecker, Mountain Bluebird, Western Bluebird, and White-headed Woodpecker were all greater than five (Figure 3). The average decay class of all trees available was three. The number of available snags in the decay classes selected for by cavity nesting birds did not vary between high severity (>50% overstory tree mortality) and moderate to low severity areas of the two recent fires (Cub and Moonlight; Figure 7).

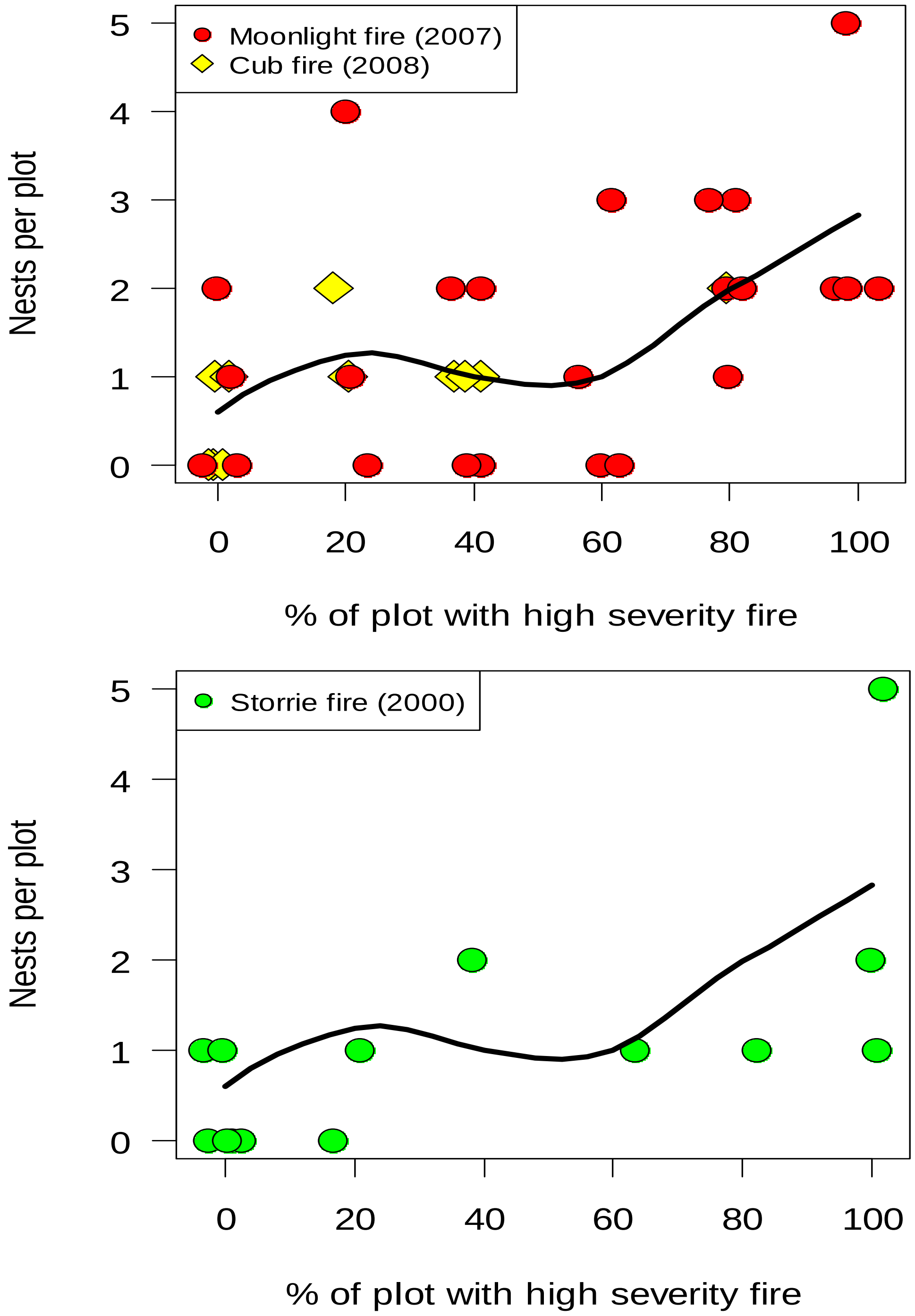
Figure 7. The distribution of decay classes in random snag plots in low-moderate and high severity fire areas compared to the distribution of snags by decay class where active nests were found in the Cub and Moonlight fires in 2009. Snags used for nesting were usually more decayed than the random sample of snags in high and low severity areas.



Nest Densities

Nest densities were highest in the Moonlight Fire and lowest in the Cub fire though the difference between any of the fires was not statistically significant (*P* = 0.3). Four out of the five plots with nest densities greater then two per plot were in the Moonlight fire with one in the Storrie Fire. Nest density did vary with respect to fire severity across the three fires and the effect was not linear (Figure 8). Nest densities were lowest in plots with no high severity fire, increased with modest amounts of high severity fire but varied little between 20% and 60% high severity. Once the amount of the plot that was high severity was over 60% the density of cavity nests increased substantially.

Figure 8. The number of cavity nests per plot in relation to the percent of the plot categorized as high severity based on vegetation surveys at point count stations in 2009 with fitted trend line. Trend line in Storrie fire graph is that predicted for the Cub and Moonlight fires for comparison. High severity was defined as overstory tree mortality greater than 50%.



Discussion

Overview

Post-fire habitat in the Plumas-Lassen study area supports a unique and moderately diverse bird community - compared to green forest in the region - including a number of rare and declining species. The disparity between green forest and burned avian assemblages was greatest between the two fires with the most high severity habitat (Storrie and Moonlight). However, all of the twenty most abundant bird species found in PLAS green forests were also detected in all three burned areas. Avian species associated with larger patches of mature green forest (e.g. Pileated Woodpecker, Hermit Thrush, Hammond's Flycatcher, Golden-crowned Kinglet, and Hermit Warbler) were detected in each of the fire areas suggesting these burned areas still contain enough green forest to support these species. Factors such as tree species, tree size, high severity patch size, and especially decay class influenced cavity nesting species nest tree choice and likely their densities across the three burned areas. High severity fire does not result - in the first two years following fire – in snags with the preferred decay condition used by cavity nesting species. Thus, pre-fire snag densities appear an important consideration for determining the occupancy of sites by cavity nesting birds immediately following fire. Post-fire habitat should be managed as a unique component in the Sierra Nevada as part of a balanced ecosystem approach in order to sustain biological diversity.

Avian Community Composition Burned vs. Green Forest

The difference in avian species diversity and total bird abundance between green forest and post-fire habitat in our study area was not clear cut. Per point species diversity and total bird abundance was generally greater in unburned forest than post-fire habitat. However, more total species were detected in the Moonlight fire which covers a much smaller geographic area and had far fewer sampling locations than the PLAS green forest. Green forest in our study area likely have greater structural diversity and total foliage volume at the point level which can support a greater diversity of species (McCarthur et al. 1966, Verner and Larson 1989), while post-fire habitats in our study area appear more heterogeneous on a landscape scale. Therefore, whether green forest or post-fire habitat supports greater avian diversity depends on the resolution at which the question is considered.

Further, when sites were stratified by fire severity, we found avian species richness in high severity areas in the oldest fire (Storrie) was equal to green forest and the total bird abundance was significantly greater at the point scale. Thus, our results suggest that within our study area, areas burned by wildfire, especially those with older high severity patches, may in some cases support equal or greater landbird diversity and total bird abundance at the point or patch scale which is consistent with what others have found in the region (Bock and Lynch 1970, Raphael et al. 1987, Fontaine et al. 2009).

While post-fire habitats clearly support a number of avian species, its most important attribute may be the unique species - those uncommon or rare in green forest - that are abundant in post-fire habitat. A few of these species (e.g. Chipping Sparrow, Western Wood-Pewee) are experiencing significant population decreases in the Sierra Nevada (Sauer et al. 2008).

The value of post-fire habitat for many species extends well beyond the first few years following fire. The abundance of some bark foraging birds has been shown to decline approximately five years after fire (Saab et al. 2004, Saab et al. 2009), presumably as a result of a decline in wood boaring beetle abundance (McCullough et al. 1998), and possibly as mammalian nest predators recolonize burn areas (Saab and Vierling 2001). However, as successional processes bring about change to burned areas the suitability of habitat is likely to increase for a number of aerial foraging cavity nesting species such as bluebirds, Lewis' Woodpecker, and swallows (Hobson and Schieck 1999, Saab et al. 2007) as well as shrub-dependent species such as Fox Sparrow and Mountain Quail (Bock and Lynch 1970, Raphael et al. 1987). In our study area, Fox Sparrow was more abundant in green forest than either the Cub or Moonlight fires but 1.6 times less abundant in green forest than in the Storrie fire. We observed similar patterns for several other shrub-dependent species including MacGillivray's Warbler and Spotted Towhee. As large patches of dense shrub cover develop in high severity patches of the Moonlight and Cub fires the abundance of many shrub dependent species will likely increase. The dense, relatively large patches of shrub habitat that develop following moderate to high severity fire support a number of species and, when interspersed with areas of green forest, are a key component for maximizing avian diversity in post-fire environments.

The abundance of different foraging guilds varied considerably between burned and unburned forest in our study area similar to patterns documented elsewhere in the western mountains (Bock and Lynch 1970, Raphael et al. 1987, Hutto 1995, Fontaine et al. 2009). Generally, we found foliage gleaning birds were more abundant in unburned forest and bark gleaning, aerial insectivores, and shrub and ground foraging species were more abundant in post-fire habitat. However, very few foliage gleaning birds present in green forest were absent from fire areas but a few in the fire associated guilds were absent from green forest (Lewis' Woodpecker, Black-backed Woodpecker, and Mountain Bluebird). In addition, a number of species quite common in post-fire areas were exceedingly rare in green forests of the PLAS. Lazuli Bunting, a species associated with shrubs and herbaceous understory vegetation, was the most abundant bird in the Moonlight fire, where it was 21 times more abundant than in the PLAS green forest. Two mature forest foliage gleaning birds, Hermit Warbler and Golden-crowned Kinglet, were five and four times more abundant respectively in green forest in 2009 compared to the Moonlight and Storrie fires. There is little question that these fires have resulted in a significant decrease in the suitable habitat and thus populations of many green forest species in these fire areas but, in 2009, almost all green forest associated species found in the PLAS were still occupying habitat within both of these fires. Thus, the needs of mature green forest species species should be considered in post-fire management.

Cavity Nesting Densities and Nest Snag Characteristics

The importance of post-fire habitats for cavity nesting and bark foraging birds is well established (Raphael et al. 1987, Hutto 1995, Saab and Dudley 1998). However, little information exists for the Sierra Nevada describing the important characteristics in post-fire snag-dominated habitats that determine the density and diversity of cavity nesting species. Our results here provide some of the first detailed information for a whole suite of cavity nesting species in post-fire habitat in the Sierra Nevada.

Almost all species selected for more decayed trees and generally used the existing pre-fire snags in the recent burns. In at least the first two years following fire, the majority of available snags were not in the decay classes selected for by cavity nesting species regardless of fire severity. This suggests that in post-fire habitat the presence of existing snags is critical for providing suitable nesting substrate until fire killed trees obtain a greater amount of decay which aligns well with what Hutto (1995) found in the Northern Rockies. This lack of suitable nest substrate is a likely the reason for the observed peak in woodpecker density four to five years after a fire (Saab et al. 2004), even though wood boring beetle abundance may peak sooner (McCullough et al. 1998). Thus, the necessary tree decay for suitable nesting substrate may not be obtained until after food availability has begun to wane. Maximizing habitat suitability for bark foraging-cavity nesting birds appears a function of managing for sufficient snag densities in green forest prior to fire.

The density of snags and size of high severity patches has been shown to influence the density of cavity nesting species in post-fire habitat (Saab et al. 2004, Dudley and Saab 2007, Hutto 2006). Nest densities in all three fires in our study appeared increase with the proportion of the plot that burned with high severity. This pattern continues to support the hypothesis that cavity nesting birds favor larger patches of high fire severity. Our plots were 20ha (49 acres), which probably represents a minimum patch size of high severity to maintain. However, more detailed analysis has suggested a minimum 200ha (494 acres) of high severity high snag density (>250 snags/acre) habitat is a minimum patch size for species such as Black-backed Woodpecker (Dudley and Saab 2007). Further investigation using GIS based analysis of patch size in future years will provide a greater understanding of the size and overall importance of landscape scale patterns on species associated with post-fire habitat.

In addition to managing large areas with high snag densities for cavity nesting and foraging species, when considering replanting patterns patch size is also important for shrub dependent species. Fox Sparrow, average territory size in the Northern Sierra is between two and three hectares (5-7 acres) (PRBO unpublished data), and their abundance precipitously increases as the area within a 500 m radius (78.5 ha, 196 acre) is comprised of greater amounts of shrub habitat (Howell and Burnett *In review*).

Interestingly, we found a similar pattern of high severity burn area and nest density in the nine year post-burn habitat of the Storrie fire as in the Moonlight and Cub fires. With only one year of data sample sizes are relatively low to discern a clear pattern but, we found that at least some areas of the Storrie fire that burned at high severity are still supporting relatively high densities of many cavity nesting birds. This suggest post-fire treatments being planned and carried out in the Storrie fire should still consider the needs of species dependent upon relatively large high severity and high snag density patches.

Conclusions

In the Sierra Nevada considerable debate surrounds the management of post-fire habitat. After nearly a century of fire suppression policies in the Sierra Nevada, the area affected by wildfire each year appears to be increasing back towards pre-suppression levels (Miller et al. 2007). Thus, there is a growing need to understand the value of the habitats created by wildfire and the critical elements required by the unique and relatively diverse avian community in the Sierra Nevada. It is clear from our first year of monitoring three burned areas that post-fire habitat, especially high severity areas, are an important component of the Sierra Nevada ecosystem.

Wildfires provide a unique opportunity to mold a landscape into the forest composition that will exist there for the decades to come. However, post-fire areas are not blank slates or catastrophic wastelands; they are a unique component of the ecosystem that supports a diverse and abundant avian community that should be considered in planning post-fire management. The results from this ongoing study, especially with several more years of monitoring, can provide some important information on factors influencing the unique bird assemblages in post-fire habitat in order to ensure their needs are met while meeting other post-fire objectives.

In future years we will conduct a more detailed analysis of fire severity at different scales and using standard measures used to classify burn severity (e.g. composite burn index). Additionally, we will use available remotely sensed data on burn severity and possibly LiDAR data to better understand the importance of severity class, patch size, and snag densities for the various species associated with post-fire habitat. Finally, we will compare bird assemblages within fuel reduction treatments in green forest, post-fire treatments, and untreated post-fire habitat to determine the effects of various treatments on a broad range of avian species to provide insight for a balanced approach to management of these forest ecosystems.

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Appendix A. Outreach and Publications

In Press
Nesting Ecology of Yellow Warblers in Montane Chaparral Habitat in the Northern Sierra Nevada – Western North American Naturalist

In Revision
Landbird community composition in the context of Spotted Owl management in the Sierra Nevada – resubmitting summer 2010.

In prep
Short-term response of avian species to fuel treatments in the Sierra Nevada – draft submitting for publication spring 2010.

Presentations
Short-term response of avian species to fuel treatments in the Sierra Nevada – oral presentation at the Annual PLAS symposium Quincy, CA April 2009

Managing post-fire habitat for birds in the Sierra Nevada – poster presentation – Pre and Post-Wildfire Forest Management for Ecological Restoration and Fire Resiliency Conference – Sacramento, CA 2/10/10.

PRBO's Sierra Nevada Program – Management Indicator Species Online Tools – Regional Biologist Training – Susanville, CA. 2/2/10

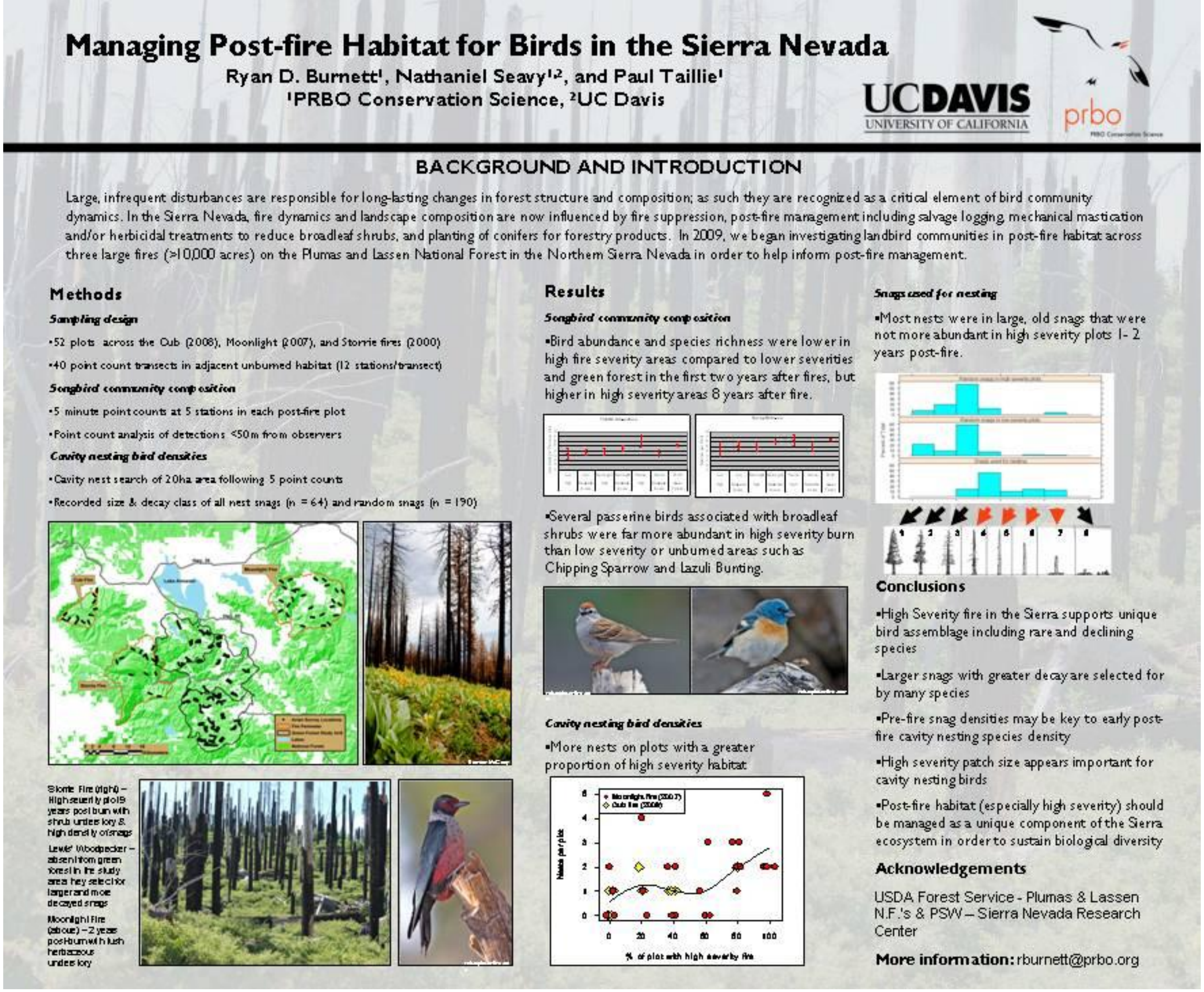
Forest Management, Fire, and Climate Change in the Sierra Nevada – PRBO Board of Directors and Friends Holiday meeting – San Francisco, CA – 12/9/09

Other Outreach
"Birds in the Park" – presentation on managing coniferous forest for birds and bird banding demonstration in collaboration with Lassen Volcanic National Park – over 200 park visitors participated 7/20/08.

"Habitat for Birds and Humanity" – Sierra Institute for Community and Environment sponsored field trip. – 6/28/2009

Participated in several Lassen National Forest field trips to discuss fuel reduction projects.

Appendix B. Managing Post-fire Habitat for Birds in the Sierra Nevada Poster



Managing Post-fire Habitat for Birds in the Sierra Nevada

Ryan D. Burnett[1], Nathaniel Seavy[1,2], and Paul Taillie[1]
[1]PRBO Conservation Science, [2]UC Davis

BACKGROUND AND INTRODUCTION

Large, infrequent disturbances are responsible for long-lasting changes in forest structure and composition; as such they are recognized as a critical element of bird community dynamics. In the Sierra Nevada, fire dynamics and landscape composition are now influenced by fire suppression, post-fire management including salvage logging, mechanical mastication and/or herbicidal treatments to reduce broadleaf shrubs, and planting of conifers for forestry products. In 2009, we began investigating landbird communities in post-fire habitat across three large fires (>10,000 acres) on the Plumas and Lassen National Forest in the Northern Sierra Nevada in order to help inform post-fire management.

Methods

Sampling design

- 52 plots across the Cub (2008), Moonlight (2007), and Storrie fires (2000)
- 40 point count transects in adjacent unburned habitat (12 stations/transect)

Songbird community composition

- 5 minute point counts at 5 stations in each post-fire plot
- Point count analysis of detections <50m from observers

Cavity nesting bird densities

- Cavity nest search of 20ha area following 5 point counts
- Recorded size & decay class of all nest snags (n = 64) and random snags (n = 190)

Storrie Fire (right) – High severity plot 9 years post burn with shrub understory & high density of snags

Lewis' Woodpecker – absent from green forest in the study area they select for larger and more decayed snags

Moonlight Fire (above) – 2 years post-burn with lush herbaceous understory

Results

Songbird community composition

- Bird abundance and species richness were lower in high fire severity areas compared to lower severities and green forest in the first two years after fires, but higher in high severity areas 8 years after fire.
- Several passerine birds associated with broadleaf shrubs were far more abundant in high severity burn than low severity or unburned areas such as Chipping Sparrow and Lazuli Bunting.

Cavity nesting bird densities

- More nests on plots with a greater proportion of high severity habitat

Snags used for nesting

- Most nests were in large, old snags that were not more abundant in high severity plots 1-2 years post-fire.

Conclusions

- High Severity fire in the Sierra supports unique bird assemblage including rare and declining species
- Larger snags with greater decay are selected for by many species
- Pre-fire snag densities may be key to early post-fire cavity nesting species density
- High severity patch size appears important for cavity nesting birds
- Post-fire habitat (especially high severity) should be managed as a unique component of the Sierra ecosystem in order to sustain biological diversity

Acknowledgements

USDA Forest Service - Plumas & Lassen N.F.'s & PSW – Sierra Nevada Research Center

More information: rburnett@prbo.org

Appendix C. PRBO's Plumas-Lassen Study sites with dates surveyed in 2009.

Transect	Transect Code	1st Visit	2nd Visit
Cub 01	CB01	5/28/2009	6/23/2009
Cub 02	CB02	5/28/2009	6/23/2009
Cub 03	CB03	6/1/2009	6/24/2009
Cub 05	CB05	6/1/2009	6/24/2009
Cub 07	CB07	6/3/2009	6/25/2009
Cub 08	CB08	6/2/2009	6/24/2009
Cub 09	CB09	6/2/2009	6/25/2009
Cub 10	CB10	6/2/2009	6/25/2009
Cub 11	CB11	6/2/2009	6/25/2009
Cub 12	CB12	6/2/2009	6/24/2009
Cub 13	CB13	6/13/2009	6/29/2009
Cub 14	CB14	6/9/2009	6/24/2009
Moonlight 01	ML01	5/28/2009	6/17/2009
Moonlight 02	ML02	5/26/2009	6/18/2009
Moonlight 03	ML03	5/28/2009	6/19/2009
Moonlight 04	ML04	5/27/2009	6/22/2009
Moonlight 05	ML05	5/26/2009	6/17/2009
Moonlight 06	ML06	5/31/2009	6/20/2009
Moonlight 07	ML07	6/1/2009	6/22/2009
Moonlight 09	ML09	6/1/2009	6/20/2009
Moonlight 10	ML10	5/31/2009	6/20/2009
Moonlight 11	ML11	5/27/2009	6/22/2009
Moonlight 12	ML12	5/27/2009	6/23/2009
Moonlight 13	ML13	5/27/2009	6/22/2009
Moonlight 14	ML14	6/1/2009	6/22/2009
Moonlight 15	ML15	5/27/2009	6/20/2009
Moonlight 16	ML16	5/25/2009	6/19/2009
Moonlight 17	ML17	5/30/2009	6/19/2009
Moonlight 18	ML18	5/28/2009	6/19/2009
Moonlight 19	ML19	5/28/2009	6/19/2009
Moonlight 20	ML20	5/29/2009	6/20/2009
Moonlight 21	ML21	5/29/2009	6/23/2009
Moonlight 22	ML22	5/29/2009	6/23/2009
Moonlight 23	ML23	5/26/2009	6/18/2009
Moonlight 24	ML24	6/1/2009	6/20/2009
Moonlight 25	ML25	5/26/2009	6/19/2009
Moonlight 26	ML26	5/27/2009	6/22/2009
Moonlight 27	ML27	5/26/2009	6/17/2009
Storrie 01	ST01	6/17/2009	6/26/2009
Storrie 02	ST02	6/15/2009	7/1/2009

Transect	Transect Code	1st Visit	2nd Visit
Storrie 03	ST03	6/15/2009	7/1/2009
Storrie 04	ST04	6/9/2009	6/29/2009
Storrie 05	ST05	6/12/2009	6/30/2009
Storrie 06	ST06	6/3/2009	6/29/2009
Storrie 07	ST07	6/13/2009	7/1/2009
Storrie 08	ST08	6/16/2009	6/30/2009
Storrie 09	ST09	6/8/2009	7/1/2009
Storrie 10	ST10	6/8/2009	6/26/2009
Storrie 11	ST11	6/15/2009	6/27/2009
Storrie 12	ST12	6/10/2009	6/29/2009
Storrie 13	ST13	6/8/2009	6/27/2009
Storrie 15	ST15	6/9/2009	6/25/2009
Unit 2 13	213	5/22/2009	6/18/2009
Unit 2 14	214	5/23/2009	6/15/2009
Unit 2 22	222	5/25/2009	6/16/2009
Unit 2 24	224	5/22/2009	6/18/2009
Unit 3 13	313	5/25/2009	6/16/2009
Unit 3 22	322	5/25/2009	6/16/2009
Unit 4 13	413	5/20/2009	6/11/2009
Unit 4 14	414	5/19/2009	6/17/2009
Unit 4 22	422	5/20/2009	6/23/2009
Unit 4 24	424	5/18/2009	NS
Black Hawk Creek 1	BLH1	5/20/2009	6/10/2009
Butt Valley Reservoir 3	BVR3	5/19/2009	6/13/2009
Caribou 2	CAR2	5/30/2009	6/26/2009
Caribou 3	CAR3	5/30/2009	6/26/2009
DFPZ Unit 4 01	D401	5/19/2009	6/11/2009
DFPZ Unit 4 02	D402	5/16/2009	6/5/2009
DFPZ Unit 4 03	D403	5/18/2009	6/12/2009
DFPZ Unit 4 04	D404	5/18/2009	6/8/2009
DFPZ Unit 4 05	D405	5/16/2009	6/6/2009
DFPZ Unit 4 07	D407	5/18/2009	6/5/2009
DFPZ Unit 4 08	D408	5/23/2009	6/11/2009
Halsted Flat 3	HAL3	5/18/2009	6/6/2009
Lower Knox Flat 1	LKF1	5/20/2009	6/12/2009
Lower Knox Flat 3	LKF3	5/20/2009	6/16/2009
Miller’s Fork 1	MIF1	5/23/2009	6/11/2009
Miller’s Fork 2	MIF2	5/25/2009	6/11/2009
Miller’s Fork 3	MIF3	5/19/2009	6/11/2009
Meadow Valley 1	MVY1	5/16/2009	6/5/2009
Meadow Valley 2	MVY2	5/23/2009	6/17/2009
Ohio Creek 2	OHC2	5/22/2009	6/18/2009

Transect	**Transect Code**	**1st Visit**	**2nd Visit**
Pine Leaf Creek 1	PLC1	5/16/2009	6/5/2009
Seneca 1	SEN1	5/22/2009	6/18/2009
Silver Lake 1	SIL1	5/30/2009	6/26/2009
Silver Lake 2	SIL2	5/30/2009	6/26/2009
Silver Lake 3	SIL3	5/19/2009	6/15/2009
Snake Lake 2	SNK2	5/18/2009	6/6/2009
Snake Lake 3	SNK3	5/16/2009	6/6/2009
Soda Creek 1	SOD1	5/21/2009	6/12/2009
Soda Creek 2	SOD2	5/21/2009	6/12/2009
Spanish Creek 2	SPC2	5/20/2009	6/10/2009

Plumas-Lassen Administrative Study Vegetation Module

Forest Restoration in the Northern Sierra Nevada: Impacts on Structure, Fire Climate, and Ecosystem Resilience.

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OBJECTIVES

The vegetation module of the Plumas-Lassen Administrative Study studies how changes in the forest canopy affect ecosystem functioning, including microclimate, growth and competition of shrubs and juvenile trees, understory diversity, and landscape continuity. Research approaches include stand-level experimental manipulations, measurement of plant growth and survival along existing environmental gradients, and assessment of impacts of routine (i.e., non-experimental) forest management activities.

Research Activity Summary 2009

A five-person field crew carried out a study on tree regeneration in landscapes undergoing fires of varying severity, and a study on how growth of large wildlife trees responds to neighboring tree density and to current and past climate. A new member to the research team, Dr. Michael Papaik (formerly of the University of Quebec) is working on development and evaluation of a model of light transmission through trees that will allow prediction of stand development under novel silvicultural treatments and conditions.

Two papers were published in the peer-reviewed literature. One of these establishes the light-dependent growth responses of saplings of six common trees; this information will be integrated with the model Dr. Papaik is developing. The other carries out a landscape connectivity analysis of group-selection silviculture in patchy East-side forests.

Publications

Bigelow, S. W., and S. A. Parks. 2009. Predicting altered connectivity of patchy forests under group selection silviculture. Landscape Ecology, published online Oct 2009, print version pending. DOI: 10.1007/s10980-009-9421-1 .

Bigelow, S. W., M. P. North, and W. R. Horwath. 2009. Models of resource-dependent growth for Sierran mixed-conifer saplings. The Open Forest Science Journal 2: 31-40.

Description of Research and Preliminary Results

Growth of large trees in response to tree neighborhood and climate

Large trees are one of the most valuable resources of the national forests of the Sierra Nevada, for their role in supporting biodiversity, providing beauty, sequestering carbon, and resisting large wildfires. Further, by virtue of their great age and distinct growth rings they provide a window on past climates. In the last two decades forest management has become increasingly focused on sustaining existing large trees and accelerating the development of more. Yet we know comparatively little about increasing recruitment of large trees and maintaining existing ones: there are few studies that have manipulated stand conditions or collected extensive retrospective data on large tree growth. For example, the Forest Vegetation Simulator (FVS) is the principal stand dynamics model used for almost all National Forest planning, yet most regional variants are based on datasets that include few large trees. FVS models tree growth for a stand average and does not account for spatial location or smaller-tree density effects on large-tree growth. As fuels treatments are more widely applied and wildfire intensity increases, identifying site, burn, and tree conditions associated with large tree survival and growth becomes essential for preserving and increasing the number of large, old trees that provide essential habitat that supports biodiversity. The goal of our field study is to learn how density of neighboring trees affects large tree growth, and how growth of large trees responds to past and current climate conditions.

During the 2009 field season, our crew collected data at 102 large (i.e., ≥ 30 inches or 76 cm in breast-height diameter) tree plots in the Plumas and Lassen national forests. Crews measured the distance, species, and size (DBH) of all neighboring trees within 33 feet (10 m) of the focal big tree. Several growth-increment cores were extracted from the large tree and two neighboring trees in each plot.

Annual growth increment over the past ~100 years was measured in a dendrochronology laboratory. Preliminary results indicate a marked decline in growth rates of large trees over the past 100 yrs.

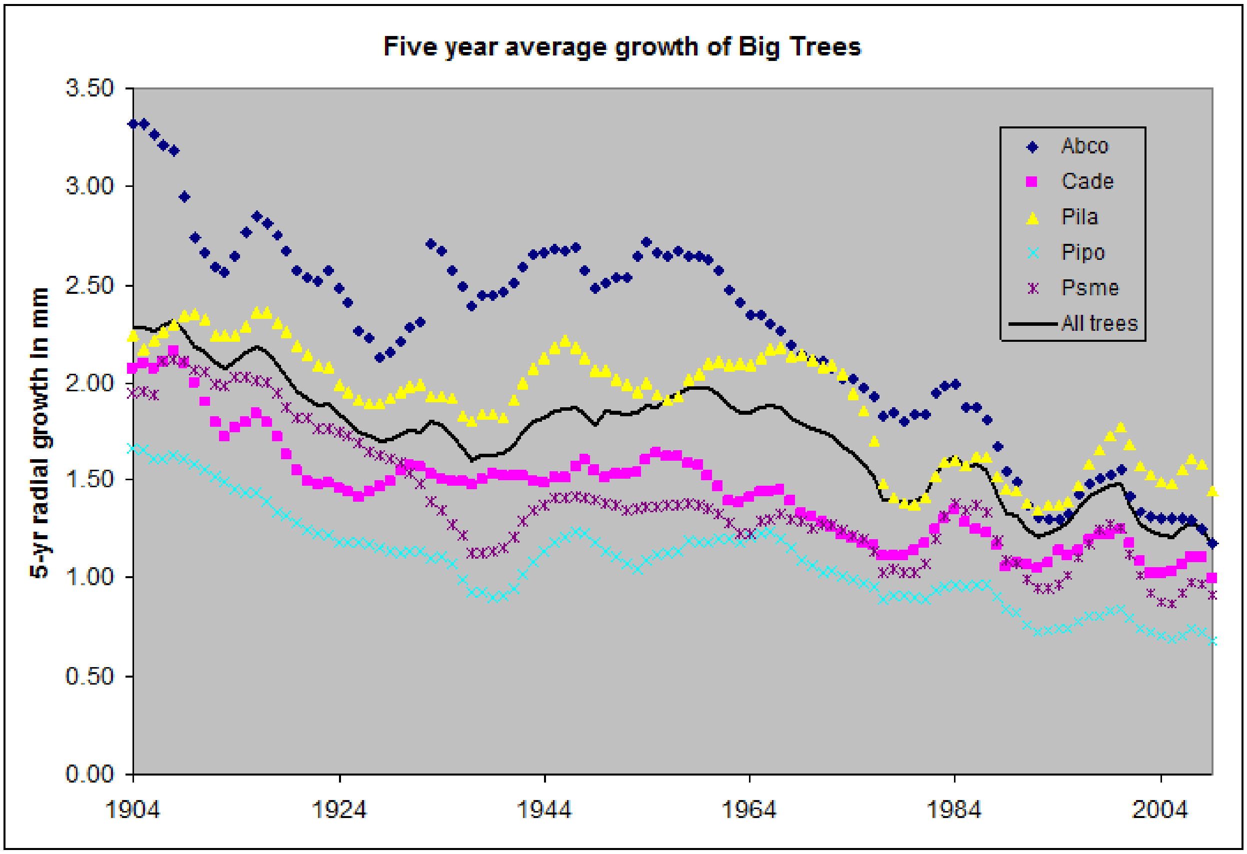

Figure 1. Five-year running average of radial growth of large (>30" DBH in 2009) trees.

The growth rate decrease may be related to simple size effects on growth, or there may be an underlying effect due to climate or increasing neighborhood density over time. Further analyses of the data will provide the answer. We will than assess whether local climate patterns correspond with regional (i.e., the Palmer Drought Severity Index) climate measures, using a same nested calibration approach. This method involves gathering climate records from the nearest weather station and then sequentially adding longer weather records but from further locations. We will then compare these data to the North American Drought Variability (PDSI) reconstructions using the gridpoint nearest the sample locations. To examine species-specific response to climate and the resultant climate reconstruction differences we will use the Kalman filter, which forecasts time (t) using all information at time *t*-1. As new information becomes available, the estimate is recursively updated.

There is need for this research as recent studies suggest accelerating mortality rates in western U.S. large trees, possibly due to climate change, pest damage, and/or fire suppression activities. Regardless of mechanism, the probability of mortality increases once basal area growth drops below species-specific thresholds. Widely used fuels treatments such as mechanical thinning and prescribed fire can potentially ‘release’ the growth of moderate-sized to large leave-trees. Contrary to this supposition several recent studies have reported increases in post-treatment large tree mortality and different growth responses between leave-trees within the same stand. In fire-suppressed forests, large tree growth and mortality response to fuels treatments may differ from our assumptions based on growth and yield studies with smaller trees. We also do not know what factors make some trees more resilient to high intensity fire and whether stand conditions can be manipulated to enhance large tree survival.

Seedling dispersion away from parent trees in salvaged, unsalvaged, fuels-treated, and group-harvested stands

To predict natural regeneration after canopy–opening disturbance such as high-intensity fire or group-selection silviculture, foresters need to know distances seedlings will become established away from parent trees. We censused naturally established conifer seedlings along transects where we also mapped adult trees. Twenty-one sites were visited, located in post-high and low-intensity wildfire stands, salvage-logged stands, and group-selection stands. The availability of substrates (e.g., conifer litter, bare soil) was noted, along with the type of substrate each recruit occurred in. The probable parent tree for each seedling is being identified with maximum likelihood techniques, yielding information on seedling dispersal distances. This information will also be used to develop parameters for a spatially explicit stand regeneration model that can be used to predict regeneration in fuels-reduction-thinned stands, wild-fire-burned stands, and silvicultural situations.

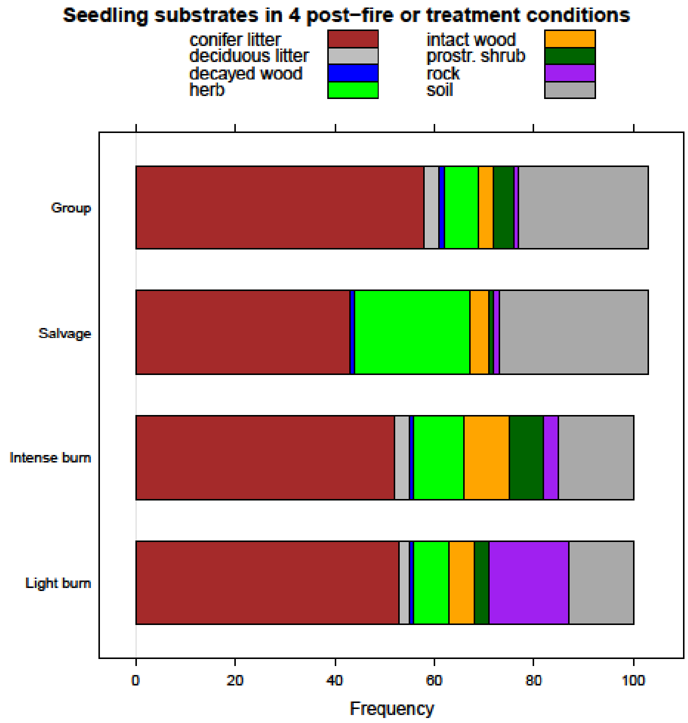

Figure 2. Frequency of ground substrate types encountered after four types of canopy-opening disturbance: Plumas and Lassen National Forests.

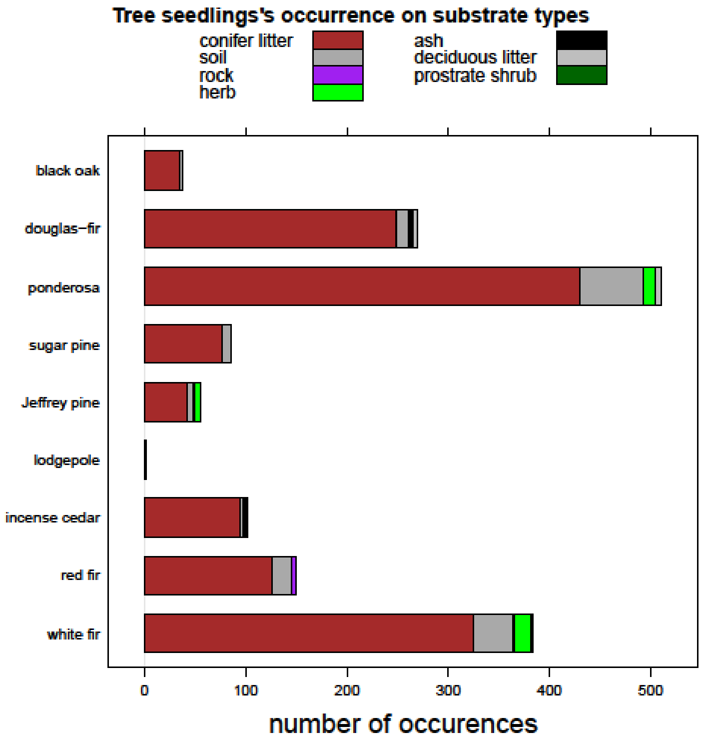

Figure 3. Number of naturally established seedling occurrences on various substrates. Seedlings occured much more commonly on conifer litter than the abundance of this substrate would suggest (Fig. 2).

Light Modeling to Predict Effects of Fuels Treatments and Wildfires

Knowing how forests recover and develop after fire remains a critical challenge for meeting forestry and conservation goals. Historically, fire happened often enough to control the abundance of shrubs and fire intolerant tree species. More recently, after almost a century of fire suppression practices, there is often an immediate flush of new growth of shrubs that may impede regeneration of earlier successional tree species such as pine in such a way as to short-circuit well-known successional patterns that allow for early successional trees to flourish after a disturbance. Concerns for the loss of valuable economic and habitat resources supplied by early successional tree species have led to the implementation of a large scale thinning experiment designed to increase the presence in the Sierra landscape of these early successional tree species.

Whether thinning treatments will actually result in increased abundance of light demanding tree species remains an open question. Traditional stand simulators such as FVS have limited ability to predict the outcome of novel silvicultural treatments like those of the HFQLG Pilot Project because they do not account for position of individual trees within a stand and cannot calculate regeneration as a function of spatial stand characteristics and available light. These details are critical to estimating the actual effects of the thinning treatments. We are using data from the thinning experiment, both pre-thinning and post-thinning light measurements, to characterize the light environment under a wide range of stand conditions. These data will be used as input into an existing spatially-explicit forest simulator called SORTIE-ND that has been successfully used in conifer forests in British Columbia and eastern Canada, and in mixed temperate forests of northeastern US to forecast forest development under harvest and natural disturbance regimes.

Typically, SORTIE-ND requires detailed field studies to determine growth as a function of light and tree neighborhood competition, mortality as a function of growth and regeneration as a function of available substrate and understory light. Mapping tree stands to estimate growth is one of the most time- consuming tasks: our study is designed to use FIA plot data to estimate the spatial structure of the understory, obviating the need for detailed mapping of tree stands. The primacy of light in determining forest regeneration and succession is unquestioned. However, the understory light environment is very heterogenous both horizontally and vertically. SORTIE-ND has two simple light models that adequately capture horizontal heterogeneity of forest stands, but not so vertical heterogeneity. tRAYci is a more realistic (i.e., complex) light model that captures much of both horizontal and vertical heterogeneity of forest stands, and has been used in TASS, a forest simulation model used in British Columbia.

Light models for stand simulation under FIA mapping protocol

Two questions underlie our modeling work. The first is: Which of three light models is best at combining ease of use with realistic representation of the understory light environment? The second question is: Can we make use of the individual-tree spatial information embedded in the

Forest Inventory and Analysis (FIA) data so that this vast database can be used as a resource for prediction of fuels treatments and wildfire on stand regeneration dynamics?

The three light models vary in complexity and reality. They are Sail, Lext, and tRAYci. For all three models the sky is divided into a grid, of which each cell is evaluated at its center. If a tree is present at the center-point, light is attenuated for the whole cell. If the point is clear full light passes from that cell to the point/tree in question. For both Sail and Lext the tree crown is modeled as a cylinder with one dimension defined as the crown depth (top of tree to base of leafing branches) and the radial dimension defined by the ½ maximum crown diameter. For tRACYci the tree crown is modeled as an ellipsoid defined by crown depth and two radial crown width measures that define the ellipsoid. Thus, tRAYci is capable of modeling a wide range of realistic crown geometries. Sail is the simplest model and the fastest to compute during a simulation. It models the effects of the tree crown on light as a "billboard" or "sail" that blocks all light. The presence of a nearby tree crown in any given sky cell blocks all light from that cell and obviates the need to search further for the presence of additional crowns along that path. Lext also models the tree crown geometry as a cylinder, but allows light transmission as calculated from hemispherical photographs for each species using the program GLA. The total light extinguished along any path is the product of the proportion of light extinguished by each crown that intercepts the center of the cell. tRAYci is a more complex crown light interception model that uses a more realistic model of tree crown geometry that varies depending on the tree species and size. It is similar to the SORTIE-ND crown light transmission model in that it models light decay through multiple crowns, but it also parameterizes light decay within each crown following a Beers law decay function that is parameterized from field data. All three models use mapped plots to estimate the light at any given point, in which the surrounding trees intercept light accordingly. From our experimentally treated plots in the Meadow Valley area we have light data in our mapped stands to validate all three models. Answering this question will help us decide when a more complex and realistic model (e.g., tRACYci) may be needed to evaluate forest dynamics and when a simpler more phenomenological model (e.g., Sail) can suffice. Thus we will be better positioned to evaluate the tradeoffs between loss of precision due to too simple a model and excess computing time due to using too complex a model. One possible result is a hybrid approach that incorporates two or more models that can be tailored to answer specific application questions.

The second question addresses our ability to use FIA permanent plot data to support the light and growth models. Creating a map of a plot of trees is a time intensive procedure that represents the largest single cost in both time and personnel resources in a study that assesses the role of light in regeneration, growth and mortality of trees. The FIA data maps plots of trees in a complex way that seeks to address this difficulty using a nested plot, sub-plot design that maps smaller trees in small sub-plots and larger trees in larger sub-plots or the whole plot. FIA is a nationwide tree inventory: data collection includes spatial information to help field crews find trees once they have located a plot, but this spatial information is rarely, if ever, used to improve predictions of growth or species replacement dynamics. The main drawback to the FIA data is that the map of trees is incomplete because the protocol calls for four circular plots that are rather small compared to the usual dimensions of mapped plots. The predicted light environment will therefore be biased relative to the actual measured light environment.

We will assess the performance of the above models in our experimental plots using trees mapped following the FIA plot design. First, we are developing an algorithm that estimates the mean density, diameter distribution, and radius of aggregation of seven species in 4 mapped plots. Based on these a virtual tree stand is generated and a K function is fitted to the observed data to estimate the maximum radius of aggregation (dispersal kernel). Then the code generates a distribution with this radius of aggregation, after specifying the number of clusters and their maximum radius. Then the K function is checked for similarity to the original data. This approach is then used to generate virtual tree maps with the same statistical distribution properties as the mapped plots. Our model expands on this approach to take advantage of the additional mapping of large trees outside the FIA subplots. We use the FIA subplot data to generate the same information but with the additional co-variance of understory tree distribution to overstory tree distribution. The second approach uses imputation, a statistical methodology that estimates a best guess of missing data values from interpolation of measured data. We will try several imputation algorithms to evaluate whether FIA data can be used as the mapped stands for estimating tree regeneration, growth and mortality. The importance and utility of this is obvious for any forest simulation studies. If the FIA field protocol can predict light using either method, then that opens the way to robust predictive forest change modeling.

Experimental Thinning and Group Selection Experiment

No field work was conducted in 2009 on the experimental thinning and group selection experiment in Meadow valley. Work continues on preparing the first round of post-treatment measurements on light, soil temperature, and soil moisture, for publication. Final measurements of forest stand structure, regeneration, fuels, and understory plant community will take place in 2010.

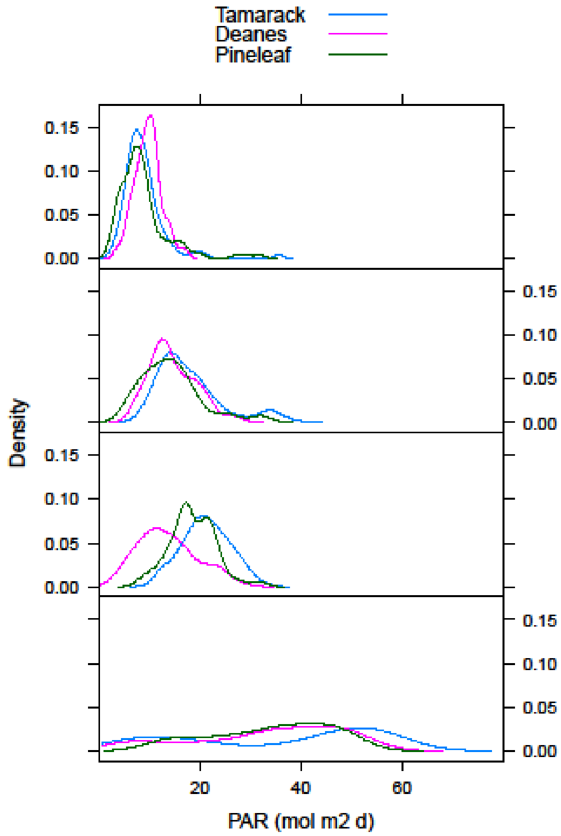

Figure 4. Post-treatment light density in unthinned stands (upper panel), fuels-reduction-thinned stands (two middle panels) and group-selection opening (bottom panel).

Personnel development

Joe Hulbert is an undergraduate student at Washington State University who has spent two summers as an intern doing field work in support of the big tree growth study. He expressed interest in doing additional work to further his ambition of attending graduate school and has enrolled in a senior thesis class. He has developed a working hypothesis that big tree growth responds to local understory tree distributions as well as local climatic variables. He will use SAS/Minitab to run anova and some other regression models to measure correlations between temperature/precipitation, local stand structure and tree growth rates. He is using tree growth data derived from the tree cores and mapped stem data within 10 meters of each tree that he collected together with interpolated NCDC and NOAA data from several stations in the sample region. It is expected that this project will be completed by May 2010.

Plumas-Lassen Administrative Study Fuels and Fire Module:

Fuels and Fire at the Landscape Scale

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Background:

Starting in May, 2009 Dr. Brandon Collins joined the Pacific Southwest Research Station with the specific intent of developing and implementing a research work plan to investigate landscape-level fire within the Plumas-Lassen Administration Study (PLAS). Dr. Scott Stephens remains involved as a co-investigator and participated in developing this research work plan. The work plan involves completing ongoing analyses within the fire and fuels module, integrating fire and fuels results with other PLAS modules, and designing new projects to investigate recent large wildfires that occurred within the greater PLAS region (including Storrie, Antelope complex, and Moonlight complex). In the following we present our accomplishments towards the newly developed plan for 2009. We highlight three specific projects, two of which were completed in 2009 and one is ongoing:

1. Imagery-based assessment of fuel treatment effects on landscape-level fire behavior in Meadow Valley
2. Watershed impacts of wildfires and simulated fuel reduction treatments for the Antelope lake watershed
3. Landscape fuel treatment effects on potential crown fire and long-term forest dynamics in Meadow Valley

1. Imagery-based assessment of fuel treatment effects on landscape-level fire behavior in Meadow Valley

Status: in peer-review, CANADIAN JOURNAL OF FOREST RESEARCH

Introduction

The Sierra Nevada landscape contains many areas of high density mixed conifer forests that are dominated by shade-tolerant species (Parsons and Debenedetti 1979; Ansley and Battles 1998; Taylor 2004) that are increasingly prone to high severity fires (Miller et al. 2009a). Factors contributing to these conditions include a reduction of anthropogenic burning by Native Americans during the mid to late 19th century(Anderson 2005), removal of large trees through the early 20th century via railroad logging (Stephens 2000), a nearly 100 year policy of fire exclusion (Stephens and Ruth 2005), and extensive use of even aged management and overstory removals on public lands through the 1980's (Hirt 1994). Trends of increased uncharacteristically severe burning are expected to continue under changing climates (McKenzie et al. 2004; Westerling et al. 2006; Millar et al. 2007). For decades, scientists and managers have understood the threat fire would pose to forests in this condition (Biswell 1989). However, it was not until the early 1990's that the US Forest Service was given direction to manipulate stands, using a combination of silvicultural prescriptions, with the specific objective of modifying landscape-level fire behavior.

One approach for implementing a landscape fuel treatment relies on creating Defensible Fuel Profile Zones (DFPZs). A DFPZ is an area approximately 0.4–0.8 km (0.2-0.5 mi) wide where surface, ladder, and crown fuel loadings are reduced. They are usually constructed along roads or ridge tops to reduce fuel continuity across the landscape and provide a defensible zone for fire suppression resources. DFPZs are designed to provide three primary functions: 1) safe access for firefighters to conduct suppression activities, 2) limit fire behavior to prescribed levels (e.g., limit flame lengths at the 90th percentile weather condition to 122 cm (48 in or 1.2 m), and 3) create a well-spaced canopy and conditions in which canopy fires are less likely to spread (Agee and Skinner 2005). DFPZs are generally designed to be used in conjunction with suppression actions including fire line construction, application of aerial retardant, and burnout activities. DFPZs are designed to function under 90th percentile weather conditions and their successful performance has been documented in many instances (Moghaddas and Craggs 2007). Under more extreme weather conditions, DFPZs have reduced fire severity at the stand level, but the performance of DFPZs at the landscape scale in more extreme conditions is less understood (Hardy 2005) and has not been systematically studied.

We report the modeled effects of implemented landscape level DFPZs and group selection (GS) treatments on crown fire potential, flame length, and burn probabilities of the treated areas themselves and the untreated landscape surrounding them. This is one of the first studies we are aware of to analyze effects of a fully implemented landscape fuel treatment. We perform this analysis for the Meadow Valley study area, which is within the Mt. Hough Ranger District, Plumas National Forest (Figure 1). This study area is within PLAS Treatment Unit 4.

Methods

Creating fuels layers

We acquired high resolution IKONOS imagery for the study area from the Space Imaging Corporation in 2003 and 2004. We completed radiometric corrections to minimize backscatter and distortion due to atmospheric moisture and haze. We then evaluated the orthorectified image using twelve independent ground reference points. This analysis indicated the five scenes were accurate with an overall average of 2.9 m (9.5 ft). Each of these measures is within a single 4 m (13 ft) pixel of the multispectral imagery, resulting in precise and consistent orthorectification.

Vegetation/fuel characteristics were mapped from the IKONOS imagery using supervised classification. We created the five layers needed to run the spatial fire models FARSITE (Finney 1998) and FlamMap (Finney 2006): fuel model, canopy cover, canopy base height, canopy height, and canopy bulk density. We used vegetation spectral signatures to apply initial fuel models based on vegetation structural characteristics described in Scott and Burgan (2005). Additionally, we used expertise of local and regional fire managers, along with our own familiarity of the vegetation characteristics acquired from field time in the area to aid in fuel model selection.

Calibrating fuel layers

We were fortunate to have an actual wildfire burn largely within the area captured by our IKONOS imagery following the acquisition dates. The Rich fire, which started 29 July 2008 and was contained on 09 Aug. 2008, burned just north of our study area (Figure 1). This provided an excellent opportunity to calibrate our vegetation/fuel layers. Through multiple iterations we adjusted not only fuel model assignments, but canopy height and canopy base height values as well. The intent of these adjustments was to reasonably approximate the size and post-fire effects observed for the Rich fire using the fire spread models FARSITE and FlamMap. Our initial fuel model and canopy layers substantially under predicted fire spread and crown fire activity. Our final iteration resulted in approximately 80% agreement between the modeled and actual fire size. This agreement suggests reasonable validity with respect to our ability to model fire over the Meadow Valley landscape.

Landscape fuel treatments

Fuel treatments within the analysis area implemented prior to the Meadow Valley Project (“Spanish Camp”, “Waters”, and “Ridge” projects) were captured in the IKONOS imagery. Together, these treatments made up 9% of the analysis area *prior to* the implementation of the Meadow Valley Project. For the Meadow Valley Project, an additional 1,650 ha (4,077 acres) of DFPZ’s and 231 ha (571 acres) of GS were implemented, resulting in 19% of the study area being treated at the time of this analysis. Post-treatment (DFPZ and GS) adjustments were made for each of the five layers developed for spatial fire modeling: fuel model, canopy cover, canopy base height, canopy height, and canopy bulk density. These adjustments were applied only to those raster cells that were within treatment polygons. Treatment polygons were generated by US Forest Service personnel (pers. comm., C. Dillingham, Plumas National Forest) and represent actual treatment boundaries, not simply planned or proposed treatments as described in the Meadow Valley Decision Memo and Environmental Analysis (USDA 2004). The post treatment area differed from the planning documents as units were removed or modified due to access or other issues. Modified values for the four canopy layers were based on post-treatment field data collected within the analysis area 1-3 years following implementation.

Fire modeling

We use the Minimum Travel Time algorithm in FlamMap (Finney 2006) to derive conditional burn probability estimates based on 1000 random ignitions for both the pre- and post-treatment landscapes (Ager et al. 2007; Finney et al. 2007). The maximum simulation time for each ignition was 900 minutes, or three 5-hour burning periods. Simulation times were 7.4 and 5.8 hr for pre- and post-treatment model runs, respectively, using 4-processor, 4GB RAM machine. We buffered our study area by 2 km (1.2 mi) where possible, then extracted conditional burn probability estimates for the analysis area (not including the buffer – see Figure 1).

We used FARSITE to simulate the potential fire behavior and size of a single “problem fire” (sensu Bahro et al. 2007) ignited on the upwind edge of the study area. We used a line ignition for this simulation based on an actual fire that occurred in 1999 (the “Pigeon Fire”), which if burning under dry fuel moisture conditions and moderate to high southwest winds, would pose significant fire management difficulties within the Meadow Valley study area (Figure 1). We modeled this supposed “problem fire” for three days under the same conditions that existed during the previously mentioned Rich fire. The Rich fire, which burned just north of the Meadow Valley study area, demonstrated substantial growth during the first two days (48 h), and as such suppression efforts were less effective on fire growth during this time (USDA 2008). We simulated the same fire on the pre- and post-treatment landscape to examine the effectiveness of the DFPZ network under this theoretical “problem fire” scenario.

Results

Conditional burn probability

Minimum Travel Time output from FlamMap resulted in over 200,000 30 m (98 ft) pixels. Average conditional burn probability across all pixels was 0.13 (sd 0.09) and 0.10 (sd 0.07) for pre- and post-treatment landscapes, respectively. These estimates are ‘conditional’ in that they represent the probability of a fire burning in a given pixel given an ignition within the buffered study area (Figure 2). Average pixel-to-pixel difference (post- minus pre-treatment) was -0.03 (sd 0.04). Within DFPZ treatments, overall conditional burn probabilities were decreased by 62% after treatment; across all other untreated areas and within group selection units, conditional burn probabilities were decreased by 17-36%.

Conditional burn probabilities are related to fire size, i.e., higher burn probabilities over a landscape are indicative of the potential for larger fires (Finney et al. 2007; Seli et al. 2008). As such, it is clear that the pre-treatment landscape is more conducive to large fire growth relative to the post-treatment landscape. The map of pixel-to-pixel differences in burn probability demonstrates that the reduction in burn probability following full treatment implementation gets larger moving from the southwest to the northeast, or moving from upwind to downwind (Figure 2).

"Problem fire" scenario

In the "problem fire" scenario, modeling the treated landscape reduced fire growth, and therefore final fire size, by 39% compared to the pre-treatment condition (Figure 3). It is important to note the "problem fire" was modeled without the effects of suppression activities, which could further reduce the final fire size by a greater percentage then reported here. Average flame length was lower throughout the "problem fire" in the post-treatment landscape (Table 1). In addition, both the total area and the proportion of the final fire size burned at a flame length greater than 2.44 and 3.35 m (8 and 11 ft) were lower in the post-treatment landscape (Table 1). On the post-treatment landscape, the modeled "problem fire" intersected 1833 ha (4529 acre) of DFPZ treatments and pre-2005 fuel treatments (Table 1).

2. Watershed impacts of wildfires and simulated fuel reduction treatments for the Antelope lake watershed

Status: to be submitted February, 2010, INTERNATIONAL JOURNAL OF WILDLAND FIRE

Introduction

Erosion rates can increase by several orders of magnitude after wildfire (Morris and Moses 1987). Postfire soil losses are influenced by the amount of vegetative cover and surface fuels removed, soil type, fire intensity, topography, and intensity of post-fire rainfall (Cannon and Reneau 2000; Larsen *et al.* 2009). If post-fire rainfall is intense the critical factors in determining erosion rates are exposed soil (from removal of vegetation and surface/ground fuels) and slope steepness (Prosser and Williams 1998; Larsen *et al.* 2009). Due to strong influence of post-fire hydrologic events, and the erratic nature of such events, wildfire-related increases in soil erosion, and thus sedimentation of water channels and impoundments can be highly stochastic (Robichaud *et al.* 2000; Carroll *et al.* 2007). Despite this stochasticity the relationship between fire severity and post-fire erosion/sedimentation is predictable (Robichaud and Waldrop 1994; Cannon and Reneau 2000). Higher fire severity indicates a greater magnitude of fire-induced change in vegetation and fuel cover, thus greater potential to negatively impact water resources (Sugihara *et al.* 2006).

In addition to the impacts associated with soil erosion and sedimentation, higher fire severity creates an aesthetically degraded landscape that negatively affects several forms of recreation in and around multiple use water resources. Given the recent findings demonstrating increasing trends in the proportion of high severity fire throughout the Sierra Nevada (Miller *et al.* 2009a) it is becoming clear that the potential negative impacts of fire on a variety of natural resources are growing.

Mechanical treatments including thinning of lower- and intermediate-canopy trees and/or mastication of ladder fuels and shrubs, as well as prescribed fire treatments not only reduce the amount of fuel in a forest stand, they reduce the vertical and horizontal continuity fuel (Agee and Skinner 2005; Stephens and Moghaddas 2005b). The net effect of such treatments is a reduction in the fire-induced effects on the system, or fire severity. Recent studies have demonstrated this type of reduction in fire severity within wildfires due to properly designed mechanical fuel treatments (Strom and Fulé 2007; Safford *et al.* 2009), prescribed fire treatments (Finney *et al.* 2005), and a combination of mechanical and prescribed fire treatments (Raymond and Peterson 2005; Ritchie *et al.* 2007). While properly designed fuel treatments reduce severity of wildfires, the activities on the ground associated with these treatments (harvesting, skidding, burning, etc.) have the potential to accelerate erosion and augment sedimentation. It is unclear how this potential for increased erosion and sedimentation as a result of installing fuel treatments compares to that following high-severity wildfire in untreated areas.

Our objectives with this project were to: 1) characterize the fire severity patterns, in particular high-severity patches, resulting from a suite of recent wildfires that burned within the Antelope Lake watershed, and 2) model runoff and erosion in sub-watersheds within the larger Antelope Lake watershed under four scenarios: wildfire (based on actual fires that occurred), thinning alone, prescribed fire alone and no treatment/no fire. Four wildfires burned into the Antelope Lake watershed between 2001 and 2007, ranging in size from 1390 ha to 26390 ha (3,435 to 65,211 acre). We analyzed these fires using satellite-derived fire severity images with the intent of characterizing the fire-induced effects on the forests surrounding Antelope Lake (Figure 4).

Methods

Fire severity assessments

The fire severity images we used were developed from a relative version of the differenced Normalized Burn Ratio (RdNBR). RdNBR has be shown to robustly capture fire effects across a range of forest types and physiographic provinces, particularly high severity effects (Miller *et al.* 2009b). Using ArcGIS® we extracted RdNBR values from each of the four fires within the Antelope Lake watershed. For each fire we used FRAGSTATS (McGarigal *et al.* 2002) to compute mean and area-weighted mean patch sizes for each fire severity class using the threshold values reported in Miller and Thode (2007). We did this for just the portion of each fire that burned within our defined watershed. Area-weighted means place more emphasis on larger patches and less on the numerous smaller patches (one to four 30 m cells) that account for a larger portion of patches in each severity class (Collins *et al.* 2007).

Hydrologic models

To evaluate the erosion risk associated with the wildfires, and compare that to risks associated with fuel management, we use the Water Erosion Prediction Project (WEPP) Model (Flanagan *et al.* 1995; Laflen *et al.* 1997), Version 2009.6, to develop vegetation files from the observed severity, and to carry out the watershed predictions. We used two ARC GIS Wizards to develop the links between the digital elevation model (DEM) files, the fire severity layers and the WEPP Watershed Version. The GeoWEPP, 2008 (version dated 3/10/08) Wizard modeled the undisturbed and managed forest conditions (thinning and prescribed burn). For the wildfire scenario, we used the GeoWEPPBAER, 2008 (Alpha version) ArcGIS Wizard. These GIS wizards delineated the watersheds, and divided the watersheds into hillslope polygons and channel reaches. The wizards were also useful for developing maps of the shape sources of sediment in watersheds. The online Rock:Clime interface (Elliot 2004) was used to generate a stochastic climate file typical of the area.

Calibration process

In order to determine sediment sources within a watershed, it was necessary to first calibrate WEPP to get the correct runoff. Once the total annual runoff and monthly values were reasonably well predicted, then the distribution of sediment sources from hillslope polygons or channel segments could be determined. In climates where there can be snow on the ground on the first day of simulation (January 1, Year 1), it is necessary to have an initial year that ends with a snowpack on December 31 the previous year or to specify the snowpack in WEPP's initial conditions file. For this study, we had the precipitation records, but not any snow pack data, so we included an initial year to generate this snowpack, but did not use the output for that year. As all we had was precipitation records, we chose to run an initial year (2005) to build a snowpack, and then focus on predicted values for 2006-2008. This technique addresses another problem in that WEPP has a tendency to wash all unconsolidated material out of the watershed in the first year, making the sediment yield results for this year unreliable (Zhang *et al.* 2009). It also made it easier to address the differences between the observed data in water years, but the predicted data in calendar years. Therefore, the first year estimates for WEPP simulations were disregarded in the 4-yr climate simulated results. However, the first year climate file was needed so the snow/water yield for the following year would be more accurate.

The GeoWEPP BAER wizard utilized the burned area reflectance classification (BARC) map, which allowed the generation of both management and soil files reflecting the dominant fire severity in each hillslope polygon based on the BARC map. The GeoWEPP BAER software allows users to specify desired ground cover for low, moderate and high severity fire conditions. For the managed conditions, standard calibration procedures within the WEPP Windows software were followed to make sure the above ground biomass and ground cover values were as previously noted.

Results

Fire severity patterns

High severity accounted for 53% of the burned area, while 26% of the watershed burned at moderate severity (Figure 5A). This equates to 5542 ha (13,695 acre) burned at high severity and 2708 ha (6,692 acres) burned at moderate severity within the watershed between 2001 and 2007. Area in the low fire severity and unchanged classes within the watershed was 13% and 8%, or 1363 and 805 ha (3,368 and 1,989 acres), respectively. When resolved into area and fire severity class by fire it is clear that the most recent fire (Moonlight – 09/2007) accounts for not only the largest proportion of burned area within the watershed, but the largest proportion of area burned at high severity (Figure 5B). Figure 5B also shows that 3 of the 4 fires burned predominantly at high severity within the watershed.

In addition to the large amount of area affected by high severity fire within the Antelope Lake watershed, contiguous patches of area burned at high severity were generally very large. These high severity patches were on average much larger than those for the moderate, low, and unchanged classes within the same Antelope watershed fires (Figure 6), as well as those reported for other Sierra Nevada fires. Mean high severity patches ranged from 12.8 ha in the Stream fire to 49.9 ha in the Antelope fire, while area-weighted mean patches ranged from 148.5 ha in the Stream fie to 2407.3 ha in the Moonlight fire (Figure 4). In the WFU fires studied by Collins et al (2007) area-weighted mean high severity patches ranged from 1.2 to 9.1 ha (3 to 22 acres), and in the fires by Safford et al. (2008) mean high severity patches ranged from 11.3 to 17.6 ha (28 to 43 acres). In forest types historically associated with frequent low to moderate severity fires, patches on the order of hundreds and even thousands of acres with few or no surviving trees were unprecedented (Agee 1998). Due to the absence of extensive high severity fire historically, these forests have limited capacity to recover following high severity fire of this scale. As such, reforestation by planting and/or competing vegetation control becomes a necessity for managers to ensure relatively forest recovery in these extensive high severity patches. It is apparent in this watershed that cultural treatments are needed to keep reforested areas from developing high fire hazards (Stephens and Moghaddas 2005a). Planting seedlings in widely-spaced clumps rather than a regular grid pattern may reduce both the homogeneity of the developing forest, as well as the need for future cultural treatments.

Hydrology/erosion patterns

In the Antelope Lake area the majority of the precipitation is received from October through May. The remaining months, June-September, received less than 40 mm (1.6 in), or 2 %, of total precipitation between 2004 and 2008. Table 2 reports results from the GeoWEPP and GeoWEPP BAER modeling under four scenarios: no treatment or fire, thin only, prescribed fire only, and the actual wildfire (based on BARC fire severity maps). Variability in predicted runoff and erosion among the years is much greater than the variability among modeled scenarios (Table 2). Predicted runoff rates were 10 to 100 times greater in 2006 than in the much drier years of 2007 and 2008 for each of the modeled sub-watershed. The high precipitation observed in 2005-2006, which was more than twice the average, is likely responsible for these large discrepancies among years. Sediment delivery responded similarly, indicating the influence of yearly precipitation amount on erosion overrides influences of vegetation condition or watershed configuration in these forests. Despite the small differences in runoff among the modeled forest scenarios, the wildfire and prescribed fire both resulted greater sediment delivery than the no treatment/fire or the thin only simulations. This is likely due to the differences in soil properties and residual ground cover among scenarios.

Most of the predicted soil loss occurred in channel and not on the hillslopes. Very little sediment is generated from the hillslopes. Given the high conductivity of the soil and little surface runoff, it seems reasonable that only the areas of concentrated flow would have the erosion. The sediment yields from the simulated channel networks were scrutinized and it was found that most of the sediment was coming from first order streams within the watershed. The sediment yield events occurred in the winter months, in particularly January and February, when a rain-on-snow event is likely. Onsite observations supported this prediction, with little evidence of upland erosion following these fires, but some increased channel erosion (*personal communication*, C. Reilly, Ca Dept. Water Resour., 2008).

Table 3 reports the predicted runoff and sediment loss immediately following each of the three fires. No sediment and very little runoff were modeled for any fire for this critical period. Most runoff and sediment occurred during the winters of the high precipitation years of 2005 and 2006, before the fires had occurred.

Overall, the results from the simulations indicate that the soil loss for this area is more dependent upon the weather than on vegetation conditions. During the wet years, erosion can be anticipated regardless of treatment. During the dry years, excessive erosion is unlikely, regardless of vegetation conditions.

3. Landscape fuel treatment effects on potential crown fire and long-term forest dynamics in Meadow Valley

Status: analysis to be completed February, manuscript April, 2010 BIOLOGICAL CONSERVATION

Introduction

Much of the knowledge of landscape fuel treatment impacts on potential fire behavior relies on modeling not only potential fire behavior, but fuel treatments as well. Modeled fuel treatments can differ substantially from 'on the ground' implementations, both in terms of where treatments are located and in treatment intensity (Collins *et al.* 2010). As such, inferences from modeled treatments can be limited. Recently, a landscape fuel treatment has been implemented in the Meadow Valley area on the Plumas National Forest. This consists of a Defensible Fuel Profile Zone (DFPZ) network constructed across the Meadow Valley landscape. A DFPZ is an area approximately 0.4–0.8 km wide where surface, ladder, and crown fuel loadings are reduced. DFPZs are usually constructed along roads or ridge tops to reduce fuel continuity across the landscape and provide a defensible zone for fire suppression resources. In this analysis we seek to quantify change in potential crown fire following actual treatment implementation. In addition, we 'grow' the forests out for 30 years to compare crown fire potential in a theoretical untreated landscape to that of the treated landscape.

Methods

The available geographic information system (GIS) layers for treatments in the Meadow Valley area consisted mostly of individual timber sales, although in the case of the Meadow Valley project several sales were combined in one GIS layer. We made a number or manipulations to make a single continuous layer of meaningful treated stand polygons. First, we separated group selection (GS) treatment areas from the other treatments, and created a new GS-only layer. Due to their small size and treatment intensity, the boundaries of these polygons were not altered, and would be added at the end of spatial manipulation. The next step was to correct for overlap in GIS treatment layers. Two layers (Spanish Camp and Ridge projects) had overlapping polygons within the individual layers. We edited these layers such that individual polygons were discrete, i.e., individual polygons were only represented by a single record in the attribute database. To correct for overlap between layers the order of treatment importance, from greatest to least was DFPZ Service Contract, DFPZ (both from Meadow Valley Project), Spanish Camp, Stanley, Ridge, McFarland, and Waters. These steps resulted in a single, non-overlapping layer of all treatments in the study area, accompanied by a concise table of dates, activity, sale, project, and ID. Table 4 summarizes the total treated area within our study area and proportion of the study area by timber sale/treatment type as represented in our final treatment layer.

We then merged this treatment layer with the existing VSTRA layer to have a continuous coverage of the Meadow Valley area. The GS polygons were then added to the layer, ensuring they did not change as a result of the above alterations. The next step was to generate tree lists for each polygon or 'stand' in the final layer combined layer. Tree lists came from field plots collected as part of earlier PLAS Fire and Fuels Module efforts, as well as from the Treated Stand Structure Monitoring (TSSM) program. Stands with field plots located within the polygon were 'populated' with trees using only those plot(s) within them. For stands without plots directly located within the polygon, which was by far the majority of stands, we used a non-statistical imputation approach based on the combination of elevation, slope, aspect, vegetation type, and vegetation size classes to generate tree lists. The rules for imputing went as follows for each stand: 1) if one or more plots matched a given stand based on the classes described, a random one was chosen; 2) if no plots matched exactly, classes were eliminated in the following order such that a plot did match a stand: aspect, slope, elevation, vegetation type, vegetation size class. Post-treatment stand tree lists came primarily from TSSM plots, with a small contribution from PLAS plots. Table 4 describes the five treatment types we used to apply plots, and thus tree lists, to treated stands.

After 'populating' stands both pre- and post-treatment with trees, we utilized the Forest Vegetation Simulator (FVS) to model growth of the treated and untreated landscape over 30 years, with data recorded every 10 yr cycle. We used the integrated platform ArcFuels (Ager *et al.* 2006) to run FVS and generate the necessary metrics needed to run spatial fire behavior models: canopy height, canopy base height, canopy cover, and canopy bulk density.

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Table 1. Characteristics of a modeled “problem fire” for the pre- and post-treatment Meadow Valley landscape. Area burned is summarized using two flame length (F_L) break values. These F_L values are based on operational constraints for fire suppression activities. The area intersecting treatments includes defensible fuel profile zone (DFPZ) and group selection (GS) units that were both burned over by the modeled fire and immediately adjacent to the fire perimeter. For both F_L classes and the area intersecting treatments, we also report the proportion of the final fire size represented in the respective classes.

Treatment phase	Fire size (ha)	Average F_L (m) (std. dev)	Area burned > 2.44 m F_L (ha) (proportion)	Area burned > 3.35 m F_L (ha) (proportion)	Modeled problem fire area intersecting treatments (ha) (proportion)		
					DFPZ	GS	Pre-Meadow Valley Project Treatments
Pre	9176	3.1 (3.3)	3772 (0.41)	2893 (0.31)	-	-	
Post	5569	2.7 (3.0)	2075 (0.37)	1546 (0.28)	809 (0.15)	100 (0.02)	1024 (0.18)

Table 2. Modeled runoff and sediment yield predicted using GeoWEPP (no treatment or fire, thin only, prescribed fire only) and GeoWEPP BAER (wildfire). Results are from sub-watersheds within the Boulder Complex (2006), Antelope Complex (2007), and Moonlight (2007) fire areas. Results are based on 4-year climate simulations, in which 2005 is used to initialize precipitation and generate snowpack needed to begin GeoWEPP simulations.

Sub-watershed	**Year**	**Area (ha)**	**Runoff (mm)**				**Sediment yield from the outlet (Mg/yr)**				**Sediment yield per unit area of watershed (Mg/ha/yr)**			
			No treatment or fire	Thin only	Prescribed fire only	Actual wildfire	No treatment or fire	Thin only	Prescribed fire only	Actual wildfire	No treatment or fire	Thin only	Prescribed fire only	Actual wildfire
Boulder	2006	148.0	109.7	116.7	97.7	90.3	164.2	132.4	224.2	193.5	1.1	0.9	1.5	1.3
Antelope		156.3	134.55	140.0	105.8	-	466.8	327.1	436.9	-	3.0	2.1	2.8	-
Moonlight		163.9	120.2	120.8	91.6	-	234.6	243.4	191.2	-	1.4	1.5	1.2	-
Boulder	2007	148.0	10.1	6.1	10.5	12.6	7.2	0.5	10.0	8.7	0.0	0.0	0.1	0.1
Antelope		156.3	14.1	10.7	15.4	19.6	27.8	14.9	38.2	38.0	0.2	0.1	0.2	0.2
Moonlight		163.9	8.3	2.18	7.7	15.3	1.6	0.0	1.3	5.7	0.0	0.0	0.0	0.0
Boulder	2008	148.0	0.3	1.2	0.2	0.7	0.0	0.0	0.0	0.1	0.0	0.0	0.0	0.0
Antelope		156.3	0.3	0.4	0.6	1.5	0.0	0.0	0.1	0.2	0.0	0.0	0.0	0.0
Moonlight		163.9	0.2	0.2	0.2	0.3	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
Sub-watershed averages														
Boulder		148.0	40.0	41.3	36.1	34.5	57.1	44.3	78.1	67.4	0.4	0.3	0.5	0.5
Antelope		156.3	49.7	50.4	40.6	41.2	164.9	114.0	158.4	145.6	1.1	0.7	1.0	0.9
Moonlight		163.9	42.9	41.1	33.2	34.0	78.7	81.1	64.2	59.6	0.5	0.5	0.4	0.4
Overall averages														
		468.2	44.2	44.3	36.6	36.6	100.2	79.8	100.2	90.9	0.6	0.5	0.6	0.6

Table 3. Modeled runoff and sediment yield predicted using GeoWEPP (no treatment or fire, thin only, prescribed fire only) and GeoWEPP BAER (wildfire) for the months immediately following the three fires: Boulder Complex (2006), Antelope Complex (2007), and Moonlight (2007). Results are reported for sub-watershed within each of the fire areas.

Sub-watershed	**Dates**	**Area (ha)**	**Runoff (mm)**				**Sediment yield from the outlet (Mg/yr)**				**Sediment yield per unit area of watershed (Mg/ha/yr)**			
			No treatment or fire	Thin only	Prescribed fire only	Actual wildfire	No treatment or fire	Thin only	Prescribed fire only	Actual wildfire	No treatment or fire	Thin only	Prescribed fire only	Actual wildfire
Boulder	Jun – Dec 2006	148.0	1.7E-4	1.7E-4	2.3E-4	2.3E-4	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
Antelope	Jul – Dec 2007	156.3	0.0	0.0	0.0	5.6E-4	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
Moonlight	Sep – Dec 2007	163.9	0.0	0.0	1.1E-3	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0

Table 4. Summary of both treated area within the Meadow Valley study area by treatment type and specific timber sale and the number of post-treatment plots actually used to ‘populate’ stands within each timber sale/treatment type.

	Sales	**TSSM plots**	**PLAS post plots**	**Area (ha)**	**Proportion of treated area (%)**	**Proportion of total area (%)**
BURN	McFarland	0	0	238.4	2.61	0.50
	Ridge	0	1	161.8	1.77	0.34
	Spanish Camp	0	3	1068.9	11.71	2.25
	Stanley	0	0	24.3	0.27	0.05
	Waters	0	1	1167.8	12.80	2.46
GS	Deans	0	0	124.3	1.36	0.26
	Guard	2	0	215	2.36	0.45
	PSW	0	0	5.7	0.06	0.01
	Silver	0	0	86.5	0.95	0.18
	Snake	1	0	159.8	1.75	0.34
HANDPILE	ServiceCon	0	0	262.2	2.87	0.55
	Waters	0	5	904.8	9.92	1.90
MAST	Waters	6	0	766	8.39	1.61
THIN	Deans	0	0	1044.2	11.44	2.20
	Guard	21	0	1426.8	15.64	3.01
	PSW	0	0	93.3	1.02	0.20
	Service	0	0	24.9	0.27	0.05
	Silver	5	0	639.8	7.01	1.35
	Snake	11	0	710.9	7.79	1.50
TOTAL		**46**	**10**	**9124.4**	**100**	**19.20**

Figure 1. Meadow Valley study area within the Plumas National Forest. The edge of acquired IKONOS imagery is identified to point out where the study area could not be buffered. We used weather data from both the Quincy and Cashman Remote Automated Weather Stations (RAWS).The towns are displayed for reference and to show proximity of the study area to communities.

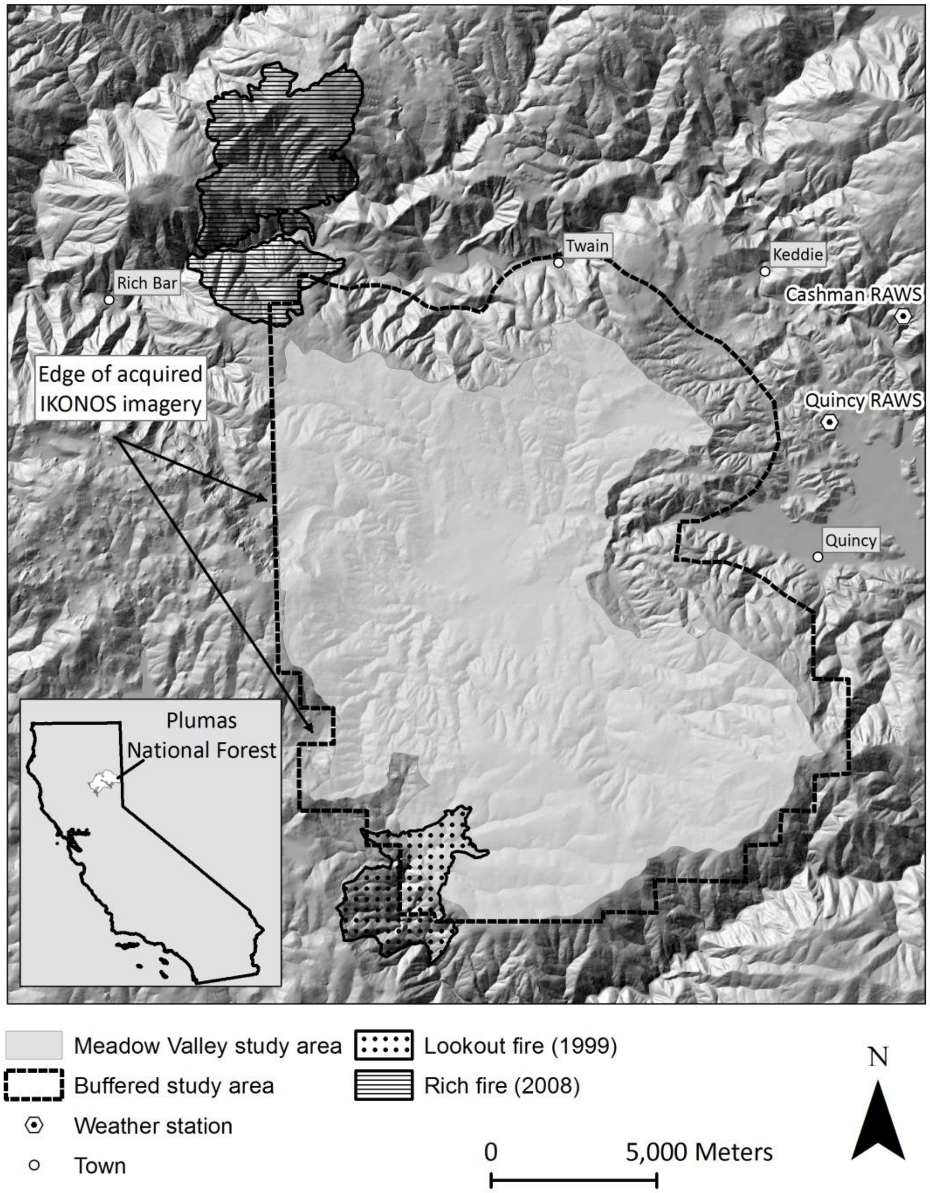


Figure 2. Conditional burn probabilities based on 1000 randomly placed ignitions using the Minimum Travel Time function in FlamMap (Finney 2006). Polygon outlines for both Defensible Fuel Profile Zone (DFPZ) and Group Selection (GS) treatments implemented as part of the Meadow Valley coordinated landscape fuel treatment are also identified.

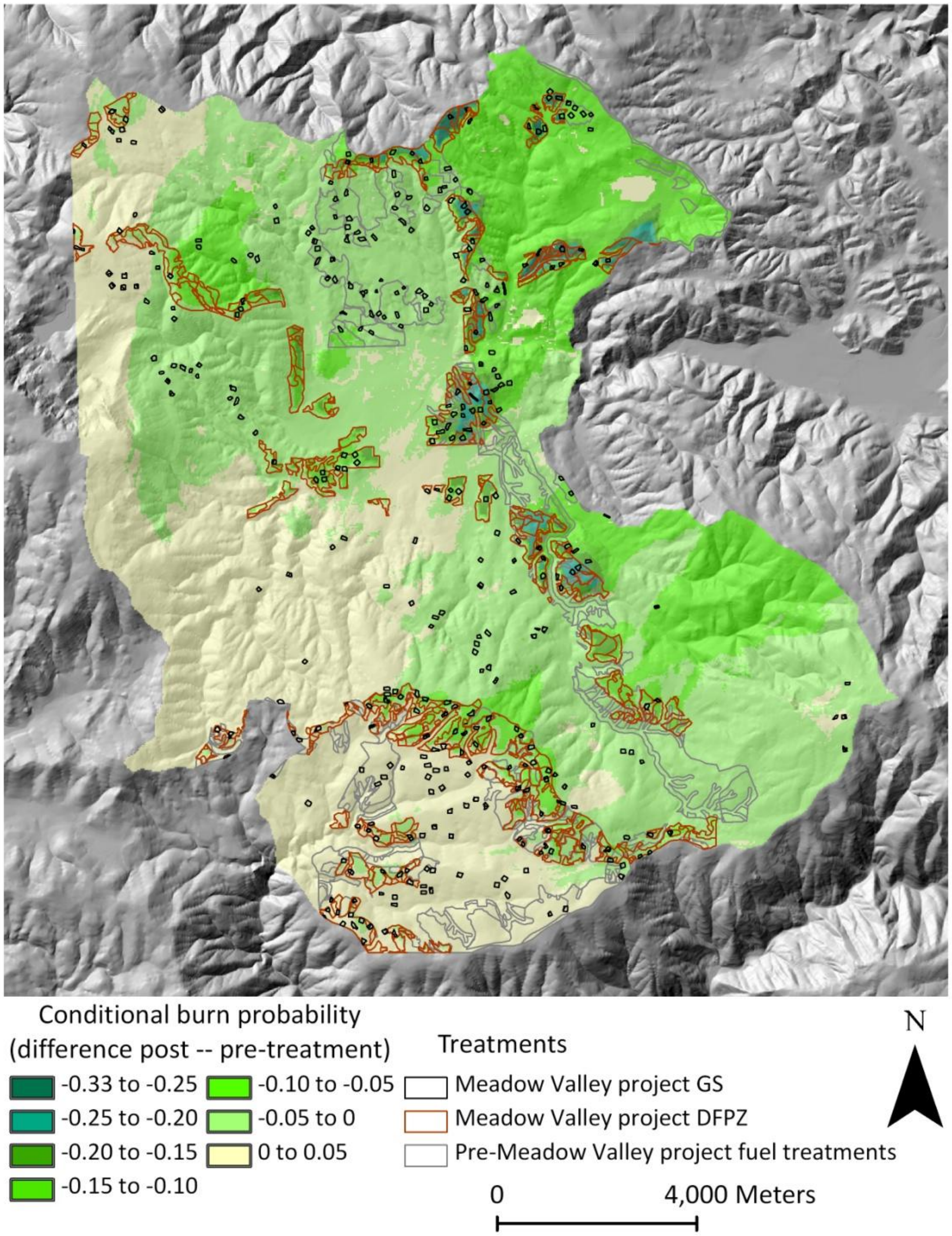


Figure 3. Modeled area burned, classified into flame length categories, based on a single "problem fire" scenario. This scenario consisted of an ignition on the southwest edge of the landscape (for which we used and actual fire that occurred in 1999) burning under fairly severe conditions. This "problem fire" was simulated using FARSITE for the pre- (left) and post-treatment (right) Meadow Valley landscape. Polygon outlines for both Defensible Fuel Profile Zone (DFPZ) and Group Selection (GS) treatments implemented as part of the Meadow Valley coordinated landscape fuel treatment are also identified.

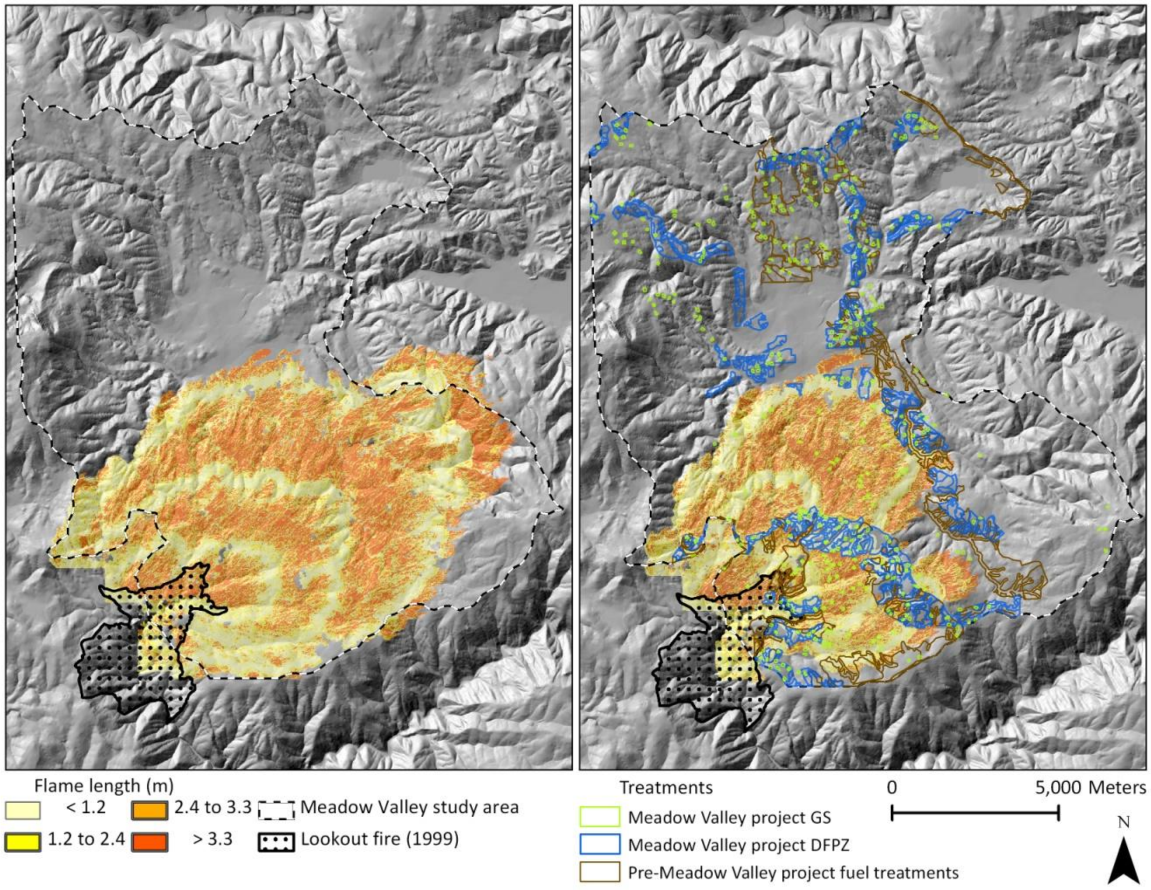


Figure 4. Ground-based and aerial photos of Antelope Lake, California, USA. The aerial photo includes approximate perimeters of recent wildfires burning immediately adjacent to the lake.

Figure 5. (A) Frequency distribution of RdNBR fire severity pixels for the area burned by all 4 recent wildfires (2001-2007) within the Antelope Lake watershed. (B) Proportion of watershed burned, by fire severity class for each of the fires individually. Thresholds for the fire severity classifications are based on Miller and Thode (2007).

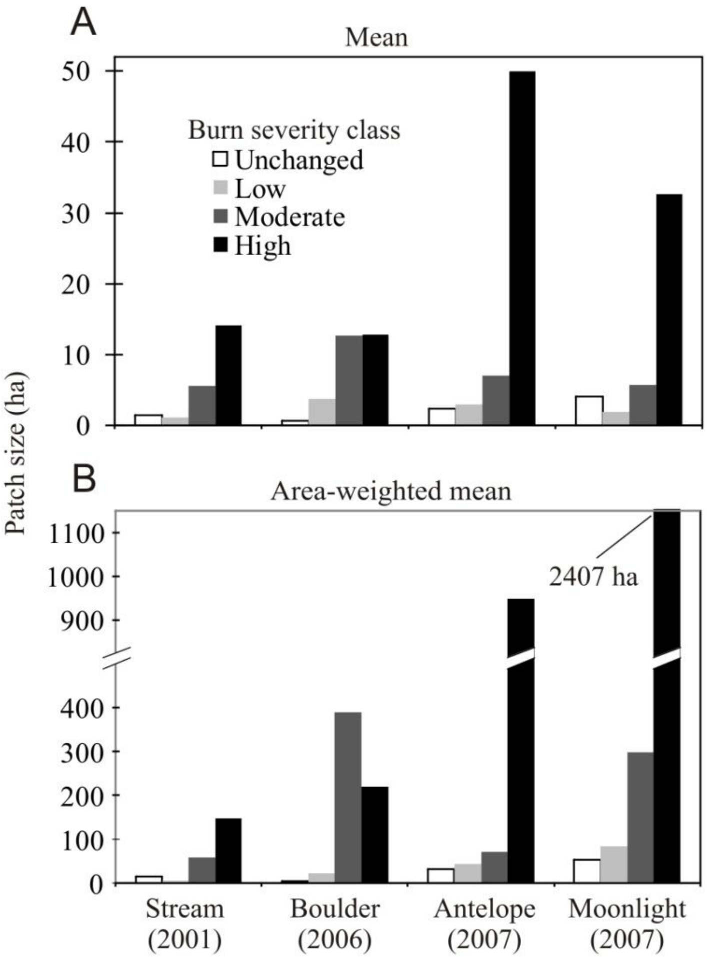
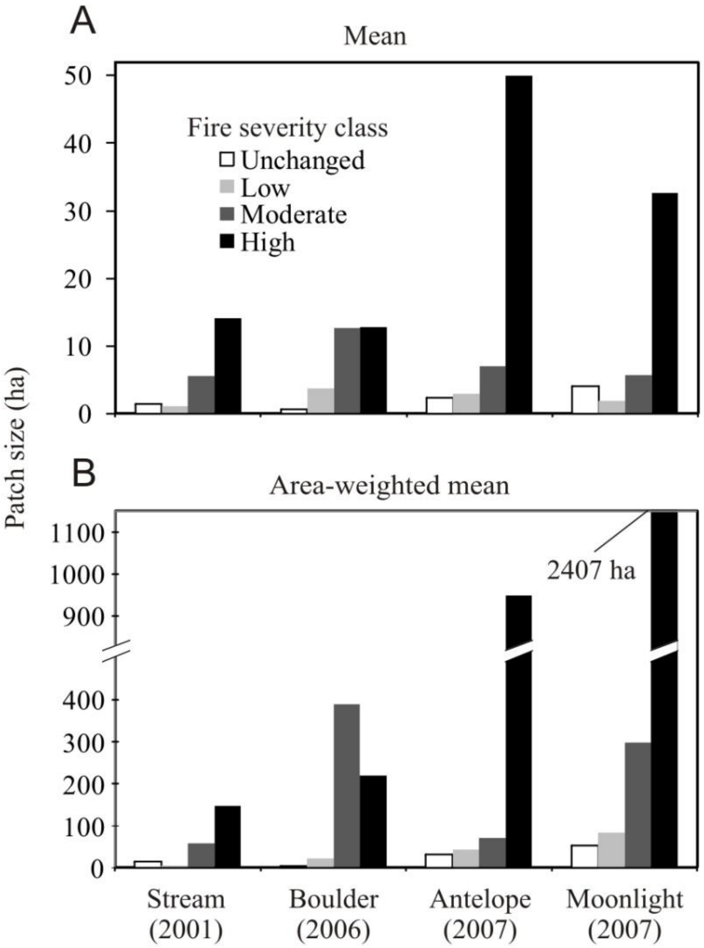


Figure 6. (A) Mean and (B) area-weighted mean patch sizes, within the Antelope Lake watershed, for each of the 4 recent wildfires by fire severity class. Thresholds for the fire severity classifications are based on Miller and Thode (2007).



Plumas-Lassen Administrative Study Small Mammal Module:

Distribution, Abundance, and Habitat Relationships

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EXECUTIVE SUMMARY

The Mammal Module of the Plumas-Lassen Administrative Study (PLAS) completed its seventh year of field efforts in 2009. Field efforts extended from the beginning of May through the end of October. We resurveyed 21 permanent small mammal live trapping grids and quantified conifer cone production for use in models to predict abundance of deer mice (*Peromyscus*), an important prey species of spotted owls. We pursued a third year of sampling a series of shorter transects initially established by the Land Bird Module (hence referred to as "land-bird transects"), surveying 30 transects to bring our total sample to 74 of the ca. 95 transects established by the LB Module (remaining transects are too isolated for small mammal surveying, which requires large numbers of heavy traps, bait, and twice daily visits for numerous days). We completed the most comprehensive study to date on home range characteristics and habitat selection by northern flying squirrels, and Mr. Jaya Smith completed his Master of Science degree at UC Davis (Graduate Group in Ecology).

In 14,448 trap-nights of effort on 21 permanent live-trapping grids we recorded 1,672 captures of 823 individual small mammals representing 10 species. As in past years, predominant species were deer mice, chipmunks (both Allen's and long-eared), ground squirrels (California and golden-mantled), Douglas squirrels, long-tailed voles, California red-backed voles, and northern flying squirrels. *Peromyscus* numbers, so far, are best modeled as a function of three external parameters – annual snowfall and cone productivity by firs and sugar pine; model fit increased with cone productivity by Douglas fir, and moderately so with current year rainfall. *Peromyscus* availability may be important in spotted owl reproductive success, giving simple models such as this potential management importance. Cone counts in 2009 were the greatest recorded in this study, suggesting that 2010 field season may be characterized by very high deer mouse numbers if the 2009-2010 winter is not severe enough to limit *Peromyscus* numbers. An increase in deer mouse abundance is potentially very good for spotted owls but problematic for humans, as positive density-dependent transmission of hantavirus may result in large numbers of infected deer mice.

Land-bird transects also were very successful in 2009, with 23,040 trap-nights of effort on 30 transects resulting in 2,016 captures of 1,367 individual small mammals, representing 11 species. Species captured included those listed above, plus dusky-footed woodrat (*Neotoma fuscipes*).

Mr. Jaya Smith completed his comprehensive study of spatial and denning behavior of northern flying squirrels, and has two manuscripts submitted from his MS degree. One of these describes home range size, overlap, and ecological characteristics. The other applies patterns of home range overlap to infer patterns of mating behavior in this species. Both chapters address issues of importance to land managers, as northern flying squirrels are key prey species for spotted owls.

The mammal module has completed three MS projects and published 6 papers thus far. We have three additional papers submitted and another in preparation. We have data in hand for one more paper on forest-wide habitat associations of small mammal species, and after a final abbreviated field season in 2010 will have data for a final mammal paper on short-term (2-3 yr) responses of small mammals to thinning treatments. We are eager to interface further with other modules in this project, in particular the land-bird module with which we anticipate sharing data on small mammal and bird species composition at 74 transects.

INTRODUCTION

Small mammals play vital roles in forest ecosystems, serving as important consumers and dispersers of seeds, fruits, and fungi (Carey et al. 1999; Gunther et al. 1983; Maser and Maser 1988; Pyare and Longland 2001), and as prey for mammalian and avian predators, including many species of concern in the Sierra Nevada (e.g., California spotted owl, *Strix occidentalis occidentalis*; northern goshawk, *Accipiter gentilis*; fisher, *Martes pennanti*; marten, *M. americana*; Carey et al. 1992; Forsman et al. 1984; Zielinski et al. 1983). Given their essential interactions with flora and fauna across multiple trophic levels (e.g., Carey et al. 1992; Forsman et al. 1984), changes in the distribution and abundance of small mammals could substantially affect the dynamics of forest communities. As a result, small mammals are valuable subjects for the integrative research necessary to fully understand the ecological responses of spotted owls and other species to forest management practices.

Here we report on the Mammal Module of the PLAS, one of five integrated study modules intended to evaluate land management strategies within the area covered by the Herger-Feinstein Quincy Library Group Forest Recovery Act (HFQLG) Pilot Project. Forest management practices implemented during the HFQLG Pilot Project aim to "…promote ecologic and economic health for federal lands and communities of the northern Sierra Nevada" (Title IV-HFQLG Forest Recovery Act- Sec.401). Toward this end, understanding how small mammal communities respond to different forest management regimes both at the scale of macrohabitat (i.e., stand-level, landscape scale) and microhabitat (trap-level, home range) is essential and will provide valuable feedback to other PLAS modules. Managers manipulate fuel loads through mechanical thinning of the forest in an effort to prevent catastrophic wildfire events and to create more suitable conditions for prescribed burning. Such management activities are becoming more frequent throughout western North America (Covington et al 1997, Dodge 1972, Long et al. 2007). Exactly how these management practices effect small mammal populations, particularly in the Sierra Nevada, is not well understood (Carey 2000, Meyers et al 2007, Suzuki and Hayes 2003). Most studies focus on the short-term responses of small mammal communities to habitat manipulation (e.g., Carey 2000, Carey and Wilson 2001, Converse et al. 2006, Klenner and Sullivan 2003, Meyers et al. 2007); to date, this study is similarly temporally constrained. Long-term studies have the potential to record a broader range of variability in community dynamics which will lead to a greater understanding of ecosystem processes (Beche and Resh 2007); i.e., distilling natural fluctuations in population, climatic effects, and predator responses from the effects of management practices. The value of long-term studies has been exemplified in many field studies (e.g., Beche and Resh 2007, Brook and Bradshaw 2006, Elias et al 2006, Jackson and Furender 2006, Pettorielli and Durant 2007, Tsuji et al 2006). When considering the goals of the HFQLG Forest Recovery Act, the long-term study approach is most appropriate for the mammal module. We will continue to develop predictive small mammal habitat models to forecast how individual species will respond to forest management treatments and test these models by assessing the impacts of forest management treatments on long term responses of small mammal abundance and species diversity. This is done by monitoring several independent populations of small mammals for multiple years before and after forest management treatments are applied, developing demographic profiles (e.g., survival, reproduction) of species, and obtaining detailed measurement of habitat characteristics. To sample and monitor these small mammal populations, we have established both permanent trapping grids and more ephemeral trapping transects distributed in a stratified random manner throughout the Plumas National Forest (PNF). The former trapping grids allow us to document temporal dynamics in small mammal numbers in response to habitat manipulation. The latter allow for a more immediate "snapshot" of faunal structure across the range of habitats available throughout PNF. We anticipate that this complementary approach will provide meaningful insight to mammal habitat relations as well as longer-term temporal dynamics.

Key non-hibernating small mammals in the northern Sierra Nevada include the northern flying squirrel (*Glaucomys sabrinus*), dusky-footed woodrat (*Neotoma fuscipes*), and mice (mostly *Peromyscus* sp.). Northern flying squirrels and dusky-footed woodrats are the principle prey of the California spotted owl (Carey et al. 1992; Rosenberg et al. 2003), a species of concern in California due to its dependence upon late-seral forest ecosystems (United States Department of the Interior 2003). These late-seral ecosystems are among the most highly altered ecosystems in the Sierra Nevada (Beardsley et al. 1999; Franklin and Fites-Kaufman 1996). Because some populations of northern flying squirrel may be depressed by the intensity of spotted owl predation (Carey et al. 1992), and high woodrat biomass may reduce the area requirements of the spotted owl (Carey et al. 1990; Zabel et al. 1995), northern flying squirrels and dusky-footed woodrats has been an important focus of our study module.

<u>Northern flying squirrels</u> are nocturnal, arboreal rodents located throughout the northern latitudes of the United States and Canada (Wells-Gosling and Heaney 1984), and frequently are associated with forests possessing high densities of large trees (Smith et al. 2004, 2005). Northern flying squirrels act as a major dispersal agent for hypogeous fungal spores, which are important for nutrient and water uptake by host trees (Fogel 1980). Although typically associated with mesic northern forests, northern flying squirrels are also found throughout forested regions of the Sierra Nevada, although these typically are more xeric environments than elsewhere in their range. Because of the dryer conditions here, populations of northern flying squirrel inhabiting the Sierra Nevada may be quite different from those inhabiting the more mesic forests of Oregon, Washington, and Alaska. Specifically, northern flying squirrels may be more sensitive to the distribution of relatively mesic habitats in the Sierra Nevada since this influences the distribution of truffles, their primary food source. The more dispersed distribution of food resources may drive differences in northern flying squirrel biology, and suggest that northern flying squirrels may exhibit a more clumped distribution, lower overall densities, increased competition for suitable nest trees, and larger individual home ranges. Therefore, northern flying squirrels in the Sierra Nevada may be affected differently by forest management practices than in other parts of their range. We have incorporated both live-trapping and radiotelemetry to determine the abundance and distribution, habitat use, and home range of northern flying squirrels in the Sierra Nevada, comparing this with data from other parts of their distribution, and evaluating the effects of forest

management practices on this species within the area covered by the HFQLG Pilot Project.

Dusky-footed woodrats are nocturnal and semi-arboreal, and occur in suitable habitat throughout northern California and Oregon. Preferred habitat includes chaparral, juniper woodland, streamside thickets, and deciduous or mixed forests with well-developed undergrowth (Carraway and Verts 1991). Dusky-footed woodrats play an important role in community dynamics. As mentioned previously, they are prey for many avian and mammalian predators, including the California spotted owl. Additionally, the availability of woodrat houses may influence species richness for small mammals, reptiles, amphibians, and invertebrates (Cranford 1982; M'Closkey et al. 1990; Merritt 1974; Vestal 1938). Thus, promoting quality habitat for the dusky-footed woodrat may provide a variety of ecological values in managed forests, for example in the form of increased biodiversity, with important consequences for forest conservation (Carey et al. 1999). We used live-trapping and radiotelemetry to determine the abundance and distribution, habitat use, and home range of dusky-footed woodrats in the Sierra Nevada, and evaluate the effects of forest management practices on this species. Specifically, we tested for an association between woodrat abundance and abundance of California black oak (*Quercus kelloggii*), an important food source (Atsatt and Ingram 1983; Cameron 1971; Meserve 1974). We also evaluated the importance of microhabitat variables at 2 scales – placement of houses within mixed-conifer habitat, and use of houses within home ranges.

Deer Mice (*Peromyscus maniculatus*) and brush mice (*P. boylii*) inhabit PNF, although the former is generally more common than the latter. These species differ in both behavior and morphology, yet possess similar foraging habits. Due to the ubiquitous nature of deer mice, there is a great overlap in habitat affinity with the brush mouse (Jameson 1951). Deer mice are nocturnal, non-hibernating, terrestrial to semi-arboreal, and are distributed throughout the Sierra Nevada (King 1968; Martell and Macaulay 1981; Jameson and Peters 2004) and forage on a variety of seeds, berries, insects, and hypogeous fungi. Deer mice are by far the most common mouse-sized mammal within the study areas of the Mammal Module in PNF, comprising 98% of such captures. Because of the overwhelming dominance of this species we group these as *Peromyscus* sp.; because their skulls are difficult to distinguish in owl pellets, this is a common practice in spotted owl diet studies as well. Foraging habits of the spotted owl may vary as functions of habitat and prey availability, and surprisingly little is known regarding the diet of California spotted owls in the northern Sierra Nevada. Studies elsewhere indicate that *Peromyscus* species are common prey items although they do not dominate the percent of biomass ingested by the spotted owl (Block et al. 2005, Bravo-Venaja 2005, Forsman et al. 2004, Rosenberg et al. 2003, Smith et al. 1999, Trailkill and Bias 1989). However, reproductive success in northern spotted owls is positively correlated with trends in *Peromyscus* abundance (Rosenburg et al. 2003), and Munton and others (2002) recorded an increase in *Peromyscus* taken during the breeding season of California spotted owls in the Sierra National Forest. Hence, it may be that years with abundant *Peromyscus* provide an additional energetic margin to allow for increased reproductive success in this owl species.

Other key small mammals include the golden-mantled ground squirrel (*Spermophilus lateralis*[1]) and both Allen's and long-eared chipmunks (*Tamias senex* and *T. quadrimaculatus),* which also are important prey species of the northern goshawk (*Accipiter gentilis*), a species of increasing concern to resource managers due to its sensitivity to forest management. It is surprising, considering the abundance and wide distribution of the golden-mantled ground squirrel, that little scientific attention has been awarded to this species. Social spermophilines have been intensely investigated (Michener 1983), whereas asocial species such as the golden-mantled ground squirrel have not. We aim to evaluate the spatial distribution and social organization of the golden-mantled ground squirrel; thus promoting further investigation of habitat selection by this species. Chipmunks are forest- or shrub-associated, semi-arboreal rodents that constitute a considerable portion of the small-mammal biomass in an area, making them important prey for a variety of mammalian and avian predators (Vaughan 1974). Additionally, chipmunks are important consumers and dispersers of seeds (Briggs and Vander Wall 2004; Vander Wall 1992), and may contribute to the natural regeneration of some species of plants by caching seeds (Aldous 1941). Small mammals may cache seeds beneath the layer of decaying vegetation on the forest floor, where they stand a better chance of germinating than those remaining on the surface litter (Sumner and Dixon 1953); alternatively, they may deposit seeds in underground burrows where seeds cannot establish seedlings. If soil-moisture levels have been altered due to fire, logging, or weather patterns, the ability of chipmunks to retrieve cached seeds may be reduced, thus promoting germination of a larger proportion of seeds after disturbance (Briggs and Vander Wall 2004;Vander Wall 2000). However, when very abundant, chipmunks may prevent normal regeneration of some plants, particularly pines, by eating their seeds, which may contribute to the generation of dense brushfields that could further hinder the return of timber (Smith and Aldous 1947, Tevis 1953). Both *T. senex* and *T. quadrimaculatus o*ccur commonly throughout PNF. These sympatric species are similar in body mass, diet, and general resource utilization, and thus are likely to compete locally. Similar species often coexist by partitioning habitat. However, detecting differences in habitat affinities is influenced by spatial scale. In addition to evaluating the affect of forest management practices on these species we also are quantifying the abundance, distribution, and habitat associations of both chipmunk species at multiple spatial scales in PNF.

OBJECTIVES

The primary objective of the Mammal Module is to evaluate small mammal responses to different forest management practices, and to model these responses in terms of demography, spatial distribution, and habitat associations at local and landscape scales. To meet the primary objective, we will address the following:

1. Determine small mammal habitat associations at the scale of both macro- and microhabitat.
2. Develop demographic profiles of small mammal populations inhabiting a variety of habitat types.

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[1] Recently elevated to *Callospermophilus lateralis*; Helgen et al. 2009. J. Mammalogy 90:270-305.

3. Develop predictive small mammal habitat models, based on the results of objectives 1-2, to forecast how individual species will respond to forest management treatments.
4. Quantitatively assess the impacts of forest management treatments on small mammal abundance and species diversity.
5. Determine small mammal population trends, evaluate how populations are changing temporally, and assess the factors responsible for the observed trends.
6. Evaluate the spatial distribution (i.e., home range), social organization (i.e., home range overlap), and habitat selection (i.e., den use, house use) of the principle prey of the California spotted owl, the northern flying squirrel and dusky-footed woodrat, as well as other small mammal species.

MATERIALS AND METHODS

Live-trapping

Capture-recapture data allow us to measure population parameters such as abundance, density, and frequency of occurrence of individual small mammal species and small mammal species richness and diversity, and permit the measurement of habitat use, availability, and selection (Lancia et al. 1996, Litvaitis et al. 1996). Live-trapping methods are useful for making comparisons of small mammal communities across time, locations, habitats, and land-use treatments. We established several different live-trapping designs, each appropriate to the small mammal community or species of interest.

Long-term grids

To provide base-line information on small mammal populations inhabiting major forest types, and to quantitatively assess the impacts of forest management treatments on small mammal abundance and species diversity, we established 21 long-term grids using controls and pre- and post-treatment data (Fig. 1a; Table 1). In summer 2003 we established 18 long-term grids in 4 habitat types named after dominant tree species: white fir (n=9), red fir (n=3), Douglas fir (n=3), and pine/cedar (n=3). In 2005, 3 additional sampling grids were established in “group select” treatments located in white fir (n=3) habitats. Overall, PNF is dominated by white fir forest types and therefore have proportionally more trapping grids placed within them. Common shrubs in the region include mountain rose (*Rosa woodsii*), Sierra gooseberry (*Ribes roezlii*), serviceberry (*Amelanchier utahensis*), bush chinquapin (*Chrysolepis sempervirens*), green- and white-leaf manzanita (*Arctostaphylos patula* and *A. viscida*), mountain dogwood (*Cornus nuttallii*), mountain whitethorn and deer brush (*Ceanothus cordulatus* and *C. intigerrimus*), bitter cherry (*Prunus emarginata*), willow (*Salix* spp.), Fremont silk tassel (*Garrya fremontii*), Sierra coffee berry (*Rhamnus californica*), and huckleberry oak (*Quercus vaccinifolium*). Pinemat manzanita (*Arctostaphylos nevadensis*) occurred almost exclusively in red fir forests, and buck brush (*Ceanothus cuneatus*) predominantly in pine-cedar forests.

Twelve long-term grids were placed within the experimental management plots established by the Vegetation Module of the PLAS. These 12 study plots were placed in 3 groups (Miller Fork, Dean’s Valley, and Snake Lake) of 4 study plots each, consisting of 1 control plot and 3 experimental plots (1 group select plot, 1 light thin, and 1 heavy thin). The remaining 9 study plots were not established in groups. Most grids were ≥1 km (≥0.6 mi) apart; 4 grids were 700-900 m (766-984 yds) apart. Movement between grids was extremely rare; we documented a single golden-mantled ground squirrel that moved from red-fir grid #12 to red-fir grid #6 in 2006.

Twenty grids consist of a 10 x 10 array of Sherman traps (Model XLK, 7.6 x 9.5 x 30.5 cm, H. B. Sherman Traps, Inc., Tallahassee, FL, USA) with 10 m (33ft) spacing, nested within a larger 6 x 6 grid of 72 Tomahawk traps (Model 201, 40.6 x 12.7 x 12.7 cm, Tomahawk Live Trap, Tomahawk, WI, USA; 1 ground, 1 arboreal) with 30 m (98 ft) spacing (Fig. 1b). The final long-term grid was constrained by road configuration such that the array of Sherman traps was nested within a 4 x 9 grid of 72 Tomahawk traps (30 m [98 ft] trap spacing; 1 ground, 1 arboreal). Total area represented was identical to that of other grids – 2.25 ha (3.24 ha [8 acres] with a ½ inter-trap distance buffer). Ground traps were placed within 1 m (3 ft) of the grid point under protective cover, such as a shrub or log, at small mammal burrow entrances, and along small mammal runways, when possible. Arboreal traps were placed approximately 1.5 to 2 m (5 to 7 ft) above the ground on the largest tree within 10 m (33 ft) from the grid point; arboreal traps were not constrained to be placed on the same tree each trapping session.

To maximize resolution on seasonal dynamics, long-term grids were trapped monthly (May-October) in 2003 and 2004. We reduced this effort to biannual sampling (June, Oct) in 2005 and 2006. Canopy treatments (including logging and prescribed burning) were effected in 2006 and 2007, limiting our access to many grids in the latter year; consequently, we sampled only once in 2007 (July-August). Subsequently, however, we have sampled biannually (2008 and 2009. Arboreal Tomahawk traps were removed from all grids on August 1, 2004 because of consistently poor capture rates, but were reinstated in 2005; capture rates were improved by placing the trap entrance flush against the tree bole, fastening the trap more securely to the tree, and switching to a bait mixture recommended by Carey et al. (1991).

Prior to August 2005, all traps were baited with crimped oats and black oil sunflower seeds lightly coated in peanut butter; thereafter, traps were baited with a mixture of rolled oats, molasses, raisins, and peanut butter which was formed into a small, sticky ball (Carey et al. 1991). Small nest boxes made from waxed-paper milk cartons were placed behind the treadle in Tomahawks to minimize stress and provide thermal and protective cover (Carey et al. 1991); in addition, synthetic bedding material (nonabsorbent polyethylene batting), and natural cover (e.g., bark, moss) or cover boards, were provided as needed for thermal insulation.

Trapping sessions consisted of 4 consecutive trap-nights, totaling 688 trap-nights[2] of effort. Sherman and Tomahawk traps were set and baited every evening just before dusk, and checked just after dawn. Sherman traps were then closed during the morning trap revision, whereas Tomahawk traps were re-baited, checked again at mid-day (≥2 hrs after the first trap check), then closed until dusk. Field technicians were thoroughly trained and rotated among grids each trapping session, to reduce the variability in capture success due to differences among technicians.

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[2] One trap-night equals one trap run for a single night.

We trapped all long-term grids (n=21) in 2009.

Land-bird transects

The Land-bird Module of PLAS established a large number of U-shaped transects with 12 point-count stations for surveying birds. These were randomly distributed throughout the PNF and stratified by habitat type. To complement the data collected at our long-term grids and more fully integrate our live-trapping efforts with that of other modules, we established temporary, small-mammal trapping grids at a subset of Land-bird Module census points in 2006 – 2007 and 2009. We randomly selected 8 census points within each land-bird census transect for small mammal sampling (e.g. Figs. 2, 3); selection of census transects was stratified to include transects located throughout (former) treatment units 2-5 (Fig. 4). At each census point, a 2 x 2 array of live-traps with 50 m (164 ft) spacing was established by pacing 35 m (115 ft) from the census point in the four cardinal directions (north, south, east and west; Fig. 5). Each array covers 0.25 ha (1 ha [2.4 acre] with a ½ inter-trap distance buffer). All live-trapping methods were designed to optimize the capture and recapture of northern flying squirrels, the most difficult to capture small mammal in our study area, and in this way provide the best means of trapping for the full suite of small mammals (Carey et al. 1991). The live-trap array we used ensured that the 4 trap stations resided within the 50 m (164 ft) radius vegetation plot that was established by the Land-bird Module to access vegetation characteristics around each census point, as well as meeting recommended spacing between trap stations and the suggested minimum number of trap stations per home range area recommended for the northern flying squirrel (Carey et al. 1991).

As with long-term grids, we sampled all stations on land-bird transects with 1 Sherman and 2 Tomahawk (ground + arboreal) traps placed at each point in the array; thus, each array consisted of 12 live traps, and each transect included 96 traps. Arboreal Tomahawks were placed 1.5 to 2 m (5 to 7 ft) above the ground on the largest tree within a 10-m (33 ft) radius of the grid point. The largest tree was chosen since large trees provide better support for the trap, improving trap functionality and improving capture success (Carey et al. 1991). Ground traps were placed within 1 m (3 ft) of the grid point and were placed under protective cover, such as a shrub or log, at small mammal burrow entrances, and along small mammal run ways, when possible. Trap baiting and setting protocols were identical to those used on the long-term grids.

Because sampling all Land-bird transects could not be accomplished within a single season, we have divided this effort across 3 field seasons - May-October 2006, 2007, and 2009. Trapping session consisted of 2 4-night sessions separated by 2 nights; this allowed for suitable time off for field crews and allowed a period of rest for animals from the stress of capture and handling (Carey et al. 1991). Hence, assuming no interruptions, each transect was surveyed with 768 trap-nights of effort. All traps were set and baited every evening just before dusk; baiting was completed in 2-4 hours. Traps were checked just after dawn and completed within 4-6 hours; all traps were closed prior to 12:00 and remained closed until after 16:00 each day.

We analyzed species richness and abundance indices for each sampled land-bird census point. Species richness and abundance are defined as the total number of species detected and individuals, respectively. We utilized a restricted list of species that excluded species that are not accurately surveyed using our live-trapping method (e.g., shrews, skunks, hares). Following the completion of data collection in 2009, we plan to begin more detailed analyses of these data in close collaboration with the Land-bird Module.

Northern flying squirrels

We captured and radiocollared northern flying-squirrels in selected areas with good habitat for this species, and where triangulation was feasible. "Good" habitat was selected using a GIS-based northern flying squirrel habitat-relations model of known habitat affinities for this species, and which predicted poor, moderate, and high suitability habitat for northern flying squirrels. We established a number of live-trapping transects (>10) in three areas predicted to have high and moderate suitability (Fig. 6). Northern flying squirrels also were captured at long-term grids and land bird transects which were feasible for triangulation efforts.

We established flying squirrel transects primarily along riparian areas, due to the importance of this habitat type to northern flying squirrels (Meyer and North 2005) but also reflecting the availability of intact stands. If habitat, road configuration, and topography were suitable, we used a live-trapping grid (i.e., several parallel transects) to maximize the number of captures. We used a combination of Sherman and Tomahawk traps (typically 1 Sherman and 2 Tomahawk as with land bird transects and long-term grids), spaced 40-50 m (131-164 ft) apart. Sherman and Tomahawk traps were set and baited every evening just before dusk, and checked with the same schedule as our long-term trapping grids. As indicated above, we changed our trapping bait and protocol for flying squirrels in August 2005.

Animal handling

Captured animals were transferred to a mesh handling bag, identified to species, individually marked with numbered Monel ear tags (National Band & Tag Co., Newport, Kentucky), weighed, aged, measured (e.g., ear length, hind foot length), examined for reproductive status, and released at the point of capture. Total processing time for an experienced technician was generally <2 minutes. Reproductive condition for males was scrotal (enlarged and scrotal testes) or non-scrotal (reduced and abdominal testes); for females, the vagina was noted as either perforate (thereby receptive) or imperforate (not receptive), the vulva as either swollen or not, and the animal as lactating (nipples were enlarged and/or reddened, reflecting nursing offspring) or not. Age was assessed based on weight, pelage (juvenile: gray, subadult: intermediate, and adult: brown), and reproductive condition (juvenile/subadult: nonreproductive, adult female: pregnant/lactating, and adult male: scrotal).

Trap mortalities were very low, but any such specimens were retrieved and frozen for subsequent donation to the University of California, Davis, Museum of Wildlife and Fish Biology. All field work and handling procedures are approved by the University of

California, Davis Animal Use and Care Administrative Advisory Committee protocol, and meet guidelines recommended by the American Society of Mammalogists (Animal Care and Use Committee 1998; Gannon et al. 2007).

Radiotelemetry

Movement and location data obtained via radiotelemetry allow quantification of home range area and characteristics, habitat selection, and movement patterns, and documentation of denning, nesting, and resting sites. Additionally, social organization may be inferred by patterns of home range overlap (Lancia et al. 1996, Litvaitis et al. 1996). Radiotelemetry methods are useful for making comparisons of small mammal movements and space use across time, locations, habitats, and land-use treatments.

Radiotransmitter application

Between 2003 and 2007 we applied radiotransmitters to three species – golden-mantled ground squirrels, northern flying squirrels, and dusky-footed woodrats. A 4.0 g (0.1 oz) collar-type radio transmitter (Holohil Systems Ltd., Model PD-2C) was fitted to target individuals after light sedation with ketamine hydrochloride (100mg/ml) injected into the thigh muscle. Flying squirrels were not anesthetized prior to radiocollaring in earlier years; however in 2007, the use of anesthesia was implemented to avoid handling-related stress. A mild level of anesthesia was achieved by administering 0.8-1.0ml (0.02-0.03 oz) of a low concentration mixture of ketamine hydrochloride (1.4mg, 100mg/ml) and xylazine (0.06 mg, 100mg/ml) in a saline solution. Animals were allowed to fully recover from anesthesia prior to being released at the point of capture. Radiotelemetry activities of newly collared individuals were initiated only after a 24-hour acclimation period.

Homing

To document the location and frequency of use of denning, nesting, and resting sites we used homing techniques. Adult golden-mantled ground squirrel burrows were located 1-2 times a week in 2003, whereas juveniles and adults were located sporadically in 2004 and 2005. For northern flying squirrels, diurnal locations were determined sporadically in 2003-2005, 1-2 days per week in 2006, and 1 day per week in 2007. Locations were marked and accurately (≤ 1 m or 3 ft) mapped using a Trimble GPS unit.

Triangulation

We pursued diurnal telemetry sessions on golden-mantled ground squirrels for 5 days/mo from July to September 2003. Nocturnal triangulation of radiocollared northern flying squirrels was conducted for 5 nights per month in 2003 and 8-10 nights per month during 2004-2007. Radiolocations were obtained for each animal 2-3 times per session at a minimum of 2 hours apart to avoid serial correlation (Swihart and Slade 1988, Taulman and Smith 2004). We used a Yagi® antenna and a hand-held radiotelemetry receiver (Model R-1000, Communications Specialists, Orange, CA, USA) to obtain the location of radiocollared animals. Compass bearings were obtained with a hand-held compass and bisecting the signal drop-offs. We established fixed telemetry stations (mapped to within 1 m accuracy using a Trimble GPS unit) that allowed us to record bearings to collared animals without approaching or otherwise disturbing them. Technicians worked in synchronized teams to achieve 3-6 directional bearings within a brief period (usu. <10 min.). The timing of telemetry efforts was varied to ensure that radiolocations were sampled at different times of activity. To reduce error among field workers, all technicians were thoroughly trained and were rotated among stations and study sites. Additionally, error was quantified using 1-2 “dummy” collars placed in known locations within each study area; technicians did not know dummy collar locations, and the dummy collars were moved weekly. To assess bearing error rates, dummy collar locations were determined through triangulation and compared to their actual location previously mapped with a Trimble GPS unit. All bearings were accompanied by field notes recording the quality of a fix (good, intermediate, poor), and all poor fixes were removed prior to analysis.

We used Locate III (Nams 2006) to calculate animal locations. We used several criteria to evaluate bearing data and determine animal locations. These included convergence of bearings, presence of outliers, number of bearings (≥ 4), and signal quality. To minimize the influence of radiosignal “bounce” in this topographically challenging environment, we removed the two most divergent bearings until ≥4 bearings were used for triangulation. Accepted locations were analyzed in Ranges VI. We estimated home range (95%) and core range (50%) using the Minimum Convex Polygons (MCP) (Mohr 1947) and Fixed Kernel (FK) methods (example in Fig. 7; Kenward 2001).

Home range analysis

We applied incremental area analysis to assess the number of fixes necessary to delineate home ranges (e.g., Fig. 8). For northern flying squirrels, home ranges stabilized after an average of 34 points were collected.

We constructed home ranges using RangesVI (Kenward et al. 2003), and used these to determine habitat preference with compositional analysis (Aebischer et al. 1993). Compositional analysis quantifies use of habitat as a function of habitat availability. We applied compositional analysis at two spatial scales. The first compares the habitat in an animal’s home range to that available in the broader study area, whereas the second compares the habitat actually used within a home range (based on telemetry fixes) with the habitat available within the home range (e.g., Fig. 9). Compositional analysis applies Multivariate Analysis of Variance (MANOVA) to compare the proportions of available habitat to the proportions of used habitat (Aebischer et al. 1993).

Analysis of Spatial Data

We compared both static and dynamic overlap (Powell 2000) for radiocollared animals. Static overlap (e.g., Fig. 10) simply quantifies overlap of two home ranges, and was compared across sexes using a one-way analysis of variance (ANOVA) in JMP version 7 (SAS Institute Inc., Cary, NC, 1989-2007). Because two individuals could have high static overlap yet behaviorally avoid each other, dynamic overlap assesses the proximity of two animals from the cloud of known points (telemetry fixes) to assess if they occur further apart in simultaneous observations than expected assuming random distribution.

Static interactions were determined by generating an overlap matrix of HR in Ranges 6.1.2 (Kenward et al. 2003), from which we calculated a geometric mean of overlap for each dyad. Dynamic interactions involved comparing observed and possible distances between simultaneous locations for two animals with overlapping home ranges. The distribution of observed pairwise distances was compared against that of possible pairwise distances, and compared using a standardized Jacob's Index, which ranges from -1 (avoidance) to +1 (attraction; Kenward et al. 2003). We assessed whether the Jacob's index values were significantly different from zero using a Wilcoxon-sign-rank test (as per Walls and Kenward 2001).

Northern flying squirrels-- We confirmed normality of residuals using Shapiro-Wilks test ($p > 0.10$), and we tested for homoscedasticity using multiple tests; two (Levine test, $p = 0.025$; Brown-Forsythe test, $p = 0.036$) suggested heteroscedasticity, while two (O'Brien test, $p = 0.104$; Bartlett tests, $p = 0.156$) indicated homoscedasticity. Since the parametric tests applied here are relatively robust to mild heteroscedasticity, and these tests yielded conflicting results, we proceed with parametric analyses but interpret results conservatively.

Golden-mantled ground squirrels-- We quantified static overlap using Minta's index (Minta 1992) as well as dynamic interaction for adult females using fixed kernel home ranges. Only adjacent females were included in our analysis. Minta's index ranges from 0 to 1, and was calculated using both the 95% home range and the 50% core area. Dynamic interaction calculations followed the same methodology as those used with northern flying squirrels.

Vegetation

Long-term grids

Cone Counts.—To evaluate the effects of conifer seed production on small mammal abundance, we measured cone production during fall of 2003,2004, and 2006-2009 using 10 randomly selected individual trees of each species on each long-term grid; the same trees were measured each year and within the same 2-week period to avoid confounding temporal factors. For this we selected mature dominant or codominant trees with pointed crowns, as tall as or taller than the surrounding canopy, sufficiently far apart that their crowns did not touch. For grids with <10 individual trees of a given species, additional trees were found as close to the grid as possible (<500 m or 546 yd). Independent tallies of cones were conducted by two observers standing at a distance of ≥1.5x the tree height and visually counting cones using binoculars. For each tree we recorded tree height, diameter at breast height (DBH), species, and crown class.

In 2009, predictive *Peromyscus* population models were developed (objective #5). We hypothesized that yearly trends in Peromyscus abundance were strongly influenced either by resource availability (conifer cone seed), annual precipitation, or an interaction between these two factors. We applied multiple regression using SAS PROC REG with the maxr option (SAS Institute Inc., 2001) to build models. We used 7 potential predictor variables including 5 measures of cone abundance and 2 measures of precipitation. Cone abundance variables included: total annual conifer cone production of preceding year across all species and separately by (1) white-fir (*Abies concolor*; ABCO) plus red-fir (*Abies magnifica*; ABMA), (2) "yellow pines" (ponderosa pine, *Pinus ponderosa*; PIPO plus Jeffery pine, *Pinus jeffreyi*; PIJE), (3) sugar pine (*Pinus lambertiana*; PILA) plus western white pine (*Pinus monticola*; PIMO), and (4) Douglas fir (*Pseudotsuga menziesii*; PSME). Precipitation variables included total annual snowfall of the preceding winter (Nov -Mar) and total annual rain fall of the current year (Apr-Oct) (data taken from the Western Regional Climate Center 2009). The maxr option calculates the best 1, 2, 3,……,n variable models in a stepwise fashion based on parameter residuals (R^2). We selected among competing models using Akaike's information criterion (AIC), ΔAIC ($AIC_i - AIC_{min}$), and associated AIC weights (w_i) and we perform model averaging as necessary (Burnham and Anderson 2002). All variables were first rank transformed (RT; Iman and Conover 1979; Conover and Iman 1981) followed by an analysis of autocorrelation using Spearman's coefficient of rank correlation prior to conducting regression analyses.

Land-bird transects

We measured habitat characteristics through the implementation of 1 m (3 ft) fixed-radius plots (3.14 m^2 or 33.8 ft^2)centered about trap locations at all trap locations in 2006, 2007, and 2009 (n=2,397). As previously mentioned, each of these points falls within the landbird module's 50 meter (164 ft) vegetation plots and therefore our data completely overlap. In the future we hope to examine how landbird species richness compares to small mammal species richness at a site (i.e., are habitats associated with areas of high landbird species richness and abundance also areas of high small mammal species richness?). Our fixed-radius vegetation sampling methods are consistent with method we employed in the long-term grids (Coppeto et el. 2006 - Publications #1 and #4) with two exceptions. Due to logistical and sample size constraints canopy cover was measured with a moose-horn (Garrison 1949) rather than digital photography, and tree sampling was conducted with a PANAMA Basal Area Angle Gauge (Panama Pump Company, Hattiesburg, MS) rather than through the point-centered quarter method.

Northern flying squirrels

Habitat Selection— Habitat availability was obtained from high-resolution aerial mapping of the Plumas National Forest available in a GIS format. We quantified habitat selection at 2 spatial scales. We compared the proportion of selected habitat types in an animal's home range relative to that available in the study area (2nd order selection; Aebischer et al. 1993) defined by a 2 km buffer around 95% home ranges; in the absence of any biologically defensible distance, we used a 2 km buffer as this is the mean dispersal distance of the northern flying squirrel (W. P. Smith, personal communication, November 2008). Additionally, we compared the use of habitat within an animal's home range (based on telemetry "fixes") relative to the available habitat (3rd order selection; Aebischer et al. 1993). At both scales we used compositional analysis (Compos-Analysis 6.2 plus, Smith Ecology Ltd., Wales, UK) to quantify habitat selection.

After determining important habitat types in the 2nd order analysis, we combined 3 categories in the 3rd order analysis (see "Missing habitat types" in Aebischer et al. 1993). In addition, because the northern flying squirrel is a forest-dwelling animal that is known to be positively associated with old-growth (Waters and Zabel 1995), we combined chaparral, small Sierra mixed-conifer, and monocultures into a single habitat category.

Den Trees-- We located den trees by homing in on telemetry signals during the day, 1-2 times per week in 2006, and weekly in 2007. Den sites were marked with labeled flagging for later identification, and UTM coordinates were recorded using a GPS device accurate to ± 1m (GeoXH handheld and backpack antenna; Trimble Navigation, Limited; Sunnyvale, CA). We applied compositional analysis to compare characteristics of den trees with those of all trees available in an 18 m (59 ft) radius around the den tree. We characterized these trees initially as hardwood or conifer. Conifers were further characterized as saplings (0-10 cm or 0-4 in diameter breast height), poletimber (10-27.9 cm or 4-11 in dbh), small sawtimber (28-53.3 cm or 11-21 in dbh), or large sawtimber (>53.3 cm or 21 in dbh). Hardwoods were characterized as small or large (< 33 cm or 13 in dbh vs. ≥ 33 cm or 13 in dbh) based on increased mast production in the larger size class (Innes et al. 2007).

RESULTS AND DISCUSSION

We continue making steady progress towards our objectives, and in 2009 we completed several projects. In addition to successfully completing an extensive (1 May-1 October) field season, our study module has produced quality peer-reviewed publications and other products. In 2009, we completed 1 Masters of Science thesis on the habitat ecology of the northern flying squirrel (Smith 2009), submitted 1 manuscript, and we have 3 manuscripts in preparation. In the following we present the abstracts of our published, submitted, or in preparation manuscripts where appropriate, as a representation of the work that we have completed to date.

2009 Field Season

Long-term grids

During the 2009 field season we marked a total of 823 individuals over 1,672 captures of 10 species. Predominant species in the study area were deer mice, brush mice, long-eared and Allen's chipmunks, California ground squirrels (*Spermophilus beecheyi*), golden-mantled ground squirrels, Douglas squirrels (*Tamiasciurus douglasii*), long-tailed voles (*Microtus longicaudus*), California red-backed vole (*Clethrionomys californicus*[3]), and northern flying squirrels. Incidental mammals captured included shrews (*Sorex* sp., primarily *S. trowbridgii*), striped skunk (*Mephitis mephitis*), western spotted skunk (*Spilogale gracilis),* snowshoe hare (*Lepus americanus*), and western gray squirrels (*Sciurus griseus*).

In 2009, abundance of *Peromyscus* sp. and *Tamias* sp. increased from those documented in 2008 (Fig. 11). We noticed a marked increase in capture rate of northern flying squirrels at long-term grids from 2005-2009 (Fig. 11). Correspondingly during this time overall woodrat abundance at long-term grids across habitat and treatment types has declined steadily (Figs. 11, 12, 13). We did not capture any dusky-footed woodrats in long-term grids in 2009, although a single bushy-tailed woodrat (*Neotoma cinerea*) was trapped in a red-fir grid. The 2009 field season represented our second full year, and third fall session, of post-treatment data collection. Overall, trends in abundance of

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[3] *Clethrionomys* currently is treated as *Myodes*; we use the former here as it is more widely recognized.

Peromyscus, *Tamias*, northern flying squirrels, and dusky-footed woodrats within treatment plots have mimicked trends observed within control plots (Figs. 14, 15, 16, and 11 respectively). Relative to other habitat types, Douglas fir and red-fir habitats continue to support greater abundances of *Peromyscus* and *Tamias* (Fig. 17 and Fig. 18 respectively). In 2009 northern flying squirrel abundance was also greatest in Douglas fir and red-fir habitats (Fig. 19). *Peromyscus* numbers continue to vary strongly across years but are very well explained by only three variables (Fig. 20, also see cone count results below). Across all years sampled, biomass of PNF small mammal communities was greatest within red fir habitat (Fig. 21a). Large abundances of relatively larger small mammal species, such as golden mantled ground squirrels and chipmunks, resulted in this habitat supporting 47-61% greater total biomass than other forest types sampled (Table 2). Total biomass of key California spotted owl prey species was also greatest in red fir habitat (Fig. 21b, Table 3).

Cone counts and Peromyscus— Our data indicate that the cone crop in 2009 was the largest since the inception of the PLAS (Fig. 20). All conifer species with the exception of PIPO and PIJE experienced record cone abundance (Fig. 22).

The best predictive model of *Peromyscus* numbers included 3 parameters - in declining order of importance, these were annual total snow fall of the preceding winter, annual abundance of fir (*Abies*) cones in the preceding year, and annual abundance of sugar pine and western white pine cones the preceding year (Table 4). This model accounts for 45% of the yearly variation in observed *Peromyscus* abundance in PNF and performs very well when compared to actual observed *Peromyscus* abundance values (Fig. 23). A simple (2 parameter) model (total annual snow fall and *Abies* cone abundance) accounted for 41% of the yearly variation in observed *Peromyscus* abundance. Surprisingly, a model with a single parameter (total annual snow fall) accounted for 31% of the yearly variation in observed *Peromyscus* abundance. Total annual rainfall, PSME cone abundance, and PIPO/PIJE cone abundance contributed minimally to the model. Relative to the 3 parameter model these additional parameters did not substantially increase the fit of the model.

Land-bird transects

In 2009 we sampled 30 transects and 232 census stations across 4 treatment units, bringing our sample to 74 transects and 599 census stations. We recorded 2,016 captures of 1,367 individuals of 11 species. In total, 3 years of land-bird transect sampling has produced the identification of 3937 individuals over 6166 captures of 13 species. Predominant species captured were deer and brush mice (n=1,489 (38%) and 332 (8%), respectively), chipmunks (n=1,181 (30%) Allen's and 536 (14%) long-eared), California and golden-mantled ground squirrels (87 (2%) and 135 (3%), respectively), and northern flying squirrels (76 = 2%). Douglas squirrels, long-tailed voles, California red-backed voles, montane voles (*Microtus montanus*), and western jumping mouse (*Zapus princeps*) each comprised of <2% of the total capture. Incidental mammals captured included shrews (*Sorex* sp.), striped skunk (*Mephitis mephitis*), western spotted skunk (*Spilogale gracilis)*, snowshoe hare (*Lepus americanus*), western gray squirrels (*Sciurus griseus*), ring-tailed cat (*Bassariscus astutus*), and grey fox (*Urocyon cinereoargenteus*).

Mean richness and abundance indices per point across treatment units 2-5 is 2.02 ± 0.05 SE and 6.58 ± 0.24 SE, respectively. Treatment unit 2 ranked highest in both abundance and richness (Figs. 2, 3). Treatment unit 2 yielded 32% greater abundance than treatment units 3 and 4, and 26% greater than treatment unit 5. Treatment unit 2 displayed 25%, 35%, and 17% greater richness than treatment units 3, 4, and 5, respectively.

Long-term grids

One of our objectives for the long-term grid data is to characterize small mammal habitat associations at macro- and microhabitat scales (Objective #1). We have examined this at our long-term grids and include summaries herein (Publications #1 and #4). Another objective for our long-term grid data was to determine small mammal population trends, evaluate how populations are changing temporally, and assess the factors responsible for the observed trends (Objective #5). We have documented the dynamics of small mammal abundance at long-term grids since 2003, and we have evaluated trends using data from 2003-2004, and include this summary herein (Publication #8). In 2007, the planned treatments were implemented and data on small mammals were collected immediately after the treatments were completed. We will analyze both population and habitat data obtained at long-term grids pre-treatment (2003-2006) and post-treatment (2007-??) to assess the impacts of forests management treatments on small mammal abundance and species diversity (Objectives #2, 3, and 4).

Publications #1 and #4: Habitat associations of small mammals at two spatial scales in the northern Sierra Nevada

Effective management strategies require an understanding of the spatial scale at which fauna use their habitat. Towards this end, small mammals were sampled in the northern Sierra Nevada, California, over 2 years (2003-2004) at 18 live-trapping grids among 5 forest types. Macrohabitats were defined by overstory tree composition, and 19 microhabitat variables were measured at all trap stations. Macrohabitat and year explained 93% of variation in abundance of deer mice (*Peromyscus maniculatus*), whereas 69% was explained by microhabitat and year. Variation in abundance of *Tamias* sp. (long-eared and Allen's chipmunk) was slightly better explained by microhabitat and year (70%) than by macrohabitat and year (67%). Red fir forests supported significantly more mice and chipmunks than mixed conifer and pine-cedar forests, and more chipmunks than mixed fir forests. Five of 6 uncommon species were significantly associated with macrohabitat type; golden-mantled ground squirrels, northern flying squirrels, and *Microtus* sp. (long-tailed vole–*M. longicaudus*; Mountain vole–*M. montanus*) were captured almost exclusively in red fir forests, whereas dusky-footed woodrats and California ground squirrels were found in pine-cedar, mixed fir, and mixed-conifer forests. The first 2 axes of a canonical correspondence analysis on microhabitat variables explained 71% of variation in combined small mammal abundance. Microhabitat associations varied among species but were driven primarily by canopy openness, shrub cover, and shrub richness. Although much of the small mammal fauna appeared to select habitat at both spatial scales studied, CCA using macrohabitat as a covariate revealed that microhabitat explained much less of the variation in small mammal abundance than did macrohabitat. Still, the strongest scale of association may be species-dependent and hierarchical in nature.

Publication #8: Population dynamics of small mammals in relation to cone production in four forest types in the northern Sierra Nevada

We studied the small mammal assemblage in 4 forest types (white fir, red fir, Douglas fir, and ponderosa pine) in the Sierra Nevada of California for 2 consecutive field seasons (2003-2004). We also assessed cone production by dominant conifer species in both years. Cone production was greater overall in fall 2003, but varied within forest type and between conifer species. Parallel to this, mean maximum densities of deer mice increased in 2004 (from 0.7 - 7.3 ind./ha to 65.7 - 112.7 ind./ha). Numbers of golden-mantled ground squirrels were similar in both years, and displayed the typical pattern of a hibernating species, with low densities in May (6.6 ± 0.2), peak densities in September (24.5 – 32.5 ind./ha), and declines in October (9.2 ± 4.8). Long-eared chipmunks reached higher densities in red fir (48.2 ± 13.4 ind./ha) and Douglas-fir forests (36.0 ± 13.5 ind./ha) than in white fir forests (7.6 ± 2.7 ind./ha), and all populations peaked in September. Allen's chipmunk remained at lower densities than long-eared chipmunks except during September 2004, when populations of the former reached high densities (54.6 ± 26.8 ind./ha). Survival of deer mice was dependant on an interaction between forest type and month with additive effects of winter and 2003 fall mean cone production. Golden-mantled ground squirrel survival varied by month whereas survival in both species of chipmunk varied by an interaction of forest type and month + winter. Dusky-footed woodrats were present at lower elevations and reached greatest densities in ponderosa pine forests. Northern flying squirrels were uncommonly captured and found predominantly in red fir forests.

Northern flying squirrels

We have captured and radiotracked northern flying squirrels since 2004 in an effort to evaluate the abundance and distribution, habitat use, and home range of this important species (Objective #6). We have examined data from 2004-2005 and provide a summary below (Publication #7). We continued these efforts during 2006 and 2007 to increase our sample size and improve our statistical power; 2007 marked the final year of northern flying squirrel radiotracking. Data from 2006 and 2007 will be included in additional publications (Publications #3, #12, and #13).

Publication #7: Home range and activity of northern flying squirrels in the northern Sierra Nevada

We studied the northern flying squirrel in PNF using radiotelemetry. Fourteen northern flying squirrels from 2 forest types (mixed conifer and red fir) were fitted with radiocollars and provided sufficient locations for home range analysis. We used 95% adaptive kernel and 95% minimum convex polygon (MCP) analysis to determine home ranges. No sex differences and no differences in forest type were observed for home range size. Mean kernel home range size was 25.7 ha for all squirrels. Mean distance to the nearest nest tree did not vary throughout the night; however, females tended to travel greater distances from nest trees.

Publication #3 and #12: Home range and habitat selection of northern flying squirrels in the northern Sierra Nevada

Northern flying squirrels (*Glaucomys sabrinus*) are a key prey species for threatened spotted owls (*Strix occidentalis*), and thought to be keystone dispersers of mycorrhizal fungal spores in the forests of the Pacific Northwest. Understanding their habitat and spatial requirements is essential to enlightened management of these forests. We quantified home range size for 22 northern flying squirrels over 2 years in the northern Sierra Nevada using radiotelemetry. We applied compositional analysis to assess habitat preferences at 2 spatial scales, and to quantify den tree selection. Mean home range sizes were 8.56 ± 1.36 (SE) ha (Table 5), and did not differ between sexes or across years. Habitat selection was scale-independent; home ranges included disproportionate cover by large mixed-conifer stands, and habitat use within home ranges also favored use of areas with large mixed-conifer stands (Table 6, Fig. 24). Squirrels selected large conifers and hardwoods for den sites preferentially over smaller conifers and saplings (Fig. 25). These results suggest that retention of viable populations of this key prey species would be facilitated if forest managers could retain stands of large mixed conifers as well as hardwoods.

Publication #13: Spatial organization and mating behavior of northern flying squirrels (*Glaucomys sabrinus*)

Spatial organization of female mammals may be influenced either by high energetic needs when raising young or by the need to defend altricial young; either factor can lead to territoriality. The resource defense hypothesis postulates that home ranges should be structured to provide access to key resources, including food and, for males, access to females. The offspring defense hypothesis postulates that females will structure their home ranges to provide for protection of offspring when they are young and vulnerable. Northern flying squirrels (Glaucomys sabrinus) are highly social and aggregate in communal nests, but females have been reported to segregate from conspecifics when rearing young, suggesting that they may fit the offspring defense hypothesis. We used extensive home range data to assess both spatial hypotheses in northern flying squirrels. Home range sizes across sexes were similar, and female-female overlap was much greater than male-female overlap, suggesting that males do not fit the resource defense hypothesis. Although small sample sizes limit our conclusions, females appeared to defend territories when with young, matching predictions of the offspring defense hypothesis.

Dusky-footed woodrats

We have captured and radiotracked dusky-footed woodrats in an effort to evaluate the abundance and distribution, habitat use, and home range of this important species (Objective #6). To date, we have examined vegetation data obtained during 2004-2005 and include this summary herein (Publications #2, #5, and #6). In 2008 we published a manuscript on the spatial organization of dusky-footed woodrats (Publication #9). The 2006 field season marked the final year of data collection, so that we might focus our efforts on northern flying squirrel ecology during 2007 and analyze data obtained on woodrats from previous years.

Publication #2 and #5: Habitat selection by dusky-footed woodrats in managed mixed-conifer forest of the northern Sierra Nevada

Dusky-footed woodrats are important components of forest communities, including serving as a primary prey of the California spotted owl, a species of concern in California. We examined the macro- and microhabitat associations of the dusky-footed woodrat at 4 study sites within mixed-conifer forest of the northern Sierra Nevada, California, during 2003–2005. We investigated the importance of California black oak as a macrohabitat component for woodrats, and we examined microhabitat selection at 2 levels, house location and house use, by comparing house-site (n = 144) characteristics to random sites (n = 144) and characteristics of used and unused houses, respectively. We found a strong trend towards a positive relationship between woodrat density and large (≥33 cm diameter at breast height) oak density, suggesting that large oaks are an important macrohabitat component for woodrats, probably because of their value as a food resource. At the microhabitat scale, house location was strongly influenced by the presence of large (≥30 cm diameter at root collar) stumps, but also by abundance of logs, steeper slopes, and lack of bare ground and mat-forming shrub cover. Houses used by adults were not distinguishable from unused houses on the basis of microhabitat variables, suggesting that woodrats make decisions about microhabitat conditions at the time a house is built. Adult and subadult woodrats selected houses with different microhabitat characteristics, but this pattern was not consistent between years. In 2005, adults chose larger houses that were characterized by more logs and less poletimber, but we detected no such differences in 2004. Dusky-footed woodrats in the northern Sierra Nevada would benefit from management techniques that promote the growth and retention of large California black oaks and create abundant dead wood within a stand.

Publication #6: Characteristics and use of tree houses by dusky-footed woodrats in managed mixed-conifer forest of the northern Sierra Nevada

We examined habitat associations of the dusky-footed woodrat at 4 study sites in mixed-conifer forest of the northern Sierra Nevada, California, during 2003–2005. We investigated the importance of California black oak (*Quercus kelloggii*) as a macrohabitat component for woodrats, and we examined microhabitat association at 2 levels, house location and house use, by comparing house-site (n = 144) characteristics to random sites (n = 144) and characteristics of used and unused houses, respectively. Woodrat density was positively related to density of large (≥33-cm-diameter at breast height) oaks, probably because large oaks are valuable food resources. Location of woodrat houses was strongly influenced by the presence of large (≥30-cm-mean diameter) logs and large (≥30-cm-diameter at root collar) stumps, but also by steeper slopes, and lack of bare ground and mat-forming shrub cover. We found little evidence that adult woodrats distinguished among houses with respect to microhabitat, suggesting that woodrats make decisions about microhabitat suitability when the house is built. In 2005, adults chose larger houses that were characterized by more logs and less poletimber than those of subadults, but we detected no such differences in 2004. Dusky-footed woodrats in the northern Sierra Nevada would benefit from management techniques that promote the growth and retention of large California black oaks and create large logs and stumps within a stand.

Publication #9: Spatial organization of dusky-footed woodrats in managed mixed-conifer forest of the northern Sierra Nevada

We studied the spatial organization of dusky-footed woodrats (Neotoma fuscipes) in mixed-conifer forest of the northern Sierra Nevada, California by radiotracking 63 adult woodrats at 2 study sites during May–Oct, 2004–2006. Home range and core area estimates differed between study sites, but they were within the range reported elsewhere; variability in home range size was explained in part by density. Woodrat home ranges overlapped with multiple neighboring woodrats, both same-sex and opposite-sex, suggesting that foraging areas were shared. However, core areas showed little overlap between same-sex neighbors. Woodrats occupied multiple houses and frequently moved among them, and sharing of houses (either simultaneously or nonsimultaneously) with neighboring woodrats was common but occurred mostly between male-female pairs. Females typically shared their core area and houses with 1 male, whereas males shared core areas and houses with multiple females; further, males moved more than females. Our results suggest that dusky-footed woodrats are semi-territorial, maintaining near-exclusive use of their core area and houses against same-sex conspecifics, and that the mating system likely is polygynous.

Golden-mantled ground squirrels

We captured and radiotracked golden-mantled ground squirrels during 2003-2005 in an effort to evaluate the home range and social organization of this species (Objective #6). Data analysis and manuscript preparation took place in 2009; no additional data has been collected since 2005. The following summary (Publication #10) represents the culmination of this work.

Publication #10: Spatial-organization of golden-mantled ground squirrels

Social organization and dispersal patterns indicate that golden-mantled ground squirrels exhibit asocial behavior as suggested by Armitage (1981) and Michener (1983). Although a great deal of home range overlap (FK- mean IO 0.31 ± 0.20 SD, range 0.00-0.69) is observed, core areas were defended heavily (FK- mean IO 0.03 ± 0.08 SD, range 0.00-0.43) (Fig. 10) indicating territoriality. Many (75%) juveniles dispersed in their first year. Dynamic interactions revealed Jacob's index scores that did not differ from zero (Wilcoxon sign-rank test, $P>0.05$) for either the 95% FK home range ($\bar{x} = 0.003 \pm 0.052$ SD) or the 50% FK core area ($\bar{x} = -0.001 \pm 0.067$ SD), indicating that dynamic interactions were neutral. Most juveniles (14 of 20), including both dispersers and those that remained philopatric, made exploratory excursions averaging 1.9 ± 0.6 SE per juvenile (range = 0-9) and extending as far as 1142 m (0.7 mi). Juveniles often made exploratory excursions to locations they later dispersed to. Mean home ranges based on 95% MCP ($\bar{x} = 3.57$ ha or 8.8 acres, range = 1.1-9.3) were similar to those based on 95% FK ($\bar{x} = 3.75$ ha or 9 acres, range = 1.1-13.5). Core areas based on 50% FK averaged 1.03 ha or 2.5 acres (range = 0.25-4.24). To the best of our knowledge, this study represents the first account of spatial organization, dispersal behavior, and home range size of the golden-mantled ground squirrel.

Chipmunks

We have live-trapped chipmunks at long-term grids, land bird grids, and flying squirrel transects since 2003. In the face of evidence that Sierra mammals are being impacted by global climate change (e.g., Moritz et al. 2008), one of our objectives was to evaluate the habitat affinities of 2 species found commonly in PNF and using similar habitats, using data obtained from long-term grids during 2003-2004 (Objective #7). The following (Publication In prep #11) is a summary of these results.

Publication #11: A multiple spatial scale perspective of the habitat affinities of sympatric long-eared and Allen's chipmunks.

Sympatric species that are similar in body mass, diet, and general resource utilization are likely to compete locally. Similar species often coexist by partitioning habitat. However, detecting differences in habitat affinities is influenced by spatial scale. We investigated the habitat associations of two ecologically similar chipmunk species – the long-eared chipmunk and the Allen's chipmunks – at three spatial scales in the northern Sierra Nevada, California. Locally, we censused these species over two years at 18 trapping grids, and recorded 19 microhabitat metrics at all trap stations. At a macrohabitat scale, we assessed relative abundances at different study sites as a function of forest type. Finally, at a landscape (e.g., geographic range) scale we examined digital vegetation information and calculated extent of range overlap. At this largest spatial scale, both species showed similar habitat affinities, with extensive overlap in distribution within the Sierra Nevada. At the macrohabitat scale, both the species reached their highest mean abundance in red fir forests but showed divergent secondary affinities. At the microhabitat scale, however, habitat affinities differed significantly. Logistic regression models indicate that microhabitat presence of long-eared chipmunks was associated positively with open canopies, cover by rocks, and multiple sapling species, and negatively with east and south facing, steep slopes. Allen's chipmunks shared the affinity for open canopies but differed in exhibiting a preference for traps on south facing slopes with multiple shrub species, and aversion to traps on hard substrates covered by litter and vegetation mats (e.g., Mahala mat—*Ceanothus prostratus*). Affinities at micro- and macrohabitat scales varied between sampling years, indicating that these species retain a degree of flexibility in habitat associations while maintaining segregation and minimizing the potential for competition.

2009 Field Season

We continue to collect tissue samples from chipmunks (small sections (< 1 mm or 0.04 in) of ear pinna, frozen in 95% ethanol) while performing live-trapping duties at long-term grids and land bird transects. We collaborate with Dr. Jack Sullivan (University of Idaho) to develop molecular markers for non-lethal identification of chipmunk species by external morphological characteristics. For each chipmunk captured we record the presence of six external morphological characteristics that have been suggested to visually distinguish between chipmunk species. These are: ear patch size and color; face stripe color and curvature; length and shape of the ear; and body color. We will use these data to determine characteristics that reliably distinguish these species in the field, thereby allowing us to proceed with analyses of habitat use.

COLLABORATION

We have continued to maintain and improve collaborative efforts with other PLAS Modules. Vegetation and Fuels Modules have collected and continue to collect vegetation, fire and fuels, and microclimate data within some portion of our long-term and land-bird trapping grids. In 2009, we continued to collaborate closely with the directors of the University of California Davis McLaughlin Reserve, Cathy Koehler and Paul Aigner, who provided space to train our field crew prior to our housing becoming available at the University of California, Berkeley Forestry Camp. In exchange for housing and training facilities, we provided information on the abundance and distribution of small mammal species within a long-term study grid established on the reserve. We collaborate with the University of Idaho for molecular analyses to determine chipmunk species identification and worked together with them to secure outside funding for these analyses. Lastly, we work closely with the University of California Davis Museum of Wildlife and Fish Biology to preserve specimens for research and educational purposes.

PUBLICATIONS

Theses

1. Coppeto, S. A. 2005. Habitat associations of small mammals at two spatial scales in the northern Sierra Nevada, California. M.S. Thesis, University of California, Davis, 39 pp.

2. Innes, R. J. 2006. Habitat selection by dusky-footed woodrats in managed, mixed-conifer forest of the northern Sierra Nevada. M.S. Thesis, University of California, Davis, 31 pp.

3. Smith, J. R. 2009. Home range and habitat selection of the northern flying squirrel (*Glaucomys sabrinus*) in northeastern California. M.S. Thesis, University of California, Davis, 29 pp.

Peer-reviewed

1. Coppeto, S. A., D. A. Kelt, D. H. Van Vuren, J. A. Wilson, S. Bigelow, and M. L. Johnson. 2006. Habitat associations of small mammals at two spatial scales in the northern Sierra Nevada. Journal of Mammalogy 87:402-416.

2. Innes, R. J., D. H. Van Vuren, D. A. Kelt, M. L. Johnson, J. A. Wilson, P. A. Stine. 2007. Habitat selection by dusky-footed woodrats in managed, mixed-conifer forest of the northern Sierra Nevada. Journal of Mammalogy 88: 1523-1531.

3. Innes, R. J., D. H. Van Vuren, D. A. Kelt. 2008. Characteristics and use of tree houses by dusky-footed woodrats in the northern Sierra Nevada. Northwestern Naturalist 89:109-112.

4. Wilson, J. A., D. A. Kelt, and D. H. Van Vuren. 2008. Home range and activity of northern flying squirrels (*Glaucomys sabrinus*) in the Sierra Nevada. Southwestern Naturalist 53:21-28.

5. Wilson, J. A., D. A. Kelt, D, H, Van Vuren, and M. Johnson. 2008. Population dynamics of small mammals in relation to cone production in four forest types in the northern Sierra Nevada, California. The Southwestern Naturalist 53: 346-356.
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Submitted

1. Jesmer, B. J., D. A. Kelt, and D. H. Van Vuren. Spatial-organization of golden-mantled ground squirrels *(Spermophilus lateralis*). American Midland Naturalist.

2. Smith, J. R., D. A. Kelt, D. H. Van Vuren. Home range and habitat selection of the northern flying squirrel (*Glaucomys sabrinus*) in northeastern California. Journal of Mammalogy.

3. Smith, J. R., D. A. Kelt, D. H. Van Vuren. Spatial organization and mating behavior of northern flying squirrels (*Glaucomys sabrinus*). Southwestern Naturalist.

In Preparation

1. Coppeto, S. A., D. A. Kelt, D. H. Van Vuren, J. Sullivan, J. A. Wilson, and N. Reid. Different scales tell different tales: niche conservatism vs. niche differentiation in chipmunks in the northern Sierra Nevada. To be determined. Spring 2010.

PRESENTATIONS

1. Coppeto, S. A., D. A. Kelt, J. A. Wilson, D. H. Van Vuren, and M. L. Johnson. 2004. Habitat selection by small mammals in the northern Sierra Nevada, California. Poster to the American Society of Mammalogists, Annual Meeting, Arcata, CA.

2. Coppeto, S. A., D. A. Kelt, D. H. Van Vuren, J. A. Wilson, S. Bigelow, and M. L. Johnson. 2005. Spatial scale and habitat use of small mammals in the northern Sierra Nevada, California. Poster to the American Society of Mammalogists, Annual Meeting, Springfield, MO.

3. Innes, R. J., D. H. Van Vuren, J. A. Wilson, D. A. Kelt, and M. B. Johnson. 2004. Factors affecting the distribution and use of dusky-footed woodrat (*Neotoma fuscipes*) houses. Poster to the American Society of Mammalogists, Annual Meeting, Arcata, CA.

4. Innes, R. J., D. H. Van Vuren, J. A. Wilson, D. A. Kelt, and M. B. Johnson. 2005. Space use and social organization of dusky-footed woodrats (*Neotoma fuscipes*) in mixed-conifer forests of the northern Sierra Nevada. Poster to the American Society of Mammalogists, Annual Meeting, Springfield, MO.

5. Innes, R. J., D. H. Van Vuren, D. A. Kelt, M. B. Johnson, J.A. Wilson. 2006. Habitat relations of dusky-footed woodrats (*Neotoma fuscipes*) in mixed-conifer forests of the northern Sierra Nevada. Poster to the American Society of Mammalogists, Annual Meeting, Amherst, MA.

6. Smith, J.R.. 2006. Ecology of *Glaucomys sabrinus*: habitat, demography, and community relations. Presentation to the American Society of Mammalogists, Annual Meeting, Springfield, MO.

7. Wilson, J.A., and K.E. Mabry. 2005. Trap disinfection to reduce Hantavirus risk: does it also reduce small mammal trapability? Presentation to the American Society of Mammalogists, Annual Meeting, Springfield, MO.

8. Wilson, J. A., D. A. Kelt, and D. H. VanVuren. 2005. Effects of maternal body condition on offspring dispersal in golden-mantled ground squirrels (*Spermophilus lateralis*). Presentation to the American Society of Mammalogists, Annual Meeting, Springfield, MO.

9. Wilson, J. A., D. A. Kelt, and D. H. VanVuren. 2005. Effects of maternal body condition on offspring dispersal in golden-mantled ground squirrels (*Spermophilus lateralis*). Presentation to the IX International Mammalogical Conference, Sapporo, Japan.

10. Wilson, J. A., D. A. Kelt, and D. H. Van Vuren. 2006. Home range and activity of the northern flying squirrel (*Glaucomys sabrinus*) in the northern Sierra Nevada. Poster to the American Society of Mammalogists, Annual Meeting, Amherst, MA.

11. Smith, J.R., D.A. Kelt, D.H. Van Vuren, and M.L. Johnson 2009. Spatial organization of *Glaucomys sabrinus*: territory and habitat use. Presentation to the American Society of Mammalogists, Annual Meeting, Fairbanks, AK.

PERSONNEL

This project is currently coordinated and supervised by Brett Jesmer. Field work in 2009 was conducted by Hillary Squires, Travis Lewis, Chelsea Beebe, Jon Draper, Kate Ingram, and Brett Jesmer. This study was carried out under the guidance of Dr. Douglas Kelt, Dr. Dirk Van Vuren, and Dr. Michael Johnson, University of California Davis.

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Table and Figure Legends

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Figure 15. Mean abundance of chipmunks (*Tamias* sp.) pre and post treatment within the Plumas National Forest, California, during 2003-2009.

Figure16. Mean abundance of northern flying squirrels (*Glaucomys sabrinus*) pre and post treatment within the Plumas National Forest, California, during 2005-2009.

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Figure 20. General relationships between mean mouse (*Peromyscus* sp.) mean abundance, mean annual cone production, and mean annual snow fall from 2003-2009. Note: lag in response to mean conifer cone abundance and immediate response to annual snow fall.

Figure 21. Mean annual biomass of a) small mammal species and b) spotted owl principle prey species from 2003-2009; summarized across 4 forest types (Douglas-fir (DF), Pine/Cedar (PO), Red-fir (RF), and White-fir (WF)).

Figure 22. General relationship between mean mouse (*Peromyscus* sp.) abundance and mean annual cone production by species from 2003-2009.

Figure 23. Multiple regression model #3 predicted values versus actual observed values of *Peromyscus* populations during 2004, 2005, and 2007 – 2009 across all long-term grids (n=21).

Figure 24. Percentage of used and available habitat types for northern flying squirrels (*Glaucomys sabrinus*) at 2 spatial scales: a) home range relative to that available in a 2 km buffer around the home range (e.g., Aebischer's (1993) 2nd order), and b) use of habitats within the home range relative to that available (e.g., Aebischer's (1993) 3rd order). BAR=barren, MCP=montane chaparral, MON=monoculture, MRI=montane riparian, SC0=small Sierra mixed conifer, SC1= medium Sierra mixed conifer, SC2=large Sierra mixed conifer, WAT=water, WTM=wet meadow.

Figure 25. Distribution of available (*n*=11,602) and used (*n*=91) den trees of northern flying squirrel (*Glaucomys sabrinus*) in the northern Sierra Nevada, classified by tree type (hardwood vs. softwood) and size (in cm dbh; Sapling=0-10, poletimber=conifers 10-27.9, small sawtimber=conifers 28-53.3, large sawtimber=conifer≥53.4, small hardwood=10-33, large hardwood>33). Available trees are those trees within a 0.1 ha or 0.2 acre circle centered on den tree. Used trees are den trees.

Table 1. Characteristics of 21 long-term trapping grids in the Plumas National Forest.

GRID #	SITE NAME	LOCATION	FOREST TYPE AS PRESENTED IN COPPETO ET AL.	FOREST TYPE AS PRESENTED IN WILSON ET AL.	EXPERIMENT TYPE*
1	BLACKOAK	SNAKE LAKE	MIXED-CONIFER	WHITE FIR	LIGHT THIN
2	BOA	SCHNEIDER CREEK	MIXED-FIR	MIXED-FIR	HABITAT
3	CABIN	MILLER FORK	WHITE FIR	WHITE FIR	CONTROL
4	DOGWOOD	LITTLE SCHNEIDER CREEK	MIXED-FIR	MIXED-FIR	HABITAT
5	GIMP	DEAN'S VALLEY	MIXED-FIR	WHITE FIR	HEAVY THIN
6	GREENBOTTOM	GRIZZLY MOUNTAIN	RED FIR	RED FIR	HABITAT
7	GULCH	BEAN CREEK	PINE-CEDAR	PINE-CEDAR	HABITAT
8	MONO	GRIZZLY MOUNTAIN	RED FIR	RED FIR	HABITAT
9	NOGO	SNAKE LAKE	MIXED-CONIFER	WHITE FIR	CONTROL
10	NONAME	DEAN'S VALLEY	WHITE FIR	WHITE FIR	LIGHT THIN
11	OASIS	BEAN CREEK	PINE-CEDAR	PINE-CEDAR	HABITAT
12	RALPH	GRIZZLY MOUNTAIN	RED FIR	RED FIR	HABITAT
13	RIPPER	MILLER FORK	WHITE FIR	WHITE FIR	HEAVY THIN
14	RUTT	SNAKE LAKE	MIXED-CONIFER	WHITE FIR	GROUP SELECT
15	STEEP	TAYLOR CREEK	MIXED-FIR	MIXED-FIR	HABITAT
16	SWARM	BEAN CREEK	PINE-CEDAR	PINE-CEDAR	HABITAT
17	TEEPEE	DEAN'S VALLEY	WHITE FIR	WHITE FIR	CONTROL
18	THICK	DEAN'S VALLEY	WHITE FIR	WHITE FIR	GROUP SELECT
19	TOAST	MILLER FORK	WHITE FIR	WHITE FIR	GROUP SELECT
20	TRIANGLE	MILLER FORK	WHITE FIR	WHITE FIR	LIGHT THIN
21	VIEW	SNAKE LAKE	MIXED-CONIFER	WHITE FIR	HEAVY THIN

* treatments include no thinning (Control), light thinning (to xx% canopy cover), and heavy thinning (to xx% canopy cover); "Habitat" indicates grids established to document habitat associations across major habitat types in the PNF.

Table 2. Mean annual biomass (g) of small mammal species from 2003-2009; summarized across 4 forest types. PESP=*Peromyscus* sp., TASP=*Tamias* sp.; SPLA=*Spermophilus lateralis*; GLSA=*Glaucomys sabrinus*; NEFU=*Neotoma fuscipes*; MISP=*Microtus* sp.; CLCA=*Clethrionomys californicus*.

Habitat	PESP	TASP	SPLA	GLSA	NEFU	TADO	MISP	CLCA
Douglas fir	205.8	970.2	215.7	103.8	33.2	100.3	5.5	4.0
Pine-Cedar	96.0	3.9	159.4	60.1	99.7	30.1	0.0	0.0
Red fir	208.3	2439.0	2766.3	153.0	8.3	0.0	4.1	1.3
White fir	117.9	474.1	25.2	88.3	20.1	35.1	1.5	3.3

Table 3. Mean annual biomass (g) of spotted owl principle prey species from 2003-2009; summarized across 4 forest types.

Habitat	PESP	GLSA	NEFU
Douglas fir	205.848	103.826	33.226
Pine-Cedar	96.008	60.110	99.679
Red fir	208.289	153.006	8.306
White fir	117.851	88.273	20.127

Table 4. Results of models to assess factors influencing number of deer mice (*Peromyscus maniculatus*) across study sites. R^2 (correlation coefficient), Akaike information criterion (AIC), ΔAIC (AIC_i – AIC_{min}), and W_i (AIC weight) values used in model selection and certainty; along with number of conifer cone and precipitation parameters used model building.

					Model Parameters					
Model	**R^2**	**AIC**	**ΔAIC**	***w_i***	**1**	**2**	**3**	**4**	**5**	**6**
3	0.4505	540.15	0	0.414	ats*	abies*	pila*			
4	0.4594	540.58	0.524	0.319	ats*	abies*	pila*	pseu		
5	0.4628	542.11	1.960	0.155	ats*	abies*	pila*	pseu	tsp	
6	0.4642	543.88	3.727	0.064	ats*	abies*	pila*	pseu	tsp	pipo
2	0.4105	544.48	4.321	0.048	ats*	abies*				
1	0.3126	556.29	16.139	0	ats*					

ats=annual (Nov.-Mar.) total snowfall in preceding winter; tsp=active season (Apr.-Oct.) precipitation of current year; tca=total mean cone count in preceding year (across all species and grids); pila=total mean cone abundance for sugar pine (*Pinus lambertiana*) and western white pine (*Pinus monticola*) in preceding year (all grids); abies= total mean cone abundance for white and red fir **(***Abies concolor* and *Abies magnifica*) in preceding year (all grids); pipo= total mean cone abundance for yellow pine (*Pinus ponderosa* and *P. jeffreyi*) in preceding year (all grids); pseu= total mean cone abundance for Douglas fir (*Pseudotsuga menziesii*) in preceding year (all grids). * = $P<0.5$.

Table 5. Home range areas (ha) generated by minimum convex polygon (MCP) and fixed kernel (FK) home range estimation methods for all northern flying squirrels (*Glaucomys sabrinus*) from 2006 and 2007.

ID	# locations	95% MCP (ha)	95% Fixed Kernel (ha)	year
2	36	13.97	11.2	2006
27	37	20.05	24.9	2006
26	41	3.2	4.47	2006
8	50	3.55	1.82	2006
6	51	10.92	9.51	2006
1	53	20.44	22.43	2006
7	60	1.9	1.99	2006
27	34	2.61	4.3	2007
20	50	12.95	11.11	2007
19	53	12.74	12.3	2007
7	56	2.25	1.52	2007
22	56	7.81	5.69	2007
16	57	13.27	11.05	2007
17	57	11.55	8.45	2007
18	59	5.32	7.37	2007
6	66	9.48	9.64	2007
12	69	4.83	3.84	2007
24	50	7.1	4.33	2007
21	56	14.57	9.83	2007
23	61	2.82	2.05	2007
15	62	5.36	3.5	2007
25	62	14.9	17.05	2007

Table 6. Ranking of habitat types at 3 scales of analysis using compositional analysis. The 2nd order reflects disproportionate cover by "preferred" habitats within home ranges relative to that available in a 2 km buffer around each home range. The 3rd order reflects differential use of habitat types within home ranges (based on telemetry locations) relative to their distribution within the home range. The den tree analysis compares trees used for diurnal nests relative to those available within 18 m of each den tree. Habitat types as in Fig. 6a, 6b. Sizes of den trees are given in Fig. 24. Habitats are ordered from left to right in decreasing order of selection; > reflects preference that is not statistically significant, whereas >>> represents statistically significant differences between adjacent variables.

Habitat Type Ranking	
2nd order	SC2>SC1>>>MRI>>>MCP>SC0=BAR=WTM>MON
3rd order	SC2>SC1>(SC0+MON+MCP)
Den trees	LgHardwood>LgSawtimber>SmHardwood >>>SmSawtimber>>>Poletimber>>>Sapling

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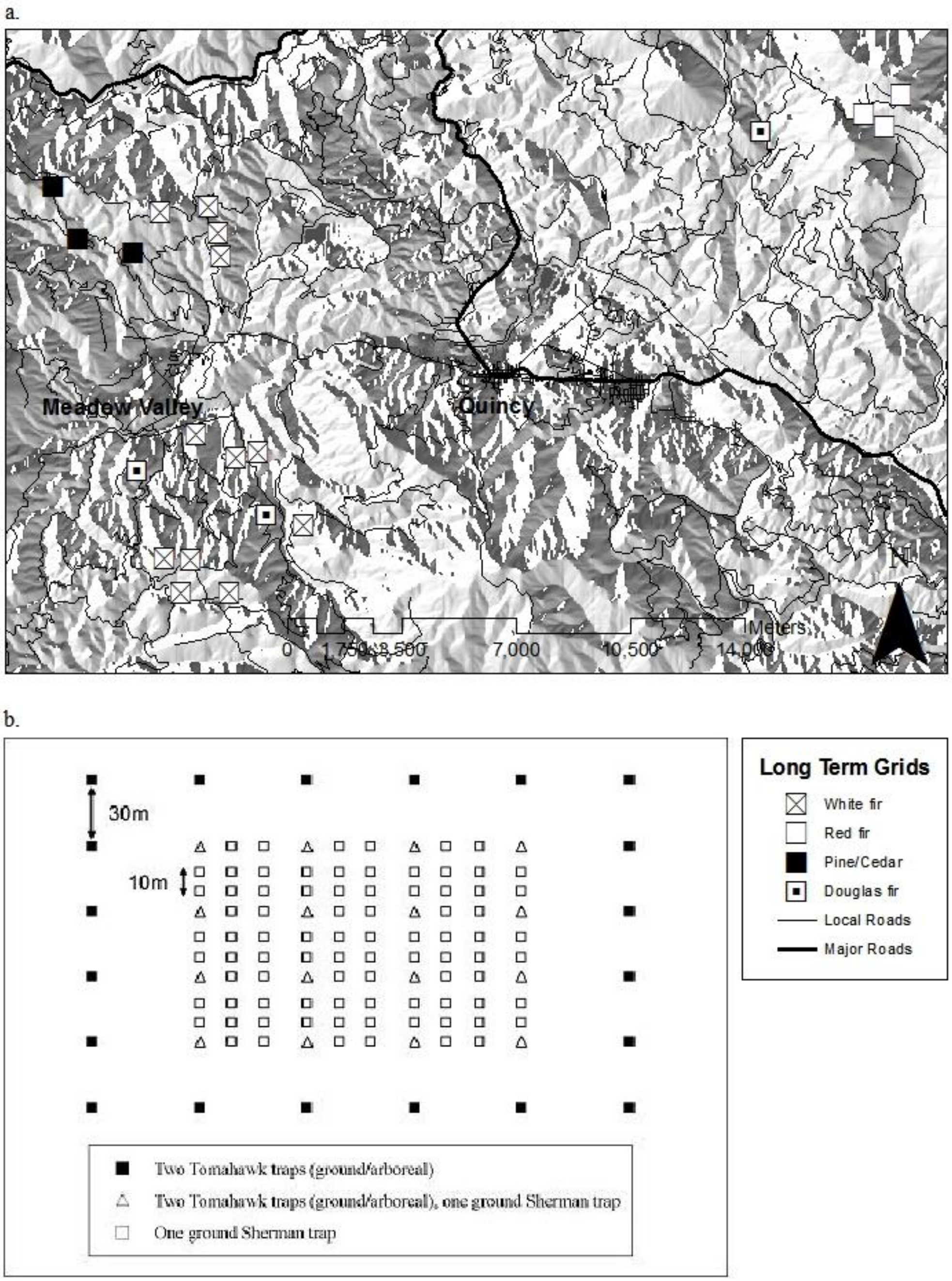

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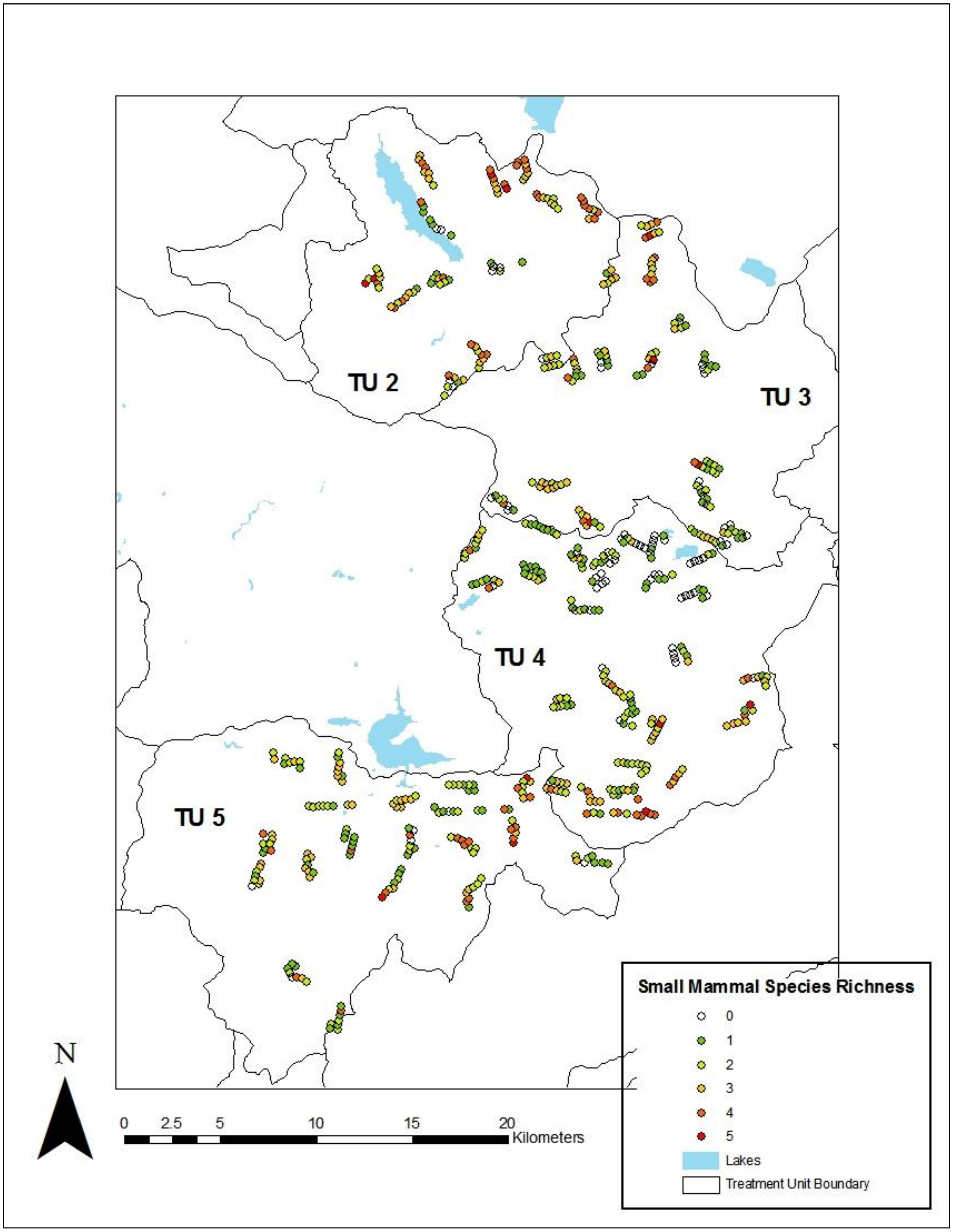

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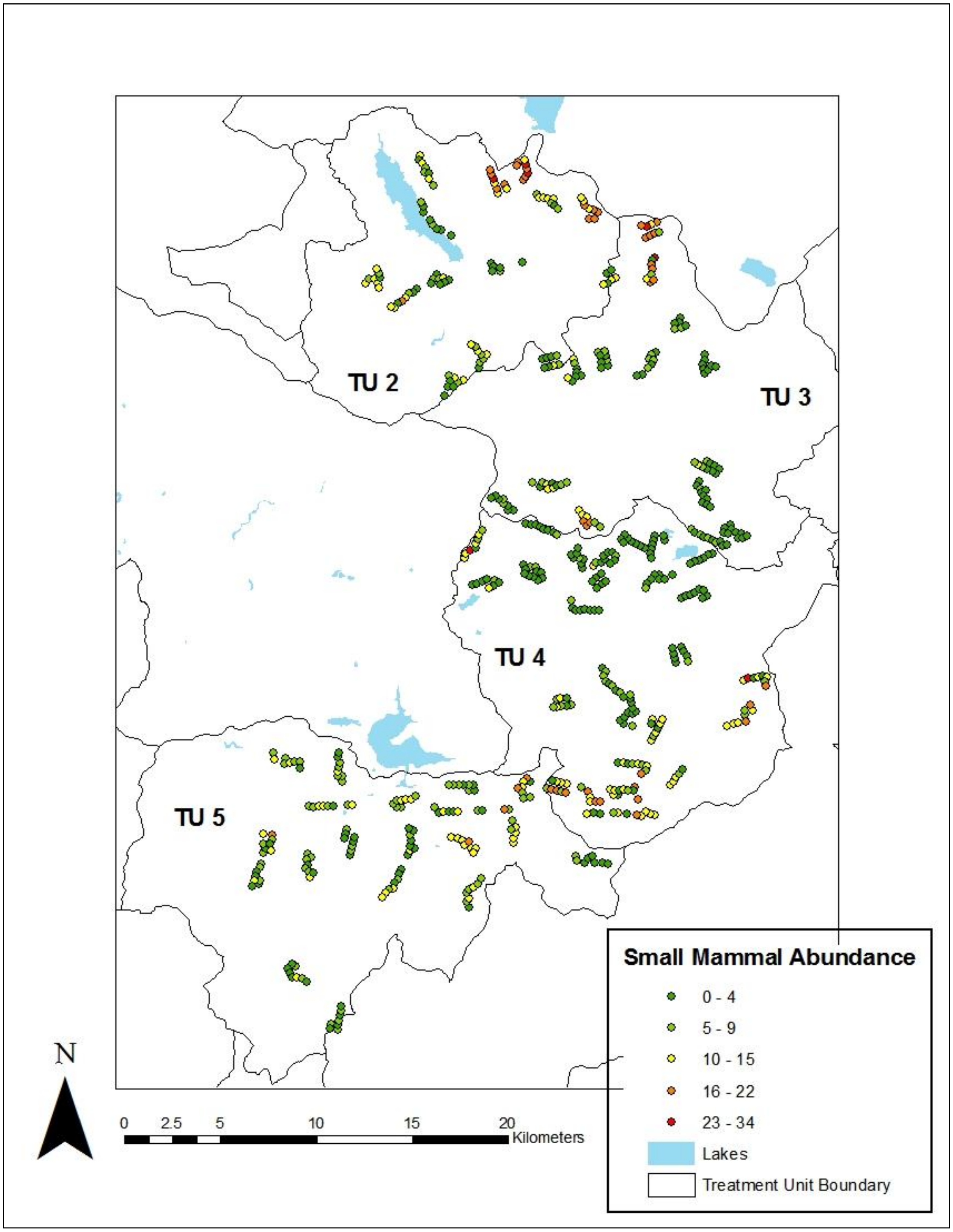

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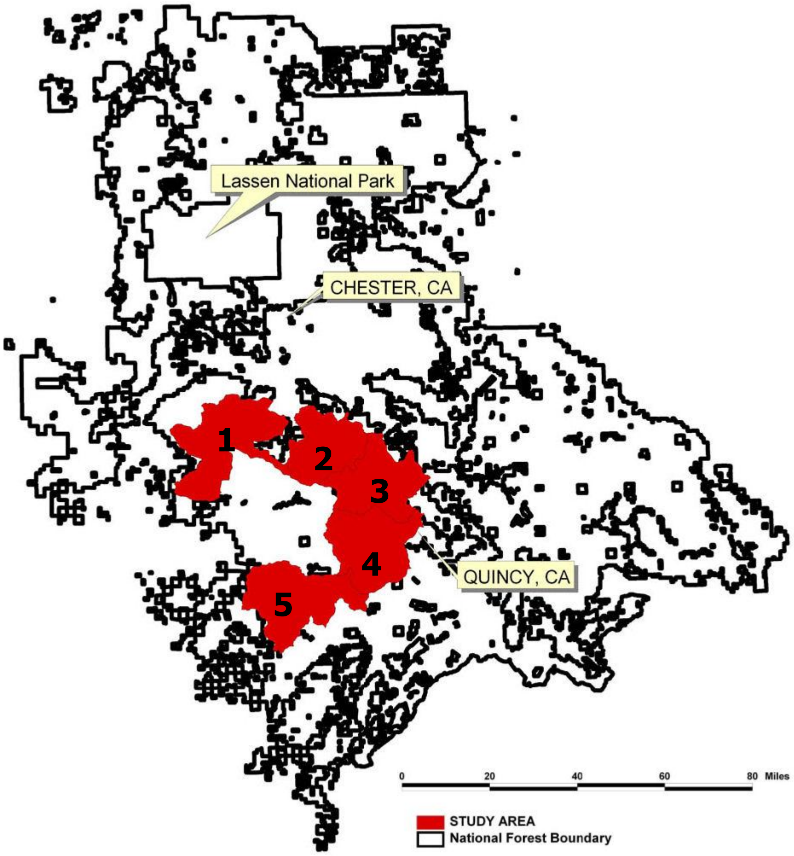

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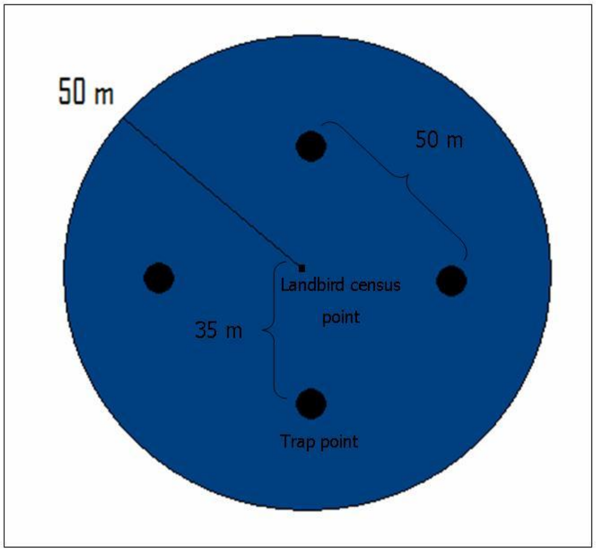

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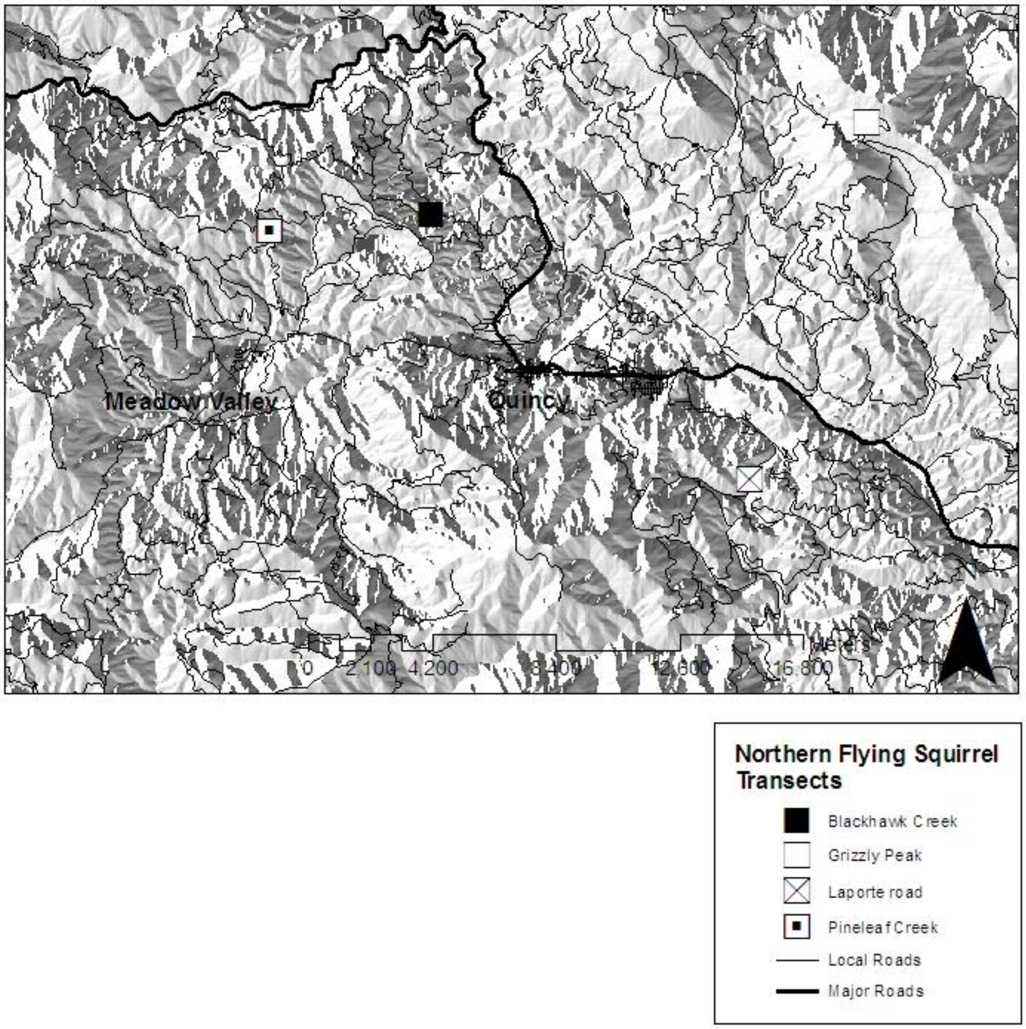

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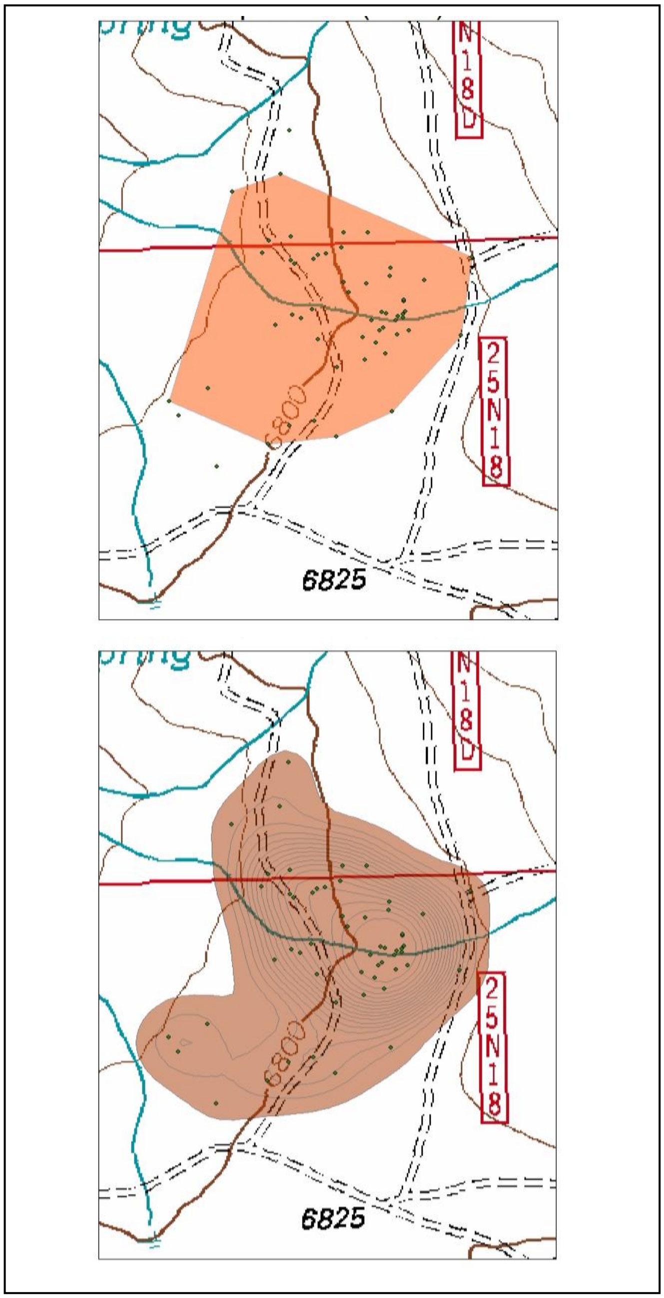

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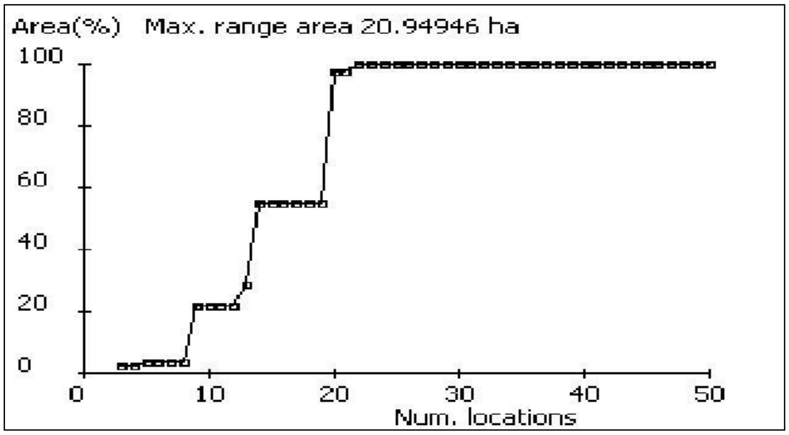

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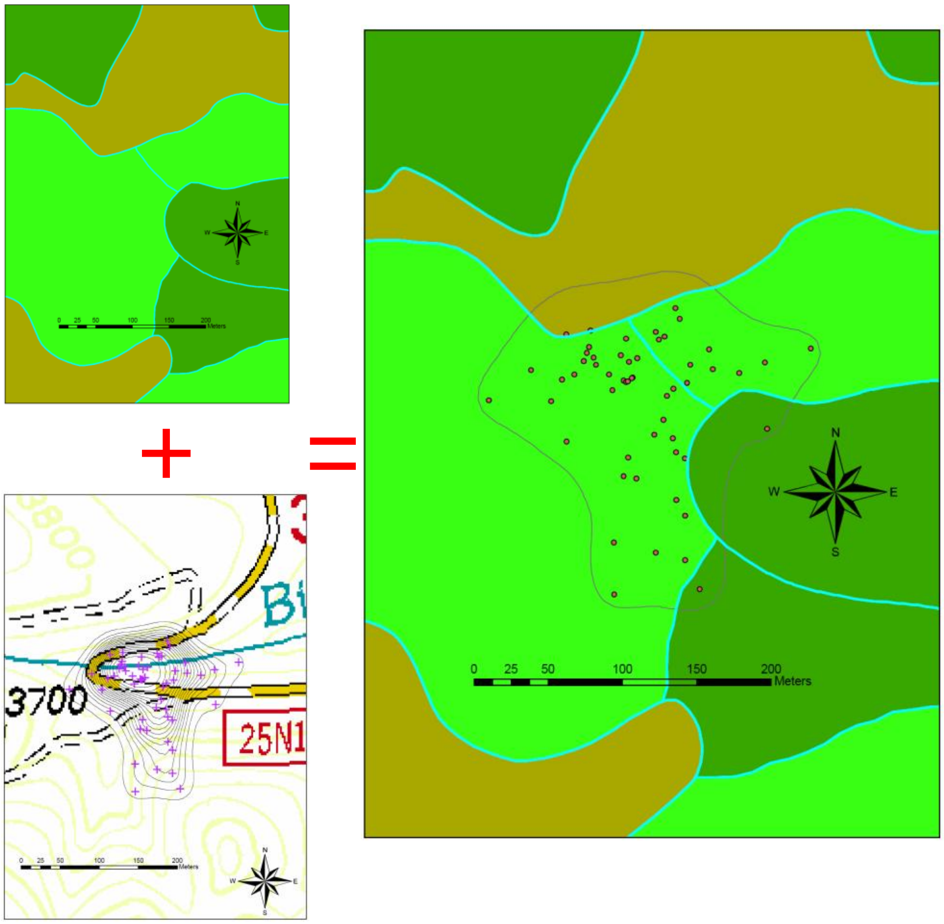

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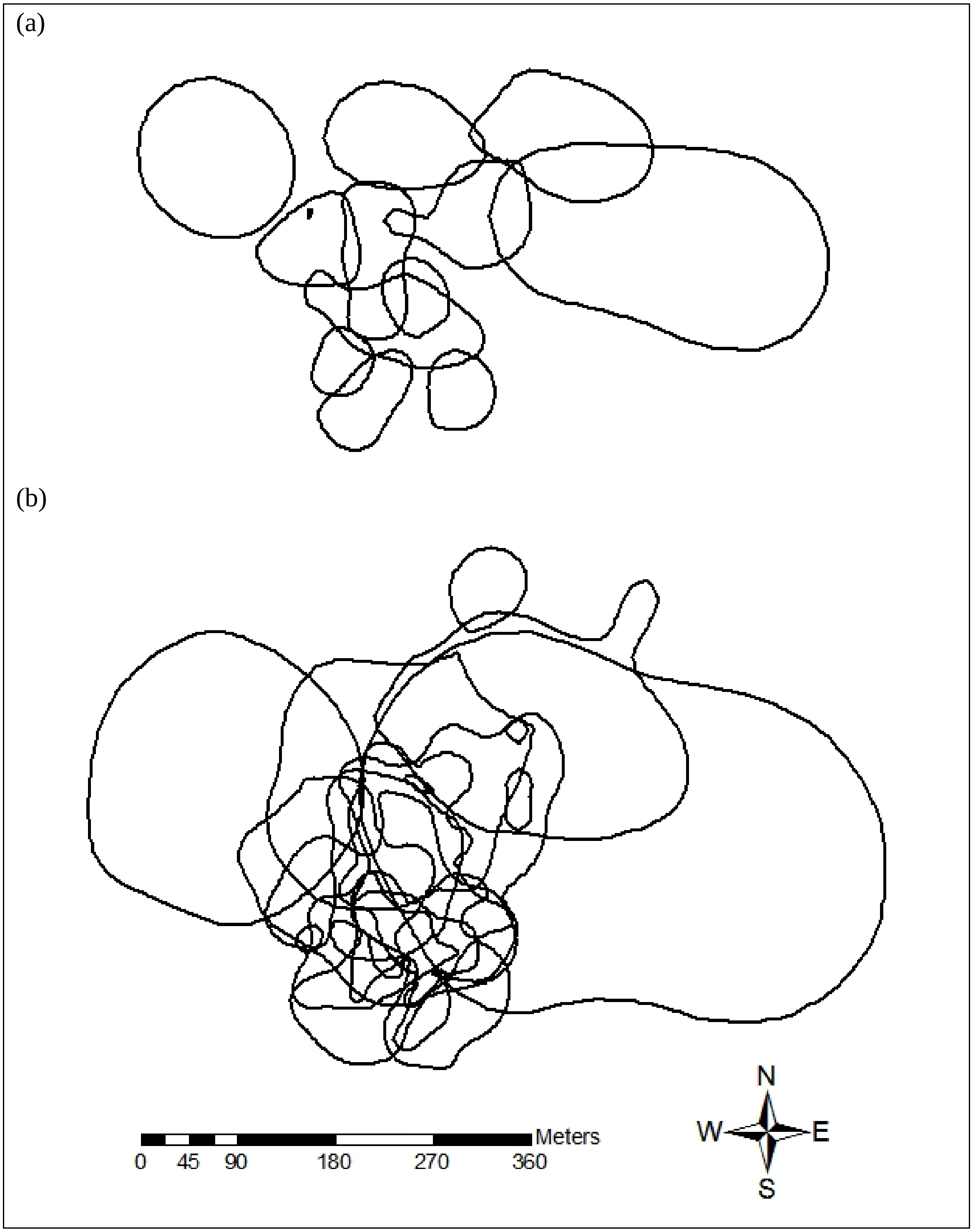

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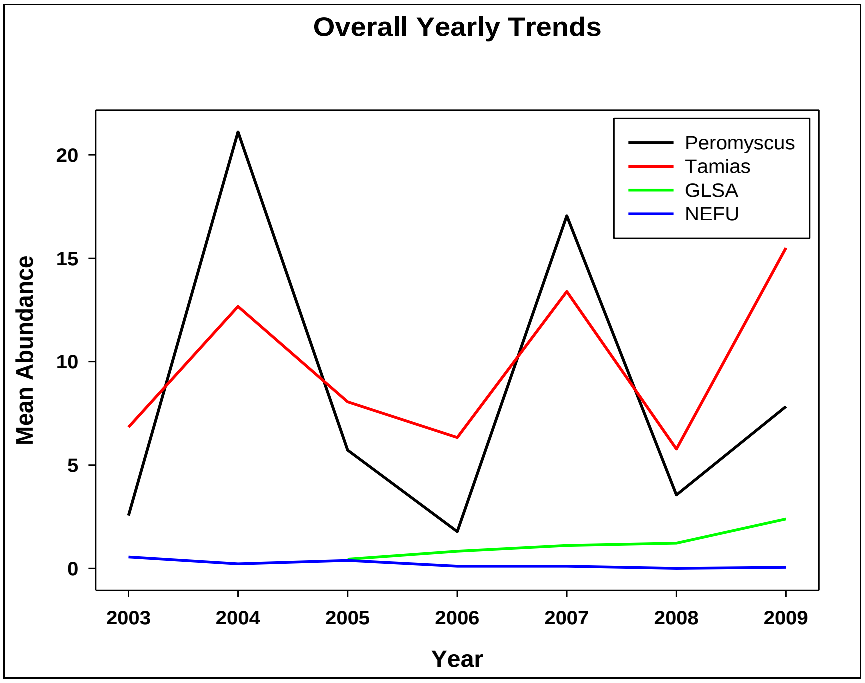

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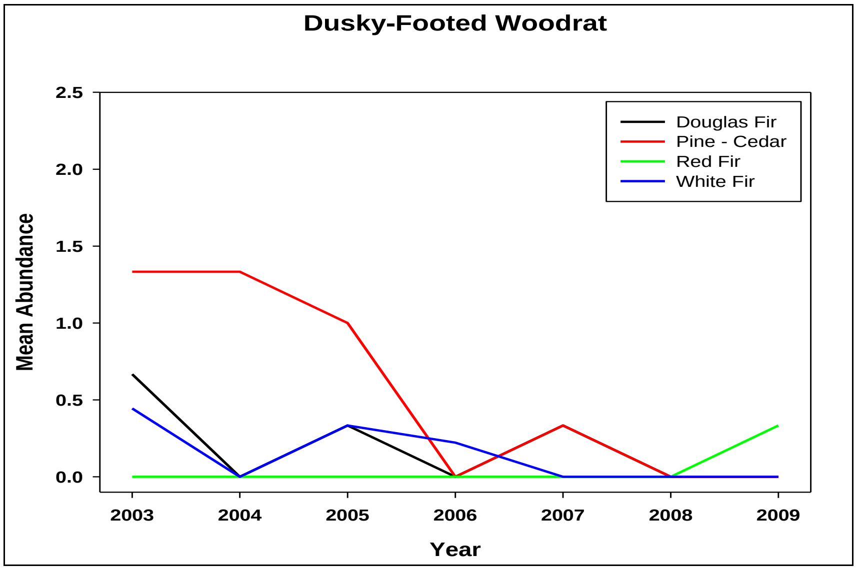

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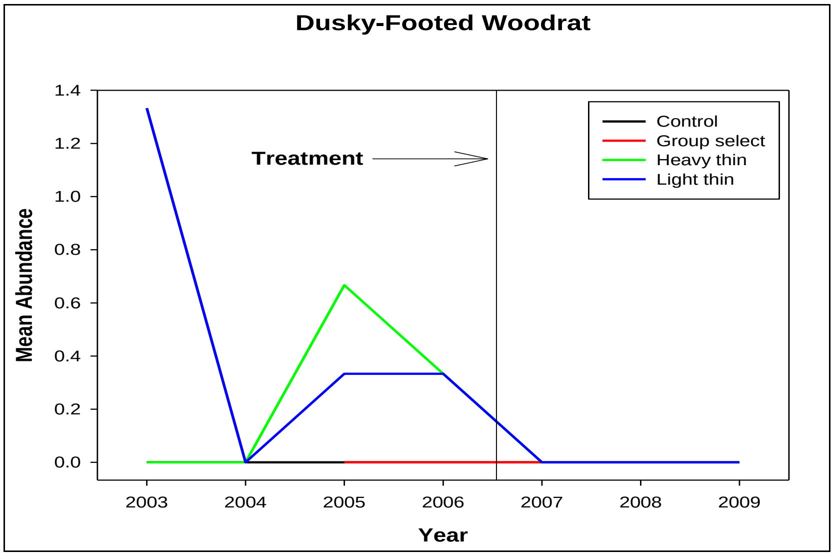

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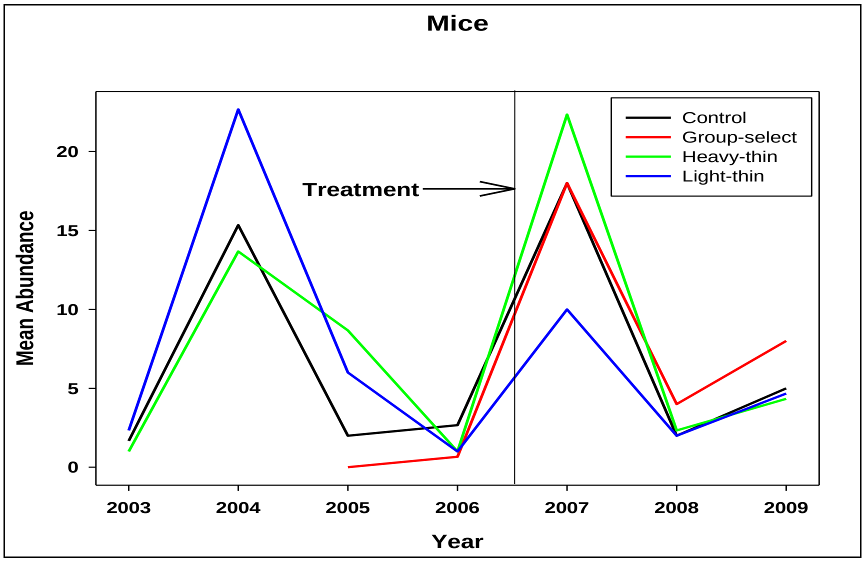

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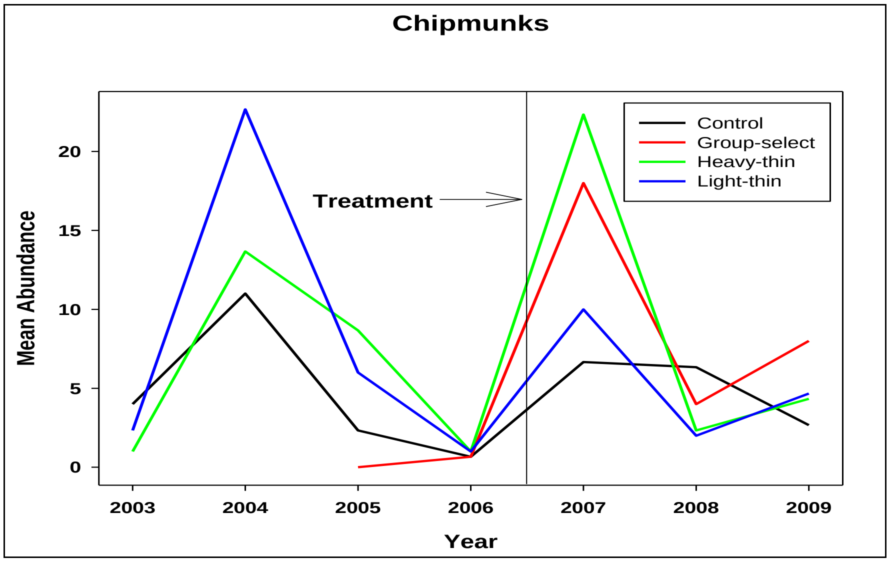


Figure16. Mean abundance of northern flying squirrels (*Glaucomys sabrinus*) pre and post treatment within the Plumas National Forest, California, during 2005-2009.

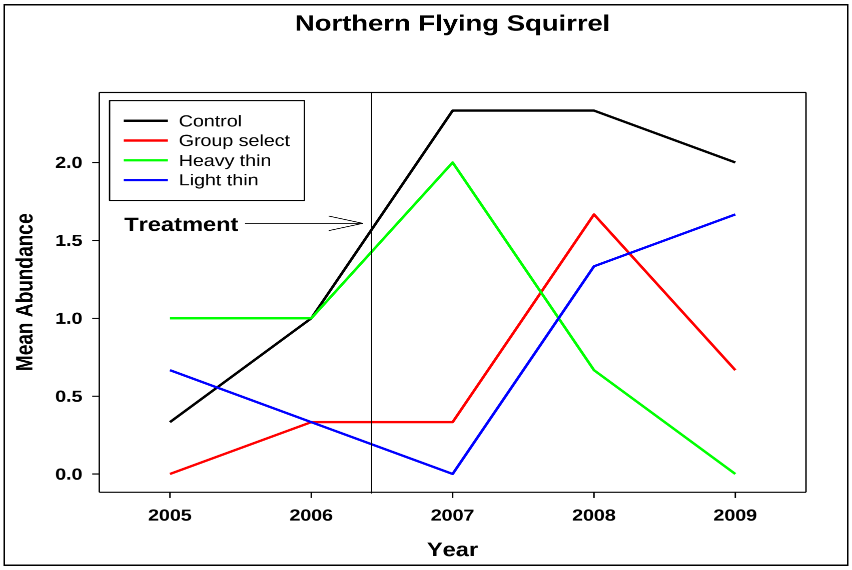


Figure 17. Mean abundance of mice (*Peromyscus* sp.) across 4 forest types within Plumas National Forest, California, during 2003-2009.

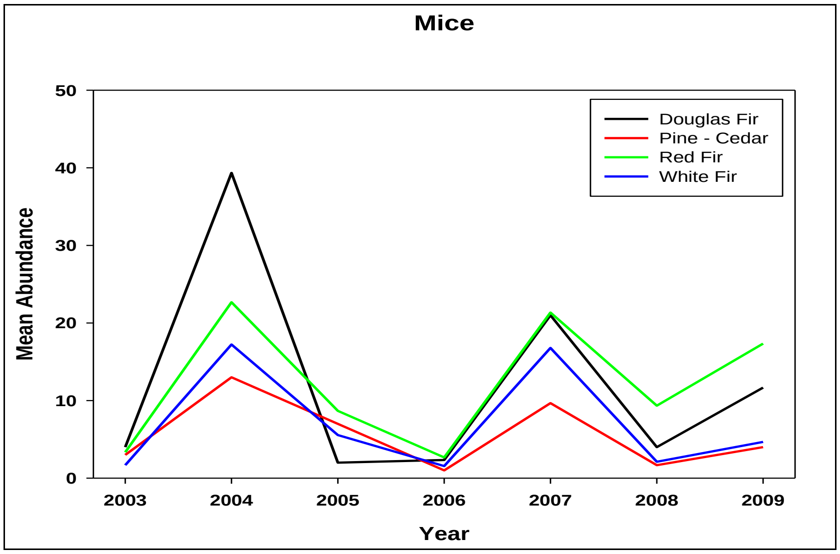


Figure 18. Mean abundance of chipmunks (*Tamias* sp.) across 4 forest types within the Plumas National Forest, California, during 2003-2009.

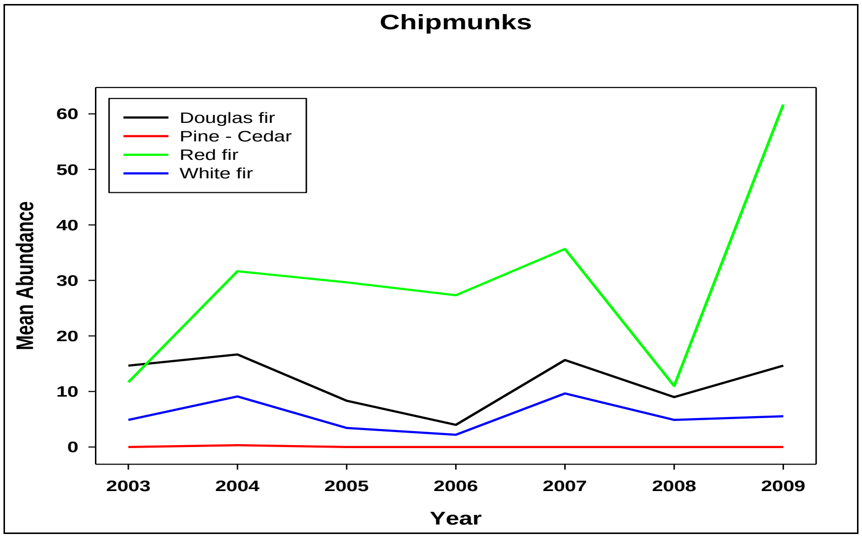


Figure 19. Mean abundance of northern flying squirrels (*Glaucomys sabrinus*) across 4 forest types within the Plumas National Forest, California, during 2005-2009.

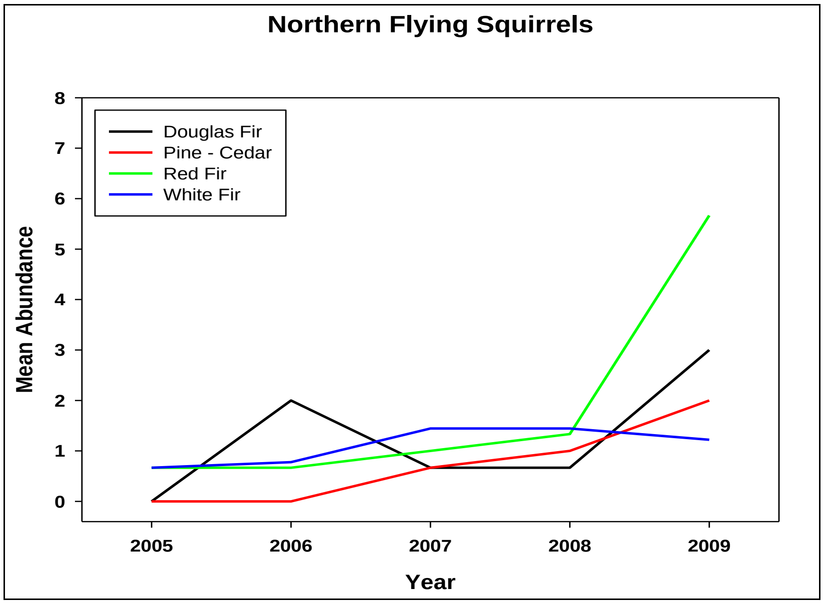


Figure 20. General relationships between mean mouse (*Peromyscus* sp.) mean abundance, mean annual cone production, and mean annual snow fall from 2003-2009. Note: lag in response to mean conifer cone abundance and immediate response to annual snow fall.

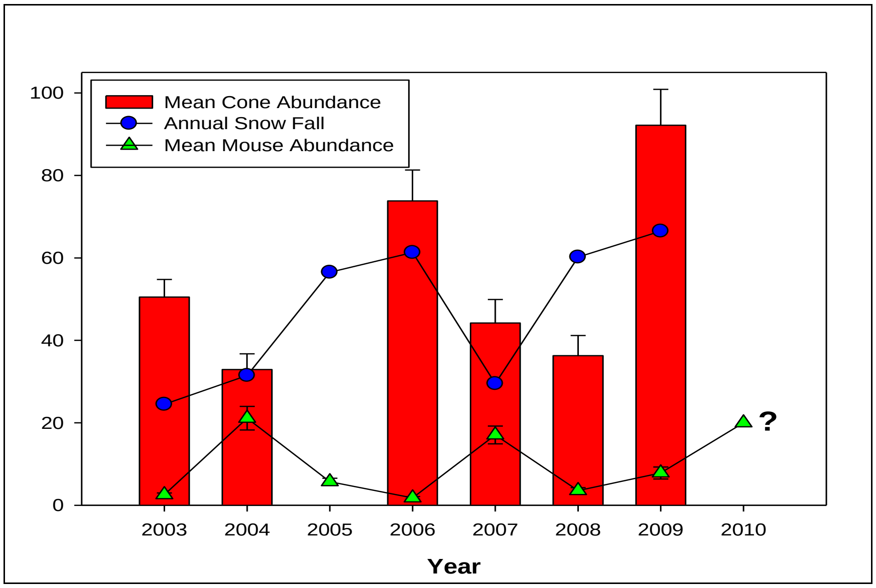


Figure 21. Mean annual biomass of a) small mammal species and b) spotted owl principle prey species from 2003-2009; summarized across 4 forest types (Douglas-fir (DF), Pine/Cedar (PO), Red-fir (RF), and White-fir (WF)).

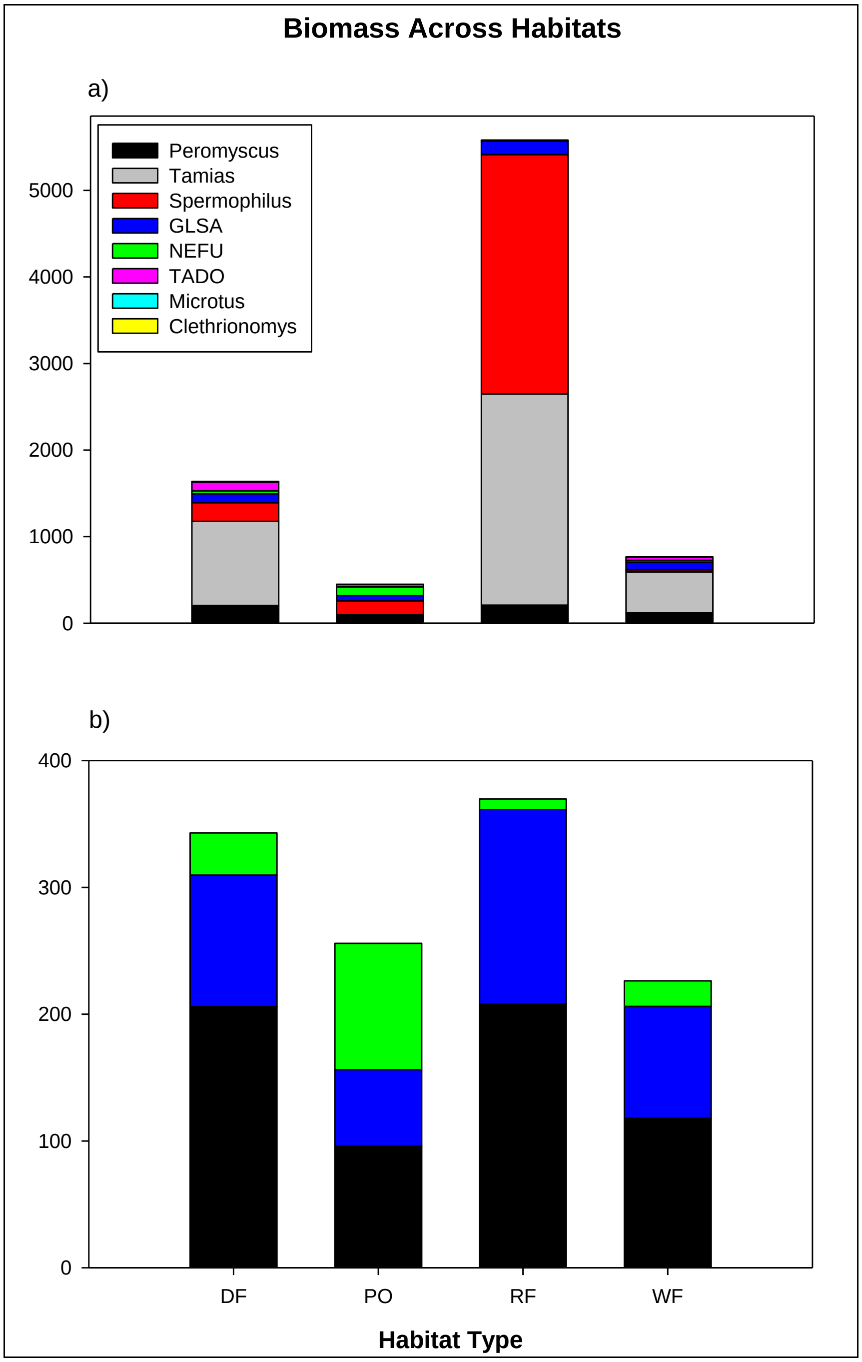


Figure 22. General relationship between mean mouse (*Peromyscus* sp.) abundance and mean annual cone production by species from 2003-2009.

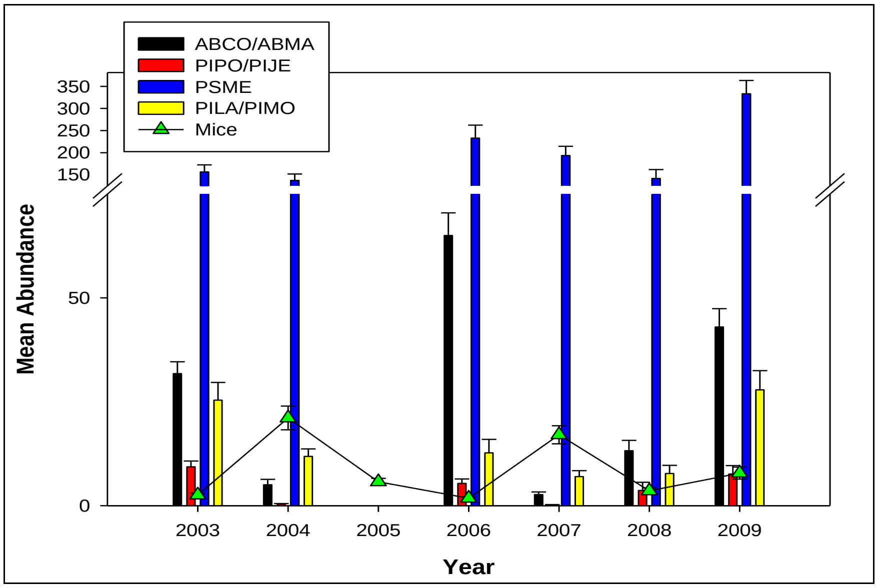


Figure 23. Multiple regression model #3 predicted values versus actual observed values of *Peromyscus* populations during 2004, 2005, and 2007 – 2009 across all long-term grids (n=21).

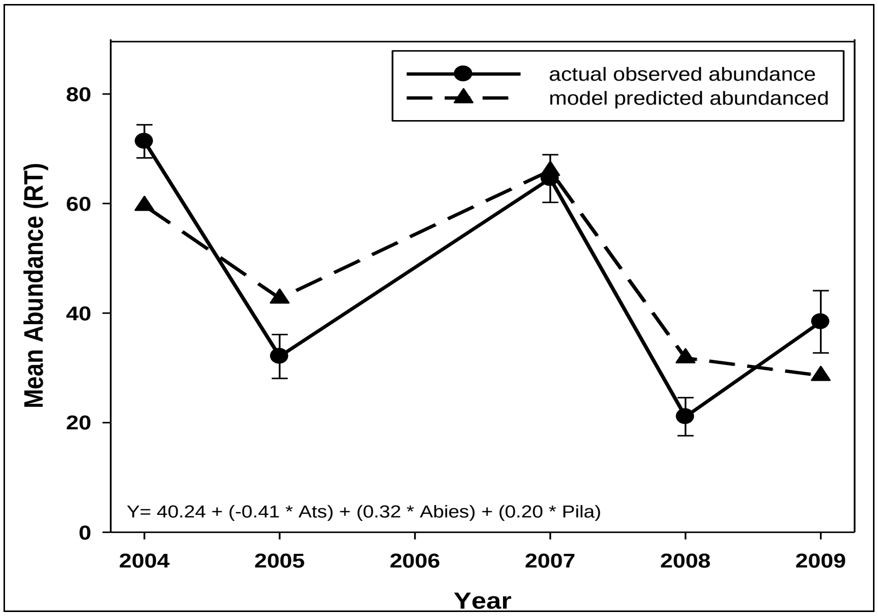


Figure 24. Percentage of used and available habitat types for northern flying squirrels (*Glaucomys sabrinus*) at 2 spatial scales: a) home range relative to that available in a 2 km buffer around the home range (e.g., Aebischer's (1993) 2nd order), and b) use of habitats within the home range relative to that available (e.g., Aebischer's (1993) 3rd order). BAR=barren, MCP=montane chaparral, MON=monoculture, MRI=montane riparian, SC0=small Sierra mixed conifer, SC1= medium Sierra mixed conifer, SC2=large Sierra mixed conifer, WAT=water, WTM=wet meadow.

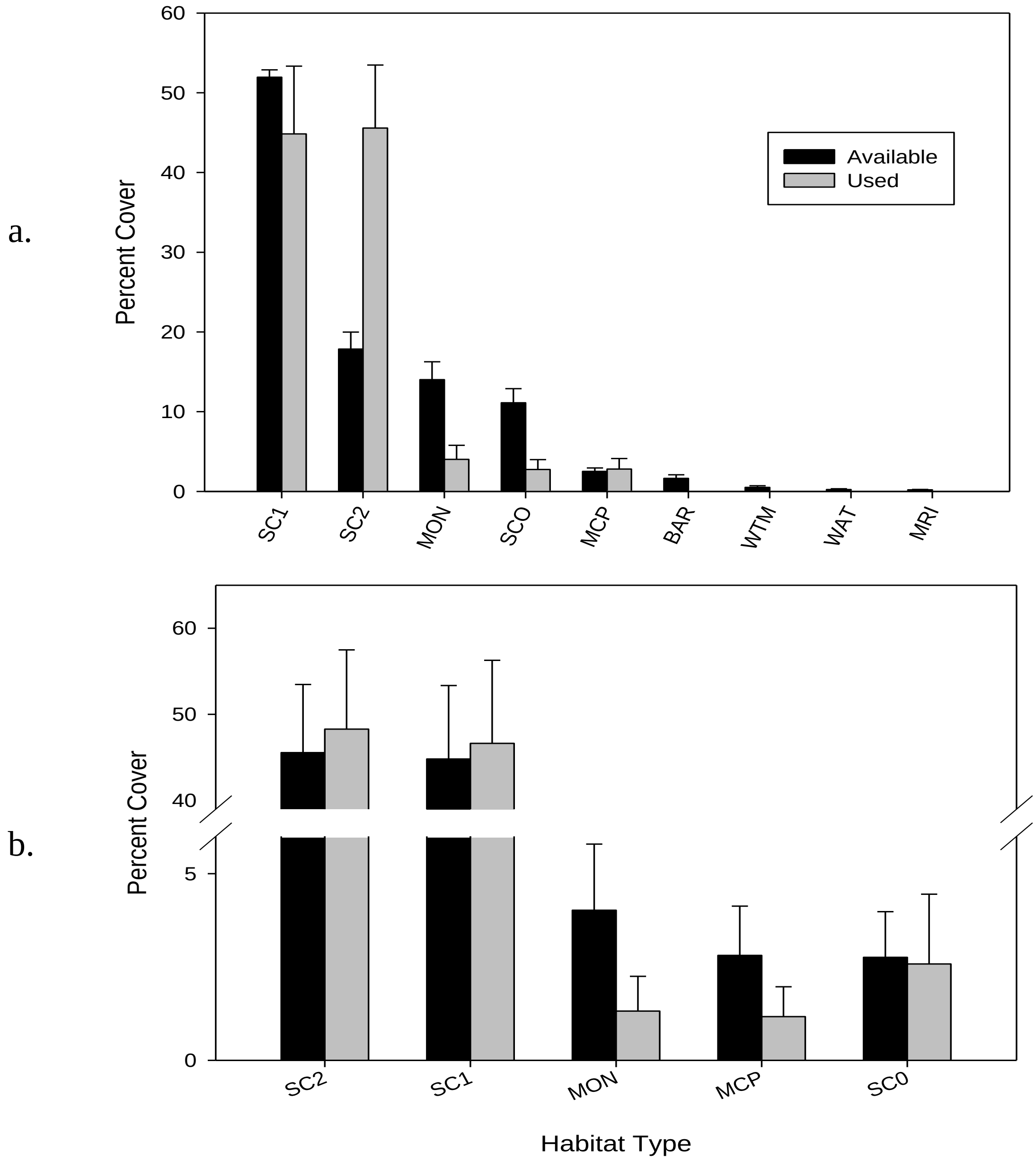
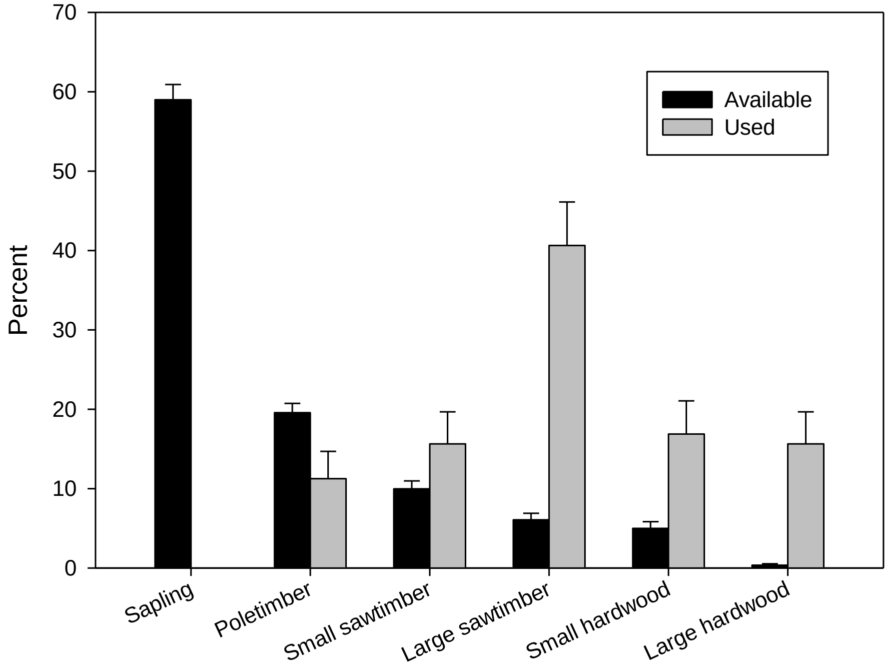


Figure 25. Distribution of available (*n*=11,602) and used (*n*=91) den trees of northern flying squirrel (*Glaucomys sabrinus*) in the northern Sierra Nevada, classified by tree type (hardwood vs. softwood) and size (in cm dbh; Sapling=0-10, poletimber=conifers 10-27.9, small sawtimber=conifers 28-53.3, large sawtimber=conifer≥53.4, small hardwood=10-33, large hardwood>33). Available trees are those trees within a 0.1 ha circle centered on den tree. Used trees are den trees.



Plumas-Lassen Administrative Study California Spotted Owl Module:

2009 Annual Report

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Introduction

Knowledge regarding the effects of fuels and vegetation management on California spotted owls (***Strix occidentalis occidentalis;*** CSOs) and their habitat is a primary information need for addressing conservation and management objectives in Sierra Nevada forests. The specific research objectives of the California spotted owl module as identified and described in the Plumas-Lassen Study (PLS) Plan are:

1) What are the associations among landscape fuels treatments and CSO density, distribution, population trends and habitat suitability at the landscape-scale?

2) What are the associations among landscape fuels treatments and CSO reproduction, survival, and habitat fitness potential at the core area/home range scales?

3) What are the associations among landscape fuels treatments and CSO habitat use and home range configuration at the core area/home range scale?

4) What is the population trend of CSO in the northern Sierra Nevada and which factors account for variation in population trend?

5) Are barred owls increasing in the northern Sierra Nevada, what factors are associated with their distribution and abundance, and are they associated with reduced CSO territory occupancy?

6) Does West Nile Virus affect the survival, distribution and abundance of California spotted owls in the study area?

7) What are the effects of wildfire on California spotted owls and their habitat?

Current information on the distribution and density of CSOs across the HFQLG study area is required to provide the data necessary to build predictive habitat models and provide baseline population information against which we will assess post-treatment changes in CSO populations and habitat. Continued monitoring on the Lassen Demographic Study Area is critical for estimating CSO population trends and status. Our focus in 2009 was to conduct landscape inventories of CSO distribution and abundance, and continue banding to provide the required data and baseline information to meet the objectives of Research Questions 1-4 identified above. Complete landscape inventory surveys were conducted across 9 of the 11 original project survey areas in 2009 (Figure 1). Surveys were not conducted in 2 survey areas in 2006-2009 (SA-5, SA-7, Figure 1). Surveys were not conducted in these 2 study areas in 2006-2009 because sufficient data for determining the number and distribution of CSO sites for initial habitat modeling efforts was collected in 2004-2005. We conducted surveys in two new survey areas in 2009. These two new survey areas were in proposed project areas on the Lassen NF (Scotts John) and Plumas NF (Empire). We added these two new survey areas to the project to bring our sample size of survey areas slated for treatment to four (Meadow Valley, Creeks, Scotts John, Empire).

Details on survey methods are described in the study plan. Efforts were made to monitor the pair and reproductive status of each owl, and to capture, uniquely color-mark, and collect blood samples from each individual owl across the study area. Capture and color-marking is necessary to estimate survival and population trend, and to assess exposure to West Nile Virus (WNV)(Research Question #5). We also recorded all barred and hybrid barred-spotted owls encountered in the study area and synthesized all existing barred owl records for the northern Sierra Nevada to address Research Question #6. Additionally, we conducted the second year of a radio-telemetry study on CSOs within SA-4 in the Meadow Valley project area to document home range size and configuration, and to assess habitat selection relative to the recently implemented treatments. In response to a need for information on the association between CSOs and wildfire we conducted a second year of surveys to assess CSO distribution, abundance and habitat associations in the Moonlight and Antelope Complex fire area (MAC) on the Plumas NF and initiated surveys in the Cub/Onion fire area (COF) on the Lassen NF. The MAC fires burned in 2007 and we conducted surveys in 2008 and 2009 to assess the immediate post-fire response of CSOs. The COF burned in 2008We have added a seventh research questions to reflect this new research direction.

Results

CSO Numbers, Reproductive Success, Density and Population Trends:

A total of 72 territorial CSO sites were documented across the core PLS study area in 2009 (Figure 2). This total consisted of 63 confirmed pairs, 5 unconfirmed pairs (i.e., one member of pair confirmed as territorial single plus single detection of opposite sex bird), and 5 territorial single CSOs (single owl detected multiple times with no pair-mate detected). Thirty pairs successfully reproduced in 2009 (47.6% of confirmed/unconfirmed pairs). Of these thirty pairs, 23 were located on the Lassen NF and 7 were located on the Plumas NF. A total of 47 fledged young were documented in 2009 (1.57 young per successful nest) (Table 1). Across the recent five years of the study, CSO reproduction has been highest in 2004, 2007 and 2009 in terms of the percent of CSO pairs that successfully reproduced, and in terms of the number of young fledged per successful nest. Approximately 50% of CSO pairs successfully reproduced in 2004, 2007 and 2009 whereas the proportion of pairs successfully reproducing ranged between 14%-18% in 2005, 2006 and 2008. The number of young produced per successful nest was more similar across years, ranging between 1.47 -1.81. Our first year of surveys in two projected project areas yielded 2 pairs of CSOs in the Empire project area and 3 pairs in the Scotts John project area. Neither of the two pairs in the Empire project area reproduced in 2009, whereas 2 pairs did reproduce in the Scotts John project area, producing 4 fledglings.

CSO reproduction is known to vary with spring weather: precipitation patterns were more similar in 2004 and 2007, with total precipitation relatively low during March-April of 2004 and 2007 as compared to 2005 and 2006 (Figure 3). However, this pattern between spring weather and precipitation varied in 2008 and 2009. In 2008 spring precipitation was low in March-April, yet CSO reproduction was also low. In contrast, during 2009 spring precipitation was high in February-March, low in April, and CSO reproduction was high on the Lassen portion of the study area and low on the Plumas portion of the study area. These patterns suggest that additional factors influence CSO energetics and are associated with annual variation in CSO reproduction. Potential factors include elevational variation in cold and hot temperatures, precipitation, duration of spring/summer and snowpack, in addition to annual variation in prey populations.

The Lassen Demographic Study Area (SA-1A, SA-11, SA-12, SA-13, SA-14, SA-15) and Plumas NF Survey Areas (SA-2, SA-3, SA-4, SA-5, SA-7) were fully integrated in 2005 to define the overall Plumas-Lassen Study project area and provide consistent CSO survey effort across the project area. (Figures 1 & 2). We estimated the crude density of CSOs based on the number of territorial owls detected across 9 survey areas during 2008 surveys at the Survey Area spatial scales (Tables 2 and 3). The estimated crude density across the overall study area in 2009 was 0.069 territorial owls/km^2. Overall study area crude densities are not directly comparable across years because different total areas were surveyed in each year. However, crude density estimates within individual Survey Areas indicate similar densities and number of territorial sites (pair sites plus territorial single sites) between 2004-2009 for the survey areas on the Plumas NF (SA-2, SA-3, SA-4). while numbers have declined somewhat on the Lassen survey areas (SA-1A, SA-11, SA-12, SA-13, SA-14, SA-15) between 2005-2007 and then stabilized or slightly increased between 2008-2009 (Tables 2 and 3).

The most recent information on CSO population trends is included in the January 2006 meta-analysis, conducted to estimate CSO population trends and to assess population status in response to a petition to list the CSO under the Endangered Species Act (Blakesley et al. 2006). These data continue to provide the best estimates of CSO population trends. Data collected between 1990-2005 from four CSO demographic studies across the Sierra Nevada and southern Cascades, including the Lassen Demographic Study Area, were analyzed as part of the meta-analysis workshop. The Lassen Demographic Study Area is contained within the overall PLS study area and consists of survey areas SA-1A, SA-11, SA-12, SA-13, SA-14 and SA-15 in Figure 1. Full details on meta-analysis methods and results are provided in Blakesley et al. (2006). In synopsis, across the four study areas, results indicated that the Lassen Study CSO population exhibited the strongest evidence for a population decline between 1990-2005. Mean lambda for the Lassen Demographic Study was 0.973, with 95% confidence limits ranging from 0.946-1.001 (Table 4).

Habitat Assessment – Nest/Roost Plot Scale

We documented a total of 103 CSO territorial sites between 2004-2006. We overlayed the primary nest/roost locations for each of the 103 CSO sites with the CWHR vegetation classes available within the VESTRA photo-interpreted vegetation map for the PLS to examine nest/roost-site habitat association patterns. Approximately 53% of the nest sites were located within CWHR 5M, 5D and 6 size classes (Table 5, Figure 4). An additional 37% of the sites were located within CWHR size class 4M and 4D polygons. CWHR size class 4 is defined as stands with average tree sizes of 12-24 inch (30-61 cm) diameter-at-breast-height (dbh) trees. Of the 38 sites located in size class 4 polygons, 25 (66%) were in size class 4 polygons with a large tree component (i.e., presence of >24 inch or 61 cm dbh trees). Overall, about 90% of the sites were located within CWHR 4M, 4D, 5M, 5D, and 6 size classes. The remaining 10 sites were located in more open, smaller-tree size polygons, with nests or roosts located within remnant, scattered larger trees (Table 5, Figure 4).

While the distribution of nest site locations relative to broad vegetation classes provides insight into patterns of nest-site habitat, we also conducted vegetation sampling at nest or primary roost sites to describe vegetation structure and composition. Vegetation plot sampling was conducted at 80 CSO territories across 2005-2007. Vegetation plots were centered on CSO nest trees, or on a primary roost tree for sites where no nest has been documented, and were measured using the national Forest and Inventory Assessment (FIA) protocol. The FIA protocol is used nationally by the USDA Forest Service for inventorying and monitoring vegetation. FIA sampling consists of measuring vegetation structural and compositional variables within a 1-ha plot centered on a CSO nest or roost tree. Only one plot was collected from each CSO territory, with the most frequently used nest tree serving as the plot center location, or the most recent nest tree used at sites where no nest tree was used more frequently than another. CSO nest sites were characterized by mean total basal areas of 260.8 ft^2/acre, 7.4 snags (>15 inch or 38 cm dbh)/acre, and 10.7 trees (>30 inch or 76 cm dbh)/acre (Table 6). Under the FIA protocol, canopy cover is modeled based on the tree inventory list. The modeled canopy cover for these plots averaged 64.1%. Shrub cover averaged 7.7%. Fuel loads averaged 0.75 tons/acre for 1-hr fuels, 4.0 tons/acre for 10-hr fuels and 4.44 tons/acre for 100-hr fuels (Table 6). Use of the FIA sampling protocol will facilitate monitoring of vegetation and development of CSO habitat models that can be used as adaptive management planning tools. Habitat models are currently being evaluated that can be used to assess projected changes in CSO nesting habitat suitability under varying fuels and vegetation treatment scenarios.

Habitat Assessment – Core Area/Home Range Scale

Core area habitat associations around 102 CSO nest/roost sites was assessed by using a Geographic Information System (GIS) and the VESTRA photo-interpreted vegetation map to determine the vegetation patterns within a 500 acre (201 ha) circle centered on each of the CSO territory sites. To compare the CSO sites with the general availability of habitat across the study area we also assessed the same vegetation patterns around 130 points determined by placing a systematic grid across the study area. For this summary we assessed vegetation using the USDA Forest Service Region 5 classification system. Overall, CSO core areas averaged 75.7% suitable habitat (classes 3N, 3G, 4N, 4G) whereas the grid points averaged 61.9% (Table 7, Figure 5). Approximately 32% of CSO core areas was composed of large tree polygons (>24inch or 61 cm dbh, >=40% canopy cover) compared to 19.6% of the grid points (Table 7, Figure 6).

Radio-Telemetry – Meadow Valley Project Area

Ten adult territorial CSOs were radio-tracked within SA-4 in the Meadow Valley Project Area for the 2007 and 2008 breeding seasons. The sample included 3 males and 5 females during each breeding season; six birds were followed for the telemetry study duration, two birds were radio-tagged in early 2008, and one male and one female died of apparent natural causes during the winter of 2007-8. We attempted to locate each radio-tagged CSO 5 times over each 2-week sampling period, resulting in approximately 30 locations per bird for the breeding season defined as April-September. The eight birds initially radio-tagged in 2007 were followed at reduced effort during the 2007-2008 nonbreeding period to determine wintering locations and post-breeding movements.

These data will be used to investigate CSO home range sizes and configurations, as well as habitat selection within home ranges relative to available vegetation and fuels treatments. Efforts to asses CSO use of the post-treated landscape in SA-4 (Meadow Valley) has been severely hampered by the lack of post-treatment vegetation data. The post-treatment vegetation dataset was recently completed March 1, 2010, and the radio-telemetry data will now be analyzed for these relationships.

The plot-scale vegetation structure and composition will also be analyzed at a subsample of CSO radio-telemetry bird locations. Eighty-seven vegetation plots were measured to the standard FIA protocol between August-November 2008, and forty-five additional vegetation plots were measured to the same protocol in September-October 2009.

Meadow Valley Project Area Case Study:

The Meadow Valley Project Area (MVPA) is the first area within the PLS where the full implementation of HFQLG treatments has occurred. Treatments were implemented on the ground within this project area during 2001-2008, with primarily light-thinning and underburning occurring in 2001-2005, and Defensible Fuel Profile Zones and Group Selections implemented during 2005-2008. The MVPA corresponds closely with the boundaries of SA-4 of the PLS.

We began monitoring CSOs SA-4 in 2003 and have annually monitored the distribution, abundance and reproduction of CSOs within SA-4. Additionally, we have color-banded all individuals within this area, with the exception of one male who could not be captured. Full survey methods are described in detail in our study plan (available from field project leaders) and are consistent with USDA Forest Service R5 survey methods. Briefly, we conduct 3 nocturnal broadcast surveys during the breeding period (April-August) across a network of survey points to detect CSOs. When a CSO is detected we then conduct dusk status surveys to pinpoint roost and nest locations for each bird. Status surveys are used to determine the social status of each bird (pair or single), nesting and reproductive status (breeding, non-breeding, unknown), and to identify color-banded individual birds.

In general, in years of higher CSO reproduction, such as occurred in 2004 and 2007, it is easier to establish pair and reproductive status and to identify individual birds as they are more vocal and exhibit stronger ties to their core areas. In years of lower reproduction, such as occurred in 2005, 2006, 2008, and 2009 (reproduction was low on the Plumas NF, while high on the Lassen NF portion of the study area) it is more difficult to determine the status of birds as they tend to range more widely and are not as vocal and territorial, particularly the females. Based on our cumulative survey results, we then use accepted, standardized methods for estimating the overall number of territorial sites (confirmed pairs, unconfirmed pairs and territorial singles) for each year. Confirmed pairs consist of a reproductive pair of CSOs or, at non-reproductive sites, the detection of a male and female on more than one occasion within 1/2-mile (0.8 km) of each other across the breeding period. Unconfirmed pairs consist of two sightings of one sex and one detection of the opposite sex within 1/2-mile (0.8 km) of each other across the breeding period. Territorial singles are considered to be individuals that are detected on at least 2 occasions within a 1/2-mile (0.8 km) distance across the breeding period without a detection of the opposite sex. Birds detected on only a single occasion across the breeding period are not considered to be territorial.

Figure 7 illustrates the proposed treatment locations and the cumulative number and distribution of CSO territorial sites across the seven years between 2003-2009. The number of territorial sites across SA-4 varied annually between 6-9 (Table 8, Fig. 8a&b). Overall, the numbers of territorial sites was fairly similar with 7 sites documented between 2004-2006, an increase to 9 territorial sites during the high reproductive year that occurred in 2007, and then decreasing to 6 territorial sites in both 2008 and 2009.

Whereas we have not detected large change in the number of CSO territorial sites across SA-4, we have documented changes in occupancy status and spatial movements of individual sites that may be associated with treatments. The Maple Flat site, located in the NW corner of SA-4 was occupied from 2004-2007, not occupied in 2008 following treatments in Fall 2007, and was colonized by an unconfirmed pair of new CSOs who were present in the area during 2009. Whether or not the treatments caused the site to be unoccupied in 2008 is uncertain as the male also died during the winter of 2008-2009 (determined by radio-telemetry). The female from 2007 visited the site in early 2008 then moved and summered 14.5 km (9 miles) from Maple Flat near Seneca in early June 2008. She remained in this area through October 2008 when she was recaptured and the radio-transmitter was removed. No new CSOs were detected or colonized the Maple Flat site in 2008.

We also observed movement of a site in the SW corner of SA-4 that corresponded to the timing of treatments in the nest core. This site (Miller Fork) was occupied by CSO pairs in 2003-2005, a single male in 2006 following treatments in 2005-2006, and then was not occupied between 2007-2009. However, a new CSO pair established a site (Big Creek) in

2007 about 2km to the NW of this site. This new site has been occupied by a pair between 2007-2009.

We also documented changes in occupancy at 2 sites in the eastern portion of SA-4 that were outside the area of treatments. Slate Creek was occupied by a male in 2003 and a pair in 2004, yet has not been occupied since. The female from 2004 was detected on the Lassen NF and then back on the Plumas in subsequent years but was not detected in 2009. CSO pairs were present at Deer Creek between 2003-2007 and fledged triplets in 2007. However, this site has been unoccupied in 2008-2009. Neither of these sites has experienced treatments.

We documented the colonization of a new site (Pineleaf Creek) during and following treatments in the north portion of SA-4. A single territorial male was present in this area in 2006, with a pair of CSOs then present between 2007-2009.

To date, we have not observed dramatic changes in the numbers of territorial CSO sites within SA-4 as an immediate acute response to treatments. These initial findings should be tempered by the need to assess possible chronic, or longer-term, responses by CSOs. Of importance, 2008 and 2009 were low reproductive years on the Plumas NF, with only 2 nests in 2008 and 7 nests in 2009 documented across all of the Plumas NF sites. Thus, the conditions leading to the low reproductive activity in 2008 and 2009 may have resulted in a low probability of recruitment and occupancy of sites in both years. For example, a higher number of territorial sites (9) were documented within SA-4 in the higher breeding year of 2007 as compared to the 7 territorial sites documented in the low reproductive years of 2005-2006. Also, higher CSO reproduction in 2007 may result in increased number of recruits available to colonize sites in 2009-2010. We recommend that monitoring be continued to assess: (1) long-term occupancy, abundance and distribution of CSOs across the project area to document longer-term responses to address concerns that site fidelity in such a long-lived species may obscure possible negative effects of habitat change over the short term; and (2) to continue to monitor color-banded birds to assess longer-term associations between CSO survival, reproduction, and recruitment related to changes in habitat. Each of the pieces of above information is necessary to fully assess the potential acute and chronic responses of CSOs to landscape treatments.

Whereas, we have not observed dramatic short-term changes in CSO numbers across the broader MVPA in response to treatments, however, we have documented some changes in the distribution and occupancy of CSO territories where treatments have occurred within SA-4 that may be associated with treatments. Accurate spatial post-treatment maps for the MVPA have been recently completed (March 1, 2010) that document: (1) the specific locations where treatments were actually implemented on the ground; (2) the specific site-specific treatments that were implemented on a piece of ground; (3) when the treatments were implemented on the ground (which year at minimum); and (4) the resulting post-treatment vegetation structure and composition. Understanding the what, where, when, and effects of treatments is the foundation on which subsequent adaptive management assessments will be constructed. With this information we will now be able to explore the associations between treatments and CSO responses at the landscape and home range spatial scales, in addition to relating within home range habitat use through our telemetry studies.

Wildfire –California Spotted Owl Case Studies

A primary source of uncertainty regarding the effects of fuels treatments is an assessment of risk to CSOs and their habitat from treatments versus the risk from wildfire that occurs across untreated landscapes. Prior to 2008 our PLS work had focused on assessing CSO distribution, abundance and habitat associations across the untreated overall project area landscape and being in position to monitor effects as treatments are implemented within specific project areas, as illustrated by the MVPA case study described above. Beginning in 2008 we were fortunate to have the opportunity and funding support from the Plumas and Lassen National Forests to extend our work to inventory CSO distribution, abundance, and status across the Moonlight and Antelope Complex Fire Area (MACFA) that burned on the Plumas National Forest in 2007. In 2009 we conducted a second year of surveys in the MACFA and also conducted the first year of similar surveys in the Cub-Onion Complex Fire Area (COCFA) that burned on the Lassen National Forest in 2008. As described below, the MACFA was largely a high severity wildfire while the COCFA burned largely at low-moderate severity. Incorporating these two study areas which differ in wildfire severity allow us to directly assess response of CSOs to landscapes that burned with low-moderate severity versus high severity wildfire.

The MACFA consists of two fires burned adjacent to each other in 2007 and both were primarily high severity fires (Fig. 9). The MACFA covers approximately 88,000 acres (35,612 ha). The COCFA consist of two low-moderate severity fires that burned adjacent to each other during June 2008 over approximately 21,000 acres (8,498 ha)(Fig 9). About 52% of the MACFA burned at high severity whereas only 11% of the COFFA burned at high severity (Fig. 10).

In both wildfire study areas we conducted CSO surveys during the breeding period across the entire landscape and within a 1.6 km (1 mile) unburned buffer surrounding the fire perimeter. We used our standardized survey protocol and conducted 3 nocturnal surveys across the landscape with follow-up visits to attempt to located nest/roost locations for birds detected on nocturnal surveys. These methods are described in the section above and fully in protocol described in the study plan. We also contracted to obtain pre- and post-fire vegetation maps to be able to assess changes to the vegetation and CSO habitat for the MACFA, we still need to obtain post-fire vegetation maps for the COCFA.

The high-severity fires that burned in the MACFA resulted in significant changes to the vegetation (Fig. 11 & 12). The amount of suitable CSO habitat (CWHR classes 4M, 4D, 5M, 5D) within the 88,000 acre (35,612 ha) MACFA decreased from 70.1% of the pre-fire landscape to 5.8% of the landscape following the fires. The largest increase in the post-fire landscape occurred in the CWHR classes <= 2D which increased from 8.2% to 64.9%. The remaining forested areas across the post-fire landscape were predominantly classified as either 4P (18.5%) or 4S (7.9%).

We are still in the process of synthesizing all of the pre-fire CSO survey information for the MACFA as there is not a solid baseline of consistently collected survey information prior to the fire such as exists for our core PLS project area. Nevertheless, this synthesis may provide us with a reasonable estimate of the pre-fire distribution and abundance of CSO sites across the MACFA. All or parts of at least 23 PACs were located within the pre-fire MACFA. Given the lack of continuous annual CSO survey effort we are uncertain what proportion of those PACs were occupied in 2007 prior to the fires.

During our 2008 surveys we documented a single confirmed pair of CSOs (non-breeding) within the MACFA, with the female from this pair being the only female we detected within the fire area (Fig. 13). We had 10 single detections of male CSOs across the burned area. In each of these ten cases we were not able to locate the birds at nests or roosts on follow-up status surveys. Each of these ten locations occurred primarily in the middle of the night when birds are out foraging and none of the detections occurred within 1/2-mile (0.8 km) of each other as required to classify these individuals as territorial birds under currently accepted protocols. Within the unburned 1-mile (1.6 km) buffer area surrounding the burned area we documented 5 confirmed pairs, 1 unconfirmed pair, 1 territorial male single, and 6 single detections (4 males, 2 sex unknown). Thus, in the immediate unburned buffer area we observed territorial sites whereas we only were able to document the single confirmed territorial pair within the burned area.

During our 2009 surveys within the MACFA we documented a single confirmed pair of CSOs in the same location as the pair documented in 2008 (Fig. 13). Within the 1-mile (1.6 km) buffer area we documented 7 confirmed pairs, 0 unconfirmed pairs, 2 territorial single males, and 3 single detections. In contrast to 2008, in 2009 we did not record single detections of apparently non-territorial single birds within the fire perimeter across the MACFA landscape. Rather, we only recorded 3 detections of CSOs near the perimeter of the fire in the vicinity of confirmed pairs located within the buffer around the fire.

In our two years of work we were able to document significant changes to the vegetation and amounts and distribution of CSO habitat within the MACFA as a result of the high-severity wildfires. Our CSO survey work suggests that the immediate post-fire landscape may not support territorial CSO sites as evidenced by the single confirmed pair of owls that we documented in 2008 in 2009. In 2009 we did not document single male CSOs across the burned landscape, suggesting that the apparently non-territorial single males observed in 2008 may have been present because of previous site fidelity or were perhaps opportunistically utilizing a flush of prey in the first year following the fire. In both years, territorial CSOs were present in similar numbers and distributed at expected spacing within the buffer area surrounding the fire. Thus, our results from our 2 years of work suggest that the primarily high-severity MACFA does not support CSOs other than a single pair that is using the landscape. Further, territorial CSO sites are well-distributed within the buffer area outside of the fire perimeter. Our 3 detections of individual CSOs just within the perimeter of the burned areas suggest that some CSOs are able to exploit the edge between the burned and unburned areas for foraging.

Our first year of surveys from 2009 in the COCFA documented 3 confirmed pairs, 1 unconfirmed, and 2 territorial single male CSOs, for a total of 6 territorial CSOs sites within the fire perimeter (Fig. 14). Additionally, we had 6 single detections (3 male, 3 unknown sex) of individual CSOs within the fire perimeter. Within the buffer area we documented 3 confirmed pairs and 3 single detections (2 male, 1 female). These results and distribution patterns suggest that CSOs were able to persist in the post-fire COCFA landscape with similar abundance and spacing as has been observed in unburned forests outside the burned areas. A second year of surveys will be conducted in 2010 within the COCFA to confirm these patterns.

It is important to determine both the acute and chronic responses of CSOs and their habitat to wildfire as it is unknown if CSOs can persist over both the short-term and long-term in these areas. Whether a landscape that has experienced wildfire can support CSOs likely depends on the pre-fire habitat suitability and variable fire severity patterns both within individual fires and across different fires. Largely low-moderate severity fires may have positive or neutral effects on CSOs and their habitat while high severity fires may result in greater negative effects. Our results into the acute, short-term response of CSOs to wildfire from the primarily high-severity MACFA and primarily low-moderate severity COCFA support this hypothesis.

Banding, Blood Sampling, West Nile Virus Monitoring

Eighty-two owls were captured and banded in 2009. Blood samples were collected from 17 individuals that will be screened at the University of California, Davis for West Nile Virus (WNV) antibodies. None of the 175 individual blood samples collected from 2004-2008 have tested positive for WNV antibodies. The 2009 samples have not been analyzed to date.

Barred and Sparred (Spotted x Barred hybrid) Distributional Records

We detected the presence of 4 barred owl and 3 sparred owls during 2009 surveys within our PLS study area. Our synthesis and update of barred-sparred owl records through 2009 based on Forest Service and California Department of Fish and Game databases indicates that there are a minimum of 39 individual records across the HFQLG Project Area and a minimum total of 53 across the Sierra Nevada (Figure 15). This includes a minimum total of 19 records that have been documented within our intensively surveyed PLS study area. The first barred owl in the region was reported in 1989. The first documented breeding in the PLS survey area was in 2007. The pattern of records suggests that barred/sparred owls have been increasing in the northern Sierra Nevada between 1989-2009 and are now present in low, stable numbers over the past 5 years on our study area. This pattern is consistent with that observed in other areas as barred owls have expanded their range in western North America. Initially barred owls first colonize an area and persist at low population numbers during which this period they will hybridize with spotted owls. At some threshold population size or when ecological conditions allow they are then poised for, and capable of, exhibiting exponential population growth.

California Spotted Owl Diet

A single diet survey plot was established at a CSO nest or roost location at each CSO territory on the Plumas National Forest during 2003-2007. Systematic searches for pellets and prey remains were conducted in each plot during each year. A total of approximately 3398 pellets have been collected during 2003-2007 (2003 = 606; 2004 = 807; 2005 = 838; 2006 = 516; 2007 = 552). We completed sorting of all pellets and identification of all prey remains in January 2010. All prey items are identified to species, or taxonomic group when species identification could not be ascertained. A total of 8,595 prey items have been recorded from the pellets. Mammals are the dominant taxonomic group and comprise of 96.5% of the total biomass identified in the diet. Across years the highest biomass contributions were from the dusky-footed woodrat (contributed 45% of the estimated total biomass) and northern flying squirrel (10.8%). Our objective has been to sample over several years to assess temporal variation in diets and possible relationships to variation in CSO reproduction, and to sample widely over space in order to investigate potential variation in CSO diets associated with elevation and vegetation conditions. We will now be able to address these questions given completion of the sorting and prey identification from the pellet samples.

Summary 2003-2009

Our efforts from 2003-2007 have focused on collecting the initial data to address our primary research objectives and provide the baseline data for monitoring HFQLG implementation. In conjunction with the now fully integrated Lassen Demographic Study we have collected landscape-scale information on the distribution and abundance of CSOs across approximately 650,000 acres (263,046 ha) of land. Determining the accurate number and distribution of CSO sites requires multiple years of survey and marking of individual CSOs to delineate separate territories and identify individual birds that move among multiple sites within and across years. These baseline data are fundamental for developing empirically based habitat models for understanding CSO habitat associations and developing adaptive management tools and models. The near completion of the Meadow Valley area projects in 2007-2008 marked the first landscape series of HFQLG treatments to be implemented within the study area, providing the first opportunity to address treatment effects within a case study framework. Our baseline information on CSO distribution and habitat associations, coupled with our 2007-2008 radio-telemetry work, will allow us to assess associations between CSOs and vegetation changes. In 2008-2009 we were now able to begin monitoring CSO distribution and abundance in the Meadow Valley project area, providing the first empirical data from a treated landscape.

Additionally we were able to expand our work to address the effects of wildfire and CSOs and their habitat through our 2008-2009 survey work in the Moonlight-Antelope Complex and Cub-Onion Complex fire area. In summary, we are working towards being able to broadly address CSO management questions across a gradient of landscape conditions ranging across untreated landscapes, landscapes treated to meet desired fuels/vegetation conditions, and landscapes that have experienced wildfire in order to address primary management issues.

Dedicated monitoring of CSOs on the Lassen Demographic study continues to provide critically valuable demographic and population trend information for determining the status of CSOs. The declining population trend estimated through the meta-analysis of the Lassen Demographic Study data and the apparent decline in numbers of CSOs observed between 2005-2007 within the Lassen NF survey areas warrant close continued monitoring of the status of CSOs within the study area, along with continued management focus on providing high-quality CSO habitat during the planning and implementation of HFQLG treatments. We lack similar long-term demographic data for the Plumas NF study areas, but our baseline information on CSO distribution and abundance suggests that numbers of territorial CSOs and sites have been similar across 2004-2009.

Our focused diet analyses have broadened and deepened our understanding of CSO diets and sources of variation in CSO diets among pairs and across environmental gradients. Monitoring of WNV exposure coupled with demographic monitoring has provided an opportunity to assess if WNV may ultimately be a factor influencing CSO viability. To date we have not had a positive detection for WNV within CSOs. Finally, through our research into historical and current occurrence records, in conjunction with our field surveys, we have been able to document the colonization of the northern Sierra Nevada by barred owls. Our results indicate that barred owls are increasing in the northern Sierra Nevada and may become an increasing risk factor to CSOs.

Current Research: 2010

In 2010 we will continue monitoring owl distribution, abundance, demography, and population trend across the core PLS study area. We will also conduct our second year of CSO surveys to document distribution, abundance and habitat associations within the Moonlight and Antelope Complex fire area. We will augment our existing work with two additions. First, we will conduct CSO surveys in Empire (Plumas NF) and Scotts John Creek (Lassen NF) proposed project areas. This new work, coupled with our ongoing work in the Meadow Valley (Plumas NF) and Creeks (Lassen NF) will provide the baseline data for four of the first project areas that are scheduled for implementation and position us to assess effects to CSOs and their habitat as these first projects are implemented on the ground. Together this work will provide a more comprehensive base of knowledge regarding CSO habitat associations and the effects of treatments and wildfires.

In addition to continuing field surveys in 2010 designed to address our six research questions, we have broadened our emphasis on the development of predictive habitat relationship models as described in the module study plan. We have continued to work closely with biologists on the Plumas and Lassen National Forests, and the R5 Regional

Office, to identify and define the types of analyses and tools that would best address management needs. Baseline information collected during this study forms the foundation for this phase of the research. The combination of broad-scale landscape CSO distribution data, in conjunction with detailed demographic information available from the Lassen Demographic Study, will facilitate exploration and development of predictive habitat models for use in an adaptive management framework and to directly monitor implementation of the HFQLG project.

Literature Cited

Blakesley, J.A., M.E. Seamans, M.M. Connor, A.B. Franklin, G.C. White, R.J. Gutierrez, J.E. Hines, J.D. Nichols, T.E. Munton, D.W.H. Shaw, J.J. Keane, G.N. Steger, B.R. Noon, T.L. McDonald, S. Britting. 2006. Demography of the California Spotted Owl in the Sierra Nevada: Report to the US Fish and Wildlife Service on the January 2006 Meta-Analysis. February 2006

Table 1. California spotted owl reproduction on the Plumas and Lassen National Forests 2004-2009.

Year	**Percent of confirmed/unconfirmed pairs with successful nests**	**Young fledged per successful nest**
2004	49.4%	1.68
2005	17.7%	1.47
2006	13.8%	1.50
2007	55.4%	1.81
2008	16.4%	1.70
2009	47.6%	1.57

Table 2. Crude density of territorial California spotted owls across survey areas on the Plumas and Lassen National Forests 2004-2009. Locations of survey areas are identified in Figure 1.

		Crude Density of Territorial Owls (#/km^2)					
Survey Area	**Size (km^2)**	**2004***	**2005***	**2006***	**2007***	**2008***	**2009***
SA-2	182.4	0.126	0.143	0.115	0.115	0.132	0.121
SA-3	214.4	0.075	0.093	0.089	0.103	0.098	0.089
SA-4	238.2	0.059	0.050	0.046	0.071	0.046	0.046
SA-5	260.2	0.069	0.069	NS****	NS****	NS****	NS****
SA-7	210.3	0.071	0.062	NS	NS	NS	NS
SA-1A	190.4	NI***	0.042	0.042	0.053	0.042	0.058
SA-1B**	130.3	NI	0.023	NS	NS	NS	NS
SA-11	179.4	NI	0.045	0.033	0.033	0.045	0.33
SA-12	215.8	NI	0.097	0.070	0.074	0.070	0.74
SA-13	152.9	NI	0.105	0.085	0.065	0.050*****	0.99
SA-14	318.7	NI	0.053	0.044	0.035	0.047	0.44
SA-15	196.8	NI	0.086	0.036	0.056	0.081	0.76
Total Study Area	2489.8	0.078	0.073	0.060	0.066	0.067	0.69

*Total Area surveyed each year: 2004 = 1,106 km^2; 2005 = 2,490 km^2; 2006 = 1,889 km^2; 2007 = 1,889 km^2; 2008 = 1,877 km^2; 2009 = 1,889 km^2

**NI = not included. Project level area surveyed only in 2005. Included for comparative purposes.

***Lassen Demographic Study Area – incorporated into the overall study in 2005.

****Survey areas not surveyed in 2006-2008.

*****This survey area was not completely surveyed during 2008 because of wildfire activity in the area. Two CSO territories within the study area could not be surveyed.

Table 3. Number of pairs (confirmed and unconfirmed) and territorial single California spotted owls across the Plumas-Lassen Study survey areas on the Plumas and Lassen National Forests, California, 2004-2009.

	2004	**2005**	**2006**	**2007**	**2008**	**2009**
Survey Area	**Pairs/ TS***	**Pairs/TS***	**Pairs/TS***	**Pairs/TS***	**Pairs/TS***	**Pairs/TS***
SA-2	11/1	12/2	10/1	10/1	12/0	11/0
SA-3	7/2	10/0	9/1	11/0	9/3	9/1
SA-4	7/0	5/2	4/3	8/1	5/1	5/1
SA-5	8/2	9/0	NS****	NS****	NS****	NS****
SA-7	7/1	6/1	NS	NS	NS	NS
SA-1A	NI***	4/0	4/0	5/0	4/0	5/1
SA-1B**	NI	3/0	NS	NS	NS	NS
SA-11	NI	4/0	3/0	3/0	3/2	3/0
SA-12	NI	10/1	1/7	8/0	7/1	8/0
SA-13	NI	8/0	6/1	5/0	3*****/1	7/0
SA-14	NI	8/1	7/0	5/1	7/1	7/0
SA-15	NI	8/1	3/1	4/3	8/0	7/1

*TS = Territorial Single.

**NI = not included. Project level area surveyed only in 2005. Included for comparative purposes.

***Lassen Demographic Study Area – incorporated into the overall study in 2005.

****Survey areas not surveyed in 2006-2009.

***** This survey area was not completely surveyed during 2008 because of wildfire activity in the area. Two CSO territories within the study area could not be surveyed.

Table 4. Mean estimated population lambda (population change) for California spotted owls on four study areas in the southern cascades and Sierra Nevada, 1990-2005 (Blakesley et al. 2006)

Study Area	**Lambda**	**Standard Error**	**95% Confidence Interval**
Lassen National Forest	0.973	0.014	0.946-1.001
Sierra National Forest	0.992	0.013	0.966-1.018
Sequoia-King Canyon National Park	1.006	0.031	0.947-1.068
Eldorado National Forest	1.007	0.029	0.952-1.066

Table 5. Distribution of California spotted owl nest/primary roost sites (n = 103) across CWHR tree size classes within the Plumas-Lassen Study on the Plumas and Lassen National Forests, 2004-2006.

CWHR Size Class*	CWHR Size Class Description	Number of Nests	Percent
Barren	Open, sparse tree coverage	1	1.0
3S	6-12 inch dbh, ,20% CC	1	1.0
3M-LT	6-12 inch dbh, 40-60% CC, large trees recorded	1	1.0
3D	6-12 inch dbh, >60% CC	4	3.9
4P	12-24 inch dbh, 20-40% CC	3	2.9
4M	12-24 inch dbh, 40-60% CC	3	2.9
4M-LT	12-24 inch dbh, 40-60% CC, large trees recorded	12	11.7
4D	12-24 inch dbh, >60% CC	10	9.7
4D-LT	12-24 inch dbh, >60% CC, large trees recorded	13	12.6
5M	>24 inch dbh, 40-60% CC	25	24.3
5D	>24 inch dbh, >60% CC	9	8.7
6	>24 inch dbh, >60% CC, multi-layer canopy	21	20.1

*defined by average tree size (dbh = diameter at breast-height) and average percent canopy cover (CC).

Table 6. Nest-site (1 ha (2.47 acres)) habitat characteristics collected using the Forest Inventory and Analysis sampling protocol at California spotted owl nest sites (n = 80) on the Plumas and Lassen National Forests, California, 2005-2006.

Variable	**Mean**	**SE**
Total Basal Area (ft^2/acre)	260.8	6.47
# Trees >= 30 inch dbh (#/acre)	10.7	0.58
Basal Area Trees >= 30 inch dbh (ft^2/acre)	96.0	5.70
# Trees >= 24 inch dbh (#/acre)	19.9	0.90
Basal Area Trees >= 24 inch dbh (ft^2/acre)	131.7	6.29
# Trees <12 inch dbh (#/acre)	383.5	26.36
Basal Area Trees , <12 inch dbh (ft^2/acre)	50.1	2.71
# Snags >=15 inch dbh (#/acre)	7.4	0.80
Mean Duff Depth (inches)	3.0	0.16
Duff (tons/acre)	67.4	3.64
Mean Litter Depth (inches)	2.3	0.18
Litter (tons/acre)	23.7	1.81
1 Hour Fuels (tons/acre)	0.75	0.03
10 Hour Fuels (tons/acre)	4.0	0.21
100 Hour Fuels (tons/acre)	4.4	0.28
Shrub Cover (%)	7.7	1.16
Canopy Cover (%)*	64.1	1.24

* estimated through Forest Vegetation Simulator modeling of plot-based tree lists.

Table 7. Distribution of USDA Region 5 vegetation classes (Mean (SE)) within 500 acre (201 ha) circles centered on California spotted owl (CSO) territories (n = 102) and systematic grid (Grid) points (n = 130) within the Plumas-Lassen Study on the Plumas and Lassen National Forests, 2004-2006.

R5 Size Class*	R5 Size Class Description	CSO	Grid
Non-forest	Sum of non-forest land types	4.4 (1.0)	8.4 (1.2)
Total Size 1	Sum of 1G,1N, 1P, 1S: <6 inch dbh, all %CC classes	1.7 (0.3)	1.6 (0.3)
2P & 2S	6-12 inch dbh, 10-39% CC	3.4 (0.4)	4.1 (0.5)
2N	6-12 inch dbh, 40-69% CC	3.8 (0.6)	4.4 (0.9)
2G	6-12-24 inch dbh, >=70% CC	1.6 (0.5)	0.5 (0.1)
3P&3S	12-24 inch dbh, >10-39% CC	9.2 (0.8)	16.1 (1.3)
3N	12-24 inch dbh, 40-69% CC	37.2 (2.4)	38.5 (1.8)
3G	12-24 inch dbh, >=70% CC	6.2 (1.0)	3.8 (0.7)
4P&4S	>24 inch dbh, >10-39% CC	1.0 (0.3)	2.1 (0.4)
4N	>24 inch dbh, 40-69% CC	25.8 (2.0)	17.3 (1.6)
4G	>24 inch dbh, >=70% CC	6.5 (0.1)	2.4 (0.8)
Total 4N & 4G	Sum of 4N & 4G: >24 inch dbh, >= 40% CC	32.4 (2.3)	19.6 (1.8)
Total Suitable habitat	Sum of classes 3N, 3G, 4N, 4G = >12 inch dbh, >40% CC	75.7 (2.19)	61.9 (1.75)

*defined by average tree size (dbh = diameter at breast-height) and average percent canopy cover (CC).

Table 8. Annual number of California spotted owls documented during the breeding period (April-August) in SA-4 (Meadow Valley Project Area) between 2004-2009 on the Plumas National Forest, California..

Year	Confirmed Pairs	Unconfirmed Pairs	Territorial Singles	Total Territorial Sites
2003	7	0	1	8
2004	7	0	0	7
2005	4	1	2	7
2006	3	1	3	7
2007	8	0	1	9
2008	5	0	1	6
2009	5	0	1	6

Figure 1. (A) Location of CSO Survey Areas surveyed in 2004-2009. (B) Example of original survey plot consisting of multiple Cal-Planning watersheds. (C) Example of Primary Sampling Units for surveying for CSOs. See text and study plan for further details .

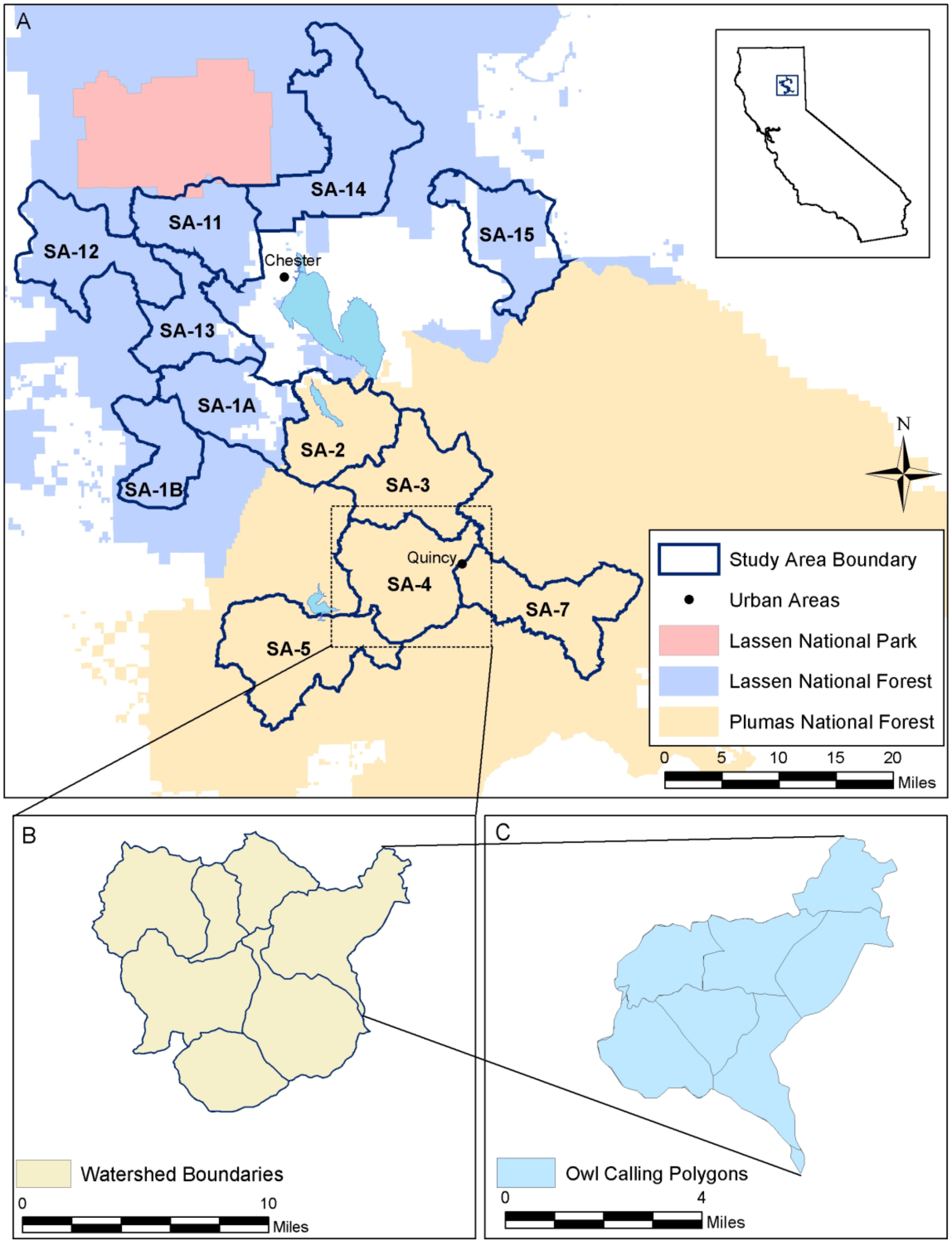


Figure 2. Distribution of California spotted owl territories within CSO survey plots across the Plumas and Lassen National Forests, 2009.

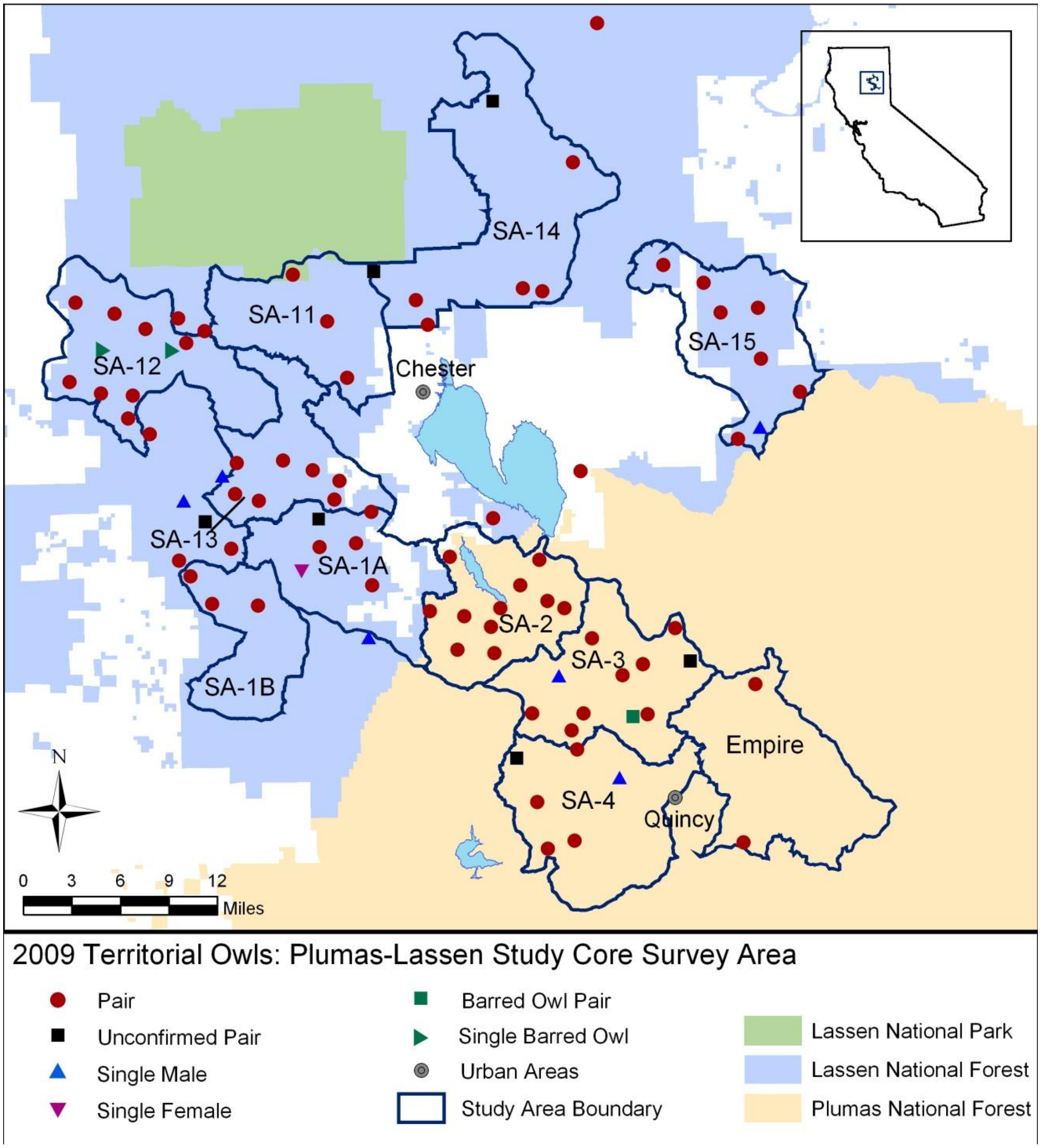


Figure 3. Monthly precipitation totals for Quincy, California, during January-May, 2004-2009 (data from Western Regional Climate Center).

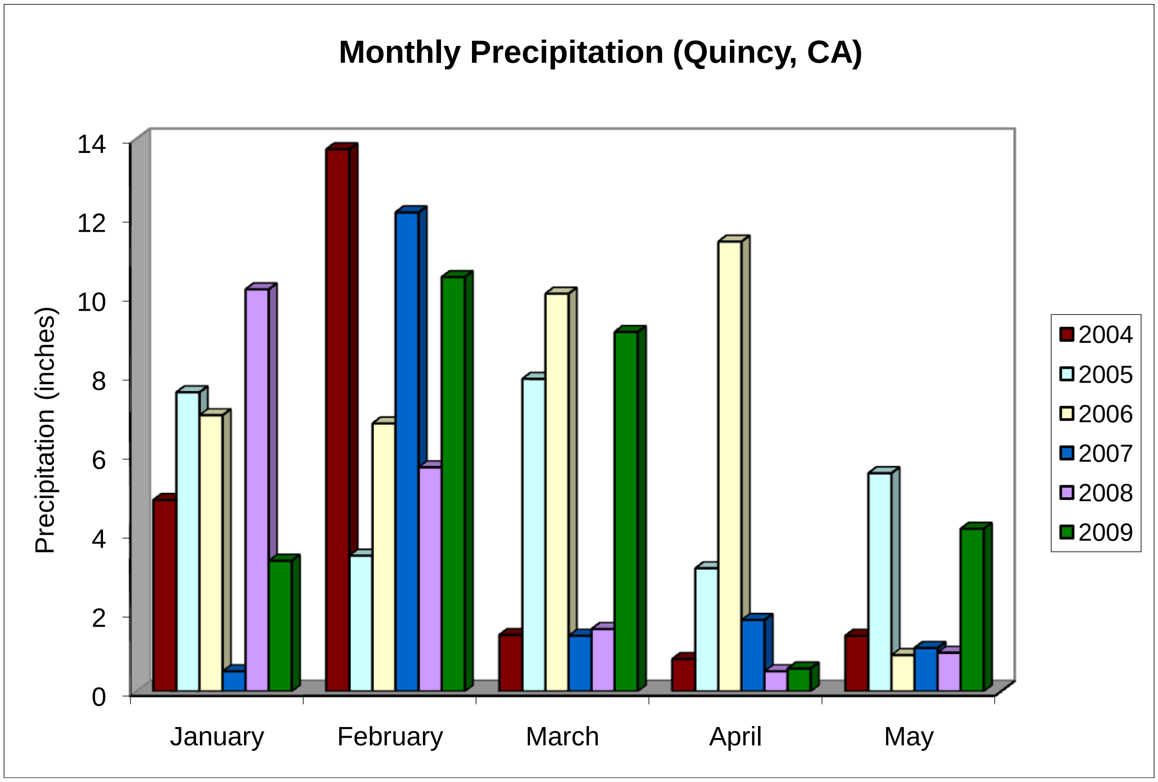


Figure 4. Distribution of California spotted owl (n = 103) nest sites by California Wildlife Habitat Relationship (CWHR) database vegetation classes on the Plumas and Lassen national Forests, California, 2004-2007. Descriptions of the CWHR classes are provided in Table 5 within the text of this document.

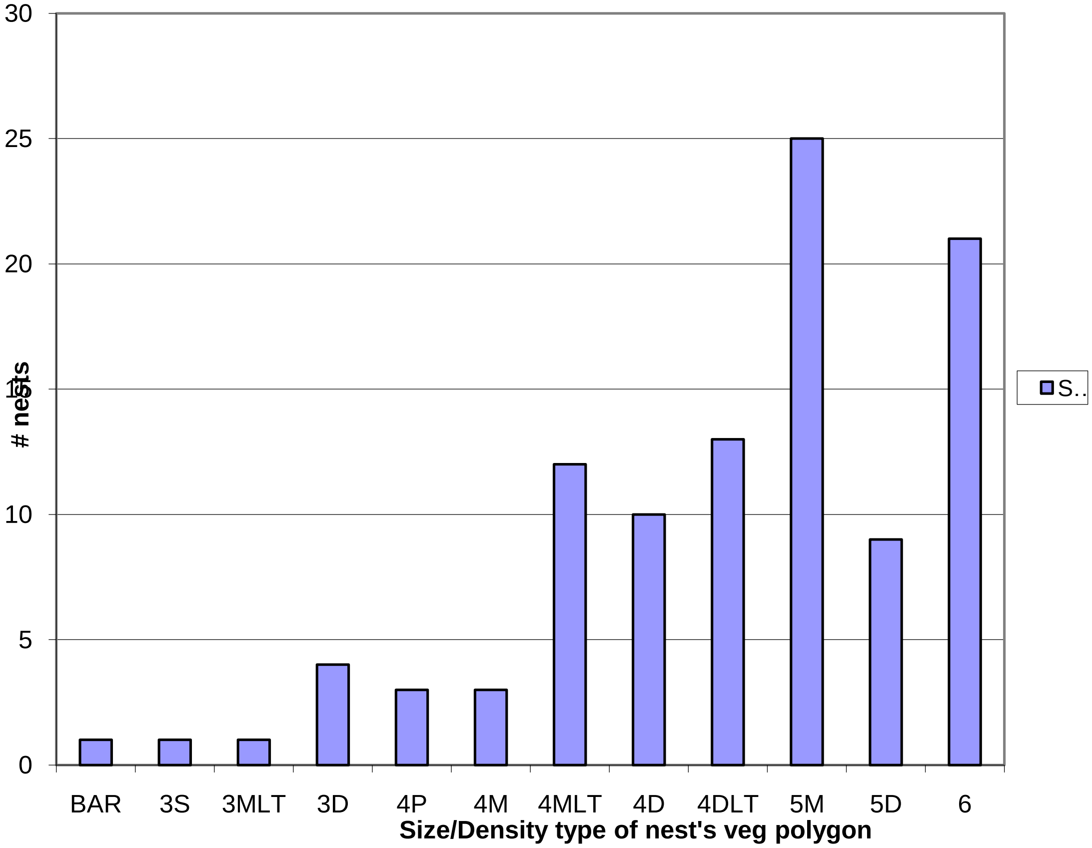


Figure 5. Percent suitable habitat (>=12 inch dbh trees with >=40% canopy cover) within 500 acre (201 ha) circles centered on California spotted owl (CSO, n = 102) and systematic grid points (Grid, n = 130) on the Plumas and Lassen National Forests, California, 2004-2007.

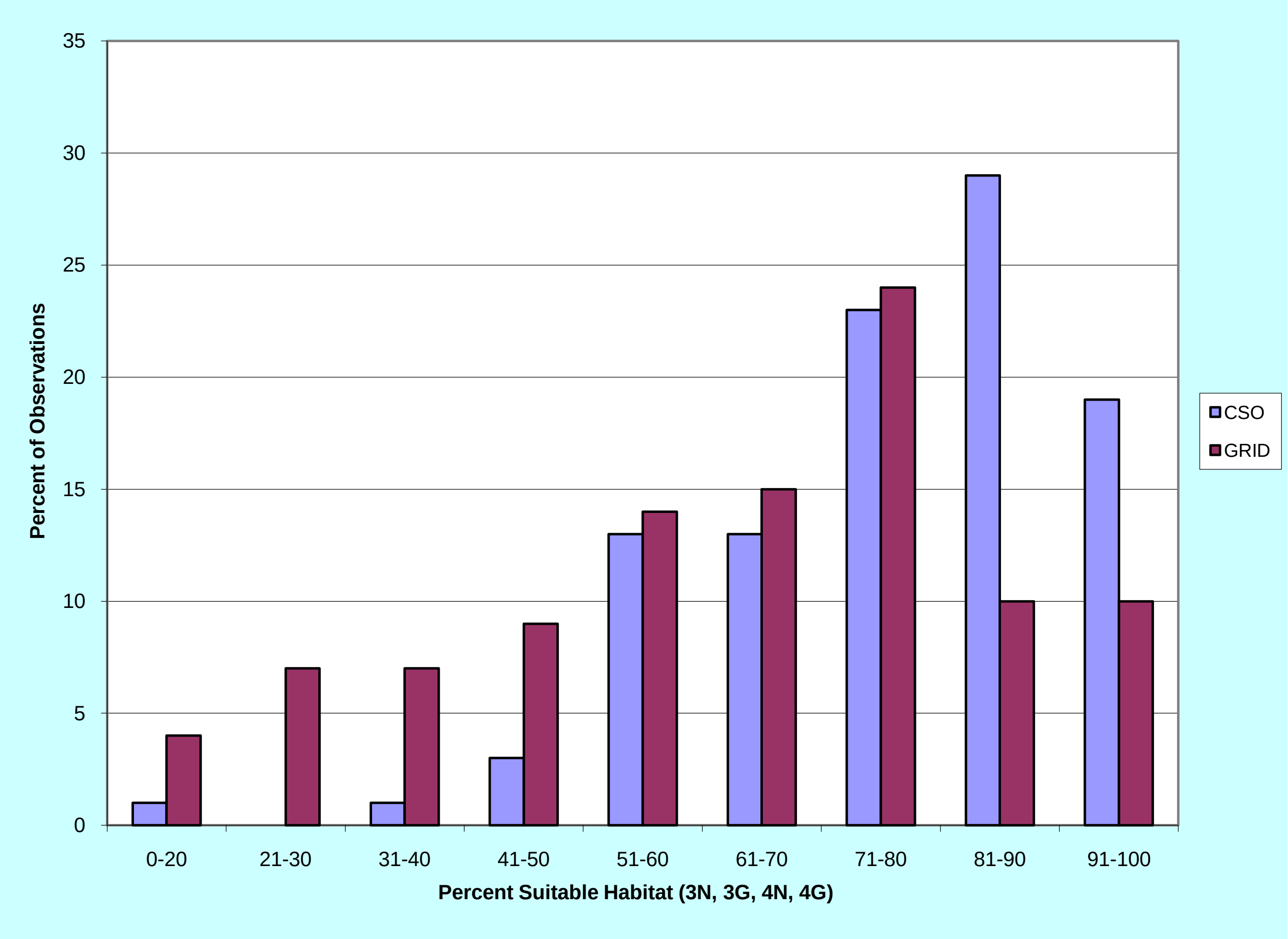


Figure 6. Percent large tree habitat (R5 classes 4N &4G: >=24 inch dbh trees with >=40% canopy cover) within 500 acre (201 ha) circles centered on California spotted owl (CSO, n = 102) and systematic grid points (Grid, n = 130) on the Plumas and Lassen National Forests, California, 2004-2007. Descriptions of R5 classes are provided in Table 7 within the text of this document.

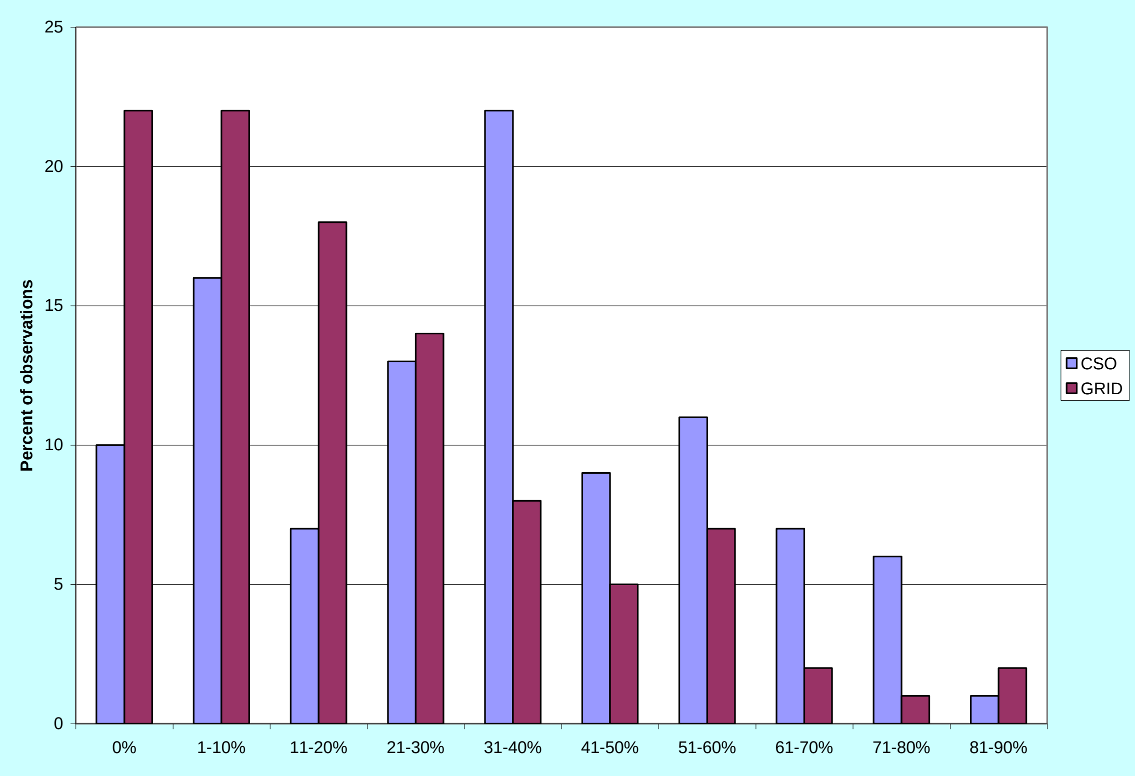


Figure 7. Distribution of proposed Meadow Valley Project Area forest management treatments and cumulative distribution of California spotted owl territorial sites between 2003-2009 in Survey Area-4 of the Plumas-Lassen Study, Plumas National Forest, California.

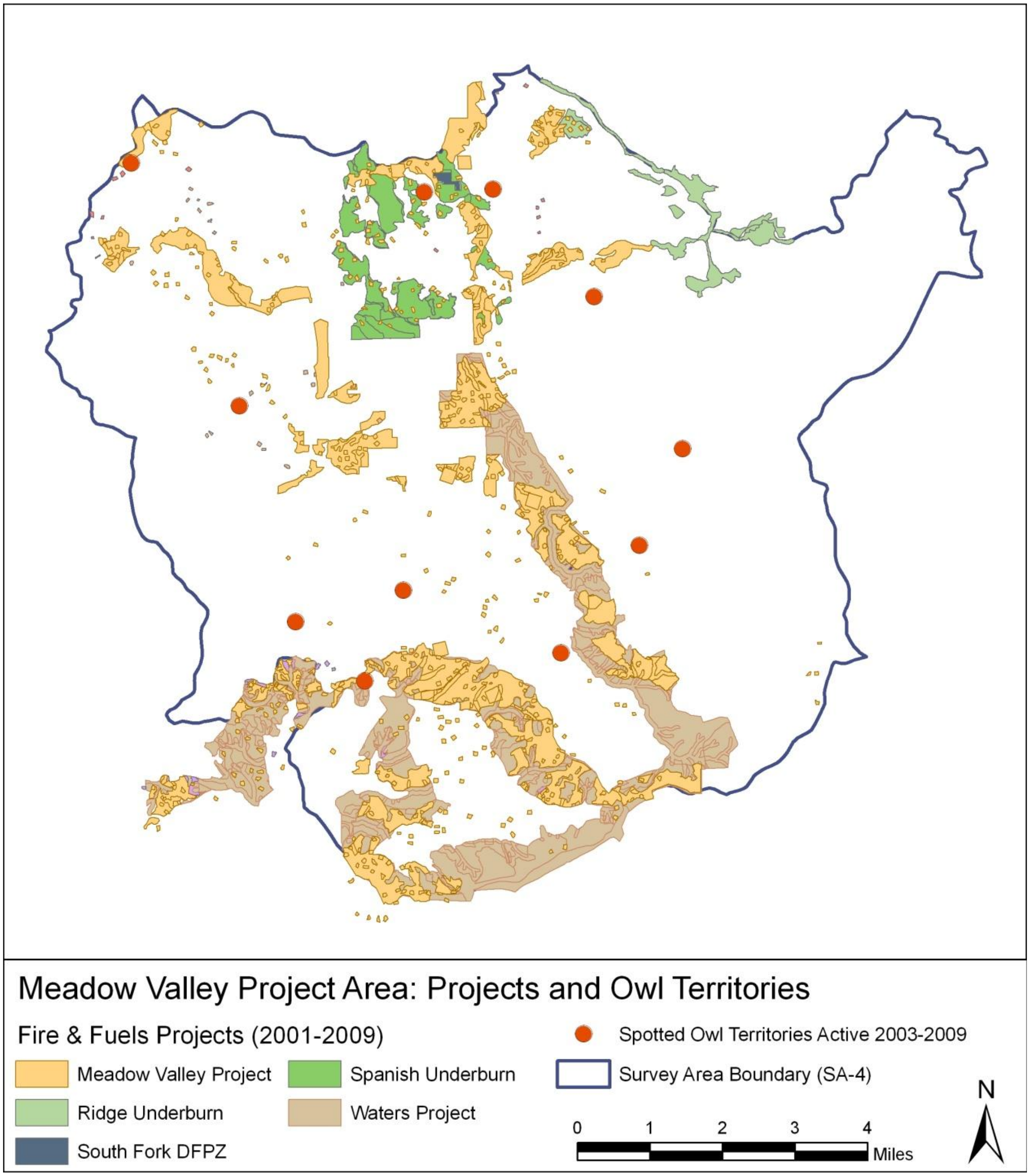


Figure 8a. Annual summary distribution of California spotted owl territorial sites between 2003-2005 across Survey Area-4 (Meadow Valley Project Area) of the Plumas-Lassen Study, Plumas National Forest, California.

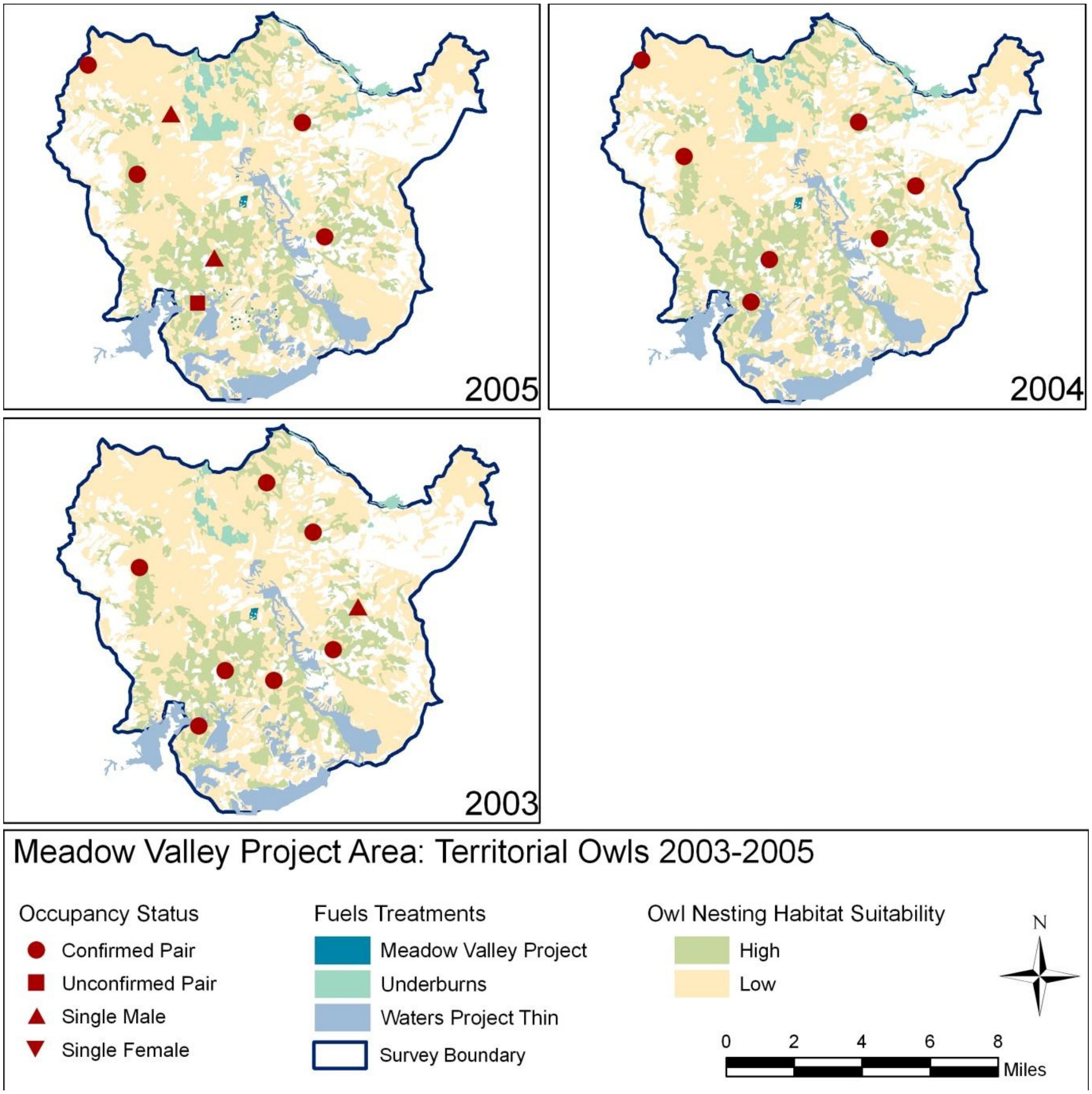


Figure 8b. Annual summary distribution of California spotted owl territorial sites between 2006-2009 across Survey Area-4 (Meadow Valley Project Area) of the Plumas-Lassen Study, Plumas National Forest, California.

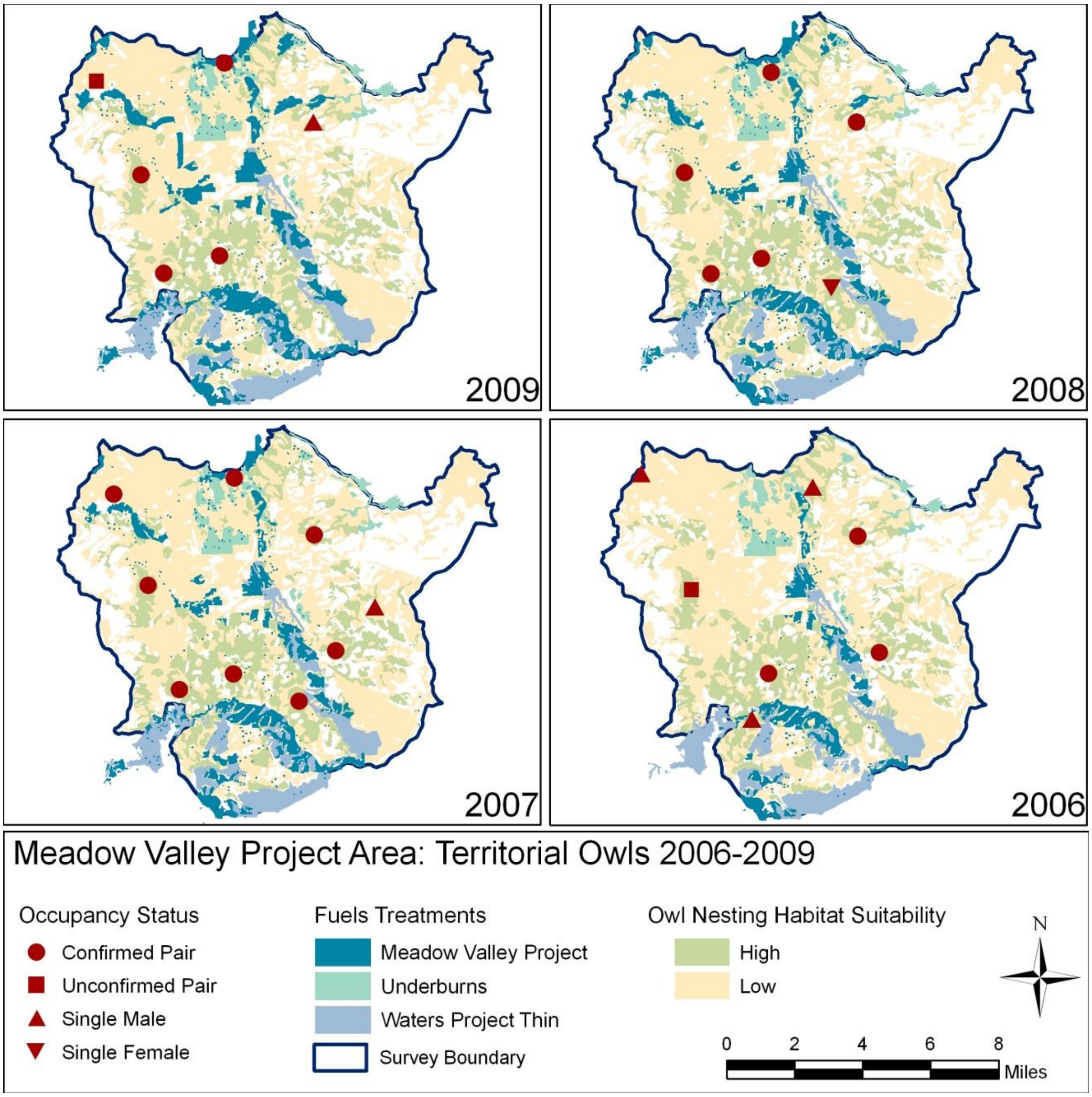
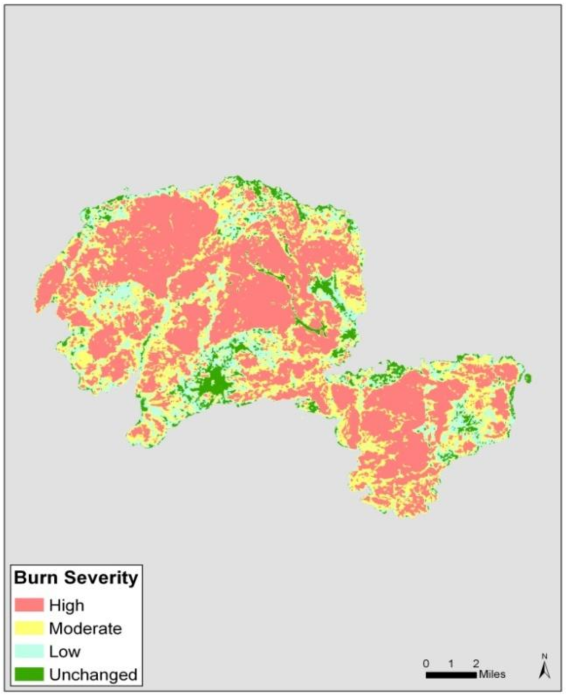


Figure 9. Maps of fire severity in the: (a) Moonlight-Antelope Complex fire (88,000 acres) that burned in 2007; and (b) the Cub-Onion Complex fire (21,000 acres) that burned in 2008 on the Plumas and Lassen National Forests, California.

(a)



(b)

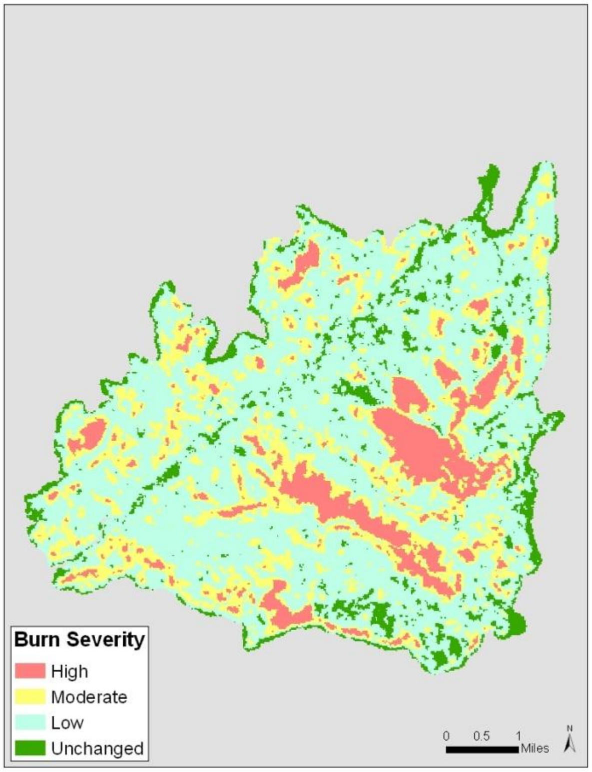


Figure 10. Distribution of the post-fire landscape by fire severity class for the Moonlight-Antelope Complex Fire Area (MACFA) and Cub-Onion Complex Fire Area (COCFA) on the Plumas and Lassen National Forests, California.

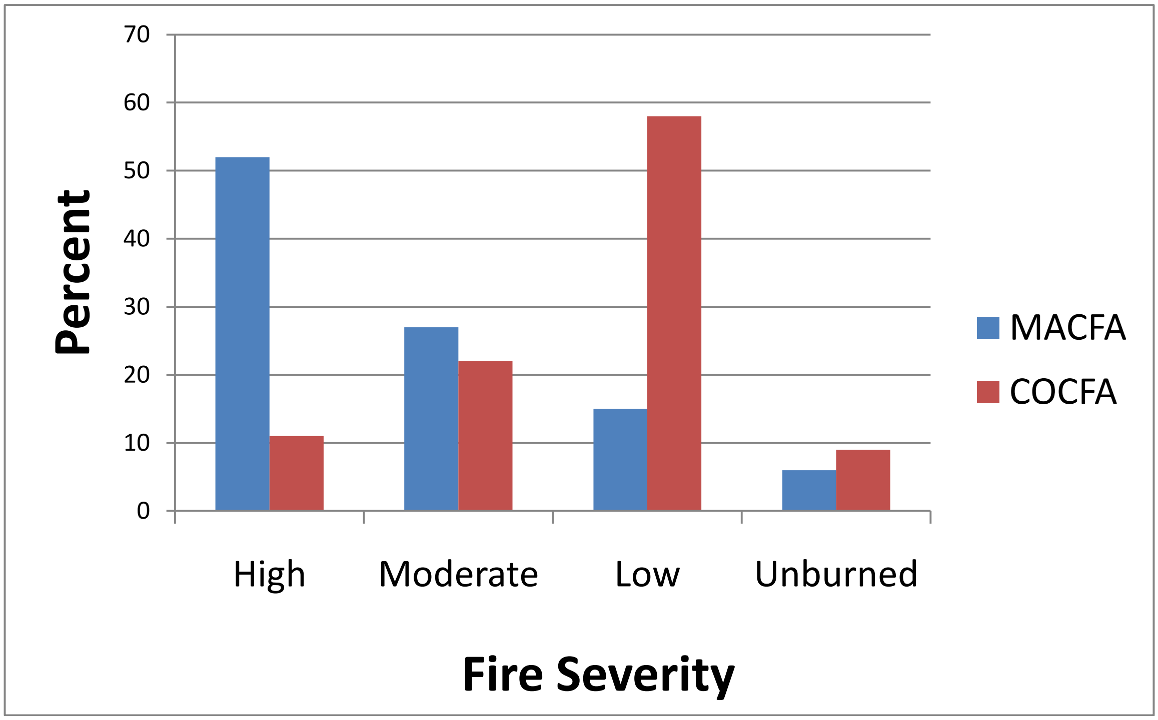


Figure 11. Distribution of pre- and post-fire California Wildlife Habitat Relationship vegetation classes within the Moonlight-Antelope Complex fire areas 2008 on the Plumas and Lassen National Forests, California.

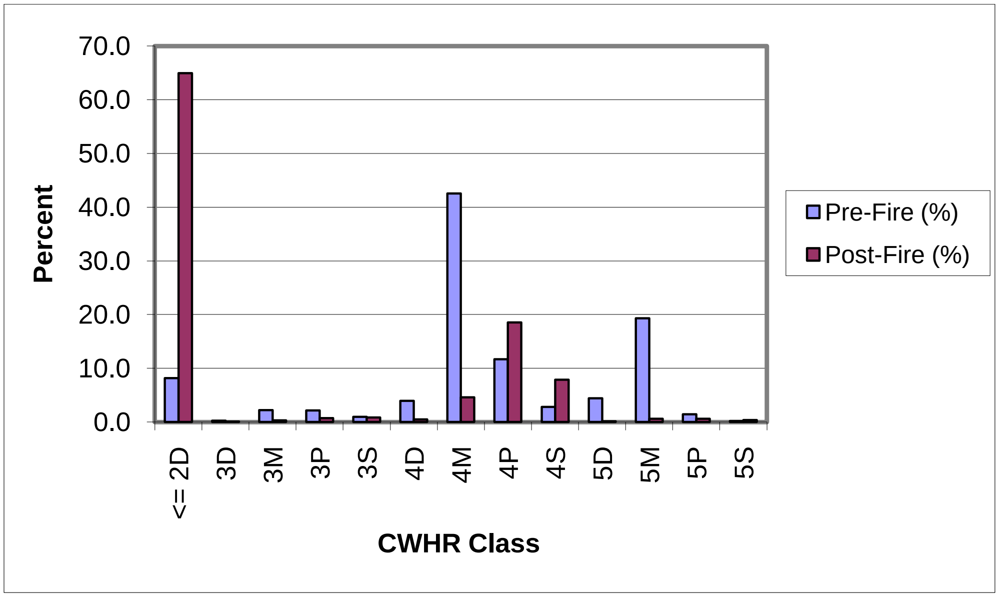
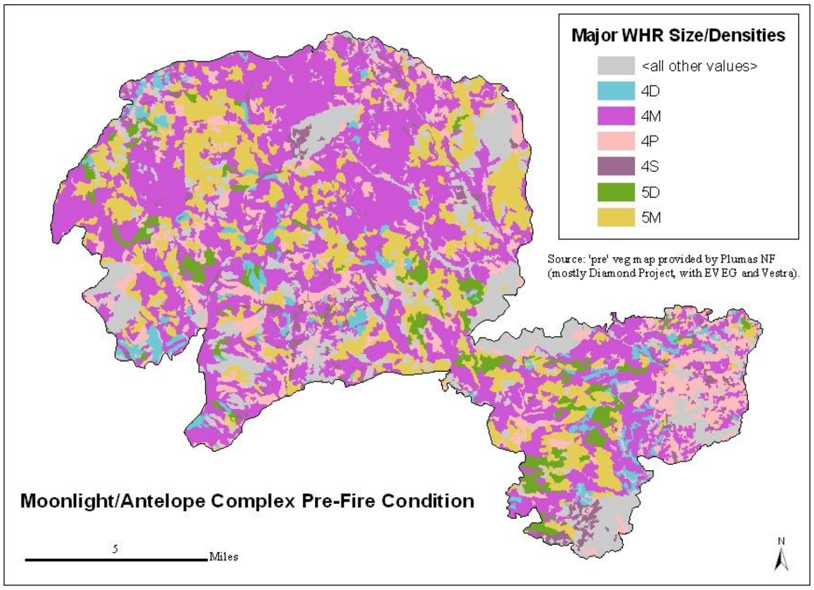


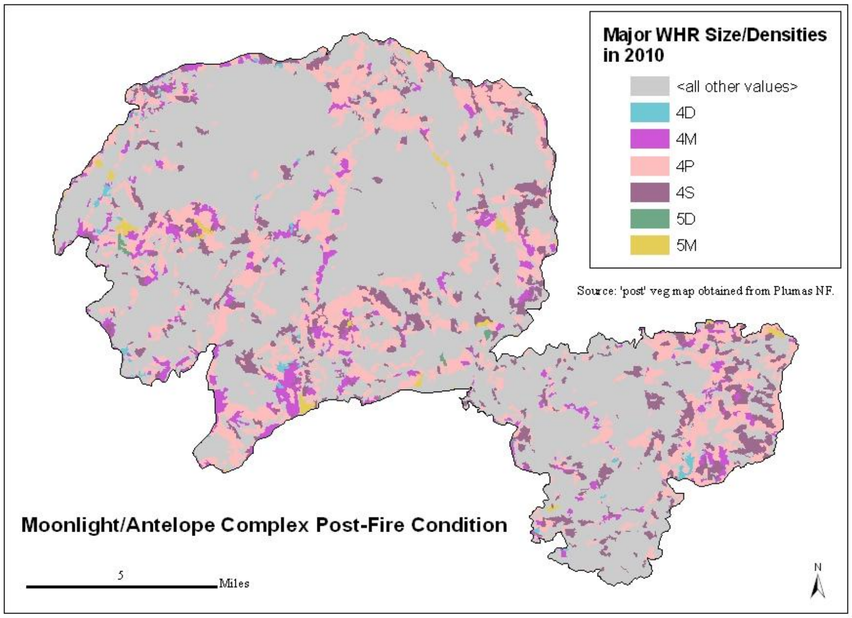
Figure 12. Maps of (a) pre-fire and (b) post-fire California Wildlife Habitat Relationship vegetation classes within the Moonlight-Antelope Complex fire areas 2008 on the Plumas and Lassen National Forests, California.

(a)



Source: 'pre' veg map provided by Plumas NF (mostly Diamond Project, with EVEG and Vestra).

(b)



Source: 'post' veg map obtained from Plumas NF.

Figure 13. Distribution of California spotted owls within the Moonlight-Antelope Complex fire area and a 1.6 km buffer during 2008 and 2009 on the Plumas and Lassen National Forests, California.

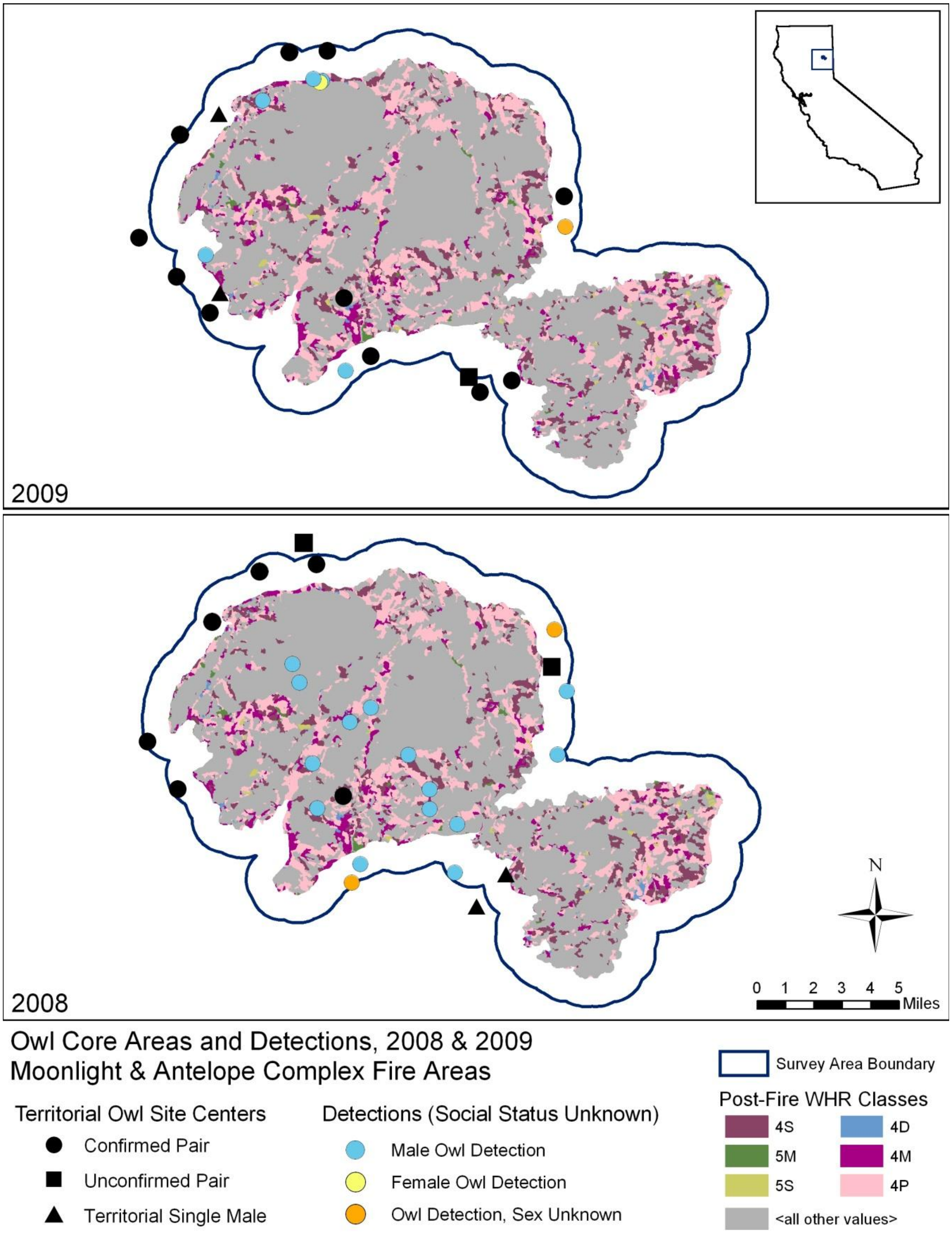


Figure 14. Distribution of California spotted owls detected in 2009 within the Cub-Onion Complex Fire Area and a 1.6 km buffer and wildfire burn severity classes on the Lassen National Forest, California.

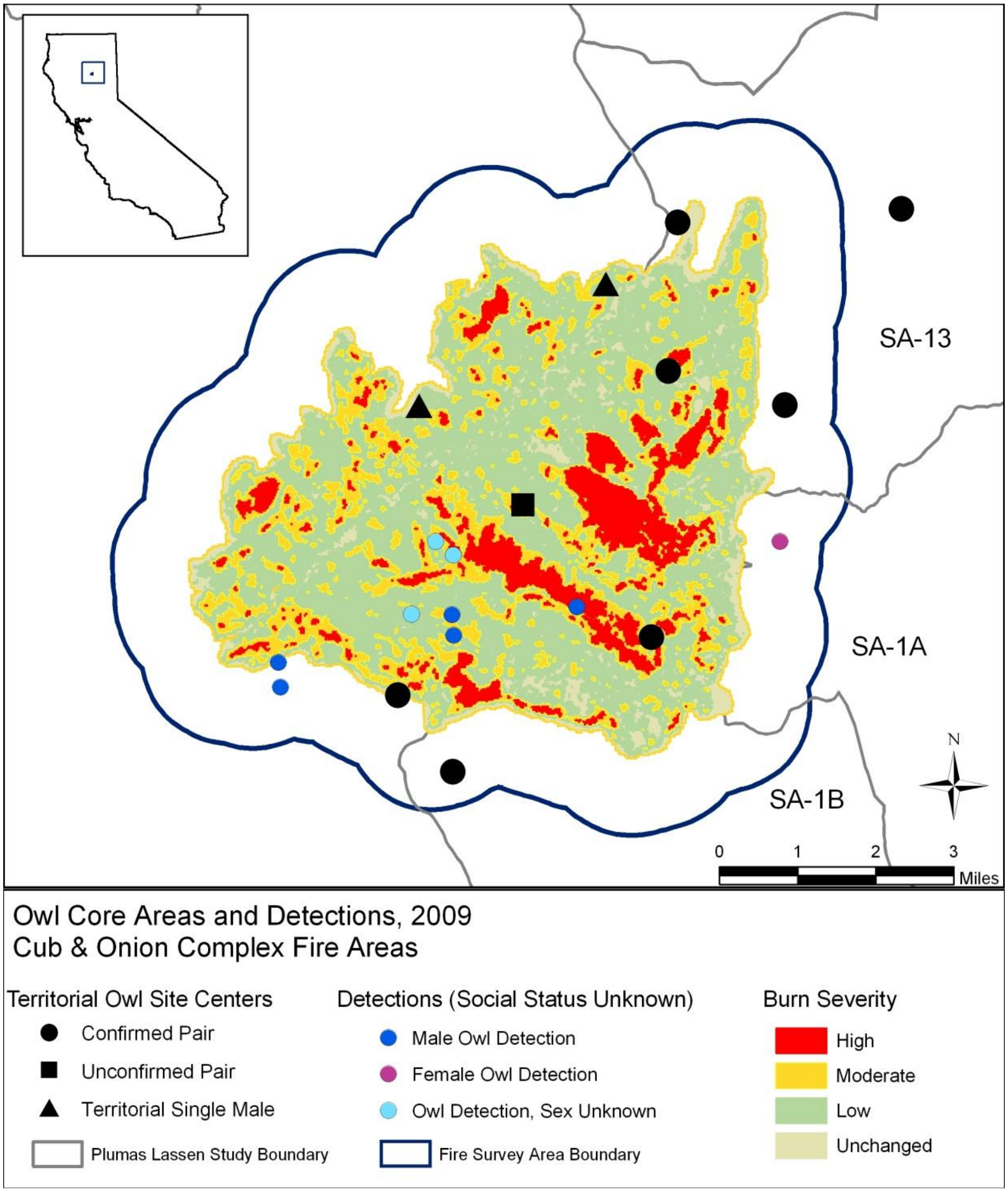
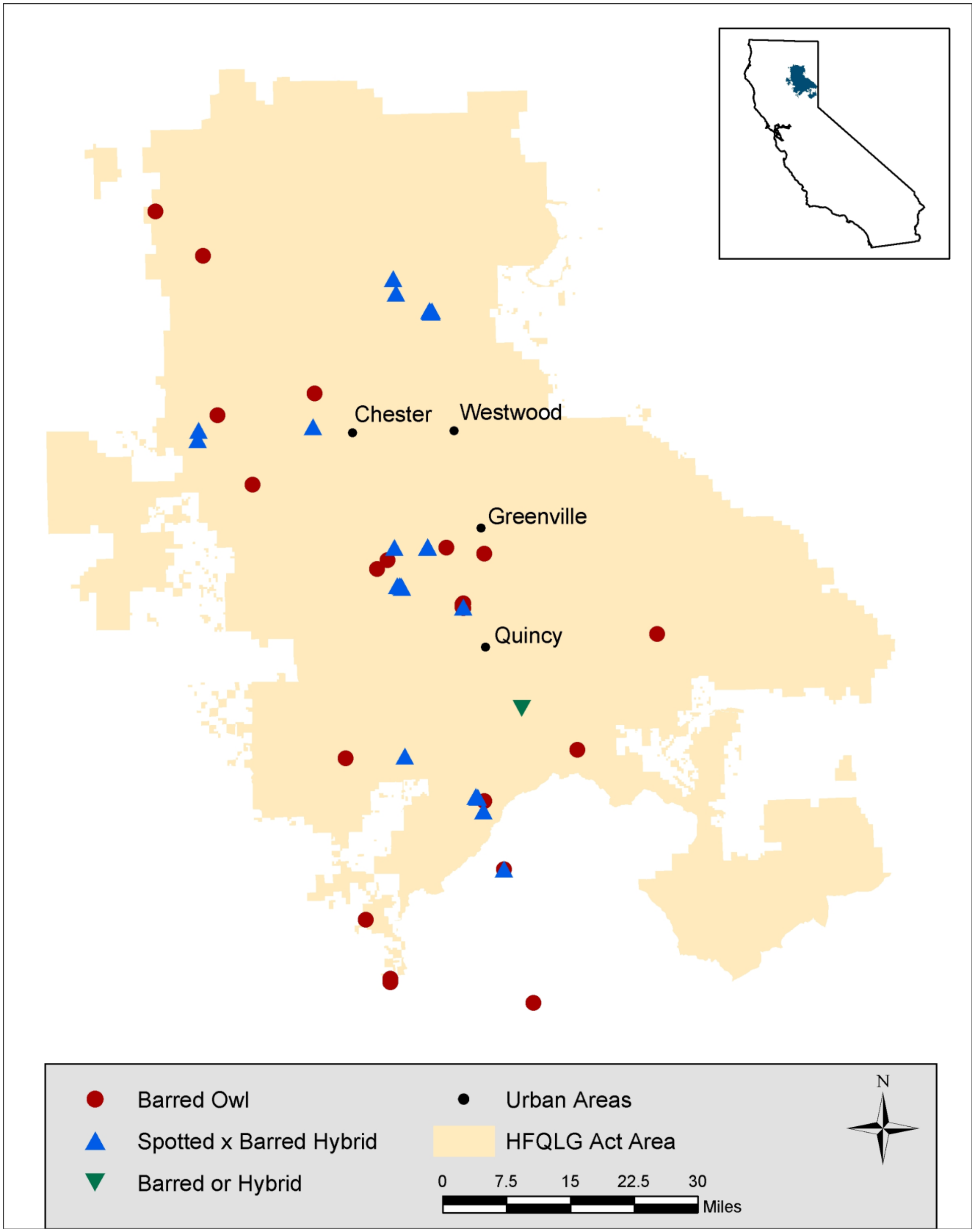


Figure 15. Distribution of Barred and Sparred (Spotted-Barred hybrids) Owls between 1989-2009 within the HFQLG Project area.



Plumas-Lassen Administrative Study:

Implication of Research Findings for Ecological Restoration

Introduction:

Nexus with the Storrie Fire and Restoration of a Severely Burned Landscape

The five modules of the Plumas Lassen Administrative Study (PLAS) have been dedicated to understanding the relationship between forest management strategies and an array of elements of the forest (response variables) with which managers and the public are concerned. Spotted owls and their habitat are a major management priority and often the fulcrum of legal challenges to proposed forest management projects. When we began the PLAS program we decided to take a landscape view of this problem and address a number of ecological issues that revolve around the conflict between managing forest to promote resilience and sustaining spotted owl populations. The specific elements we decided to examine were: 1) spotted owl populations, 2) small mammals (prey items of spotted owls) 3) terrestrial bird (both for biodiversity assessment and as prey for spotted owls, 4) vegetation dynamics (including habitat structure), and fire, the most profound threat to the forest in general. Thus the five modules of the PLAS program were established to provide an integrated understanding of forest ecology and responses to diverse management practices.

Principal among the several purposes of forest management in this area is reducing fuels to prevent large, severe fires. Large fires, like the Storrie fire of 2000, burn uncharacteristically large expanses of forests at high severity. The resulting landscape has a host of disrupted forest functions that persist for years to decades, especially with respect to habitat for species that require dense forest and large trees. With this purpose in mind managers are challenged with how to manage forests to minimize the likelihood of such a fire while simultaneously preventing and/or ameliorating any significant effects on the very resources they are attempting to conserve.

The PLAS program has a two-fold purpose that explicitly intersects with the objectives of the Storrie Fire restoration program. First, PLAS research addresses the question of "how do managers manage landscapes to restore the resiliency of the forest to the inevitable wildfire while preventing any significant short-term effects on sensitive natural resources such as spotted owl populations? " Restoration is expensive, slow, and long term success is fraught with imprecision and uncertainty. Far more desirable is the prevention of such occurrences. This includes, in part, management of landscapes so that they can withstand wildfires when they occur. PLAS research examines numerous facets of this challenge for land managers and is providing data on consequences of different strategies.

The second line of PLAS research addresses both learning what the response is of assorted biological/physical elements to fires of various intensities and the efficacy of various remedial actions. Once a fire has occurred managers have to weigh a range of restoration options. The goal of managers is restoring the landscape and the ecological balance, but the appropriate response depends on conditions of the fire and many other ecological factors. PLAS research is looking at response of vegetation, and several different groups of wildlife to varying intensities of wildfire.

To adequately address these objectives it is important to be able to begin gathering data within the first year after a fire has occurred. Although wildlife and vegetation response within the post-fire recovery period progresses through a sequence of phases, the initial trajectory is often set by immediate post-burn conditions. In particular, restoration ecology research has found that legacy structures (i.e., the number, size, and distribution of live trees, snags, and large downed logs) strongly influence forest seral development and habitat conditions for decades.

Understanding the consequences of varying degrees of fire severity requires knowledge from all phases of the post fire period. Given that the Storrie Fire occurred 10 years ago it is impossible to examine the early phases of post-fire recovery on that fire. However we can examine early post-fire response on other fires that have occurred in the region more recently such as the Moonlight fire, Antelope complex, and the Cub complex, all of which occurred within the last two to three years. By examining different locations immediately after the fire has occurred rather than go into a ten year old fire (i.e. substituting space for time) we have gained valuable insight into what happens to the wildlife and vegetation in the early years of post-fire recovery. Investigating the Storrie Fire directly can then augment these studies of early post-fire responses to gain an understanding of post-fire development/recovery over time. This knowledge can, in turn, be applied to management and recovery strategies that have relevance to the Storrie Fire as well as many other burned landscapes. Our research objectives with this line of research are to collect data and understand post fire response over time and also across levels of fire severity. This information can provide valuable insight into restoration strategies for all fires on the west slopes of the northern Sierra.

The following provides a brief summary of what each of the five PLAS modules is learning related to restoration of landscapes. When we refer to restoration we mean that this applies to both landscapes that, due to decades of fire suppression and other land management practices, are vulnerable to large, high intensity fires or landscapes that have been significantly altered due to high severity fires. We will continue to report our results with a special section devoted to how our findings inform management interests related to restoration.

Plumas-Lassen Study Terrestrial Bird Module: Restoration Implications

PRBO Conservation Science

The primary objective of the landbird module of the Plumas-Lassen Administrative Study is to assess the impact of forest management practices in sustaining a long-term ecologically stable forest ecosystem at the local and landscape scales by using a suite of landbirds as ecological indicators. We know the avian community in the Sierra Nevada is comprised of species that are associated with a wide range of forest seral stages, vegetative composition, and structures. This vegetation, and hence avian diversity, is constantly changing as a result of natural disturbances (primarily fire) that create a dynamic and diverse ecosystem. Therefore, it is imperative for managers to consider how natural disturbance events interact temporally and spatially with management actions, and how forest restoration can promote ecological integrity in an inherently dynamic system.

In the Sierra Nevada, there is a pressing need to understand the nexus of silvicultural practices, wildfire, and fuels treatments in order to maintain forest ecosystems that are ecologically diverse and resilient. In the context of a century of fire suppression, at the core of the debate over how to manage Sierra forests is how to most appropriately manage areas where natural disturbances have been disrupted. Forest Service managers need a better understanding of the suitability of habitat created through fire suppression, fuel treatments (DFPZ, groups, mastication), wildfire, and post wildfire management.

Our existing PLAS study investigating the effects of fuel treatments at the local and landscape scale is providing valuable information on the response of a suite of species to help guide future forest restoration activities.

Beginning in 2009 the avian module of the Plumas-Lassen Administrative Study (PLAS) expanded to address important questions related to post-fire habitat and its management. The primary objective of this new part of the study is to assess the influence of post-fire conditions on spatial and temporal variation in bird abundance, and use this information to inform forest management practices that can maintain avian diversity across multiple spatial scales while meeting other management objectives. We began sampling three areas affected by fire within the boundaries of the original PLAS study: the Storrie Fire that burned in the fall of 2000, the Moonlight Fire that burned in the fall of 2007, and the Cub Fire that burned in the summer of 2008. Each of these fires burned at similar elevations and through primarily mixed conifer and true fir vegetation communities but with varying intensity patterns.

In the Sierra Nevada considerable debate surrounds the management of post-fire habitat. After nearly a century of fire suppression policies in the Sierra Nevada, the area affected by wildfire each year appears to be increasing back towards pre-suppression levels (Miller et al. 2007).

Thus, there is a growing need to understand the value of the habitats created by wildfire and the critical elements required by the unique and relatively diverse avian community in the Sierra Nevada.

In addition to fire suppression, there are a number of management activities that influence post-fire vegetation characteristics and landscape composition in working forests. These activities include salvage-logging, the mechanical mastication and herbicidal treatments to reduce broadleaf shrubs, and planting of conifer species that are favored by forestry. As a result, management activities may have profound influences on post-fire conditions- locally and across the landscape.

Wildfires provide a unique opportunity to mold a landscape into the forest composition that will exist there for the decades to come. The results from this ongoing study, especially with several more years of monitoring, can provide some important information on factors influencing the unique bird assemblages in post-fire habitat in order to ensure their needs are met while shaping the future forest that will one day return to these areas.

In future years we will conduct a more detailed analysis of fire severity at different scales and using standard measures used to classify burn severity (e.g. composite burn index). Additionally, we will use available remotely sensed data on burn severity and possibly LiDAR data to better understand the importance of severity class, patch size, and snag densities for the various species associated with post-fire habitat. Finally, we will compare bird assemblages within fuel reduction treatments in green forest, post-fire treatments, and untreated post-fire habitat to determine the effects of various treatments on a broad range of avian species to provide insight for a balanced approach to management of these forest ecosystems

Plumas-Lassen Study Vegetation Module: Restoration Implications

Seth W. Bigelow, March 10, 2010

The vegetation module's work is to develop tools to assist restoration of historic tree species composition and structure in northern Sierran Forests. Such restoration is essential for to improve resilience to, and recovery from, the large wildfires that have become the dominant disturbance regime in these forests.

The first step towards restoration of mixed conifer forests is often to confer resistance to wildfire by fuels-reduction thinning; ideally such thinning should also create opportunities for regeneration of desired species. Heretofore it has been difficult to determine whether thinning is creating adequate light for regeneration of the shade-intolerant, fire-and-drought-tolerant species that confer resilience to climate change. The vegetation module has published light/growth curves for saplings of the six most-common tree species of Sierran mixed-conifer. These show predicted stem diameter growth at any given light level. A manager who wishes to determine whether s/he is creating adequate microsites for shade-intolerant regeneration can measure microsite light using the canopy-photography protocol described in our paper, and compare the resulting values to our light/growth curves. We thus provide a powerful quantitative tool for forest restoration.

Creation of one to two acre (0.4-0.8 ha) openings under the group-selection system is a controversial means of restoring shade-intolerant tree species to the landscape, and the vegetation module has several studies that bear on this practice. Our study in the Beckwourth Ranger District, which was published in the journal Landscape Ecology, examined connectivity implications of group selection silviculture in patchy East-Side forests. We developed a new method of assessing landscape vulnerability to fragmentation which can be applied to raster maps of forest cover at any scale within a geographic information system. Other studies show distance and directionality of seed dispersal and establishment into group selection openings from surrounding trees, and impacts of group selection openings on fire climate and soil moisture.

Our current studies collect information for development of a flexible, spatially explicit forest simulator that will enhance managers' capacity to restore forests by allowing accurate prediction of forest growth and species replacement dynamics in novel silvicultural situations such as post-high-intensity fire and fuels-reduction thinning. The programming framework for this model, Sortie-ND, is already in place but further information tailored to Sierran tree species is required. Our study on regeneration in areas of the Storrie Fire footprint that have experienced high- or low-intensity fire is providing information on seedling dispersal direction, distance from parent tree, and abundance according to substrate type. In addition to providing a snapshot of forest recovery at the stand level, these data also provide dispersal parameters that go directly into the stand simulator. Our study on effects of tree neighborhood density on growth rate of large wildlife trees similarly serves two restoration purposes: it informs managers how to manipulate tree neighborhood density and composition to increase the rate of recruitment into the large-tree size class (vastly underrepresented on the landscape compared with historic levels), and provides growth parameters for the new forest simulator.

Spatially explicit forest simulators represent the state of the art for prediction of forest restoration treatments effects on tree species dynamics, but they do require mapped stands. Stand mapping is time-consuming and technically challenging, so we are investigating the ability of a ubiquitous data source, the FIA plot, to provide mapped plot data. A good representation of the understory light environment is essential for prediction of shade-tolerant vs. intolerant tree species dynamics, and the light measurements we have taken in our network of experimentally thinned stands provides the necessary data to assess the adequacy of the FIA data when used with several competing light models. If we can demonstrate the usefulness of FIA data for creating mapped stands, we expect these data in combination with the Sortie stand simulator to garner widespread use among forest managers.

The most effective restoration approach is to restore ecosystem function as well as structure. Our main effort, the network of stands that underwent experimental thinning or groups selection cutting, based in the Meadow Valley area, continues to provide an assessment of the ecosystem function implication of current management treatments. How does opening up the canopy to mimic historic forest structure affect the understory microclimate? The effect of opening up the overstory canopy on factors affecting fire in the understory have until now been largely conjectural. We collected continuous wind speed, air temperature, and relative humidity data in our plots for several years after treatment. These data do show an increase in peak wind gust speed in the understory subsequent to fuels reduction thinning, and, particularly, in group selection openings. We will conclude this study in the coming field season with collection of data on seedling recruitment, understory plant composition, and post-treatment fuels accumulation.

Restoration of Forests in the Plumas-Lassen Administrative Study: A Fuels and Fire Perspective

3/11/2010

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The Sierra Nevada contains many areas of high density mixed conifer forests that are dominated by shade-tolerant species (Parsons and Debenedetti 1979; Agee 1998; Taylor 2004) that are increasingly prone to large, high severity fires (Miller et al. 2009). Trends of increased uncharacteristically severe burning are expected to continue under changing climates (McKenzie et al. 2004; Westerling et al. 2006; Millar et al. 2007) and this is a major management issue. For decades, scientists and managers have understood the threat fire would pose to forests in this condition (Biswell 1989). However, it was not until the 1990's that the US Forest Service was given direction to manipulate stands, using a combination of silvicultural prescriptions, with the specific objective of modifying landscape-level fire behavior.

Much of the knowledge of landscape fuel treatment impacts on potential fire behavior relies on modeling not only potential fire behavior, but fuel treatments as well. Modeled fuel treatments can differ substantially from 'on the ground' implementations, both in terms of where treatments are located and in treatment intensity (Collins et al. 2010). As such, inferences from modeled treatments can be limited. Recently, a landscape fuel treatment has been implemented in the Meadow Valley area on the Plumas National Forest. This consists of a Defensible Fuel Profile Zone (DFPZ) network constructed across the Meadow Valley landscape. DFPZs are usually constructed along roads or ridge tops to reduce fuel continuity across the landscape and provide a defensible zone for fire suppression resources. In the analysis done in the last year our objectives were to quantify change in potential crown fire following actual treatment implementation. In addition, we 'grow' the forests out for 30 years to compare crown fire potential in a theoretical untreated landscape to that of the treated landscape.

Within DFPZ treatments, overall conditional burn probabilities were decreased by 62% after treatment; across all other untreated areas and within group selection units, conditional burn probabilities were decreased by 17-36%. In a ”problem fire” scenario, modeling the treated landscape reduced fire growth, and therefore final fire size, by 39% compared to the pre-treatment condition. It is important to note the “problem fire” was modeled without the effects of suppression activities, which could further reduce the final fire size by a greater percentage then reported here. Average flame length was lower throughout the “problem fire” in the post-treatment landscape. In addition, both the total area and the proportion of the final fire size burned at lower flame lengths in the post treatment landscape.

Results from this study demonstrate that the installation of landscape-level fuel reduction treatments can modify wildfire size and intensity. This is the first study to evaluate an actual installation of fuel treatments at the landscape scale. Treatment longevity was approximately 20-25 years highlighting the need of maintenance operations to keep treated areas effective. Reduction of fire size and intensity will positively effect other forest values such as habitat for rare species, reduction of losses of structures, improved fire fighter safety, and maintenance of forest ecosystems.

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Plumas-Lassen Study Small Mammal Module:

Relevance of data from the small mammal module to ecological restoration

The **Study Module on Small Mammal Distribution, Abundance, and Habitat Relationships** has collected data on the habitat associations, population dynamics, and spatial ecology (e.g., home ranges, territories) of key small mammal species in the Plumas National Forest and is applying these to provisional management recommendations. Our efforts have emphasized multiple approaches to understanding mammalian responses to habitat variation, and in associated studies we have focused out attention on particular species or species groups that are particularly important to the food web of this system.

Long-term live-trapping grids. In 2003 we established 18 live-trapping grids on which we proposed to monitor small mammal populations annually to quantify temporal patterns in numbers and productivity. Trapping grids were large (100 Sherman live traps + 72 Tomahawk live traps over 2.25 ha or 5.6 acres) and were established in forest dominated by white fir (*Abies concolor*; n = 9), red fir (*Abies magnifica*; n = 3), Douglas fir (*Pseudotsuga menziesii*; n = 3), and mixed pine-cedar (n = 3). In 2005 we established an additional 3 sampling grids in “group select” treatments located in white fir. Forest treatments (controls plus two levels of canopy thinning) were applied in 2006 and 2007; as such, we have sampled these grids for 4 years pre-treatment, and another 2-3 years after implementation of treatments. We have documented tremendous annual variation in certain species, but much more tempered variation in others. Perhaps most notably, deer mice (*Peromyscus maniculatus*) have varied greatly in mean abundance. To exemplify this, their numbers on Douglas fir grids declined from ca. 40 individuals / ha in 2004 to roughly 2-3 individuals / ha in 2005. In general, numbers were very high in 2004 and 2007, low in 2003, 2005, 2006, and 2008, and appear to be increasing in 2009. Linear modeling of *Peromyscus* numbers as a function of winter severity (annual snow fall) and annual conifer cone production suggests that population size is strongly influenced by both factors – mild winters following years of high cone productivity appear to result in very high deer mouse numbers. This is important to managers because deer mice are key vectors for hantavirus pulmonary syndrome, and they appear to be important prey species for California spotted owls. Hence, this species plays important epidemiological roles as well as a key trophic link in this system. Additionally, early work by one Masters student (Ms. Stephanie Coppeto) on this study quantified habitat characteristics at every trapping point, and developed models to relate the spatial distribution of small mammals to habitat features at both the scale of trapping points and the broader scale of trapping grids.

What this work provides is insight to the habitat requirements of constituent species and insight to the level of annual variability in these species. Knowledge of such habitat associations and requirements is critical for goal-based restoration. Because the species studied include key prey species for California spotted owls as well as various other raptors and mesocarnivores, it is important that restoration efforts target viable populations of these taxa. We also intend to develop predictive models of the relationship between population sizes and environmental drivers such as temperature, annual snowfall, etc.

"Songbird transects". In a parallel research effort we have surveyed small mammal numbers at 74 linear transects that were established by the Songbird Module of this research program. These transects were arrayed across the Plumas National Forest in a stratified random manner, and we have now sampled all transects that are logistically feasible in terms of access and/or topographic structure (e.g., some sites were too steep and/or remote to be sampled for 2 weeks with live traps). Because of the stratified random distribution of these transects they provide a statistically defensible survey of habitats (and associated small mammal species) in proportion to their availability within the PNF. Additionally, these span a greater range of habitats than are available in the long-term live-trapping grids, and consequently also sample additional species of small mammals.

This research provides a "snapshot" perspective of habitat associations and ecological distribution over a randomized sample of the National Forest. As such, this will yield more robust habitat association models than have the live-trapping grids. Importantly, these models should allow for a more comprehensive understanding of the spatial distribution of these species as a function of habitat structure and composition; in the context of environmental/habitat restoration, these should provide targets for restoration efforts.

Research on Focal Species. Reflecting the importance of California spotted owls in this system, we targeted research efforts on two key prey species – northern flying squirrels (*Glaucomys sabrinus*) and dusky-footed woodrats (*Neotoma fuscipes*). These species became the research focus of two Masters students. Both students applied focal trapping (typically around known dens) and extensive radiotelemetry to characterize home range size and composition for woodrats, overlap among individuals (both same-sex and opposite-sex dyads).

<u>Dusky-footed woodrats</u>. Ms. Robin Innes studied the spatial ecology and habitat associations of this species. Her studies revealed a strong positive relationship between woodrat density and the number of large California black oaks (*Quercus kelloggi*), as well as a strong association between the distribution of woodrat houses and large logs and stumps. Somewhat surprisingly, arboreal houses are prevalent in the PNF, and also appear more common in large California black oaks and snags.

<u>Northern flying squirrels</u>. Mr. Jaya Smith studied this species and his publications are currently in review. Unlike many other species, home ranges of male and female flying squirrels were similar in size, but the resources that they emphasize when establishing home ranges may be different. Habitat selection for both sexes was scale-independent; home ranges included disproportionate cover by large mixed-conifer stands, and habitat use (movements) within home ranges also favored use of areas with large mixed-conifer stands. Squirrels selected large conifers

and hardwoods for den sites preferentially over smaller conifers and saplings. These results suggest that retention of viable populations of this key prey species would be facilitated if forest managers could retain stands of large mixed conifers as well as hardwoods. Finally, Mr. Smith's research documented that overlap of home ranges among females was much greater than that between sexes (we lacked sufficient data to assess male-male overlap), suggesting that males and females are defending very different resources. Of interest, females of this highly social species appeared to segregate from other females and to defend territories when with young, but not once young have become independent.

These results suggest that promotion of viable populations of these key prey species would be facilitated if forest managers could manage for stands of large mixed conifers as well as hardwoods. As such, restoration efforts should emphasize these species, and post-fire salvage operations should strive to retain these habitat elements as foundations to the more rapidly growing (softwood) elements of these forests. We note that the importance of hardwoods may be an increasing theme in Sierra Nevada mammal ecology, as similar results have obtained for some mesocarnivore species as well.

Additionally, both of these species, as well as deer mice, appear to dominate the diets of spotted owls in the Plumas National Forest (J. Keane, pers. Com.), and our data on the spatial ecology and distribution of these will be fundamental in developing restoration strategies to assure a dietary foundation for this important avian predator.

Plumas-Lassen Study: California Spotted Owl Module – Restoration Implications

The primary goal of the California spotted owl (CSO) module is to address key information needs and increase the knowledge base regarding how forest management affects CSOs and their habitat. As restoration of forest structure and function becomes a guiding concept in the evolution of forest management, the fundamental questions of how forest restoration treatments and wildfire affects CSOs and their habitats remain relevant. Information is required on how restoration or fuels treatments versus how wildfire affects CSOs and their habitats.

The CSO module is designed to provide baseline information on CSO distribution, abundance and habitat associations against which the effects of forest restoration or fuels treatments can be evaluated. Over the last two years the CSO module has expanded the focus of the research to encompass the response of CSOs and their habitat to wildfire. Specifically we are investigating CSO response to high-severity wildfire versus CSO response to low-moderate severity wildfire. Thus, we are positioned to assess CSO responses across a gradient of forest management conditions that ranges across: (1) untreated forest conditions; (2) forests treated for fuels or restoration goals; (3) landscapes that experience low-moderate severity wildfire; and (4) landscapes that experience high-severity wildfire.

To date we have annually monitored baseline CSO distribution, abundance and habitat associations across 1,889 km^2 (729 mi^2) the Plumas and Lassen National Forests between 2003-2009. Additionally the Lassen Demographic Study provides information back to 1990 on the Lassen National Forest. This data provides a rich baseline against which to assess treatment and wildfire affects. To date, only one series of landscape fuels treatments has been implemented in the Meadow Valley Project on the Plumas National Forest. Our results based on two-years post-treatment monitoring suggests that CSOs are distributed in similar numbers as compared to the pre-treatment landscape.

In 2009 we completed the second year of post-fire monitoring in the Moonlight-Antelope Complex Fire Area on the Plumas-Lassen National Forests. This 88,000 acre (35,612 ha) fire was primarily a high-severity fire and only a single pair of CSOs is present in the post-fire landscape. In contrast, in our first year of surveys (2009) on the 21,000 Cub-Onion Complex Fire Area on the Lassen National Forest, which burned at low-moderate severity, we documented 7 territorial CSO sites distributed across the post-fire landscape at similar spacing to that we observe in unburned forest landscapes.

In summary, our results to date are providing empirical support that CSOs can persist in landscapes treated for fuels or restoration treatments. Further, wildfire effects can vary depending on fire severity patterns and the resulting post-fire vegetation conditions. Our results suggest that CSOs are able to persist in landscapes that experience primarily low-moderate severity wildfires. In contrast, high severity wildfires appear to have negative effects on CSOs and their habitat.

The design of our study, and our results to date, are useful for: (1) assessing CSO response across a gradient of forest management conditions; (2) informing future forest restoration objectives; and (3) positioning us to be able to directly monitor how CSOs and their habitat respond to restoration prescriptions at the plot, stand, and landscape spatial scales. Our results to date suggest that CSOs are able to persist in at least some treated landscapes and persist in landscapes that experience low-moderate severity wildfire. Hence, carefully designed future experiments or treatments that address restoration objectives for restoring vegetation structure and the function of wildfire as the primary natural disturbance agent in Sierra forests may be viable management approaches for restoring forests and conserving CSOs. Further, our current habitat modeling efforts at plot, home range, and landscape spatial scales will prove valuable tools for predicting the affects of restoration treatments on CSOs and their habitat.