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Fisher Population Ecology on the Hoopa Valley Indian Reservation, Northwestern California

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FISHER POPULATION ECOLOGY ON THE HOOPA VALLEY INDIAN RESERVATION, NORTHWESTERN CALIFORNIA

A Dissertation Presented

by

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ABSTRACT

FISHER POPULATION ECOLOGY ON THE HOOPA VALLEY INDIAN RESERVATION, NORTHWESTERN CALIFORNIA

MAY 2012

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I studied aspects of fisher (*Martes pennanti*) population ecology on the Hoopa Valley Indian Reservation in northern California to fill critical information gaps relative to timber management and its effect on the status of fishers, a candidate for listing under the U.S. Endangered Species Act. A decline in mark-resight density estimates of fishers from 1998 (52/100 km^2; 95% CI = 43-64) to 2005 (14/100 km^2; 13-16) was likely due to changes in prey habitat suitability, increased predation pressure, and/or disease. The decline was also indicated by catch-per-unit effort indices, but not by camera station or track-plate station indices. Colleagues and I developed and tested methods of collecting mark-recapture data using genetic marking, passive integrated transponder (PIT) tag technology, and digital, passive-infrared photography that could be used in a demographic monitoring protocol. The comparatively high cost of PIT tag reading equipment and genetic analyses makes the use these methods dramatically more expensive and yield less demographic data compared to using a traditional mark-recapture approach using only live trapping. By monitoring 40 radio-marked, breeding age ($\geq$ 2 years old) females during 2005-2011, we found that 87% exhibited denning behavior and 65% of these were successful in weaning at least one kit (mean = 1.9). Of 14 kits radio-marked in their first fall, 3 died prior to dispersal, 3 lost collars, and the other 8 established home ranges 0.8-18.0 km away from natal areas. Nipple size (width multiplied by height of the largest anterior nipple), evaluated as a predictive index of female fisher reproductive success, differed among nonbreeders vs. attempted and current breeders. A predictive index for use in assigning reproductive status to females with unknown reproductive histories had an overall correct classification rate of 81% and a chance-corrected measure of prediction of 69.5%. These results illustrate the value in establishing long-term, accurate programs to monitor populations of imperiled species which strive to determine cause and affect relationships to changes in populations and ultimately, modeling habitat fitness. The relatively low reproductive rate of female fishers brings into question the species ability to demographically respond to increased rates of juvenile and adult mortality with increased reproduction and/or survival. The limited dispersal capability of juvenile fishers restricts ability to rescue vanishing local populations from extirpation, re-inhabit landscapes from which they were previously extirpated, and establish the functional connectivity of metapopulations.

PREFACE

The fisher (*Martes pennanti*) is a mid-sized, forest-dwelling carnivore in the family *Mustelidae* (Powell 1993). The geographic distribution of fishers historically included the boreal forests of southern Canada, the northern Rocky Mountains, the northeastern and upper-midwestern United States, and south through the Cascade Range and coastal mountains, northern California, and the western slopes of the Sierra Nevada Range (Powell 1993). However, trapping for fur during the early twentieth century, predator and pest control campaigns, and forest management practices resulted in population declines and range contractions across the distribution. In the Pacific states the fisher was considered extirpated in Washington until recent reintroduction efforts; it exists in two relatively small populations in southern Oregon and occurs in less than 50% of its historic range in two isolated populations in California (Buck et al. 1994, Gibilisco 1994, Powell and Zielinski 1994, Zielinski et al. 1995, Aubry and Lewis 2003, Zielinski et al. 2005). Consequently, and in light of current threats, the U.S. Fish and Wildlife Service concluded the distinct population segment historically occurring in Washington, Oregon, and California was warranted, but precluded for listing, under the Endangered Species Act in 2004 (U.S. Fish and Wildlife Service 2004).

Following the U.S. Fish and Wildlife Service listing decision, the Interagency Fisher Biology Team was formed and tasked to develop a Fisher Conservation Assessment and Strategy (FCAS) for the Pacific states (Lofroth et al. 2010). The Biology Team is composed of representatives from various federal, state, and tribal resource management agencies. The FCAS will be a science-based assessment of current conservation concerns and management recommendations for reducing or eliminating risks to population persistence and species recovery. The Biology Team will also assist public agencies with interpretation and implementation of the FCAS. In preparing the FCAS, the Biology Team has identified key information gaps in our understanding of fisher ecology in the Pacific states. Filling these information gaps will enable the Biology Team to develop more informed conservation strategies to more effectively engage with forest managers working toward conservation outcomes across the range.

It is suspected that timber harvest continues threaten fisher populations in the Pacific states by means of habitat fragmentation, reductions in habitat size, and changes in forest structure to be unsuitable for fishers (Carroll et al. 1999, Zielinski et al. 2004). Fishers have been described as being among the most habitat-specialized mammals in North America (Harris et al. 1982). In the Pacific states and British Columbia, Canada, fisher habitat has been described as late-successional, structurally-complex, mixed-conifer and conifer-hardwood forests with dense forest cover (Harris et al. 1982, Buck et al. 1994, Ruggiero et al. 1994, Carroll et al. 1999, Weir and Harstead 2003, Zielinski et al. 2004). Fishers use large trees and snags for resting and denning sites that provide protection from unfavorable weather and predators (Powell 1993, Kilpatrick and Rego 1994, Zielinski et al. 2004). However, it has been suggested that fishers in the West are not dependent on late-successional forests, but require closed-canopy forest with adequate prey populations and suitable structural elements associated with older forests for resting and denning (Holthausen et al. 1994, Jones and Garton 1994, Klug 1997, Lewis and Stinson 1998).

These discrepancies and gaps in our knowledge of fisher habitat requirements and population ecology in the West present challenges for western forest managers. Timber management continues to be of significant economic importance to the Pacific Northwest region, especially on the Hoopa Valley Indian Reservation in northwestern California. Timber management is the single largest source of revenue for the Hoopa Valley Tribe and is the largest, single source of employment for tribal members. In light of the extreme socio-economic conditions throughout the region and specifically on the Reservation, the economic benefits provided by managing timber resources have a direct impact on improving the quality of life and the means to achieve self-sufficiency for the Hupa people. Under the Tribe's current forest management plan, the economic benefits of timber management are achieved while protecting elements of tribal cultural integrity, including fish, wildlife, plants and their habitats. This includes the culturally significant fisher, which is used in traditional ceremonial and dance regalia.

Our collaborative fisher research efforts on the Hoopa Valley Indian Reservation were designed to fill critical information gaps identified by the Hoopa Valley Tribe in its timber management and wildlife conservation operations and by the Interagency Fisher Biology Team in drafting the FCAS. Our research goals are to measure specific population parameters and develop methods to monitor fisher populations. A more complete understanding of fisher biology and methods to monitor populations will enhance the conservation capacity of the Hoopa Valley Tribe and the Interagency Fisher Biology Team's FCAS.

The following chapters were written in the form of a series of scientific papers that are structured around a common theme. Because the fisher research efforts are necessarily collaborative, the chapters resulting from this work and that are presented here use the pronoun "we" instead of "I", though I am the senior author in all respects on each chapter.

In **Chapter 1** we discuss a dramatic decline in fisher capture success and a change in sex ratio of the captured population between the first year of our research (2004-2005) and a previous effort conducted on the Reservation (1996-1999). These differences prompted us to estimate the population density of fisher on a 90-km^2 study area on the southeast corner of the Reservation using a mark-resight design. We compared the 2005 population density estimate to 1996-1999 estimates and explored possible explanations for any difference in population density and sex ratios. Our density estimation results and simultaneous population monitoring data also provided a post-hoc opportunity to evaluate the relative efficacy of three classical indexing techniques (catch-per-unit-effort, camera stations, and track-plate stations) to accurately detect population change. Our results reinforce the importance of careful thought given to the study goals and potential limitations of any technique.

The value of long-term population monitoring data was demonstrated in the forests of the Pacific states through the conservation efforts involving the northern spotted owl (*Strix occidentalis caurina*). A similar need exists for fisher to collect long-term demographic data to measure the success or failure of re-introduction efforts, determine the effects of changes in forest management practices, and monitor general population trends. In **Chapter 2** we describe a long-term population monitoring protocol, develop and test the proposed technological components, and assess the cost-efficiency of the protocol. We developed a single sampling device housing a passive integrated transponder (PIT) tag reader, a hair-snare, and a remotely-triggered camera. Testing the effectiveness of the device and its components in recording fisher visits was conducted in a captive and field setting.

Few studies have been able to document reproduction, recruitment, and dispersal for a wild fisher population in the Pacific states. In **Chapter 3** we document these important population parameters for fisher on the Reservation. These data will serve as a baseline for fisher conservation, be an element in future modeling of fecundity and habitat relationships, and inform conservation efforts across the range.

An important component in monitoring fisher population dynamics is modeling reproduction. Estimates of fisher vital rates, including reproduction, have been very difficult and costly to obtain. Cost- and labor-intensive radio-telemetry efforts have provided some information on fisher reproductive rates for a small number of landscapes. However, radio telemetry approaches are not cost effective for most managers, particularly in the context of long-term population monitoring across large ownerships. In **Chapter 4** we evaluate the efficacy of nipple size as a predictive index of female fisher reproductive success in weaning at least one kit from females with known reproductive histories. Our index could prove useful for managers hoping to model fisher reproduction and the influence of habitat and other covariates on reproductive success, particularly in timber-managed landscapes occupied by extant or reintroduced fisher populations.

Finally, **Chapter 5** is an overview of our findings as they relate to filling critical information gaps on the status of fishers. I speculate on reasons for the population decline, including habitat change, increased predation risk, and disease. I also consider the role of limited female fisher reproductive rates and juvenile dispersal capability in terms of population conservation and metapopulation dynamics.

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CHAPTER 1

DENSITY OF FISHERS AND THE EFFICACY OF THREE RELATIVE ABUNDANCE INDICES AND SMALL-SCALE OCCUPANCY ESTIMATION TO DETECT A POPULATION DECLINE ON THE HOOPA VALLEY INDIAN RESERVATION, CALIFORNIA

Abstract

We used a mark-resight design to calculate and compare density estimates of fishers, a candidate for listing under the U.S. Endangered Species Act, on the Hoopa Valley Indian Reservation in northwestern California in 1998 and 2005. Our density estimation results and simultaneous population monitoring data provided a post-hoc opportunity to evaluate the relative efficacy of three classical indexing techniques (catch-per-unit-effort, frequency of detection at camera stations, and frequency of detection at track-plate stations) and small-scale occupancy estimation to accurately detect population change. We calculated densities (and 95% confidence intervals) of 52 (43-64) and 14 (13-16) fishers/100 km^2 in 1998 and 2005, respectively. We speculate changes in prey habitat, increases in predation, disease, or some combination of these potential causes were responsible for the population decline. We also detected a decline in the relative abundance of fishers between 1998 and 2005 using catch-per-unit-effort indices ($\chi^2 \geq 10.18$, $p \leq 0.007$), but not of the same magnitude as our mark-resight density estimates. We wrongly detected an increase ($\chi^2=4.23$, $p=0.040$) and no difference ($\chi^2=1.38$, $p=0.240$) in the relative abundance of fishers between surveys using frequency of detection indices at camera stations and at track-plate stations, respectively. Occupancy estimates did not differ between 1998 and 2005. Our results reinforce the

importance of careful thought given to the study goals and potential limitations of any technique. For populations deemed valuable (e.g., at risk or sensitive), we suggest managers consider adopting more defensible, large-scale occupancy estimation or mark-recapture methods to monitor changes in population sizes.

Introduction

Managing small and isolated populations of imperiled species often involves accurately assessing changes in population sizes to evaluate extinction risk, identifying population-level threats, and determining the success of conservation measures. Often, abundance or density is the metric by which population changes are measured. However, estimation of abundance and density often require substantial effort, leading some to view indices of relative abundance (e.g., Wood 1959, Roughton and Sweeny 1982, Conner et al. 1983, Kohn et al. 1993, Sargeant et al. 2003) and occupancy estimation (MacKenzie et al. 2006, Long and Zielinski 2008) as surrogates for abundance and appropriate methods for population monitoring.

Classical indexing methods commonly involve deploying scent stations or other detection devices throughout a study area, estimating frequency of detection (number of animal visits per number of station nights), and assuming that frequency of detection is related to the number of individuals in the area (Long and Zielinski 2008). However, doubt about the relationship between classical indices and abundance has led to these indices being eclipsed by more defensible detection history methods for estimating occupancy (MacKenzie et al. 2002, Anderson 2003, Sargeant et al. 2003, MacKenzie et al. 2006, Long and Zielinski 2008). Modern occupancy estimation methods are used to estimate the proportion of a large-scale survey area that is occupied or used by the species of interest (Mackenzie et al. 2006, Long and Zielinski 2008). Occupancy estimation methods provide for the estimation of detectability directly and use it to adjust occupancy estimates to account for sites where the species was likely present but not detected (MacKenzie et al. 2002, MacKenzie et al. 2006, Long and Zielinski 2008).

However, few classical indices (e.g., Sargeant et al. 2003, Choate et al. 2006) or occupancy estimates (e.g., Sileshi et al. 2006, Gopal et al. 2010) have been applied simultaneously to populations with abundance- or density-based estimates of a change in population size. Two independent research efforts (1997-1998 and 2004-2005) on fishers (*Martes pennanti*) on the Hoopa Valley Indian Reservation (hereafter Reservation) in northwestern California provided a post-hoc opportunity to compare estimates of relative fisher abundance and occupancy to estimates of fisher population density at two points in time. Accurate monitoring of fisher populations is of particular conservation concern because of the candidate status of populations in Washington, Oregon, and California under the U.S. Endangered Species Act (U.S. Fish and Wildlife Service 2004). We calculated estimates of fisher population size using a mark-resight design on the Reservation in 1998 and 2005. We simultaneously evaluated the relative efficacy of three indexing techniques and occupancy estimation in detecting a change in relative fisher abundance between 1998 and 2005. The indexing techniques were: 1) catch-per-unit-effort, 2) frequency of fisher detections at camera stations, and 3) frequency of fisher detections at track-plate stations.

Study Area

Our study was conducted on 90 km^2 in the southeast corner of the 366-km^2 Reservation (Figure 1). The area is located within the Klamath physiographic province

(Küchler 1977) and elevations range between 98 and 1,170 m. Mean daily maximum and minimum temperatures are 22 and 6°C, respectively, and mean annual precipitation, primarily rain, is 138 cm (National Climate Data Center, http://www.ncdc.noaa.gov/oa/ncdc.html, accessed 15 January 2011). The urban zone of the Reservation occupies 25 km^2 on the western border of our study area and is the location of infrastructure for the Reservation's 2,600 human inhabitants (Figure 1; Hoopa Valley Tribe, http://www.hoopa-nsn.gov/documents/2000Census.pdf, accessed 23 January 2011).

Forests generally have an overstory dominated by Douglas-fir (*Pseudotsuga menziesii*) and a midstory dominated by hardwood trees including tanoak (*Lithocarpus densiflorus*), madrone (*Arbutus menziesii*), Oregon white oak (*Quercus garryana*), California black oak (*Quercus kelloggii*), and canyon live oak (*Quercus chrysolepis*). Pure hardwood stands occur in some areas. Past and current timber harvests created a mix of mature-old growth and early seral forests. Between 1998 and 2005 abrupt changes in habitat conditions occurred on 3% of the study area by way of timber harvest (198 ha) and wildfire (68 ha). Less obvious changes occurred through succession, as previously harvested stands grew from a brushy condition into a stem exclusion stage of dense young conifer and hardwood trees.

Methods

We applied the theoretical framework of a mark-resight model (Seber 1982, Bowden 1993, Bowden and Kufeld 1995) adjusted for density (Wilson and Anderson 1985, Maffei et al. 2004, Silver et al. 2004, Soisalo and Cavalcanti 2006, Maffei and Noss 2008) to estimate fisher population density in 1998 and 2005. Capture and handling methods were approved by the Institutional Animal Care and Use Committee of Humboldt State University, protocol 04104.W.42.A.

We selected 36 and 28 locations to place traps, cameras, and track-plates in 1998 and 2005, respectively. Selection was based on our knowledge of fisher habitat, time of year of trapping, effective coverage of the study area, road availability, and access. The goal of the selection process was to maximize the detection probability of each fisher present on the study area. Effective coverage of the study site involved maximizing the likelihood ≥1 location was placed in each potential female fisher home range (Karanth and Nichols 1998, Fuller et al. 2001, Maffei et al. 2004, Silver et al. 2004, Yaeger 2005; Figure 1). The reduction in the number of stations between 1998 and 2005 was based on an observed increase in home range size of adult female fishers on the study area between the two time periods (mean ± standard error = 168 ± 17 ha in 1998 and 728 ± 85 ha in 2005; Yaeger 2005, S. M. Matthews, Hoopa Valley Tribe, unpublished data). Thus, fewer stations were required in 2005 to maintain effective coverage of the study area.

We used Tomahawk live traps (model 207, Tomahawk Live Trap Company, Tomahawk, Wisconsin, USA) baited with chicken legs and modified with a plywood cubby box to capture fishers (Wilbert 1992, Seglund 1995). Initial trapping occurred between 1 October 1997 and 14 March 1998 over an area of 26.0 km^2. Trapping also occurred between 8 December 2004 and 11 March 2005 over an area of 36.8 km^2 that overlapped nearly all of the area trapped in 1997-1998 (Figure 1).

Captured fishers were anesthetized with ketamine hydrochloride (40 mg/kg) and diazepam (0.25 mg/kg) and handled using standard protocols (Aubry and Raley 1996, Yaeger 2005). We marked each fisher with uniquely colored plastic ear tags (Nasco

Standard Rototag Blank, Nasco, Modesto, California, USA) in both ear pinnae for future individual identification. We fitted all adult and sub-adult female fishers and select male fishers with radio transmitters (Telonics MOD80 or MOD125, Telonics, Inc., Mesa, Arizona, USA in 1998 and Holohil model MI-2, Holohil Systems Ltd., Carp, Ontario, Canada or Telonics model MOD80 in 2005) to determine the mean maximum distance moved to establish an effective trap area (Wilson and Anderson 1985, Maffei et al. 2004, Silver et al. 2004, Soisalo and Cavalcanti 2006, Maffei and Noss 2008). We released all fishers after recovery from anesthesia at their sites of capture.

Resighting devices used in 1998 included both a remote camera and a track plate housed in a single rectangular, plywood box, open on one end (Fowler and Golightly 1994, Zielinski 1995). The cameras were converted 35-mm cameras (Olympus Infinity Mini DLX, Melville, New York, USA) and were triggered by a magnetic reed-switch (HE500-ND, Hamlin, Inc., Lake Mills, Wisconsin, USA) mounted on a treadle attached to the track plate. In 2005 the cameras and track plates were housed as separate devices. The converted 35-mm cameras were triggered by infrared door alarms (Radio Shack Mini PIR Alarm Catalog Number 49-425, Fort Worth, Texas, USA; Matthews et al. 2008a) and housed in 119-cm long and 38.7-cm diameter PVC culvert pipes. Camera stations used in 1998 and 2005 required an animal to enter the device in order for the camera to be triggered and were baited with a chicken leg tied to the floor with fishing line. Track-plates in 2005 were housed in corrugated plastic, rather than plywood, boxes of the same dimensions as those used in 1998. Bait, film, and track-plates were examined every other day.

Because of equipment limitations, multiple sessions were conducted during 1998 and 2005, moving devices to alternate locations for each session. In 1998 we placed camera/track-plate devices at 12 locations for 13 nights (13 to 26 March 1998), then at 12 locations for 10 nights (30 March to 9 April 1998), and the remaining 12 locations for 10 nights (13 to 23 April 1998). In 2005 we placed remote cameras at 14 locations and track plates at the other 14 locations for 12 nights (20 March to 1 April 2005) and then moved cameras to track-plate locations and vice-versa for another 12 nights (1 to 13 April 2005).

We calculated fisher abundance for 1998 and 2005 using the Bowden estimator of population size (Bowden 1993, Bowden and Kufeld 1995) in the software program NOREMARK (White 1996). Independent resightings of the same individual was defined as photographs separated by >24 hours or the individual fisher resighted at ≥2 camera stations within 24 hours (Mace et al. 1994). A density estimate was generated by dividing fisher abundance by the effective trap area (Soisalo and Cavalcanti 2006, Maffei and Noss 2008). The effective trap area included a circular buffer around each detection station location, minus the urban zone of the Reservation. The radius of the buffer was the mean maximum distance moved (MMDM) between female fisher locations (Soisalo and Cavalcanti 2006, Maffei and Noss 2008). Fisher locations used to calculate MMDM were collected using ground-based radio-telemetry, trap, and camera locations (Soisalo and Cavalcanti 2006, Maffei and Noss 2008).

We calculated catch-per-unit-effort in 1998 and 2005 as the frequency of fisher captures at trap sites for unmarked fishers (number of unmarked fisher captures/number of trap nights) and for any fisher, including recapturing previously marked individuals (number of total fisher captures/number of trap nights). We calculated the frequency of fisher detections at camera and track-plate stations (number of fisher detections/number of station nights) in 1998 and 2005. In order to simulate relative abundance index methods, which would not include a previous marking effort, we included all detections of fishers in calculating frequencies of detection, irrespective of marked status. Binomial proportions tests were used to test for significant differences between 1998 and 2005 estimates of relative abundances for each technique using the STATS package in program R version 2.10.1 (R Development Core Team 2008).

Although we designed our study to estimate fisher density, upon reviewer suggestion, we retrospectively developed an occupancy design from our data to compare to our density results. Occupancy estimation is used as monitoring technique at larger spatial scales, following from the assumption that detection of species and detection histories at each location are independent (MacKenzie et al. 2006). We initially discounted occupancy estimation as a population monitoring method because of the small size of our study area (90 km^2) compared to estimates of average female fisher home ranges (168 ha in 1998 and 728 ha in 2005). This relationship led to violations of the above independence assumption while maintaining an effective number sample locations. Thus our results are more an expression of "use" of the space that was sampled than of true occupancy.

We analyzed detection histories as if each station were a sample unit and by combining stations with the next closest station into two-station sample units. We acknowledge that even by combining stations our design still violates the independence assumption and the precision of our occupancy estimates are probably overstated (MacKenzie et al. 2006). The 1998 effort had a single device that contained both a camera and track plate. Overall the cameras recorded many more unique detections than did the track plates. However, for each visit the occupancy data would include a 0 or a 1 (not detected or detected), therefore even if the camera recorded 2 or more distinct detections of fishers they would be consolidated into a 1 in the detection history for the sample unit. Thus, we elected to use only the track detection data to develop detection histories for the six visits in 1998. The 2005 detection devices were separated into a stand-alone camera station and a track-plate device. Detection histories were developed for both camera and track plate data and analyzed separately for the eight visits in 2005. Occupancy analyses were conducted in Program Presence 2.3 (Hines 2006).

Results

Twenty-nine individual fishers (21 females, 8 males) were captured on 67 occasions during 451 trap nights during the 1997-1998 trapping effort (Table 1). Fourteen individual fishers (8 females, 6 males) were captured on 49 occasions during 605 trap nights during the 2004-2005 trapping effort (Table 1). The measures of MMDM by female fishers in 1998 and 2005 were 2,220 and 2,997 m, respectively (Figure 1). Buffering each camera location used in 1998 by 2,220 m and those used in 2005 by 2,997 m resulted in effective trap areas of 66.8 and 102.5 km^2 during 1998 and 2005, respectively. We calculated density estimates (and 95% confidence intervals) of 52 (43-64) and 14 (13-16) fishers/100 km^2 during 1998 and 2005, respectively, indicating a population decline of 73% (Table 1).

The frequency of detection of unmarked fishers and all fishers at trap sites was greater in 1998 than in 2005, indicating a decline in relative abundance (Table 1). Fishers were detected at camera stations less frequently in 1998 than in 2005, indicating an increase in relative abundance (Table 1). Fishers were detected at track-plate stations with equal frequency in 1998 and 2005, indicating no change in relative abundance (Table 1).

Our 1998 and 2005 occupancy estimates based on either single-station or two-station sample units did not differ based on the overlap of our 95% confidence intervals (Tables 2 and 3). There was no structure on the occupancy portions of our models and only minor structure on the detection probability portions (Tables 4, 5, and 6). The structure on the detection probabilities varied from one dataset to the other; however, a time trend or quadratic trend was included in the top two models in all cases and was competitive with the top model if it was not number one based on AIC values and weights (Tables 4, 5, and 6). Estimates of occupancy were essentially identical regardless of detection probability structure. In addition, the estimates of occupancy were incredibly close to the naïve estimates (Tables 2 and 3). Detection probabilities were also fairly high with the top models for all the data sets including a time trend or quadratic trend. Detection probabilities estimated by the simplest model (constant or dot) resulting in a single probability estimate were 0.51, 0.75, and 0.71 for the 1998, 2005 photo, and 2005 track data sets, respectively.

Discussion

The inconsistent results between three relative abundance indices and occupancy estimation to detect an apparent 73% decline in fisher population density cast doubt on the reliability of these types of indices to detect large changes in population size at small scales. Our estimate of a 73% decline in fisher density was supported by a four-fold increase in female home range sizes between 1998 and 2005. Benson et al. (2006)

identified a similar relationship between density and home range size for bobcats. They documented bobcat population density explained 56% of the variation in female home range size, whereas food availability and body weight failed to explain observed changes in bobcat home range size.

Of the three relative abundance indices we considered, only the results from catch-per-unit-effort for both unmarked and total fisher captures indicated a decline, but not the magnitude, in fisher density on the study area. Fisher detection rates at camera stations were significantly greater in 2005 than in 1998, indicating an increase rather than the actual significant decrease in population size. Our occupancy estimation results illustrate the point that a purely occupancy-based approach may not be as sensitive to a population changes as a more intensive mark re-sight approach. Occupancy methods are better suited for long-term, large-scale monitoring where dramatic changes in occupancy of areas might be expected due to large-scale disturbances (Mackenzie et al. 2006, Long and Zielinski 2008).

We suspect the proximity of detection stations ($\geq$ 1 in each potential female fisher home range) did not allow us to evaluate whether visitation rates were a function of population size or repeat visitation by the same animal(s) (Zielinski and Stauffer 1996, Sargeant et al. 2003, Long and Zielinski 2008). Variation in individual fisher detection probabilities probably influenced our results. Although we considered each fisher unmarked when totaling the number of fisher visits, marked status enabled us to identify three individual fishers made up nearly 90% of marked fisher detections in 2005, whereas in 1998 the three most frequently detected marked fishers only made up 45% of marked fisher detections. Similarly, Sargeant et al. (2003) found density-dependent visitation rates (visitation rates were greater when population sizes were low) to scent stations by kit foxes (*Vulpes macrotis*) and swift foxes (*Vulpes velox*) in a comparison of scent-station population indices to concurrent and independent measures of population sizes.

Our results support the conclusions of other studies, as articulated by Ray and Zielinski (2008), that relative indices from classical detection station designs have low resolution because reliable inferences require independent and large samples (Zielinski and Stauffer 1996, Sargeant et al 2003). These results do not diminish the value of methods involving remote cameras or track-plate stations, but in fact reinforce the importance of careful thought given to the study goals and potential limitations of any technique. For populations deemed important (e.g., at risk or sensitive), we caution managers considering classical relative abundance indices to consider more defensible detection history methods for estimating occupancy at an appropriate spatial scale or a mark-recapture/resight design (MacKenzie et al. 2002, Sargeant et al. 2003, MacKenzie et al. 2006, Long and Zielinski 2008, Ray and Zielinski 2008).

Our estimates of fisher density fall within (2005) and above (1998) the upper limit of the range of estimates (5 to 38 fishers/100 km^2) from across the regional distribution of fishers (Powell 1993). Two California studies used camera-recapture methods similar to ours to estimate fisher density for two unharvested populations in different portions of the California range. On industrially-managed timberlands in northwestern, coastal California, Thompson (2008) reported fisher density estimates ranging from 15 to 21 individuals/100 km^2. On the west slope of the southern Sierra Nevada in the Sierra National Forest, Jordan (2007) reported fisher densities for an untrapped population ranging from 5.3 to 25.3 individuals/100 km^2.

Unfortunately, research activity to determine the cause of the decline did not occur on the Reservation between 1998 and 2005. Thus we are left to speculate changes in prey habitat, increases in predation, disease, or some combination of these potential causes were responsible for the population decline (Chapter 5). It is possible that habitat change adjacent to the Reservation resulting from catastrophic wildfire may have contributed to increased fisher predation risk and population declines on the Reservation. In 1999, the Megram fire burned over 505 km^2 on the southeastern border of the Reservation (Jimerson and Jones 2003). Over 63 km^2 of old-growth and late-mature forest was affected by high severity fire (>80% tree mortality), returning these areas to shrub/forb habitat (Jimerson and Jones 2003). Since 1999, reports of incidental sightings of bobcats have increased throughout the Reservation (J. M. Higley, Hoopa Tribal Forestry, unpublished data). Witmer and deCalesta (1986) reported bobcats used Doulgas-fir dominated forest cover during the day while most inactive and clear-cut units at night while most active, presumably hunting.

Predation has generally not been considered a limiting factor for fisher populations based on research conducted in New England and Great Lakes regions (Powell 1993, Kurta 1995), but recent research on west coast fisher populations indicates that predation, especially by bobcats, is a common source of mortality (G. Wengert, University of California Davis, unpublished data). An increase in bobcat predation on fishers might also explain the observed sex ratio change. Smaller body size might make female fishers more susceptible to bobcat predation to the point of skewing fisher population sex ratios.

Management Implications

The use of classical relative abundance indices and occupancy estimation are particularly attractive to managers because of their relative low cost and the belief they are effective in monitoring relative change in abundance over time. Our results cast doubt on the efficacy of these indices and small-scale occupancy estimation to adequately detect significant population change and reinforce the importance of careful thought given to the study goals and potential limitations of any technique. Managers should consider adopting more defensible large-scale occupancy estimation or mark-recapture techniques as methods to monitor changes in wildlife population sizes, especially when responsible for at-risk populations. With the absence of long-term monitoring on the Reservation between 1998 and 2005, the cause(s) of the fisher population decline will never be known. However, it illustrates the value in establishing long-term, accurate programs to monitor populations of imperiled species which strive to determine cause and affect relationships to changes in populations and ultimately, modeling habitat fitness.

Table 1. Changes in frequency of detection of fishers compared using binomial proportions tests to evaluate the relative efficacy of three indexing techniques (catch-per-unit effort, camera stations, and track-plate stations) to detect an apparent 73% fisher population decline determined using a mark-resight design in 1998 and 2005 on the Hoopa Valley Indian Reservation, Humboldt County, California, USA.

	Detections		Trap nights		Frequency of detection (%)					
Index technique	1998	2005	1998	2005	1998	2005	Delta (%)	χ^2	p	Trend
Traps: unmarked fishers	29	14	451	605	6.4	2.3	- 4.1	10.18	0.001	Decline
Traps: all fishers	67	49	451	605	14.9	8.1	- 6.8	11.38	0.007	Decline
Cameras	149	146	416	336	35.8	43.5	7.7	4.23	0.040	Increase
Track plates	100	76	229	202	43.7	37.6	- 6.1	1.38	0.240	No change

Table 2. Estimates of fisher occupancy/use and 95% confidence intervals (CI) based on track or photo detections collected at single station sample units in 1998 (six visit detection histories) and 2005 (eight visit detection histories) on the Hoopa Valley Indian Reservation, Humboldt County, California, USA.

				95% CI	
Data Set	Sample units	Naïve estimate	psi	lower	upper
1998 Track	36	0.8611	0.8717	0.7088	0.95
2005 Track	28	0.5714	0.5715	0.3868	0.7382
2005 Photo	28	0.6429	0.643	0.4539	0.7961

Table 3. Estimates of fisher occupancy/use and 95% confidence intervals (CI) based on track or photo detections collected at two station sample units in 1998 (six visit detection histories) and 2005 (eight visit detection histories) on the Hoopa Valley Indian Reservation, Humboldt County, California, USA.

				95% CI	
Data Set	Sample units	Naïve estimate	psi	lower	upper
1998 Track	18	0.8421	0.8517	0.6935	0.9358
2005 Track	14	0.7143	0.7143	0.4395	0.8886
2005 Photo	14	0.7857	0.7857	0.5057	0.9293

Table 4. Track-based fisher occupancy model results from 1998 showing that a time trend (T) was the top model on the detection probability on the Hoopa Valley Indian Reservation, Humboldt County, California, USA.

Model	AIC	delta AIC	AIC weight	Model Likelihood	Parameters	-2*Log likelihood
psi(.),p(T)	295.41	0	0.5556	1	3	289.41
psi(.),p(t)	297.32	1.91	0.2138	0.3848	7	283.32
psi(.),p(TT)	297.34	1.93	0.2117	0.381	4	289.34
psi(.),p(.)	302.17	6.76	0.0189	0.034	2	298.17

Table 5. Track-based fisher occupancy model results from 2005 with the best model including survey-specific detection probability followed by a time trend on the Hoopa Valley Indian Reservation, Humboldt County, California, USA.

Model	AIC	delta AIC	AIC weight	Model Likelihood	Parameters	-2*Log likelihood
psi(.),p(TT)	91.51	0	0.7498	1	4	83.51
psi(.),p(t)	95	3.49	0.1309	0.1746	9	77
psi(.),p(T)	95.22	3.71	0.1173	0.1565	3	89.22
psi(.),p(.)	103.41	11.9	0.002	0.0026	2	99.41

Table 6. Photo-based fisher occupancy model results from 2005 with the best model having a constant detection probability on the Hoopa Valley Indian Reservation, Humboldt County, California, USA.

Model	AIC	delta AIC	AIC weight	Model Likelihood	Parameters	-2*Log likelihood
psi(.),p(TT)	117.23	0	0.3971	1	4	109.23
psi(.),p(.)	117.52	0.29	0.3435	0.865	2	113.52
psi(.),p(T)	118.36	1.13	0.2257	0.5684	3	112.36
psi(.),p(t)	122.17	4.94	0.0336	0.0846	9	104.17

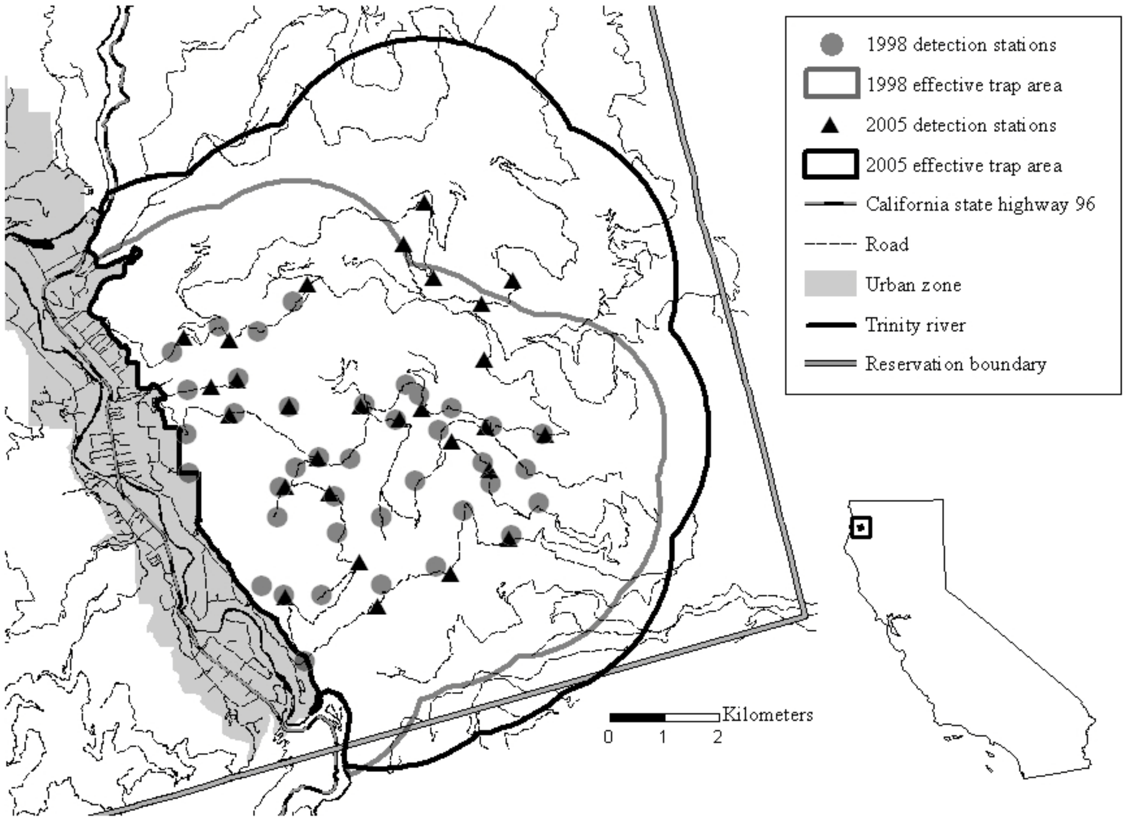

Figure 1. Locations of trap, camera, and track-plate stations and the effective trap areas in 1998 and 2005 on the Hoopa Valley Indian Reservation, Humboldt County, California, USA.

CHAPTER 2

FEASIBILITY OF USING GENETIC MARKING, PASSIVE INTEGRATED TRANSPONDERS, AND REMOTE-DIGITAL PHOTOGRAPHY TO MONITOR FISHER POPULATIONS

Abstract

We developed and tested methods of collecting mark-recapture data using genetic marking, passive integrated transponder (PIT) tag technology, and digital, passive-infrared photography that could be used in a demographic monitoring protocol for imperiled fisher (*Martes pennanti)* populations in the Pacific states. We tested these methods in captive and field settings, and assessed cost-efficiencies. We sought to overcome the biases presented by ear-tag loss and the cost constraints of a mark-recapture design using only livetrapping or genetic techniques to identify individuals. A 10-loci marker system with a mean observed heterozygosity of 55% was deemed optimal for our fisher population in northern California. Twenty-eight prototype, static hair snares yielded 66 fisher hair samples over a 14-day sampling period. Nineteen of 20 randomly selected samples had sufficient DNA to yield individual identification of 4 fishers. A single-sample hair snare proved ineffective in captive and field settings because of an unexpected behavioral response and low numbers of guard-hair follicles collected per sample. The PIT tag reader was 100% effective in recording visits of marked fishers in the captive (n=52 fisher visits) and field (n=10 fisher visits) settings. The remote-digital camera was 98% (51/52) and 100% (10/10) effective in photographing fisher visits in the captive and field settings, respectively. Projected costs of collecting six years of mark-recapture data across our 362-km^2 study area using a traditional live-trapping with PIT

tags as marks; using genetic marking; and using live-trapping and genetic marking in combination with PIT tag technology and digital, passive-infrared photography in an effort to offset costs of genetic analyses were $137,160, $329,880, and $327,160, respectively. The comparatively high cost of PIT tag reading equipment and genetic analyses makes the use these methods dramatically more expensive and yield less demographic data compared to using a traditional mark-recapture approach using only live-trapping.

Introduction

Long-term population monitoring has been an important component of the conservation effort for northern spotted owls (*Strix occidentalis caurina*) in the Pacific states (Anthony et al. 2006, Dugger et al. 2005). A similar need to collect long-term demographic data exists for fisher (*Martes pennanti*) populations because of their restricted range, small population sizes, and candidate status for federal endangered species listing in the Pacific states (U.S. Fish and Wildlife Service 2004). Further, tools to measure the success of re-introduction and other conservation efforts, determine the effects of changes in forest and fuels management practices, and monitor general population trends will assist land managers.

Camera stations (Cutler and Swann 1999) and genetic analyses (Waits and Paetkau 2005) have been widely used as methods to identify study animals and to monitor general population trends. These methods provide great potential for accurate population monitoring for fishers because of the ease of capturing and re-sighting individuals (Yaeger 2005, Zielinski et al. 2006, Jordan 2007). However, each of these methods has limitations. Ear-tags have been used for individual identification, but retention presents a problem when using remote camera stations over a long time period (Pollock et al. 1990, York and Fuller 1997, Jordan 2007). Individuals with missing tags cannot be definitively identified and could be mistaken for untagged animals. An advantage of identifying individual animals using DNA samples and genetic techniques (hereafter genetic tagging) over traditional mark-recapture or mark-resight methods, particularly with fishers, is the elimination of this dependency for ear-tag retention. However, genetic analyses can become cost prohibitive, particularly when considering the number of genetic samples collected during a long-term population-monitoring effort (e.g. $\geq$ 6 years, based on fisher life expectancy).

As an alternative marking technique, passive integrated transponder (PIT) tag technology has been used as an effective method of marking individuals and monitoring long-term population trends (York and Fuller 1997, Gibbons and Andrews 2004). Use of PIT tags serve as a means for reliably identifying marked individuals and potentially reduce the cost of genetic analyses during a long-term population monitoring study. A PIT tag reader used in conjunction with a hair-snare device could serve to identify previously marked fishers and eliminate the need to submit all hair samples for individual identification, thus reducing cost of genetic analyses.

The goals of this study were to develop and test methods of collecting mark-recapture data using genetic marking, PIT tag technology, and digital, passive-infrared photography that could be used in a demographic monitoring protocol for imperiled fisher populations in the Pacific states. We tested these methods in captive and field settings, and assessed cost-efficiencies.

Study Area

Captive trials used one adult-male fisher at the Humboldt State University Game Pens Facility in Arcata, California, USA. The fisher pen was 10.75 m x 3.75 m x 3 m (length x width x height) built of 2.5-cm square wire mesh and included a 1.2 m x 1.2 m x 0.6 m (length x width x height) wooden nest box, two 1.25-m long by 20.3-cm diameter plastic pipes, an old tire, and a small covered pavilion (1.25 m x 1.25 m) for cover. The pen was cleaned daily and the fisher was fed a diet of mink chow, hard boiled eggs, and occasionally fish, mice, or rats which were placed in the detection device during testing.

Field trials occurred on the Hoopa Valley Indian Reservation in northeastern Humboldt County, California. Elevations range from 76 m to 1170 m. The landscape is a heterogeneous mix of habitats, dominated by Douglas fir (*Pseudotsuga menziesii*) and tanoak (*Lithocarpus densiflorus*). Most of the forested habitat is classified as montane hardwood-conifer or Douglas fir (Mayer and Laudenslayer 1988). The field trials were conducted on a 53-km^2 portion of the southeastern quarter of the Reservation.

Methods

Intensive live-trapping efforts occurred between 8 December 2004 and 10 March 2005 and between 6 September 2005 and 2 March 2006 to collect samples for genetic analyses and mark each anesthetized fisher with a sterile 134.2 kHz passive integrated transponder (PIT tag, 134.2 kHz Super Tag, Sterile, Biomark, Inc., Boise, Idaho, USA) and with uniquely colored plastic ear tags (cut to 2.4 cm long x 1.0 cm wide, Nasco Standard Rototag Blank, Nasco, Modesto, California, USA) in both ear pinnae. All fishers were released after recovery from anesthesia at their sites of capture. Our methods were approved by the Institutional Animal Care and Use Committee of Humboldt State University, protocols 05/06.W.44.A (captive trials) and 04104.W.42.A (field trials).

Genetic material collected from tissue from 72 individual fishers was used to develop and test population-specific genetic markers for individual identification by Wildlife Genetics International (Nelson, British Columbia, Canada; Jordan 2007, Jordan et al. 2007). All samples were extracted using QIAGEN's DNeasy Tissue kits (QIAGEN Inc., Valencia, California, USA). We chose markers with the highest observed heterozygosity for the Hoopa population from a suite of markers previously documented using Genepop (version 3.4; Raymond & Rousset 1995, Kyle et al. 2001, Jordan et al. 2007). We empirically tested the marker system's performance by creating a mismatch distribution and looking at the number of pairs of genotypes that match at all but 1 or 2 markers (1MM or 2MM pairs).

We used prototype, static, hair snares to assess the feasibility of identifying individuals from remotely collected hair samples and to assess number of samples we might collect during a Reservation-wide effort for use in cost calculations. Hair samples were collected on mouse and insect glue traps (model 100 Series, AP&G Co., Brooklyn, New York, USA) taped to a 1.3-cm piece of wooden doweling inserted into the entry way to a track-plate box (Fowler and Golightly 1994, Zielinski 1995, Zielinski et al. 2006). Two additional pieces of doweling were used to block off enough of the entrance to increase the likelihood a fisher passed through the opening under and contacting the glue strip.

Twenty-eight locations were sampled for 14 days between 20 March and 13 April 2005 using hair snares within the study area. Location selection was based on home range estimates of adult female fishers (Yaeger 2005), ensuring ≥1 hair snare was located in each potential fisher home range (Karanth and Nichols 1998, Maffei et al. 2004, Silver et al. 2004). Hair samples were rated based on the number of guard hair follicles in the sample (poor=1-3, fair=4-6, good=7-10, and excellent>10) that was directly related to the likelihood of successful individual fisher identification (Tredick et al. 2007). Twenty samples, five of each quality rating, were randomly selected and submitted to Wildlife Genetics International (Nelson, British Columbia, Canada) for individual fisher identification.

We designed a sampling device which included a single-sample hair-snare, a PIT tag reader (model FS2001 ISO Reader and custom 45.7-cm interior diameter circular antenna, Biomark, Inc., Boise, Idaho, USA), and a digital, passive-infrared-triggered camera (model STC-WD3, Stealth Cam, Bedford, Texas, USA). The hair-snare, PIT tag reader, and camera were mounted inside a 45.7-cm interior diameter and 152-cm long corrugated, smooth-wall interior, polyethylene drainage pipe (model N-12, Advanced Drainage Systems, Inc, Hilliard, Ohio, USA).

The hair snare was a 1.27-cm thick plywood divider at the entrance of the device with an oval opening 8.5 cm high and 18 cm wide (Figure 2). A 25-cm long and 2-cm wide piece of steel strapping was mounted so it could be bent down in front of the oval opening. A 2-cm wide by 13.5-cm long piece of mouse and insect glue trap material (model 100 Series, AP&G Co., Brooklyn, New York, USA) was duct-taped to the steel strapping. A fisher entered the device through the oval opening, pushing against the bowed steel strapping and causing the tension in the strapping to trigger the snare and the strapping to bend above the oval opening while collecting hair fixed to the glue trap. The intent of the design was to allow for the collection of only one hair sample between visits by the researcher.

The PIT tag reader and camera recorded the date and time each time they were triggered. The camera documented the visit of a fisher, whether marked or unmarked. The date and time of each fisher visit recorded by the camera were compared to the PIT tag reader data to determine marked or unmarked status of the visiting fisher. These data allowed us to determine if the hair sample was from an unmarked animal and therefore whether or not the hair needed to be submitted for genetic identification. Hair samples collected from unmarked individuals could undergo genetic analyses to “mark” them and begin building capture histories before they were live-trapped and implanted with PIT tags. Total equipment cost for one device was $4,850 and required approximately 20 hours to construct.

Four sampling devices were constructed to test the efficacy of the hair snare, PIT tag reader, and camera. We documented visits by the captive fisher using a black and white, weatherproof, night-capable digital video camera (Extreme CCTV, Model EX26NX, Burnaby, British Columbia, Canada). The video camera was connected to a motion sensitive digital video recorder (DVR; Seabyrd Technologies, Arcata, California, USA; Creel 2007) that recorded each visit’s time and date. This allowed us to determine if each of the elements in the device appropriately recorded each fisher visit. A fisher visit to the device was defined the captive fisher’s shoulders passing into the device beyond the hair snare as recorded by the DVR. The device was baited with the food items usually given to the captive fisher. The captive fisher was marked with a PIT tag and with uniquely colored plastic ear tags in both ear pinnae.

We calculated the proportion of fisher visits to the device recorded by the PIT tag reader and camera and the presence of a hair sample to determine the effectiveness of each element. Effectiveness of the hair snare was rated, as above, based on the number of guard-hair follicles in the sample (Tredick et al. 2007). We note that the number of visits to a station recorded by the hair snare were lower than those recorded by the other devices because of the need to physically reset the hair snare between fisher visits.

Four locations on the Reservation were selected for the devices in a field setting. Sites were selected based on past trapping experience throughout the study area in order to increase the likelihood of fisher visits. The video camera and DVR were not used in the field trials. Rather, we calculated the total number of fisher visits recorded by any one of the device elements. Then we calculated the proportion of fisher visits recorded by the PIT tag reader and camera and the presence of a hair sample to determine effectiveness. The devices were operational at each of the 4 sites for 31 nights, checked every 2 to 3 days, and baited with a chicken leg.

We conducted a cost-efficiency analyses considering for use in a long-term, Reservation-wide (362-km^2), population monitoring study. We compared cost of a traditional livetrapping-based mark-recapture design using PIT tags as marks; a mark-recapture design using genetic marking; and a mark-recapture design using livetrapping and genetic marking in combination with PIT tag technology and digital, passive-infrared photography in an effort to offset costs of genetic analyses.

Results

A 10-loci marker system (Mp0175, Mp0144, Mp0247, Mp0182, Ggu101, Lut604, Mp0114, Mp 0085, Ma19, and Mp0059) with a mean observed heterozygosity of 55%

was deemed optimal for the Hoopa fisher population. We found no 1MM-pairs and 4 2MM-pairs using a mismatch distribution based on the 72 individual fisher samples.

Sixty-six fisher hair samples were collected using the prototype, static hair snares in 2005. We collected 0.34 hair samples per snare per check by the researcher. Forty (61%) of the samples were rated fair or higher quality. Nineteen of the 20 randomly selected, remotely collected hair samples rendered individual identifications of 4 fishers using the 10-marker system. Only 1 sample, rated as poor, failed to provide an individual identification.

The captive fisher was recorded entering the device on 52 occasions between 17 May and 14 June 2006. The captive fisher consistently removed the glue strip of the hair snare with his teeth (a behavior we rarely observed in the field); thus the device did not reliably collect hair. The PIT tag reader recorded each of the 52 visits. The digital camera photographed 51 of the 52 visits (98% of the visits). Of the 51 photographs, fisher was identified in all 51 (98% of the visits), marked or unmarked status based on the presence of ear tags was determined in 48 (92% of the visits), and individual identification (from ear tags) was determined in 43 (82% of the visits).

In the field trials, 13 fisher visits (10 marked, 3 unmarked) occurred between 5 March and 2 April 2006. The hair snare yielded samples on 12 of the 13 visits (92% of the visits). Of the 12 samples, 8 (67%) were rated poor, 3 (25%) fair, and 1 (8%) excellent. The PIT tag readers recorded each of the 10 marked-fisher visits. The camera photographed all of the 13 visits. Of the 13 visits documented by the camera, species was identified in all 13, marked or unmarked status was determined in 11 (84% of the visits), and individual identification was determined in 6 of the 10 marked-fisher visit (60% of the visits).

To effectively conduct a long-term population monitoring study across the entire Reservation we calculated that we needed 152 stations used over a 22-day sampling period. Based on our prototype hair snare results, 564 hair samples would be collected annually. DNA extraction and genotyping using the necessary 10-loci markers required for individual fisher identification for the Hoopa population has been estimated to cost $75 per sample (Wildlife Genetics International, Nelson, British Columbia, Canada), resulting in a total cost for analyzing 564 samples to be $42,300.

The purchase of 31 devices would allow for 152 sites to be surveyed for the 22-day sampling period over a 110-day field effort. The cost of the 31 devices at $4,850 per device would be $150,350 plus 20 hours of labor to construct each device. Based on remote camera mark-resight efforts conducted in 1998 and 2005, 22% of fishers visiting a camera station were unmarked. Thus, the use of the PIT tag reader in each device would reduce the number of hair samples requiring genetic identification collected annually from 564 to 124. Therefore, genetic analyses expenses are reduced from $42,300 to $9,300 with the use of a PIT tag reader used with each hair-snaring device.

A traditional live-trapping-based, mark-recapture design using PIT tags as marks during the 110-day field effort would cost $17,760 in personnel (assuming $15/hr), $2,600 in transportation, and $10,000 in equipment for the first year. Equipment costs would be $1,000 in subsequent years and other costs would remain the same. Thus, a six-year effort would cost $137,160 (Table 7).

A mark-recapture design using genetic marking would require half the personnel and transportation costs, as hair snares would only be checked every other day by the researcher rather than every day required of traps. Equipment costs would be similar to the livetrapping effort, $10,000 the first year and $1,000 subsequent years. As calculated above, genetic analyses would costs $42,300 annually. Thus, a six-year effort would cost $329,880 (Table 7).

A mark-recapture design using genetic marking in combination with PIT tag technology and digital, passive-infrared photography using an intensive trapping effort the first year followed by five years of recapture data collection using sampling devices and limited and focused trapping would require the same level of effort as the live-trapping design above the first year and three quarters the effort in subsequent years. The three quarter estimate was based on checking sampling devices every other day but occasionally needing to complete a focused trapping effort to mark previously unmarked animals, requiring checking traps every day when traps are in place. Thus, personnel and transportation costs would be $17,760 and $2,600 the first year and $13,320 and $1,950 in subsequent years, respectively. Equipment costs would be $10,000 the first year during the trapping effort and $160,650 for the materials ($150,350) and labor ($9,300) to construct the sampling devices and equipment ($1,000) for focused trapping. As calculated above, genetic analyses would costs $9,300 annually. Thus a six-year effort using our proposed methods would cost $327,160 (Table 7).

Discussion

Genetic marking, PIT tag technology and remote cameras provide significant opportunity for studying population trends over time for low density or rare mammals, including the fisher. Long-term population data could be used to measure the success or failure of fisher re-introduction efforts, determine the effects of changes in forest management practices, and monitor general population trends. Our 95% success rate in identifying individual fishers from remotely collected hair samples using a 10-loci marker system suggests that a mark-recapture approach using only hair snares is feasible.

Unfortunately the single-sample hair snare was not effective. We attribute the failure in the captive trials to the fisher removing the glue strip from the snare with his teeth. However, a large percentage of the samples collected in the field were of poor quality and thus had a reduced likelihood of successful genetic analysis. Since only 33% of our field samples were of fair or higher quality, improvements on the single sample hair snare device need to be achieved. The prototype, static hair snare device obtained higher quality samples with 61% fair or higher quality.

The use of a PIT tag system working in concert with a hair-snaring device and camera system has virtually no cost-saving benefit over using hair snares alone. The comparatively high cost of PIT tag reading equipment made the technique cost-prohibitive. We would recommend the use of PIT tag reading equipment in a long-term population monitoring study if a manager has already committed to using a mark-recapture approach using genetic tagging, particularly if the cost of PIT tag reading equipment declines in the future. Additionally, this method could prove useful if a manager needs to address issues of invasiveness and trap avoidance in developing a monitoring approach. The assumption of equal sightability of marked and unmarked animals during the recapture effort can be problematic for studies using a mark-recapture approach using only traps (Minta and Mangel 1989). Using more than one capture method (e.g. traps used for the initial capture effort and PIT tag readers used for the recapture effort) would contribute to meeting this assumption.

An alternative approach would be to remove the hair snare from the sampling device and not collect genetic data. Thus, only the remote camera would record fisher visits and the PIT tag reader would assess marked status. This approach would not allow for the development of capture histories of unmarked animals using genetics prior to being PIT tagged. However, if a large proportion of the population was marked during the initial trapping effort, not marking these individuals prior to capture would not significantly impact model results (Lebreton et al. 1992). Both cameras and PIT tags could also be used to simultaneously monitor other mesocarnivores for relatively little additional cost.

An important consideration in the development of a long-term, fisher monitoring protocol is the varying amounts of demographic data available from the different sampling methods. Live-trapping allows for individual age assessment (Strickland et al. 1982) and possibly assessment of annual female reproductive rates (Frost et al. 1999). These data are not available from a non-invasive, genetic tagging method used independent of live-trapping. Lambda and survival estimates based solely on genetic identification may be biased significantly downward due to dispersal of juvenile fisher off the study area (Burnham et al. 1996, Forsman et al. 2002, Anthony et al. 2006). The collection of age estimates allows for the calculation of non-juvenile lambda and survival estimates, removing the potential bias of juvenile dispersal.

Management Implications

Developing a sound demographic monitoring program to be used long term could be a powerful tool for the basis of future fisher population and habitat management decisions, especially when planning for species recovery in areas of its range where it has been extirpated. PIT tag technology has exceptional promise as long-term marking technique for a host of forest carnivores, and does not appear to suffer from the limitations of many external marks. However, cost of PIT tag technology still remains prohibitively high for many managers, particularly in the context of a long-term population monitoring program requiring the purchase of many sampling devices. Traditional mark-recapture continues to be the least expensive method of collecting long-term population data and provides the most demographic information. However, if trap avoidance is of concern or live-trapping is not an option and the effects of juvenile dispersal is not considered a significant bias, we would recommend the use of genetic marking. If a manager is developing a simultaneous monitoring program for a number of forest-carnivore species, we would recommend the use of PIT tag technology following an initial intensive trapping and marking effort. We would also recommend the inclusion of either hair snares or PIT tag reading stations on a limited basis in the design of a primarily traditional mark-recapture based project for the purpose of increasing recapture probabilities by providing an alternate method of “capturing” trap shy individuals.

Table 7. Cost comparisons of 3 techniques to collect fisher population demographic data as part of a mark-recapture design on the 362-km^2 Hoopa Valley Indian Reservation, California during a 110-day field effort conducted annually for 6 years. Techniques included traditional live-trapping using passive integrated transponder (PIT) tags as marks, genetic marking, and a combination of live-trapping and genetic marking in combination with PIT tag technology and digital, passive-infrared photography in an effort to offset costs of genetic analyses.

Expense category	Live-trapping only	Genetic marking only	Live-trapping and genetic marking with PIT tag readers and cameras
Personnel ($15/hr)	$106,560	$53,280	$93,660
Transportation	$15,600	$7,800	$12,350
Equipment	$15,000	$15,000	$165,350
Genetic analyses	$0	$253,800	$55,800
Total	$137,160	$329,880	$327,160

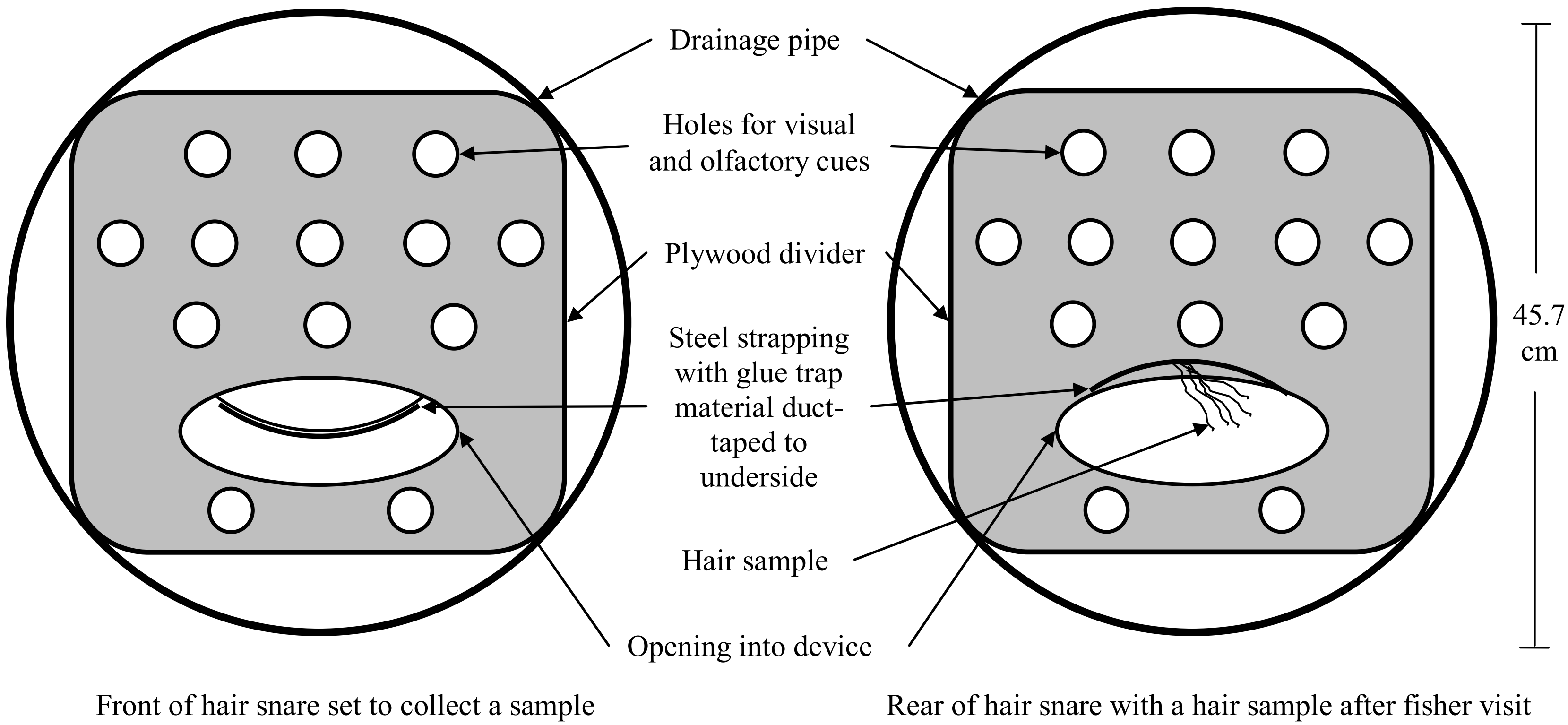

Figure 2. Single-sample hair snare using mouse and insect glue trap material duct-taped to piece of steel strapping mounted to a plywood divider inside a 45.7 cm interior diameter corrugated, smooth wall interior, polyethylene drainage pipe.

CHAPTER 3

REPRODUCTION, RECRUITMENT, AND DISPERSAL OF FISHERS IN A MANAGED, DOUGLAS-FIR DOMINATED FOREST IN NORTHWESTERN CALIFORNIA

Abstract

Many demographic parameters of imperiled fishers (*Martes pennanti*) in the Pacific Northwest remain poorly understood but are necessary to develop conservation strategies; herein we report on fisher reproduction, recruitment, and dispersal on the Hoopa Valley Indian Reservation, California to help fill key knowledge gaps. Forty radiocollared, breeding-age females exhibited denning behavior on 80 of 92 (87%) opportunities between 2005 and 2011. Twenty-eight female fishers weaned offspring in 55 of 85 (65%) adequately monitored denning opportunities. Two-year old female fishers were less likely than older females to den and wean kits. We counted 52, and extracted and marked 51, kits comprising 28 litters of 19 females between 2005 and 2008. Average litter size was 1.9 kits (27F, 24M, 1 unk.) 4-12 weeks post birth. Mean distances between natal dens and home range centroids of 14 radio-collared dispersing juvenile fishers (12F, 2M) was 4.0 km (range = 0.8-18.0 km) for 7 females vs. 1.3 km for a male. Estimated recruitment of juveniles/adult female was 0.25-0.29. Our results suggested that mangers should work toward increasing female survival rates and consider translocations to increase and expand existing fisher populations.

Introduction

Fisher (*Martes pennanti*) populations in the Pacific states have suffered significant declines and range contractions over the last 2 centuries (Aubry and Lewis

2003; Buck et al. 1994; Gibilisco 1994; Powell and Zielinski 1994; Zielinski et al. 2004). In California, fishers occur in less than 50% of their historic range in 2 isolated populations (Zielinski et al. 1995, 2005). These declines have been linked to over-trapping for furs, predator and pest control campaigns, and loss and fragmentation of forested habitats by logging, fire, and farming (Douglas and Strickland 1987; Powell 1993; Powell and Zielinski 1994).

Small and isolated populations have been identified as the most challenging threat to the conservation of fishers in the Pacific states (Lofroth et al. 2010), and the 2 extant fisher populations in California have failed to recover from range contractions (Aubry and Lewis 2003; Matthews et al. 2011; Spencer et al. 2011; Zielinski et al. 1995, 2005). Thus, current sizes and distributions of fisher populations in the Pacific states place them at inherently higher risk of extinction due to potential stochastic phenomena (Lofroth et al. 2010; Shaffer 1981). The long-term conservation strategy outlined by the Interagency Fisher Biology Team involves protecting existing populations and encouraging them to expand beyond their current boundaries (Lofroth et al. 2010).

Critical rates of reproduction, recruitment, and dispersal are needed to adequately model population dynamics, sub- and metapopulation extinction risk, the ability of populations to recolonize suitable habitat, and gene flow (Brown and Kodric-Brown 1977; Hanski 1999; Krebs 2009; Levins 1970). These are particularly important variables to understand when considering subpopulations that are increasingly isolated by habitat change and fragmentation (Hanski 1999; Levins 1970; Macdonald and Johnson 2001; Stephens et al. 2004). Only one study (Reno et al. 2008) has reported reproductive parameters for fishers in an eastern portion of the northern California extant population.

We are the first to report reproductive parameters for a western portion of the population. The results of these 2 studies representing disparate portions of the population are key elements in implementing conservation strategies for this isolated population.

Our goal was to quantify reproduction, recruitment, and dispersal of fishers in managed, Douglas-fir dominated forests on the 366-km^2 Hoopa Valley Indian Reservation in northwestern California (hereafter, Hoopa; Fig. 1) to assist in conservation efforts. In 2005, fisher density in Hoopa declined to about 14/100 km^2 from 52/100 km^2 in 1998 (Matthews et al. 2011). The causes of the decline are unknown, but we predicted that the population could recover. Thus, we hypothesized Hoopa fishers would exhibit higher reproductive, higher recruitment, and lower dispersal rate than for other unharvested fisher populations due to fisher age structure and relative habitat quality during 2005.

Study Area

Our study area was located within the Klamath physiographic province (Küchler 1977); elevations ranged between 98 and 1,170 m. Mean daily maximum and minimum temperatures were 22 and 6^{o}C, respectively, and mean annual precipitation, primarily rain, was 138 cm (National Climate Data Center, http://www.ncdc.noaa.gov/oa/ncdc.html, accessed 2 November 2011). Hoopa supported a human population of approximately 2,600 (Hoopa Valley Tribe, http://www.hoopa-nsn.gov/documents/2000Census.pdf, accessed 2 November 2011), to whom fishers are culturally significant and not harvested outside rare harvest for ceremonial regalia. Forests within the study area generally had an overstory dominated by Douglas-fir (*Pseudotsuga menziesii*) and a midstory dominated by hardwood trees including tanoak

(*Lithocarpus densiflorus*), madrone (*Arbutus menziesii*), Oregon white oak (*Quercus garryana*), California black oak (*Q. kelloggii*), and canyon live oak (*Q. chrysolepis*). Hardwood stands occurred in localized areas, whereas at higher elevations Douglas-fir was replaced by white fir (*Abies concolor*) and pine (*Pinus* spp.). Hoopa was bisected by the Trinity River into eastern and western portions.

Past and current timber harvests created a mix of mature-old growth and early seral forest. Prior to 1990, clear-cuts averaged 12-20 ha, although cuts up to 276 ha occurred. From 1960 to 1980, 30% of Hoopa was harvested, averaging over 500 ha cut/year across multiple clear-cuts. Under the direction of the Hoopa Tribe's Forest Management Plan between 1994 and 2010, tribal forest managers harvested 23,196 m^3 (9.83 million board feet) on approximately 150 ha annually, pre-commercial thinned approximately 165 ha, early released 100-175 ha, and burned for cultural-resource management 6-40 ha. Harvest was implemented using regeneration methods with green tree and snag retention on small (<10 ha) modified clear-cuts. A minor amount of commercial thinning and single-tree and group selection was also employed.

Methods

We captured fishers using cage-type live traps (model 207, Tomahawk Live Trap Company, Tomahawk, Wisconsin) baited with chicken and modified with a plywood cubby box (Seglund 1995; Wilbert 1992). Traps were not set during spring and early summer to avoid capture of lactating females or young kits. Captured fishers were anesthetized with ketamine hydrochloride (40 mg/kg) and diazepam (0.25 mg/kg) and handled using standard protocols (Aubry and Raley 1996; Yaeger 2005).

We subcutaneously injected each anesthetized fisher with a sterile 134.2 kHz passive integrated transponder for future identification (PIT tag, 134.2 kHz Super Tag, Sterile, Biomark, Inc., Boise, Idaho). All non-juvenile female fishers (based on body and dentition condition at capture) were fitted with radiotransmitters (Holohil model MI-2, Holohil Systems Ltd., Carp, Ontario, Canada or Telonics model MOD80, Telonics Inc., Mesa, Arizona). We removed a tooth, P1, for age determination by cementum annuli (Matson's Laboratory, Milltown, Montana; Arthur et al. 1992; Poole et al. 1994; Strickland et al. 1982). Each anesthetized fisher recovered for approximately 1 hour in a plywood cubby box. We released all recovered fishers at their sites of capture.

We used ground-based radiotelemetry techniques using a handheld receiver (model TR-4, Telonics, Inc., Mesa, Arizona or model R1000, Communications Specialists, Inc., Orange, California) and a 4-element antenna (model RA-14, Telonics, Inc., Mesa, Arizona) to estimate locations of fishers and identify den structures. Dens were live or dead trees used by reproductive females for birthing and nursing kits. Locations of inactive fishers were obtained using the loudest-signal method and by hiking to the source of the signal (Springer 1979). The observer circled the collared fisher until an individual tree or other structure was isolated and identified as the most likely position of the transmitter. The location of the structure was recorded using a handheld GPS unit (Garmin Rino 120, Olathe, Kansas) with a 3-dimensional fix with an estimated error $\leq$10 meters.

During the early den season, 1 March-15 April, we attempted to locate inactive female fishers 4 or 5 days/week to observe reuse of sites indicating den establishment. Denning behavior was identified by a sudden change in behavior from using numerous rest sites per week across the majority of the home range to more restricted movements in a small portion of the home range and repeated use of the same structure while inactive (Aubry and Raley 2006).

We defined denning opportunities as the total of individual, breeding-age (≥2 years old) female fishers monitored across all den seasons (March-June; Powell 1993). We defined a den as a structure used two or more times in succession over 3 or more days by an adult female within the den season (Aubry and Raley 2006; Truex et al. 1998). Dens were classified as: 1) natal - location where parturition took place or 2) maternal - any den used after the natal den and before the kits were weaned (Reno et al. 2008; Truex et al. 1998; Weir 2003). Kits were considered weaned at 10 weeks of age or by 31 May if no parturition date was established (Powell 1993). We determined if denning and weaning rates were independent of adult female age class using Fisher's exact tests using the BASE package in Program R version 2.13.2 (R Development Core Team 2011).

We investigated dens 4-12 weeks after kit parturition when the adult female was not present. Den investigations involved climbing the den tree using a single-rope technique or tree-climbing spurs and a flip rope (Jepson 2000). The presence and number of kits were determined visually, often with the aid of a Burrow Camera System (Sandpiper Technologies, Inc., Manteca, California). We extracted kits from dens without physically altering the den structure. Kits were not anesthetized and handled for <60 minutes. Kits were warmed with a towel and heat packs if necessary. Kits were marked with a PIT tag for future identification. Tissue, ectoparasites, and swabs of ocular exudate, nasal exudates and feces were collected for genetic and disease analyses. Morphometric measurements, an assessment of overall body condition, and condition of eyes (open or closed) were recorded. Following handling procedures, we placed kits back in the den structure. We closely monitored the adult female over the next 2-3 days to assess the possibility of den or litter abandonment. During the subsequent fall capture period, we attempted to capture marked kits to attach radiocollars. Techniques used to capture, handle, and mark juvenile fishers were the same as those used for adults.

Our capture and handling methods followed American Society of Mammalogists (ASM) guidelines (Sikes et al. 2011) and were approved by the Institutional Animal Care and Use Committee of Humboldt State University, protocol 04104.W.42.A. Our den investigation techniques were also approved under a National Environmental Policy Act (NEPA) compliance checklist (in cooperation with G. Falxa, U.S. Fish and Wildlife Service, Arcata, California). NEPA compliance was based on the condition that if signs of litter abandonment were detected, we were to terminate den investigations and marking of kits immediately.

We estimated recruitment (number of known births less known deaths) at 3 time intervals: 1) at weaning, 2) after the fall/winter live-trapping period, and 3) at home-range establishment (often by April of their 2nd year). We calculated recruitment at home range establishment for both genders using 2 methods, representing maximum and minimum recruitment estimates. First, we estimated the proportion of radiocollared juveniles that were known to have established home ranges from the number of all juveniles radiocollared during the fall/winter live-trapping period. This estimate of recruitment is a maximum, because some PIT-tagged kits likely died before our recapture and collaring efforts. Second, we calculated recruitment as the proportion of radiocollared juveniles that established home ranges from the number of kits PIT-tagged in dens. This estimate is a minimum because some kits survived and were not captured and radiocollared.

We determined the timing of dispersal movements, dispersal distance, and fate by conducting telemetry monitoring of each radiocollared juvenile. We used ground-based and aerial telemetry techniques to track fisher movements and dispersal patterns within and outside the mother's home range, which we defined as the natal range. All radiocollared animals were located 1 or 2 days/week from the ground. A Cessna 182, model 1000 receiver (Advanced Telemetry Systems, Inc., Isanti, Minnesota), and 4-element Yagi antennas (Advanced Telemetry Systems, Inc., Isanti, Minnesota) were used in aerial telemetry efforts within 30 days following our inability to locate a radiocollared animal from the ground.

We defined dispersal as a juvenile's movement, measured as a straight-line distance, from its natal den (Bennetts et al. 2001). We plotted all locations and measured dispersal distances using ArcGIS 9.3 (ESRI, Redlands, California USA). We measured dispersal distances as the straight-line distance between the natal den (or the first maternal den if the natal den was not located) and 1) the farthest distant location recorded for each fisher (Arthur et al. 1993; York 1996) and 2) the centroid, or geometric center, of each fisher's 95% minimum convex polygon (MCP) home range estimate. Home range estimates were calculated using an average of 55 locations/fisher (range = 39-61) weekly, diurnal, ground-based telemetry and capture locations collected between 1 April and 31 March of the fisher's 2nd year (Weir and Corbould 2008). MCP estimates were used to compare our results to those of previous studies. Home range estimates and centroids were calculated using Home Range Extension and Tools (Blue Sky Telemetry, Aberfeldy, Scotland) and XTools Pro Shapes to Centroids tool (Data East, LLC, Novosibirsk, Russia) in ArcGIS 9.3. The date range was selected based on spatial data indicating fishers in Hoopa finished dispersing and began establishing a home range by mid-March of their 2nd year. We categorized these distances by 1) juveniles that lived to their 2nd birth day to establish a home range, 2) juveniles that died prior to establishing a home range, and 3) juveniles that were lost during dispersal (Arthur et al. 1993).

Results

We captured 179 individual fishers (94 F, 85 M) on 839 occasions during 15,215 trap-nights between 2004 and 2011. Forty radiocollared, breeding-age (≥2 years old) female fishers exhibited denning behavior in 80 of 92 (87%) opportunities to produce offspring during 7 den seasons between 2005-2011 (Table 1). Denning behavior was observed on average between 22 March and 25 May, starting as early as 9 March and as late as 7 April. Eighteen of 80 (23%) whelping and rearing episodes failed prior to kits being weaned (Table 1). Five of the failures were the result of the denning female being killed by a predator. An additional failure was the result of the denning female dying of either disease or poisoning. The ultimate cause of the remaining 12 failures could not be determined. Additionally, we were unable to adequately monitor 5 females during 7 denning opportunities closely enough to determine their fate because of limitations in field staffing (Table 1). Thus, 55 of 85 (65%) adequately monitored denning opportunities of 28 adult female fishers were successful in weaning at least one kit. Denning and weaning rates were not independent of adult female age class ($P < 0.001$, Table 1). Two-year old females were less likely than expected to den and wean. Females 3-5 years old were more likely than expected to den and weaned kits equal to expected. Females ≥ 6 years old were more likely than expected to den and wean.

Female fishers monitored from den initiation until weaning ($n = 28$) used a mean of 3.1 dens per den season (n = 53 denning opportunities, range = 1-6). Successive dens were located an average of 385 m apart (n = 138, range = 18-1728). Females monitored until weaning moved an average total distance between successive dens of 873 m (n = 52, range = 0-2459).

We counted 52 kits in 28 litters born to 19 females during 4 den seasons between 2005 and 2008. Average litter size at the time of extraction was 1.9 kits (27 F, 24 M, 1 unknown). We documented the deaths of 6 kits (3 female, 3 male) prior to weaning in 4 litters. Necropsy of a dead female kit from a litter of triplets, suggested death due to starvation. We suspected the death of a female kit marked in a den when she was not found with 2 male litter mates marked 12 days after the female. Two male kits were found dead in their den and a male and female kit were presumed to have died following the predation of their mothers. A female kit from another litter was successfully rescued from its den following the predation of its mother, raised in a captive setting, and released in her mother's home range. Data on the dispersal patterns of this kit are not reported here. One kit was observed but we were unable to extract it. The fate of it and its littermate (a female) was not determined because we were unable to monitor the adult female until weaning.

Forty-three kits (22F, 21M) were suspected to have weaned from 24 adequately monitored litters of 16 adult female fishers after accounting for known and suspected adult and kit mortalities (Table 2). Fourteen PIT-tagged juveniles (12F, 2M) were captured and radiocollared during fall live-trapping sessions. Thirteen PIT-tagged juveniles were captured between 15 September and 8 January during their birth year. One juvenile female was not captured during her birth year but was captured as a 1-year old on 28 January within her mother's home range. Nine of the 14 juveniles were captured within their mother's home range. The remaining 5 (4F, 1M) were captured outside of, but within 700 m of their mother's home range (mean = 353, range = 75-694). The later 5 were captured on 12 and 15 October, 5 and 20 November, and 8 January.

We were able to confirm home range establishment for 7 females and 1 male (57%). Of the 14 radiocollared juveniles, 2 females and 1 male died during dispersal prior to establishing home ranges. One female and 1 male died of unknown disease. The cause of the other female mortality was not determined. Three females dropped their radiocollars prior to establishing home ranges. Minimum and maximum recruitment rates (i.e., proportion of PIT-tagged juveniles successfully established a new home range) were 0.22 and 0.26 for juvenile females and 0.03 for juvenile males, or a total of 0.25-0.29 juveniles/adult female (Table 3).

Mean distances between natal dens and centroids newly established ranges for the 7 juvenile females was 4.0 km (range = 0.8-18.0); this distance for the male was 1.3 km. Only 2 females dispersed beyond and did not exhibit home range overlap with their mother's home range (dispersal distances of 3.6 and 18.0 km). The mean maximum distance between a natal den and the farthest distant location recorded for juvenile fishers (including those that died during dispersal or were lost from contact) was 6.7 km (range = 2.1-20.l) for females and 8.1 km (range = 5.9-10.3) for males.

Discussion

Parturition dates reported for fishers in the Pacific states range between the last 2 weeks in March through the 1st week in April, with a mean litter size of 1.8 (range 1-3; Aubry and Raley 2002; Truex et al. 1998). Fishers in the northeastern United States parturition dates range slightly earlier (3 March-1 April), with a mean litter size of 2.4 (range 1-4; Paragi et al. 1994; York 1996). Our average estimate of parturition of 22 March (range 9 March-7 April) with an average litter size of 1.9 (range 1-3) was similar to these findings. Parturition estimates and dates during which denning behavior is exhibited could prove useful if the need to develop seasonal management restrictions becomes necessary.

Our estimate of the proportion of monitored females that exhibited denning behavior (87%) was higher than estimates presented for other areas within the Pacific states. Truex et al. (1998) reported estimates of 50 and 54% of monitored females exhibiting evidence of reproduction (evidence of lactation at capture) in the north coast ranges of California and in the southern Sierra Nevada, respectively. Aubry and Raley (2006) found an average of 59% of monitored females exhibited denning behavior (use of dens, lactating, and/or females observed with kits) in the southern Cascade range of Oregon. Reno et al. (2008) reported an average annual estimate of 51% for 2 northern California study areas combined over a 2-year period (22% of collared fishers during year 1 and 80% of collared fishers during year 2). Weir and Corbould (2008) reported an average annual estimate of 69% in north-central British Columbia. Average rates of denning behavior observed for fishers in eastern North America were similar to those observed on other study sites in the Pacific states (range = 0.45–0.65; Arthur and Krohn 1991; Paragi et al.1994; York 1996).

We suspect higher denning rates in Hoopa compared to other Pacific populations could be a function of higher quality habitat at Hoopa, more accurate age determination, population age structure of study populations, or a combination of factors. Denning rate estimates presented by Truex et al. (1998), Aubry and Raley (2006), and Reno et al. (2008) were based on age estimates determined by subjective assessments of body and dentition condition. If these researchers classified juveniles as potential breeders they would have underestimated denning rates of breeding age females. In addition, the age structure of the population at the time of each study likely influenced denning and weaning rates. Populations weighted heavily to younger animals, such as harvested populations in eastern North America (Brown et al. 2006; York 1996), would be expected to have lower denning and weaning rates based on our finding that older females are more reproductively successful than younger females.

Not all reproductive efforts were successful. Although 87% of monitored adult females on our study site exhibited denning behavior, only 65% weaned at least 1 kit. Similarly, Aubry and Raley (2006) reported an average of 44% of females monitored successfully weaned ≥1 kit, although 59% exhibited denning behavior. These results demonstrate that evidence of reproduction does not necessarily reflect successful weaning of kit(s) or recruitment. Furthermore, younger adult female fishers are less likely than older females to den and successfully wean kits. Conservation strategies that increase female survival rates (e.g., females reaching ≥5 years of age on average) might be instrumental in efforts to increase populations of fishers.

Female fishers in Hoopa typically used more than 1 den structure during the den season, moving between 18 and 1,728 m between successive structures. Despite this distance, dens used within a season were located within a small, concentrated area of each adult female's home range. This concentrated use could be related to availability or spatial distribution of den and rest structures with favorable microclimates and/or food availability within a localized area of the home range. These factors warrant further investigation to determine their relationship to fecundity and to better plan management activities.

The unexpected female bias in our captured and radiocollared sample of kits might be attributed to male fishers dispersing earlier than females, prior to the start of live-trapping in early September. This hypothesis is supported by our live-trapping data. We captured a nearly equal ratio (22 females and 18 males) of marked and previously unmarked juvenile fishers during our capture efforts. Twelve of the 22 (55%) captured juvenile females and 2 of the 18 (11%) captured juvenile males were marked as kits. Our results support the theory that fishers have male-biased dispersal and female philopatry (Aubry et al. 2004; Aubry and Raley 2006; William et al. 2000). We suspect the male fishers we marked as kits dispersed prior to being captured as juveniles and those juveniles we did capture may have been animals that immigrated onto our study area. Juvenile male fishers might also experience a disproportionate level of mortality prior to or during dispersal, and these factors may confound our interpretations. Although live-trapping in late summer may have increased our chances of catching juvenile males, we avoided live-trapping during this period because of the influence of black bears (*Ursus americanus*) on trap success and concern over properly fitting radiocollars on juveniles.

Our hypothesis regarding the earlier dispersal of juvenile males was supported by subsequent live-trapping efforts. A single female kit and a single male kit were marked in maternal dens of different adult females on 2 May 2006 and 9 May 2008, respectively. During live-trapping efforts in mothers' home ranges the following fall/winter seasons, we failed to capture either of these juveniles. However, the female was captured and radiocollared within her mother's home range (871 m from her natal den) on 14 February 2011. Telemetry efforts demonstrated her home range to be located adjacent to, but largely outside of our study area. The male was captured on 6 November 2010, 31.6 km from his natal den on a neighboring, private timber company's ownership (Green Diamond Resource Company, Korbel, California) as part of an ongoing fisher translocation effort coordinated by California Department of Fish and Game. Future research efforts might consider repeating our methodology using GPS collars on juveniles over a larger study area or neighboring study areas, while simultaneously using hair snares across a large-scale systematic grid to potentially detect juveniles marked as kits but not captured and collared. These methods would provide a more complete picture of juvenile fisher dispersal patterns, an important consideration for translocations, dispersal corridor protections, and similar conservation measures.

The dispersal distances we recorded were generally shorter than those reported in the literature. However, many of these earlier studies only reported dispersal distances of animals that were successful in leaving their mothers' home ranges and establishing a home range at least some arbitrary distance from their mothers' home ranges, typically based on mean maximum distance moved by adults. In the Cascades Range of southern Oregon, Aubry and Raley (2006) reported mean female and male dispersal distances from den locations to established home range centers of 6 km (range 0-17, n = 4) and 29 km (range 7-55, n = 3), respectively. In north-central British Columbia, Weir and Corbould (2008) reported mean female and male dispersal distances from den locations to established home range centers of 16.7 (range 0.7-32.7, n = 2) and 41.3 (n = 1). In a Massachusetts fisher population, York (1996) reported mean maximum female and male dispersal distances of 37 km (range 12-107, n = 19) and 25 km (range 10-60, n = 10), respectively. Similarly Arthur et al. (1993) in Maine reported mean maximum female and male dispersal distances of 14.9 (range 7.5-22.6, n = 2) and 17.3 km (range 10.9-23.0, n = 4), respectively.

Our comparatively shorter dispersal distances might be related to a significant decline in the density of fishers in Hoopa between 1998 and 2005 (Matthews et al. 2011). Suitable habitat might have been available in close proximity to mothers' home ranges, resulting in comparatively short dispersal distances. We would have likely observed longer dispersal movements of males if we had collared a higher percentage of those we had marked as kits.

Male-biased dispersal and female philopatry have implications for western fisher management, considering existing populations have failed to expand into suitable, unoccupied habitat (Lofroth et al 2010). Fisher populations exhibiting male-biased dispersal and female philopatry are unlikely to expand unless lambda is well above 1, forcing some females to disperse greater distances (Matthysen 2005). Male fishers might disperse outside of the general population boundary, forced to return to find females or die without opportunity to breed. This supports the hypothesis that fishers need to be translocated in order to reach suitable, unoccupied habitat even short distances away from the population boundaries (Lewis and Stinson 1998).

Estimates of recruitment rates are lacking for most fisher populations in western North America (Lofroth et al. 2010). In central interior British Columbia, Weir and Corbould (2008) estimated an average fall recruitment rate of 0.58 juveniles/adult female, which is double our estimate. Such estimates provide potential insights into population growth; however, they must be viewed with caution as they were derived by piecing together various information sources (e.g., denning rates of females, telemetry and live-trapping data, and anecdotal field observations) and making assumptions about age-specific survival rates (Lofroth et al. 2010). Evaluating recruitment derived for fisher populations in eastern North America (e.g., Paragi et al 1994) provides limited insights into the dynamics of western populations because legal harvest of fishers in the east directly affects recruitment rates (Lofroth et al. 2010).

Estimates of female reproductive rates, recruitment, and juvenile dispersal are important elements in assessing the resilience of and drafting conservation strategies for imperiled carnivore populations (Weaver et al. 1996). Following an apparent 73% fisher population decline in Hoopa between 1998 and 2005 (Matthews et al. 2011), we found higher reproduction, higher recruitment, and lower dispersal rate than in other unharvested fisher populations. These rates demonstrate the resilience of this population and future research to calculate lambda will quantify the rate of recovery. However, large home-range size, low fecundity, and limited dispersal ability across open habitat make fishers sensitive to anthropogenic habitat alteration, such as extensive logging practiced at greater intensities outside Hoopa (Carroll et al. 1999; Powell and Zielinski 1994). Therefore, it is important for wildlife managers to consider fisher demographics when developing and implementing forest management operations and conservation strategies.

Table 8. Summary of female fisher (≥2 years old) denning and weaning rates by age class and year on the Hoopa Valley Indian Reservation, California, 2005-2010.

	Females monitored[a]	Denning[b]	Proportion denned[c]	Unknown fate[d]	Failed[e]	Weaned[f]	Proportion weaned[g]
Age class (years)							
2	13	6	0.46	3	2	1	0.10
3-5	52	48	0.92	3	12	33	0.67
≥6	27	26	0.96	1	4	21	0.81
Year							
2005	9	7	0.78	0	1	6	0.67
2006	12	9	0.75	0	1	8	0.67
2007	12	10	0.83	0	6	4	0.33
2008	15	15	1.00	1	4	10	0.71
2009	14	13	0.93	4	3	6	0.60
2010	15	11	0.73	0	2	9	0.60
2011	15	15	1.00	2	1	12	0.92
Total	92	80	0.87	7	18	55	0.65

[a] Number of females (≥2 years old) monitored
[b] Number of monitored females (≥2 years old) which exhibited denning behavior
[c] Number of denning females divided by number of monitored females
[d] Number of monitored females (≥2 years old) which exhibited denning behavior and whose fate was not determined
[e] Number of monitored females (≥2 years old) which exhibited denning behavior and ceased exhibiting denning behavior prior to weaning
[f] Number of monitored female fishers (≥2 years old) which exhibited denning behavior to weaning
[g] Number of monitored females which denned to weaning divided by the number of females monitored minus the number of monitored females with unknown fates

Table 9. Summary of female fisher (≥2 years old) kit production and birth rates by age class and year on the Hoopa Valley Indian Reservation, California, 2005-2008.

	Females monitored[a]	Weaned[b]	Kits[c]	Kits weaned per litter	Birth rate[d]
Age class (years)					
2	7	1	1	1.0	0.14
3-5	29	18	36	2.0	1.24
≥6	7	5	6	1.2	0.86
Year					
2005	9	6	9	1.5	1.00
2006	11	7	13	1.9	1.18
2007	10	2	3	1.5	0.30
2008	13	9	18	2.0	1.38
Total	43	24	43	1.8	1.00

[a] Number of females (≥2 years old) monitored, excluding females whose fate was not determined and females whose kits were not counted

[b] Number of monitored female fishers (≥2 years old) which exhibited denning behavior to weaning

[c] Number of kits counted and suspected to have weaned

[d] Number of kits counted and suspected to have weaned divided by the number of females monitored

Table 10. Dispersal distances of juvenile fishers that died during dispersal prior to establishing home ranges, Hoopa Valley Indian Reservation, California.

Fisher ID	Gender	Distance between den and location of carcass (km)	Distance between den and furthest location (km)
10078	F	7.7	7.7
10080	M	2.8	10.3
10134	F	3.9	4.6

Table 11. Dispersal distances of juvenile fishers that were lost from contact after dropping their radio collars, during dispersal, prior to establishing home ranges, Hoopa Valley Indian Reservation, California.

Fisher ID	Gender	Distance between den and location of dropped collar (km)	Distance between den and furthest location (km)
10037	F	10.3	13.3
10138	F	10.7	11.4
10144	F	0.8	2.1

Table 12. Dispersal distances of juvenile fishers that established home ranges, Hoopa Valley Indian Reservation, California.

Fisher ID	Gender	Distance between den and 95% MCP centroid (km)	Distance between den and furthest location (km)
10039	M	1.3	5.9
10043	F	3.6	5.1
10079	F	1.0	3.6
10106	F	2.7	4.7
10107	F	18.0	20.1
10110	F	1.0	2.8
10137	F	0.8	2.2
10147	F	1.0	2.9

Table 13. Potential recruitment to a hypothetical population of 100 female fishers based on summary data of female fisher (>2 years old) kit production on the Hoopa Valley Indian Reservation, California, 2005-2008.

	Proportion of population	Individual fishers
Adult females exhibiting denning behavior	0.87	87
Denning females successfully producing young	0.65	65
Average weaned litter size	1.80	117
Weaned young that are females	0.51	60
Weaned young that are males	0.49	57
Weaned females captured in fall/winter	0.55	33
Weaned males captured in fall/winter	0.10	6
Maximum recruitment		
Weaned females captured in fall/winter establishing home ranges[a]	0.78	26
Weaned males captured in fall/winter establishing home ranges	0.50	3
Minimum recruitment		
All weaned females establishing home ranges[a]	0.38	22
All weaned males establishing home ranges	0.05	3

[a] Excludes three radio-marked females that dropped radio collars and fate was unknown

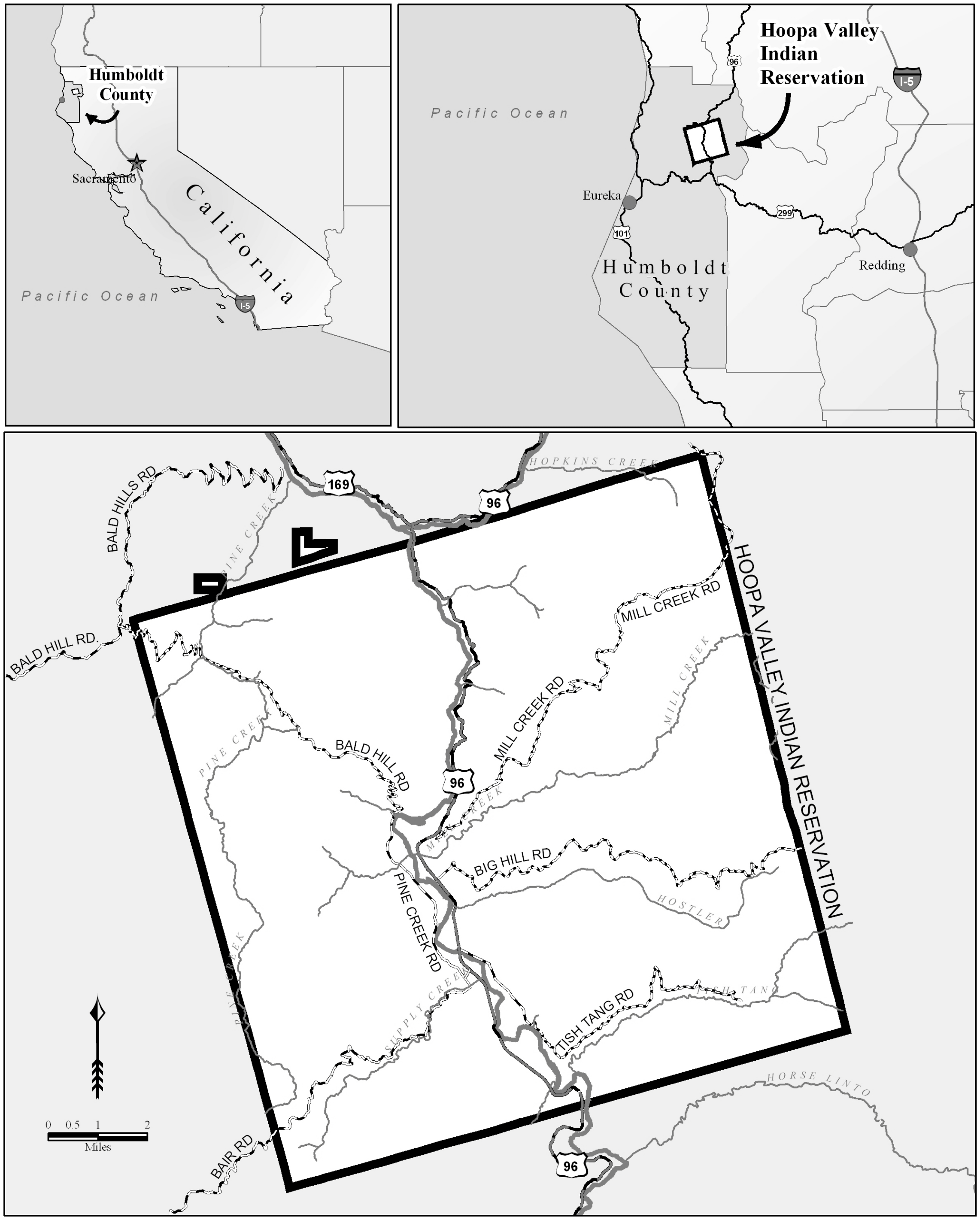

Figure 3. Hoopa Valley Indian Reservation, Humboldt County, California, USA.

CHAPTER 4

AN EVALUATION OF A WEANING INDEX FOR WILD FISHERS ON THE HOOPA VALLEY INDIAN RESERVATION, CALIFORNIA

Abstract

Conservation concern for fishers (*Martes pennanti*) in the Pacific states has highlighted a need to develop cost-effective methods of monitoring reproduction in extant and reintroduced fisher populations. We evaluated the efficacy of nipple size (width multiplied by height of the largest anterior nipple) as a predictive index of female fisher reproductive success in weaning at least 1 kit from females with known reproductive histories. Nipple size differed among classes (W=32.86, df = 2, p<0.001). Nipple sizes of non-breeders were smaller than those of attempted and current breeders (p<0.05). Nipple sizes of attempted and current breeders did not differ (p>0.05). We found the three reproductive classes were proportionately sampled throughout the capture seasons (F=0.0218, df=2, p=0.9784). We found some group separation using a quadratic discriminant analysis and jackknife cross-validation in our effort to develop a predictive index for use in assigning reproductive status to females with unknown reproductive histories. Our overall correct classification rate was 80.8% and our chance-corrected measure of prediction was 69.5%. We concluded nipple sizes of female fishers measured during a fall/winter live-trapping season can be used as an conservative index of the weaning rates of adult female fishers, although current breeders may be misclassified as attempted breeders.

Introduction

Fishers (*Martes pennanti*) have been the focus of conservation concern in the

Pacific states over the last three decades due to range contractions, their association with mature forests, and assumed sensitivity to anthropogenic habitat alteration, particularly extensive logging (Powell & Zielinski 1994, Carroll et al. 1999). The U.S. Fish and Wildlife Service concluded in 2004 that listing the west coast distinct population segment of the fisher under the U.S. Federal Endangered Species Act was warranted but precluded by higher priority listing actions (U.S. Fish and Wildlife Service 2004). Additionally, translocation efforts have reintroduced fishers to the Olympic Peninsula in Washington (Lewis et al. 2010) and the northern Sierra Mountains in California (Callas and Figura 2008). Consequently, forest and wildlife managers have expressed a need to develop cost-effective methods of monitoring existing and reintroduced fisher populations.

An important component in monitoring fisher population dynamics is modeling reproduction and the influence of habitat and other covariates on reproductive success, particularly in timber-managed landscapes, such as was done for the northern spotted owl (*Strix occidentalis caurina*, Anthony et al. 2006). Estimates of fisher vital rates, including reproduction, have been very difficult and costly to obtain (Douglas and Strickland 1987) and may vary widely based on habitat composition and prey availability (York 1996). Cost- and labor-intensive radio-telemetry efforts have provided some information on fisher reproductive rates for a small number of landscapes (Arthur and Krohn 1991, Paragi et al. 1994, York 1996, Aubry and Raley 2006, Higley and Matthews 2006, Weir and Corbould 2008). However, radio telemetry approaches are not cost effective for most managers, particularly in the context of long-term population monitoring across large ownerships.

An alternative approach to direct observation via radio telemetry is the use of nipple size as an index of fisher reproductive success (Paragi 1990, Frost et al. 1999). Paragi (1990) concluded that nipple sizes of current-year breeders and non-breeders were sufficiently distinct to assign reproductive status to unknown individuals in south-central Maine. However, Paragi's (1990) conclusion was based on live-trapped and pelt measurements, small sample sizes (nulliparous n = 26, nonparous n = 1, parous n = 7), and the parous females were captured at two distinctly different periods in the female reproductive cycle (4 captured in May and 3 between August and December). Frost et al. (1999) found nipple size to be a reliable index for captive and wild fishers in Maine and Massachusetts to distinguish current-year breeders from non-breeders, but found former breeders may be misclassified as current breeders.

The approaches of Paragi (1990) and Frost et al. (1999) demonstrate the potential of using nipple size measured during fall/winter live-trapping as an index for determining whether or not a female fisher bred the previous spring in populations of the Pacific states. We proposed an approach similar to those of Paragi (1990) and Frost et al. (1999), involving a fall/winter live-trapping effort to measure nipple size as an index of reproductive success. However, we argue that a more appropriate measure of reproductive success is whether or not a female weaned a kit, rather than simply gave birth the previous spring, particularly considering reported failure rates of denning attempts. Frost and Krohn (1997) found that during a 3-year study, 10 of 38 (26%) kits born in captivity died within a week after birth. On the Hoopa Valley Indian Reservation, we determined 11 (20%) of 55 denning attempts by wild fishers resulted in failure (Chapter 3); these 55 attempts exclude attempts made by females that died during the den season.

The objective of this study was to apply methods similar to those of Paragi (1990) and Frost et al. (1999) to a sample of wild female fishers in northwestern California with known reproductive histories to determine the efficacy of nipple size as an index of reproductive success.

Study Area

Our study was conducted on the 366-km^2 Hoopa Valley Indian Reservation in northwestern California. The area is located within the Klamath physiographic province (Küchler 1977) and elevations range between 98 and 1,170 m. Mean daily maximum and minimum temperatures are 22 and 6^{o}C, respectively, and mean annual precipitation, primarily rain, is 138 cm (National Climate Data Center, http://cdo.ncdc.noaa.gov/climatenormals/clim20/ca/049694.pdf, accessed 25 February 20011). The Reservation supports a human population of approximately 2,600 (Hoopa Valley Tribe, http://www.hoopa-nsn.gov/documents/2000Census.pdf, accessed 25 February 2011).

Forests generally have an overstory dominated by Douglas-fir (*Pseudotsuga menziesii*) and a midstory dominated by hardwood trees including tanoak (*Lithocarpus densiflorus*), madrone (*Arbutus menziesii*), and canyon live oak (*Quercus chrysolepis*). Pure hardwood stands occur in some areas and at higher elevations the Douglas-fir canopy is replaced by white fir (*Abies concolor*) and pine (*Pinus* spp.). The Reservation is split by the Trinity River into east and west portions.

Methods

Our capture and handling methods were approved by the Institutional Animal Care and Use Committee of Humboldt State University, protocol 04104.W.42.A. The current study was conducted as an element of a larger effort addressing habitat use patterns of female fishers during denning (S.M. Matthews, Hoopa Valley Tribe, unpublished data). Our initial live-trapping season was between 8 December 2004 and 11 March 2005. We used Tomahawk live traps (model 207, Tomahawk Live Trap Company, Tomahawk, Wisconsin, USA) baited with chicken legs and modified with a plywood cubby box to capture fishers (Wilbert 1992, Seglund 1995). Captured fishers were anesthetized with ketamine hydrochloride (40 mg/kg) and diazepam (0.25 mg/kg) and handled using standard protocols (Aubry and Raley 1996, Yaeger 2005). A first-upper premolar was removed for aging by cementum annuli (Matson's Laboratory, Milltown, Montana, USA; Strickland et al. 1982, Poole et al. 1994). Female fishers were radio-marked (Holohil model MI-2, Holohil Systems Ltd., Carp, Ontario, Canada or Telonics model MOD80, Telonics Inc., Mesa, Arizona, USA) to determine reproductive success during subsequent den seasons. We released all fishers after recovery from anesthesia at their sites of capture.

Radio-marked female fishers were monitored 4-5 days per week during five (2005-2009) den seasons (March-June) to determine reproductive status. We used ground-based radio telemetry techniques with handheld telemetry receivers (model TR-4 Telonics, Mesa, Arizona, USA or model R1000 Communications Specialists, Inc., Orange, California, USA) and 4-element antennas (model RA-14 Telonics, Mesa, Arizona, USA) to estimate locations of female fishers and chronicle denning behavior (Arthur and Krohn 1991, Paragi et al. 1994, York 1996, Aubry and Raley 2006). Denning behavior was characterized by a sudden change in behavior from using numerous rest sites per week across the majority of the home range to more restricted movements in a small portion of the home range and repeated use of the same structure while inactive (Aubry and Raley 2006). We determined a female weaned at least one kit if she exhibited denning behavior until 31 May (Powell 1993). Following each den season we classified each radio-marked female into one of three reproductive classes: 1) non-breeders - juveniles and subadults <2 years old and other females ≥ 2 years old that did not attempt to give birth during the last den season, 2) attempted breeders - females ≥ 2 years old that gave birth but did not wean at least 1 young during the last den season, and 3) current breeders - females ≥ 2 years old that gave birth and weaned at least 1 young during the last den season.

Live-trapping was then conducted following each den season, between September and February until February 2010, to recapture radio-marked female fishers, replace radio collars, and measure nipple sizes. We measured width and height of all 4 inguinal nipples on each captured fisher using digital calipers (Absolute Digimatic 500-196, Mitutoyo, Aurora, Illinois, USA; Paragi 1990, Frost et al. 1999). Width and height were multiplied and the largest anterior nipple measured for each female at each capture was used to index nipple size (Frost et al. 1999). We also recorded the number of days between weaning (31 May) and the capture date to account for reductions in nipple size post weaning (Frost et al. 1999). Provided the rapid decrease in fisher nipple size of current breeders following weaning to near nonbreeder levels documented by Frost et al. (1999), we felt comfortable including measurements of the same female over multiple years in our sample.

We used a Kruskal-Wallis test to examine differences in nipple size among the three reproductive classes. Then we used an analysis of variance (ANOVA) to determine if the three reproductive classes were proportionately sampled throughout the capture seasons. We used quadratic discriminant analysis and jackknife cross validation using Cohen's Kappa statistic to develop and test a predictive model for assigning reproductive class based on nipple size and days post-weaning. All statistical analyses were conducted in program R version 2.11 (R Development Core Team 2008).

Results

The reproductive history and nipple measurements during the subsequent fall/winter were collected on 30 occasions for 17 adult female fishers (Table 14; Figure 4). Nipple sizes were also measured on 22 occasions for 20 juvenile and subadult female fishers that were included in the dataset as non-breeders (Table 14; Figure 4). Two individuals were measured both as juveniles and, in subsequent years, as adults. Juvenile and subadult classification, and thus non-breeder status, was based on cementum annuli age estimates and, in some cases, confirmed with radio telemetry assessment of behavior during the den season.

Nipple size differed among classes (W=32.86, df = 2, $p<0.001$). Nipple sizes of non-breeders were smaller than those of attempted and current breeders ($p<0.05$). Nipple sizes of attempted and current breeders did not differ ($p>0.05$). We found the three reproductive classes were proportionately sampled throughout the capture seasons (F=0.0218, df=2, p=0.9784; Figure 4).

Our squared canonical correlations between the grouping variable and each canonical axis yielded adjusted R^2 values of 0.611 ($F_{2,51}$= 42.580, $p<0.001$) and -0.030 ($F_{2,51}$= 0.226, p=0.799). We found some group separation using a quadratic discriminant analysis and jackknife cross-validation (Table 15). Our overall correct classification rate was 80.8% and our chance-corrected measure of prediction (Cohen's Kappa) was 69.5%. Non-breeders had a particularly high correct classification rate of 96%, with only one incorrectly classified as an attempted breeder. Attempted breeders had a correct classification rate of 75%, with two incorrectly classified, one as a non-breeder and one as a current breeder. Current breeders had the lowest correct classification rate of 63%, with seven incorrectly classified, one as a non-breeder and six as attempted breeders.

Discussion

Our results indicate that nipple sizes measured during a fall/winter live-trapping season have some utility to distinguish reproductive status of wild female fishers during the previous spring. Although current breeders may be misclassified as attempted breeders, our results lend further support to the use of nipple size as a viable index of fisher reproduction and an alternative to radio-telemetry approaches. We also acknowledge this assessment was conducted with low group sample sizes, particularly for attempted breeders (n = 8), and recognize a larger sample might help to increase prediction classification rates.

The largest misclassification was 6 of the 19 current breeders misclassified as attempted breeders. Beyond small group sample sizes we suspect our original classification of reproductive status influenced our prediction classification rates. Our original determination of reproductive class was based on behavior we observed during the den season. Several factors may have lead to inaccurate initial determination of reproductive class and thus reduced predictive classification rates. In some cases, females may have exhibited what we interpreted to be denning behavior but were merely avoiding males during the mating season and never bred kits. Thus at the close of the mating season, these females resumed normal behavior patterns, which we interpreted as a breeding failure and classified these as attempted breeders rather than non-breeders. In other cases, a female may have exhibited denning behavior early in the den season but failed to exhibit denning behavior until weaning, although she may have successfully weaned a kit(s). In these cases we would have misclassified a current breeder as an attempted breeder. We suspect litter size and den site selection could play a role in such an outcome. Larger litters probably place more energetic demands on the female and thus the need to invest more time in securing prey away from the den as compared to a female with a smaller litter. Additionally, larger litters might provide a thermoregulatory benefit in the kits keeping each other warm, rather than depending on the female to do so, enabling the female to spend more time securing prey. Den site selection might provide similar thermoregulatory benefits in appropriate solar exposure and other microclimatic variables, allowing the female more time away from the den.

The relatively high correct classification rate for non-breeders was probably influenced by the group being entirely composed of females that had never produced young. Fifteen non-breeders were <1 year old, 9 were 1 year old, 2 were 2 years old, and 1 was 3 years old. A more complete assessment of this index would include nipple measurements of females that weaned kits some years and did not breed in other years of the assessment. Our experience in Hoopa was once females attempted to den as an adult, they attempted to den each subsequent year.

Alternatively, it has been suggested fishers could be captured immediately after weaning to assess nipple enlargement and reproductive success, as was proposed by LeCount (1986) for black bears (*Ursus americanus*). While this approach might provide greater differentiation in nipple size among females that successfully weaned young and those that did not (Frost et al. 1999), we argue the potential risk to kits by separating mothers and kits should be avoided until these risks are quantified. Additionally, fisher detection, and presumably capture success, is lower during the summer season (K.M. Slauson, USDA Forest Service Pacific Southwest Research Station, unpublished data).

Management Implications

Nipple size of female fishers measured during a fall/winter live-trapping season can be used as an conservative index of the weaning rates of adult female fishers, although current breeders may be misclassified as attempted breeders. This index could prove useful for managers hoping to model fisher reproduction and the influence of habitat and other covariates on reproductive success, particularly in timber-managed landscapes occupied by extant or reintroduced fisher populations.

Table 14. Nipple sizes (mm^2, width multiplied by height) of the largest anterior nipple for each reproductive class of wild female fishers captured on the Hoopa Valley Indian Reservation, California, USA. Females were captured and measurements taken between September and February between 2004 and 2010. Non-breeders were juveniles and subadults <2 years old and other females ≥2 years old that did not attempt to give birth during the last den season. Attempted breeders were females ≥2 years old that gave birth but did not wean at least 1 young during the last den season. Current breeders were females ≥2 years old that gave birth and weaned at least 1 young during the last den season.

Reproductive class	n	Mean	Standard deviation	Minimum	Maximum
Non-breeders	25	4.46	2.10	1.96	11.21
Attempted breeders	8	10.84	3.28	4.42	16.51
Current breeders	19	17.26	9.19	3.98	39.36

Table 15. Classification of female fishers into reproductive classes based on maximum anterior nipple size (diameter multiplied by height) and the number of days between post weaning (31 May) and capture date using quadratic discrimination analysis and jackknife cross-validation. Non-breeders were juveniles and subadults <2 years old and other females $\geq$2 years old that did not attempt to give birth during the last den season. Attempted breeders were females $\geq$2 years old that gave birth but did not wean at least 1 young during the last den season. Current breeders were females $\geq$2 years old that gave birth and weaned at least 1 young during the last den season. The rows are the known class of each female fisher and the columns are the predicted class membership. For example, the first row, 26 non-breeders were correctly classified as non-breeders, 1 non-breeder was incorrectly classified as an attempted breeder, and 0 non-breeders were incorrectly classified as current breeders.

	Nonbreeder	Attempted breeder	Current breeder
Non-breeder	24	1	0
Attempted breeder	1	6	1
Current breeder	1	6	12

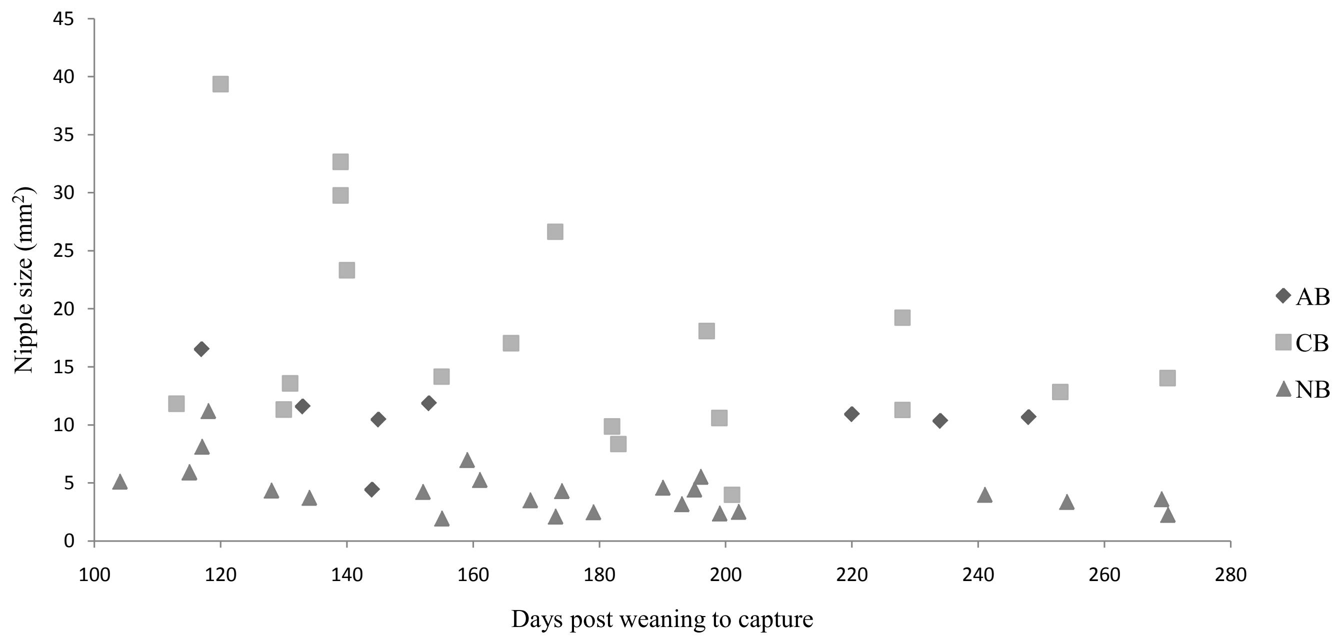

Figure 4. Nipple size (measured as width multiplied by height in mm^2) and days elapsed between weaning and capture of three reproductive classes of female fishers between 2005 and 2009 on the Hoopa Valley Indian Reservation, California, USA. Non-breeders (NB) were juveniles and subadults <2 years old and other females $\geq$2 years old that did not attempt to give birth during the last den season. Attempted breeders (AB) were females $\geq$2 years old that gave birth but did not wean at least 1 young during the last den season. Current breeders (CB) were females $\geq$2 years old that gave birth and weaned at least 1 young during the last den season.

CHAPTER 5

POPULATION ECOLOGY OF FISHERS IN MANAGED FORESTS ON THE HOOPA VALLEY INDIAN RESERVATION, CALIFORNIA

The Hoopa Valley Tribe in northwestern California has taken a leadership role in exercising significant tribal sovereignty rights won over the last two decades (Matthews et al. 2008*b*). The Hoopa Tribe was one of the first nine tribes in the U.S. to participate in the Self-Governance Demonstration Project in the 1990s. As such, the Hoopa Tribe has helped create and determine federal policy that is used in defining and determining functions of self-governance for other Native American and First Nation communities throughout the United States and Canada. These sovereignty rights included the infusion of many culturally-based conservation ethics into land management practices as the responsibility of natural resource management on many reservations transferred from federal agencies to sovereign tribal governments.

On the Hoopa Reservation, this included the transfer of forest management and a timber-extraction-based economy from the U.S. Bureau of Indian Affairs to the Hoopa Tribal Council, the governing body of the Hoopa Tribe. Today, timber management is the single largest source of revenue and employment for the Hoopa Tribe. Consequently, it has a direct impact on the Hupa people's quality of life and ability to achieve economic self-sufficiency. In assuming timber management responsibilities, the Tribe works diligently to develop a better understanding of the needs of threatened, endangered, and culturally significant wildlife and plant species to better inform their timber management practices.

The persistence of fisher (*Martes penanti*) populations has become an increasing concern for wildlife and forest managers throughout their range in the Pacific states and British Columbia (Lofroth et al. 2010). The fisher is a mid-sized, forest-dwelling carnivore in the family *Mustelidae* (Powell 1993). The geographic distribution of fishers historically included the boreal forests of southern Canada, the northern Rocky Mountains, the northeastern and upper-midwestern United States, and south through the Cascade Range and coastal mountains, northern California, and the western slopes of the Sierra Nevada Range (Powell 1993). However, trapping for fur during the early twentieth century, predator and pest control campaigns, and forest management practices resulted in population declines and range contractions across the distribution (Douglas and Strickland 1987, Powell 1993, Powell and Zielinski 1994). In the Pacific states the fisher was considered extirpated in Washington until recent reintroduction efforts; exists in two relatively small populations in southern Oregon; and occurs in less than 50% of its historic range in two isolated populations in California (Buck et al. 1994, Gibilisco 1994, Powell and Zielinski 1994, Zielinski et al. 1995, Aubry and Lewis 2003, Zielinski et al. 2005). Consequently and in light of current threats, the U.S. Fish and Wildlife Service concluded the distinct population segment historically occurring in Washington, Oregon, and California was warranted but precluded for listing under the Endangered Species Act in 2004 (U.S. Fish and Wildlife Service 2004).

The fisher is also a culturally significant species to the Hupa people and still occurs in relatively large numbers on the Reservation (Yaeger 2005, chapter one). As such, the Hoopa Tribe has taken a regionally and nationally recognized leadership role in fisher research and conservation. This role was exemplified by the Hoopa Tribe's wildlife biologist being invited to serve on the Interagency Fisher Biology Team, tasked with developing a Fisher Conservation Assessment and Strategy (FCAS) for the Pacific states (Lofroth et al. 2010). The Hoopa Tribe also hosted a regional fisher workshop in July 2009. In attendance were 45 foresters and biologists representing the U.S. Fish and Wildlife Service, U.S. Forest Service, Bureau of Land Management, National Park Service, California Fish and Game, Oregon Fish and Wildlife, Washington Fish and Wildlife, City of Arcata, three private timber companies, one private biological consultant, four universities, three non-profit conservation organizations, and Hoopa Tribal Forestry. Information shared during the workshop provided these professionals guidance during their daily management operations and during the review of as many as 1,400 timber harvesting plans submitted annually from the nearly 17 million acres of commercial forest land which overlap fisher range in California alone.

The first significant fisher research effort on the Reservation was focused on indentifying habitat characteristics associated with resting sites (Yaeger 2005). Yaeger (2005) concluded that timber harvest strategies should attempt to maintain scattered groups of the largest diameter trees, dense canopy cover, in close proximity to drainage-bottoms. Additionally, homogeneous stand management should be minimized because local structural and growth characteristics of different trees species may affect fisher resting habitat availability. Hoopa wildlife personnel identified the need to develop a better understanding of fisher reproductive ecology was the next logical step in efforts to conserve fisher on the Reservation and better inform ongoing timber management.

Our research into the reproductive ecology of fisher on the Reservation began with an initial trapping and radio collaring effort, during which we indentified a dramatic decrease in trap success between our efforts and those of Yaeger (2005). This decline prompted us to replicate a population density estimation study conducted in conjunction with Yaeger's (2005) work on a 90-km^2 study area on the southeast corner of the Reservation. We also used the opportunity to determine, post-hoc, the efficacy of three population indexing techniques and small-scale occupancy modeling as potential methods for long-term fisher population monitoring on the Reservation.

We were not surprised to find the classical relative abundance indices and the small-scale occupancy estimation failed to detect significant population change. Despite the attractiveness of these techniques to managers because of their relative low cost and the belief they are effective in monitoring relative change in abundance over time, our results reinforce the importance of careful thought given to the study goals and potential limitations of any technique. We acknowledged that our data were not collected with an occupancy framework in mind. Despite combining detection stations, our study design violated the assumption of spatial independence for occupancy estimation and the precision of our occupancy estimates were probably overstated. Thus our results cannot speak to a properly designed occupancy approach across an appropriate scale. We recommend managers consider adopting more defensible, large-scale occupancy estimation or mark-recapture techniques as methods to monitor changes in wildlife population sizes, especially when responsible for at-risk populations. Being left only to speculate on reasons for the fisher population decline in Hoopa illustrates the value in establishing long-term, accurate programs to monitor populations of imperiled species which strive to determine cause and affect relationships to changes in populations and ultimately, modeling habitat fitness.

In conjunction with the population decline reported in chapter one, and of equal conservation concern, was the finding that the population decline impacted female fishers more than males, with our captured population 72% female in 1998 and 57% in 2005. Sex ratios of unharvested fisher populations are not well known, but have been suspected to be near 50:50 (Powell 1993). Thompson (2008) and Jordan (2007) both reported 53% of their captured populations being female, suggesting a slight female bias in unharvested populations.

Unfortunately, research activity to determine the cause of the decline and the disproportionate impact on females did not occur on the Reservation between 1998 and 2005. Thus we were left to speculate changes in prey habitat, increases in predation, disease, or some combination of these potential causes were responsible for the population decline. A decline in prey habitat suitability across our study site may have influenced the fisher population decline. Mammal remains occurred in 92.2% of fisher scats (n=64) collected on the Reservation, and rodents (51.6% frequency of occurrence), especially dusky-footed woodrats (*Neotoma fuscipes*, 10.9% frequency of occurrence), were the most frequently detected prey items (Golightly et al. 2006). Whitaker (2003) reported that capture rates of dusky-footed woodrats on the Reservation were 5-10 times higher in 20-25 year old unthinned stands compared to 20-25 year old thinned stands or 35-40 year old stands whether thinned or unthinned. Thinning and growth into a stem-exclusion stage at 35-40 years of age reduced levels of understory brush and thus woodrat habitat quality (Whitaker 2003). In 1998, 19.2% of the study area was 10-29 years old, likely supporting high densities of woodrats. Between 1998 and 2005 the percentage of 10-29 year old stands decreased to 11.8%, while stands 30-45 years old increased from 18.7% to 27.9% as the 10-29 year old stands grew into stem exclusion. These changes in structural stages were by far the most prominent within the study area between 1998 and 2005. A minor decrease in the percentage of mature, multi-storied forest (43.0 to 40.6%) also occurred while no other structural stages changed by more than 1%. It is possible that the decrease in fisher population was partially or entirely the result of a decrease in woodrat abundance and availability. However we would not expect that this potential drop in prey would have affected a change in the sex ratio.

Habitat change resulting from catastrophic wildfire may have also contributed to increased fisher predation risk and population declines on the Reservation. In 1999, the Megram fire burned over 505 km^2 on the southeastern border of the Reservation (Jimerson and Jones 2003). Over 63 km^2 of old-growth and late-mature forest was affected by high severity fire (>80% tree mortality), returning these areas to shrub/forb habitat (Jimerson and Jones 2003). Witmer and deCalesta (1986) reported bobcats used Doulgas-fir dominated forest cover during the day while most inactive and clear-cut units at night while most active, presumably hunting. The clear-cut units described Witmer and deCalesta (1986) were a diverse pattern of early successional stages composed of grass/forb, shrub, and sapling/pole. A similar seral diversity was present in the Megram fire area <1 km from our study area. Since 1999, reports of incidental sightings of bobcats have increased throughout the Reservation (J. M. Higley, Hoopa Tribal Forestry, unpublished data). Predation has generally not been considered a limiting factor for fisher populations based on research conducted in New England and Great Lakes regions (Powell 1993, Kurta 1995), but recent research on west coast fisher populations indicates that predation, especially by bobcats, is a common source of mortality (G. Wengert, University of California Davis, unpublished data). An increase in bobcat predation on fishers might also explain the observed change sex ratio. Smaller body size might make female fishers more susceptible to bobcat predation to the point of skewing fisher population sex ratios.

Although little is known about disease in fishers, disease has caused significant mortality in other mustelids, including black-footed ferrets (*Mustela nigripes*), mink (*Mustela vison*), and others (Barker and Parrish 2001, Williams 2001, Langlois 2005) and may have played a role in our fisher decline. Brown et al. (2007) sampled 31 fishers between December 2004 and March 2005 on the Hoopa Valley Indian Reservation and found 1 (3%) had been exposed previously to canine distemper virus (CDV) and 13 (42%) had been exposed to canine parvovirus (CPV). Both CDV and CPV have the potential to cause immunosuppression and to work synergistically with other pathogens to increase morbidity or mortality in a susceptible population (Brown et al. 2007, M. Gabriel, University of California Davis, unpublished data). Susceptibility could be influenced by populations of sympatric mesocarnivores and unleashed dogs near development being local reservoirs for these viruses (Brown et al. 2007). Exposure risk could be greater for female fishers if transmission occurs in den sites used by other infected mesocarnivores, thus influencing the fisher population sex ratio.

The fisher population decline on the Reservation and elevated conservation status of the species throughout the Pacific states highlighted the need to assess the efficacy of methods available to collect long-term demographic data on fisher populations. This held especially true for Hoopa managers following our finding in chapter one that the use of classical indexing and small-scale occupancy proved ineffective in detecting a significant population decline. Further, tools to measure the success of re-introduction and other conservation efforts, determine the effects of changes in forest and fuels management practices, and monitor general population trends will assist other regional wildlife and forest managers.

Our conclusion in chapter one, that three indexing techniques and small-scale occupancy modeling failed to detect the 73% population decline calculated using a mark-resight framework, encouraged us to investigate alternative methods for long-term fisher population monitoring on the Reservation that could be applied to a mark-resight or large-scale occupancy design. Genetic techniques and passive integrated transponder (PIT) tag technology have demonstrated application in the monitoring of wildlife populations (York and Fuller 1997, Gibbons and Andrews 2004, Waits and Paetkau 2005). However, cost is a key consideration for many regional wildlife managers in developing demographic monitoring programs. Based on our analyses in chapter two, traditional mark-recapture continues to be the least expensive method of collecting long-term population data and provides the most demographic information.

Livetrapping allows for individual age (Strickland et al. 1982) and annual female reproductive rate (Frost et al. 1999, chapter four) assessments. These data are not available from a non-invasive, genetic tagging method used independent of livetrapping. Lambda and survival estimates based solely on genetic identification may be biased significantly downward due to dispersal of juvenile fisher off the study area (Burnham et al. 1996, Forsman et al. 2002, Anthony et al. 2006). The collection of age estimates allows for the calculation of non-juvenile lambda and survival estimates, removing the potential bias of juvenile dispersal.

Annual female reproductive rates and dispersal capabilities are important components in assessing the resilience of carnivore populations (Pimm et al. 1988, Ruggiero et al. 1994, Weaver et al. 1996). Although there is little empirical evidence, we suspect habitat degradation and fragmentation negatively influence fisher populations by loss of denning and resting structures, reductions in escape cover, and increased predation risk. We suspect the comparatively high female reproductive rates measured in Hoopa are a function of overall forest productivity and the abundance of unmanaged stands and residual legacy structures in managed stands compared to other fisher study areas. Northern and coastal California forests have the highest net primary productivity values statewide (Williams et al. 2005). This net primary productivity, largely a function of tanoak (*Lithocarpus densiflora*) mast production, provides high levels of prey availability, particularly wood rats (Whitaker 2003).

Additional research is required to determine the relationships between prey availability, forest structural characteristics, and fisher reproductive success. Although teat measurements have some value in assigning female fishers to reproductive classes, we caution managers looking to use evidence of reproduction as a metric of habitat quality. Rather, site specific assessments of the role of habitat and availability of forest structural and other habitat components on fisher reproductive success are required to better inform forest and fuels management. We emphasize the long-term conservation of this at-risk species in dynamic landscapes, including the impacts of timber/fuels management and climate change, will involve managers adopting defensible, long-term population monitoring methods which strive to determine cause and affect relationships to changes in populations and ultimately, modeling habitat fitness.

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