Evaluating the Demographic Factors that Affect the Success of Reintroducing Fishers (*Martes pennanti*), and the Effect of Removals on a Source Population.

Final Report
5 December, 2005

Roger A. Powell, Departments of Zoology and Forestry, North Carolina State University, Raleigh, North Carolina 27695-7617. newf@ncsu.edu

William J. Zielinski, USDA Forest Service, Pacific Southwest Research Station, Arcata, California 95521. bzielinski@fs.fed.us

INTRODUCTION

Early in the 20th Century, fishers occupied forested regions throughout the Sierra Nevada Mountains, Klamath Mountains and the northern Coastal Ranges of California (Grinnell et al. 1937). By 1925, fisher populations in California had decreased significantly (Hall 1942), apparently due to heavy trapping pressure and habitat loss from timber harvest and land development. The trapping season for fishers in California was open until 1946, despite being the only state that still allowed trapping during the early 1940s (Hall 1942). By the 1940s, the remnant fisher populations in northwestern California and in the southern Sierra Nevada were 2 of only 6 small but apparently viable fisher populations left in the United States (Powell 1993). A summary of sightings in the 1970s suggested that fishers may have persisted in the northern Sierra Nevada at very low densities (Schempf and White 1977) but field survey data from the 1990s verify the existence of populations only in northwestern California and the southern Sierra Nevada, separated by a distance of 300-400 km (Zielinski et al. 1995, Zielinski et al. 2005). The population in the southern Sierra Nevada is assumed to be small, but formal population estimates have not been conducted in either location. The ability of the isolated, southern Sierra Nevada population to maintain itself is uncertain. That population could potentially benefit if supplemented by dispersal of fishers re-established in the northern Sierra Nevada.

OBJECTIVES

Proposing to re-establish fishers in the northern Sierra Nevada Mountains raises 3 questions:

1) Is re-establishing a fisher population in the northern Sierra Nevada Mountains feasible?
2) Can the fisher population in northwestern California remain viable if fishers are removed for release elsewhere?
3) Might fishers dispersing from a re-established population in the northern Sierra Nevada reach the presently-isolated population in the southern Sierra Nevada and, if so, would such dispersers benefit the southern population?

Re-establishing fisher populations throughout their range is desirable, independent of its effects on the isolated population in the southern Sierra Nevada. Consequently, at least the first 2 of the questions above must have positive answers before advancing to the next stage of evaluation and planning. We have, therefore, modeled the population ecology of fishers in northwestern California and modeled the reintroduction of fishers to the northern Sierra Nevada. In addition, we modeled a metapopulation with 3 populations to explore potential benefits of a reintroduction to the fisher population in the southern Sierra Nevada. Importantly, our analysis does not include an assessment of the suitability of habitat or prey base at potential reintroduction locations. On the contrary, we explore the implications of varying the number, sex ratio and age ratio of the transplanted animals with the assumption that suitable habitat exists and that release locations provide adequate resources. Thus, our results will need to be interpreted with caution and integrated with habitat suitability and prey base assessments if a formal reintroduction plan is developed. Furthermore, we do not address the genetic implications of reintroduction strategy. Fisher populations in the Pacific region are relatively homogenous, at the level of mitochondrial DNA sequences (Wisely et al. 2004), but this is not to say that the genetic implications of fisher reintroductions should be ignored.

APPROACH

We considered several approaches to modeling fisher populations. Because removing fishers for release from the northwestern California population has the potential to have the same effect on a population as harvest of animals for fur, we considered approaches used by wildlife agencies of the states and provinces of the US and Canada to determine annual harvest. Fisher populations in southern Ontario have been evaluated and quotas established using the ratio of juveniles to adults in the annual harvest (Fryxell et al. 2001, Strickland 1994). Populations of many furbearers are estimated before and after trapping seasons using empirical data on harvest effort and the numbers of animals harvested (Lancia et al. 1994). Getz and Haight (1989) argued that the best approach to evaluating population responses to harvest is to model populations using population matrices and demographic data.

Of these potential approaches, we established that population matrices were the most applicable to answering the questions about the fisher populations in California. Age ratios, though valuable and effective for managing trapped populations, require extensive empirical data on the specific populations being evaluated (Fortin & Cantin 2004, Fryxell et al. 2001, Strickland 1994). Insufficient data exist for fisher populations in California to use this approach. From data on fishers live-trapped for research in California and data on fishers elsewhere in their range, we do have sufficient information to estimate demographic variables needed for population matrix models. We chose to use population matrix models to evaluate the northwestern California population and to explore the relative effects of removing fishers for release elsewhere.

MODEL

Because we have specific information for few demographic variables for the California fisher populations, we used data from fisher populations elsewhere where necessary (data reported by Powell 1993, Raine 1981) and explored the relative effects of changes pertinent to our 3 questions. As variables are added to population matrices, complexity multiplies and the models quickly become difficult to handle. In addition, we wished to incorporate stochasticity into our models, adding even more complexity. Fortunately, software for analyzing population viability, VORTEX (Miller and Lacy 1999), uses complex population matrices that include variables pertinent to our questions and include stochastic variability. Consequently, we used *VORTEX* for our analyses.

Because *VORTEX* is used for analyses of population viability, the program estimates the probability of extinction for populations having the demographic and environmental characteristics used as input. To gain insight into each of our 3 questions, we established baseline population characteristics (Table 1) and then explored how the probability of extinction varied as we varied those input characteristics. We chose baseline characteristics consistent with what we know about each population and that produced population projections and estimates of extinction probability that allowed us to evaluate the effects, for example, of removing fishers for release elsewhere. We used probability of extinction as an *index* of population viability, not as a dependable estimate of that probability. To understand the importance of variation in these baseline characteristics, we also calculated elasticity of most variables by varying the baseline values $\pm$10%. We did not vary values for sex ratio or age of first reproduction, because they were considered the least variable demographic characteristics.

VORTEX allows partitioning of populations into 'subpopulations' each of which experiences different conditions. Subpopulations, in this case, are an accounting unit and are not assumed to be distinct demographic entities as in formal metapopulation exercises. Across their range in northwestern California, fishers experience distinctly different probabilities of habitat change: private land managed for timber (roughly 50% of the range), USDA Forest Service land managed for multiple use (roughly 35%), and Forest Service land with wilderness and other designations that limit timber harvest and other changes in habitat (roughly 15%). Our baseline model included 50 subpopulations, 25 of which each experienced 2 timber harvests during a 100 year run (i.e., private land), 18 of which experience 1 harvest (i.e., multiple use pubic land), and 7 of which experienced no harvest (*i.e.*, wilderness). We did not specify a specific harvest program but envisioned a program that would affect animals by removing critical habitat. We considered critical habitat to be forests with complex canopy structure and, especially, complex structure on the ground below a full canopy. Clearcutting certainly removes critical habitat but other harvest regimes can do so as well. During the year after harvest, reproduction in a subpopulation was reduced by 50% and survival by 15%. *VORTEX* does not allow one to incorporate long-term changes in habitat, only 1-year changes. To gain a long-term effect, we exaggerated the reduction of reproduction and survival in the year following harvest. Juveniles dispersing from any subpopulation had a modestly higher probability of reaching adjacent subpopulations (5% probability) than of reaching other subpopulations (3% probability). Fishers are protected from trapping so we assumed that trap mortality was zero, although some incidental capture probably occurs (Lewis and Zielinski 1996).

To evaluate the potential effects of removing fishers from the northwestern California population, we decreased litter size modestly (from 2.5 to 2.0) to emphasize the potential cost of productivity to the population by losing reproductive females. By using the lower number, the model is more likely to detect negative effects from removing females. If removal was unlikely to cause trouble with a slightly lower litter size, then removal would be even less likely to cause trouble for a population at our baseline conditions. To balance this change and to retain population stability, we reduce juvenile mortality (65% to 54%). From this model population, we removed 20 fishers for 2, 3, 5 or 8 years, either 5 fishers from each of 4 different subpopulations each year or 1 fisher from each of 20 subpopulations. Again to emphasize the effects of losing reproductive females, we removed 3 adult females and 2 adult males in each removal from a subpopulation. We evaluated the potential effect on the northwestern California population through changes in the probability of extinction.

To evaluate the effectiveness of different reintroduction regimes for a potential fisher population in the northern Sierra Nevada, we simulated the release of 20 fishers, 5 each into 4 different sites, each of which could become a subpopulation of the new population. We explored the effects of releasing fishers for 2, 3, 5 or 8 years, releasing fishers into new sites each year. We again modeled 50 subpopulations but assumed that 40 (80%) would be on Forest Service land and that only 10 would be on private land. Each subpopulation on federal and private land was modeled to experience 2, 1 or no timber harvests in a 100 year run of the model, as we had modeled the northwestern California population. To explore the effects of sex and age ratios on potential release success, we varied sex ratio from 4:1 (M:F) to 1:4 and we varied the number of juveniles released from 0 to 3 in each group of 5 released at a site. Assuming that this new population of fishers will occupy a smaller area than that of the present northwestern California population, we set carrying capacity, *K*, at 1000 (vs 2000 for the northwestern California population). Values for other variables were as used for the removal runs with the northwestern California population.

Finally, we modeled the effects of immigration from a newly established population in the northern Sierra Nevada on the viability of the southern Sierra Nevada population. To do so we established a baseline using the same values for variables as were used for the removal runs with the northwestern California population, except that we set *K* at 400, a reasonable estimate of the present population size (Lamberson et al. 2000). We explored the effects of 1 female or 1 male fisher reaching the population from a new northern Sierra Nevada population at frequencies of 1/yr to 1/15 yr. We then connected the 3 populations (without subpopulations) in a metapopulation in which dispersal between adjacent populations varied from 0 to 2% of dispersing juveniles per year. This connectivity was evaluated for its effects on population ecology, not genetics.

We used the probability of extinction throughout as an index of population viability. Our index of successful reintroduction of fishers to the northern Sierra Nevada was 1 - (probability of extinction).

RESULTS

The baseline model (no fishers removed from the NW California population) predicted a 5% probability of extinction for the northwestern California fisher population. Elasticity values for the demographic variables are shown in Table 1. Population growth and viability for the population appear most affected by variation in juvenile mortality (10% variation yielded 39% change in probability of extinction) and then by variation in litter size (23% change in probability of extinction). Elasticities indicate that variations in other variables probably have small effects on the population.

In case our estimates of the present population size or carrying capacity are widely in error, we also ran the model with each reduced to half the baseline value. Halving the initial population size increased probability of extinction by 1%. Halving *K* increased the probability of extinction by 22%, which is still of lower impact than a 10% change in juvenile mortality or litter size.

Given the assumptions of the model, removal of fishers from the northwestern California population is predicted to have little effect (Table 2). The predicted probability of extinction rose <5% when 20 fishers (5 each from 4 different subpopulations each year) were removed from the model population for each of 8 years; removals of 1 fisher from each of 20 subpopulations for 3 years had no effect on probability of extinction.

The index of predicted successful reintroduction of fishers to the northern Sierra Nevada Mountains varies considerably with the introduction scenario (Table 3). Holding other variables constant, probability of success increases with the number of years that fishers are released (Table 3, Figure 1) and increases with the number of females in the release population. Although the probability of success increases with the ratio of adult females to adult males (AF:AM) in the release population (Table 3, Figure 2), the increase in the index of success responds almost entirely to the total number of females (Table 3, Figure 3, 4). In other words, releasing groups of 2 females and 3 males at each of 4 sites has the same probability of success as releasing 2 females and 2 males at each of 4 sites; the number of females released is critical, not the sex ratio *per se*.

We established the model fisher population in the southern Sierra Nevada to have an index of extinction of 15%. Allowing dispersing fishers from a re-established population in the northern Sierra Nevada to reach the southern Sierra Nevada population reduced the probability of extinction. Immigrating fishers supplemented the southern Sierra Nevada population and allowed recolonization after extinction. If fishers dispersing from a newly established northern Sierra Nevada population reached the southern Sierra Nevada with a 1% probability, the probability of extinction for the southern population decreased from 15% to 11%. With dispersal probability from the new northern Sierra Nevada population set at 2%, probability of extinction of the southern Sierra Nevada population decreased to 2%. With 1 dispersing female reaching the southern population every 5 years, probability of extinction decreased from 15% to 6%.

DISCUSSION

The models explored in this report provide information directly related to our second objective and to half of our third objective: will removing fishers from the northwestern California fisher population jeopardize that population and, if fishers disperse from a newly-established population in the northern Sierra Nevada, will that dispersal have a positive effect on the present, small, southern Sierra Nevada population? Our results predict that removal of up to 20 fishers per year for as long as 8 years from the northwestern California population, to re-establish a fisher population in the northern Sierra Nevada Mountains, will not jeopardize the northwestern California population. This conclusion is contingent on our assumptions about the effects of timber harvest, and its rate, on fisher vital rates. A shortcoming of modeling the effect of timber harvest on subpopulations in VORTEX is the difficulty of applying the depressed rates of survival and reproduction for more than the single year in which the harvest has its effects. This may lead to somewhat optimistic forecasts for the viability of the northwestern California population. Furthermore our analysis was conducted without considering recent information that suggests that fisher populations, particularly the female component, may be decreasing precipitously on several ownerships in northwestern California (M. Higley, Hoopa Tribal Forestry, pers. comm., L. Diller, Green Diamond Resource Company, pers. comm.). This observation has yet to be confirmed, but if true would call for revised assessments of the viability of this population. Specifically, we would vary adult sex ratio to evaluate the effect of a male biased adult population on the probabilities of extinction.

McLoughlin and Messier (2004) warned that in models of population viability, erroneous estimates of initial population size may cause greater errors than poor estimates of vital rates. They used grizzly bears (*Ursus arctos*) as their example, though, and grizzly bears have lower population sizes and lower mortality and reproductive rates than fishers, all of which exacerbate the effects of initial population size and de-emphasize the importance of vital rates (Lewontin 1974). We tested for the effects of over-estimating the present size of the fisher population in northwestern California and showed that over-estimating the population size by a factor of 1 (i.e., the estimated size is twice as big as the real population size) had relatively little effect on our results. Importantly, changing the initial population size would not affect the relative rankings of different strategies for removal of animals from the northwestern California population or for introduction of animals to a new site. Because of the structure of the *VORTEX* model, changes in the effects of harvest on fisher reproduction and survival, changes in proportions of public and private land, and changes in baseline conditions will also not affect the relative rankings of removal and introduction strategies.

Levins (1966) noted in his classic paper that models can have 3 important characteristics, accuracy, precision and generality, but can never have more than 2 of these characteristics at once. As models become more precise, they must become less general. In our models, we have sought accuracy and generality, knowing that to model one situation precisely required our model to become inaccurate for other situations. In the end, however, no model can be 100 accurate. Whether removing fishers from the population in northwestern California will jeopardize the population or not can only be learned by actually removing fishers.

Similarly, our models could not predict with certainty that a new population of fishers can be established in the northern Sierra Nevada. The models do predict that the 2 major factors affecting success of a release should be 1) the numbers of females released each year (as long as at least some males are released) and 2) the number of years that fishers are released (Table 3. Figures 1, 2, 3, 4). The results suggest that releasing more males than females and releasing any juvenile males will not increase the probability of success. Retaining these juvenile and adult males in the northwestern California population will decrease the impact of removals on that population and decrease the cost and effort associated with the release without changing the chances of success.

Assuming that habitat quality is sufficient in a potential reintroduction area, previous reintroduction efforts suggest that the probability of re-establishing a new fisher population may be higher than we predict. For 28 fisher releases documented in the literature, 75% succeeded in establishing new populations, some with as few as 12 or 14 fishers released (Roy 1991, Lewis and Hayes 2004; Figure 5). The literature also suggests, consistent with our results, that the more animals released, the higher the probability of success. The mean number of fishers released that established new populations, 49 fishers, was larger than the number released that failed, 22 fishers (significant at $p = 0.1$, Figure 5). Also, the number of years that fishers were released appeared to affect success ($p = 0.12$, Figure 5). Consequently, we believe that releases of 20 fishers/year for 3 to 5 years may have probabilities of success higher than the 20-42% listed in Table 3. If sex ratio of the fishers released favors females, the success rate might be considerably higher (Table 3, Figures 3, 4). Both assertions depend, however, on adequate habitat and prey resources in the reintroduction location.

Although our results suggest that dispersal of fishers from a newly-established population in the northern Sierra Nevada Mountains to the present population in the southern Sierra Nevada should decrease the probability of extinction for the southern population, we are unable to estimate the actual probability of such dispersal. The maximum dispersal distance documented for fishers is 100 km (Arthur et al. 1993) and the northwestern California and southern Sierra Nevada populations are approximately 350 km apart. Dispersal from a new population established midway between the 2 present populations would require dispersal of approximately 150 km to reach either of the present populations, a distance no fisher has been recorded to disperse, and through a landscape that has unknown suitability to fisher movements. Nonetheless, even rare dispersal of fishers to the southern population should have a positive effect, and possibly a significantly positive effect.

On the basis of our analyses, and the review of the literature, we suggest that if reintroduction is to proceed, the next steps would be: (1) a more critical examination of the characteristics of the northwestern population to validate our modeling assumptions, especially in regard to the effect of timber harvest on life history parameters and to incorporate the effects of a local population decline, should it be confirmed, (2) a thorough investigation of the distribution and abundance of habitat and preyin potential reintroduction areas, and (3) an evaluation of the genetic implications of transplantation.

LITERATURE CITED

Anderson, B. 2002. Reintroduction of fishers (*Martes pennanti*) to the Catoose Wildlife Area in Tennessee. Unpublished annual report. Tennessee Wildlife Resources Agency, Crossville, Tennessee.

Aubry, K B & J C Lewis. 2003. Extirpation and reintroduction of fishers (*Martes pennanti*) in Oregon: implications for the conservation in the Pacific states. Biological Conservation 114: 79-90.

Arthur, S M, T F Paragi & W B Krohn. 1993. Dispersal of juvenile fishers in Maine. Journal of Wildlife Management 57: 868-874.

Baird, R & S Frey. 2000. Progress report: Riding Mountain National Park fisher reintroduction program 1994-1995. Unpublished report. Riding Mountain National Park, Manitoba.

Benson, D A. 1959. The fisher in Nova Scotia. Journal of Mammalogy 40: 451.
Berg, W E. 1982. Reintroduction of fisher, pine marten and river otter. Pages 159-175 in G C Sanderson (editor). Midwest furbearer management, North Central Section of The Wildlife Society, Bloomington, Illinois.
Bradle, B J. 1957. The fisher returns to Wisconsin. Wisconsin Conservation Bulletin 22(11): 9-11.
Dodds, D G & A M Martell. 1971. The recent status of the fisher, *Martes pennanti pennanti* (Erxleben), in Nova Scotia. Canadian Field-Naturalist 85: 62-65.
Drew, R E, J G Hallett, K Aubry, K W Cummings, S M Koepfs & W J Zielinski. 2003. Conservation genetics of the fisher (*Martes pennanti*) based on mitochondrial DNA sequencing. Molecular Ecology 12: 51-52.
Fortin, C & M Cantin. 2004. Harvest status, reproduction and mortality in a population of American martens in Quebec, Canada. Pages 221-234 in D J Harrison, A K Fuller & G Proulx (editors). Martens and fishers (*Martes*) in human-altered environments. Springer Science+Business Media, Inc. New York.
Fryxell, J, J B Falls, E A Falls, R J Brooks, L Dix & M Strickland. 2001. Harvest dynamics of mustelid carnivores in Ontario, Canada. Wildlife Biology 7: 151-159.
Getz W. M. and R. G. Haight. 1989. Population harvesting: Demographic models of forest, fish, and animal resources. Princeton University Press, Princeton, New Jersey.
Grinnell, J., J. S. Dixon and L. M. Linsdale. 1937. Fur-Bearing Mammals of California: Their Natural History, Systematic Status and Relations to Man. Volume 1. University of California Press. Berkeley, California.
Hall, E. R. 1942. Gestation period of the fisher with recommendation for the animals' protection in California. California Fish & Game Journal 28: 143_147.
Heinemeyer, K S. 1993. Temporal dynamics in the movements, habitat use, activity, and spacing of reintroduced fishers in northwestern Montana. MS thesis, University of Montana, Missoula.
Irvine, G. W., L. T. Magnus and B. J. Bradle. 1964. The restocking of fishers in lake states forests. Transactions of the North American Wildlife Natural Resources Conference 29: 307_315.
Kyle, C J, J. F. Robtaille and C. Strobeck. 2001. Genetic variation and structure of fisher (*Martes pennanti*) populations across North America. Molecular Ecology 10: 2341-2347.
Lamberson, R. H., R. L. Truex, W. J. Zielinski, and D. Macfarlane. 2000. Preliminary analysis of fisher population viability in the southern Sierra Nevada. USDA Forest Service, Pacific Southwest Research Station, Arcata, California.
Lancia, R A, J D Nichols & K H Pollock. 1994. Estimating the number of animals in a wildlife populations. Pages 215-253 in Bookhout (editor). Research and management techniques for wildlife and habitats. The Wildlife Society.
Levins, R. 1966. Strategy of model building in population biology. American Scientist 54: 421-431.
Lewontin, R R. 1974. The genetic basis of evolutionary change. Columbia University Press, New York.
Lewis, J. C. and W. J. Zielinski. 1996. Historical harvest and incidental capture of fishers in California. Northwest Science 70:291-297.
Lewis, J. C. and G. E. Hayes. 2004. Feasibility assessment for reintroducing fishers to Washington. Washington Department of Fish and Wildlife, Olympia, WA.

Luque, M. 1984. The fisher: Idaho's forgotten furbearer. Idaho Wildlife 4: 12-15

McLoughlin, P. D. and F. Messier. 2004. Relative contribution of sampling error in initial population size and vital rates to outcomes of population viability analysis. Conservation Biology 18: 1665-1669.

Miller, P. M. and R. C. Lacy. 1999. VORTEX: a stochastic simulation of the extinction process; version 8 user's manual. Conservation Breeding Specialist Group, Apple Valley, MN, USA.

Pack, J. C. and J. I. Cromer. 1981. Reintroduction of fisher to West Virginia. Pages 1431-1442 in J. A. Chapman and D. Pursley (editors). Proceedings of the worldwide furbearer conference. Frostbury, MD, USA.

Petersen, L. R., M. A. Martin and C. M. Pils. 1977. Status of fishers in Wisconsin, 1975. Wisconsin Department of Natural Resources, Rep. No. 92.

Powell, R A. 1993. The fisher: life history, ecology and behavior. University of Minnesota Press, Minneapolis.

Proulx, G, A J Kolenosky, M Badry, R K Drescher, K Seidel & P J Cole. 1994. Post-release movements of translocated fishers. Pages 197-203 in S W Buskirk, A Harestad, M Raphael & R A Powell (editors). Martens, sables, and fishers: Biology and conservation. Cornell University Press, Ithaca, New York.

Raine, R. M. 1981. Winter food habits, responses to snow cover and movements of fisher (*Martes pennanti*) and marten (*Martes americana*) in southeastern Manitoba. MS thesis. University of Manitoba, Winnipeg. 145 pp.

Roy, K D. 1991. Ecology of reintroduced fishers in the Cabinet Mountains of northwest Montana. MS Thesis, University of Montana, Missoula.

Schempf, P. F. and M. White. 1977. Status of Six Furbearer Populations in the Mountains of Northern California. USDA Forest Service, San Francisco, CA. 51 p.

Serfass, T, R Brooks, W Tzilkowski, D Mitcheltree & T Hardisky. 1996. Pennsylvania fisher reintroduction project. Pennsylvania Game Commission, Bureau of Wildlife Management, Research Division, Annual Job Report.

Strickland, M. 1994. Harvest management of fishers and American martens. Pages 149-164 in S W Buskirk, A Harestad, M Raphael & R A Powell (editors). Martens, sables, and fishers: Biology and conservation. Cornell University Press, Ithaca, New York.

Wallace, K & R Henry. 1985. Return of a Catskill native. The Conservationist 40(3): 17-19.

Weckwerth, R. P. and P. L. Wright. 1968. Results of transplanting fishers in Montana. Journal of Wildlife Management 32: 977-980.

Weir, R D. 1995. Diet, spatial organization, and habitat relationships of fishers in south-central British Columbia. MS thesis, Simon Fraser University, Burnaby, British Columbia.

Weir, R D, I T Adams, G Mowat & A J Fontana. 2003. East Kootenay fisher assessment. Unpublished report, British Columbia Ministry of Water, Land, and Air Protection, Cranbrook.

Wisely, S. M., Buskirk, S. W., Russell, G. A., Aubry, K. B. & Zielinski, W. J. 2004. Phylogeography and genetic diversity of the fisher (*Martes pennanti*) in a peninsular and peripheral metapopulation. Journal of Mammalogy 85: 64-648.

Zielinski, W. J., T. E. Kucera, and R. H. Barrett. 1995. The current distribution of fisher, *Martes pennanti*, in California. California Fish and Game 81:104-112.

Zielinski, W. J., R. L. Truex, F. V. Schlexer, L. A. Campbell, and C. Carroll. 2005. Historical and contemporary distributions of carnivores in forests of the Sierra Nevada, California. Journal of Biogeography 32:1385-1407.

Table 1. Baseline conditions for population matrix analyses of the fisher population in northwestern California. Baseline conditions were run 100 times for 100 years using *VORTEX* with stochastic variation as indicated by standard deviations (SD). Elasticity indexes the percent change in the probability of extinction when input variables changed by ± 10%.

		Elasticity (% change in probability of extinction)	
Demographic Variable	Value (± SD)	-10%	+10%
Starting population size, N_0	1000	4	2
Carrying capacity, *K*	2000 ± 250	7	8
Mean litter size	2.6 ± 1.0	23	3
Age (yr) first reproduction	2	—*	—*
Exponent for density dependence, *B*	16	0	0
Exponent for Allee Effect, *A*	0.5	0	0
Mortality rates			
Juveniles (age 0-1)	65 ± 25 %	3	39
Yearlings (age 1-2)	25 ± 20%	2	8
Adults (age ≥ 2)	12 ± 20%	0	9
Reproduction after logging	50%	4	8
Survival after logging	75%	—*	—*
Local subpopulations (and timber harvest/100 yrs)			
Total number	50		
Private land	25 (2 harvests/100 yrs)		
USFS land managed for timber	17 (1 harvest/100 yrs)		

*elasticity values not calculated.

Table 2. Predicted effects of removing fishers from the northwestern California population for 3 or 5 years, predicted by 100 runs of *VORTEX* for each set of values for variables. Fifty 'subpopulations' were included, where a subpopulation is an accounting unit rather than a distinct population subcomponent.

Removal Scenario			
Number Fishers Removed From Each Subpopulation	Number Subpopulations	Number Years	Increase in Probability of Extinction
1	20	3	0%
1	20	5	3%
5	4	3	2%
5	4	5	0%
5	4	8	1%

Table 3. Predicted index of successful reintroduction of fishers to the northern Sierra Nevada Mountains, predicted by 100 runs of *VORTEX* for each set of values for variables.

Reintroduction Scenario					
Number Released at Each of 5 Release Sites					
Adult Females	Adult Males	Juvenile Females	Juvenile Males	Number Years	Index of Success
3	2	0	0	2	2
3	2	0	0	3	20
3	2	0	0	5	42
3	2	0	0	8	82
1	4	0	0	5	3
1	3	0	0	5	4
1	2	0	0	5	3
2	3	0	0	5	29
2	2	0	0	5	26

4	1	0	0	5	66
2	2	1	0	5	40
2	2	0	1	5	27
2	1	1	1	5	40
2	1	2	0	5	54
2	1	0	2	5	17
1	1	3	0	5	55

Figure Captions

Figure 1. Predicted relationship between number of years that fishers are released and success in establishing a new fisher population in the northern Sierra Nevada Mountains. The *VORTEX* model included stochastic variation and was run for 100 years. Twenty fishers were modeled to be released each year in 4 groups of 5, each group with a sex ratio of 2 males:3 females, and each group released into a different site.

Figure 2. Predicted relationship between sex ratio for fishers that are released and success in establishing a new fisher population. The *VORTEX* model included stochastic variation and was run for 100 years. Twenty fishers were modeled to be released in each of 5 years in 4 groups of 5, with each group released into a different site.

Figure 3. Predicted relationship between numbers of adult female and adult male fishers released at each of 4 subsites in each of 5 years and success in establishing a new fisher population. No juvenile fishers were released in these scenarios. The *VORTEX* model included stochastic variation and was run for 100 years. Twenty fishers were modeled to be released each of 5 years in 4 groups of 5, with each group released into a different site.

Figure 4. Predicted relationship between numbers of female and male fishers, including juveniles, released at each of 4 subsites in each of 5 years and success in establishing a new fisher population. The *VORTEX* model included stochastic variation and was run for 100 years. Twenty fishers were modeled to be released each of 5 years in 4 groups of 5, with each group released into a different site.

Figure 5. Actual relationship between the numbers of fishers that were released for 28 releases and the number of years that fishers were released (tabulated by Roy (1991) and Washington

Department of Fish & Wildlife 2004). Successful releases are shown by solid circles, failed releases by open circles. These real data suggest that releases with more fishers and that lasted for more years had greater success ($p = 0.1$ for numbers of fisher, $p = 0.12$ for number of years). (Data from Anderson 2002, Aubry & Lewis 2003, Baird & Frey 2000, Benson 1959, Berg 1982, Bradle 1957, Dodds & Martell 1971, Drew et al. 2003, Heinemeyer 1993, Irvine et al. 1964, Kyle et al. 2001, Luque 1984, Pack & Cromer 1981, Petersen et al. 1977, Proulx et al. 1994, Roy 1991, Serfass et al. 1996, Wallace & Henry 1985, Weckwerth & Wright 1968, Weir et al. 2003, Weir 1995.)

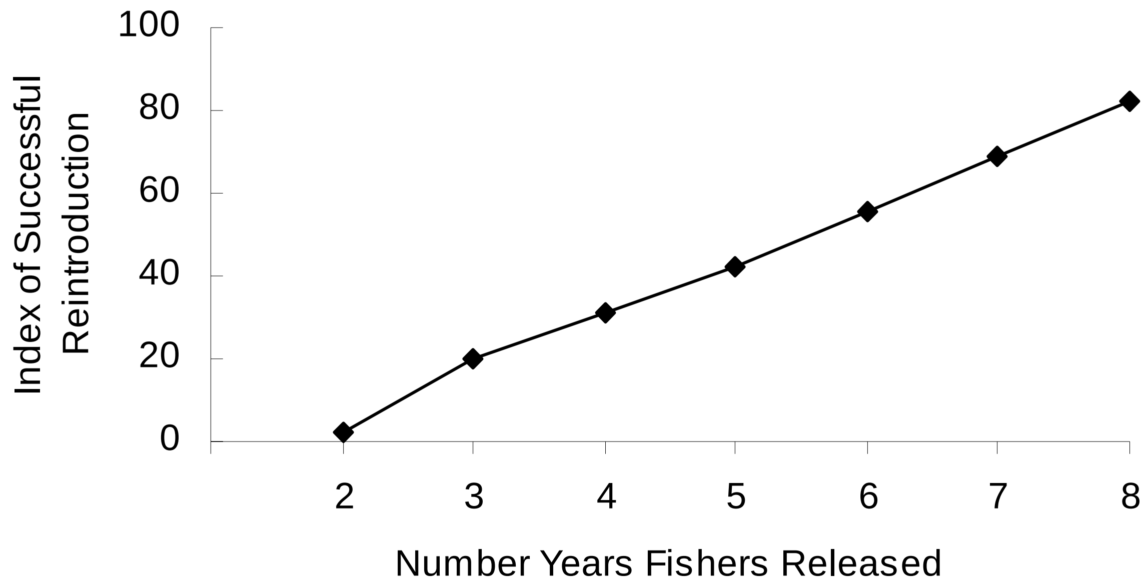

Index of Successful Reintroduction
100
80
60
40
20
0
2
3
4
5
6
7
8
Number Years Fishers Released

Figure 1

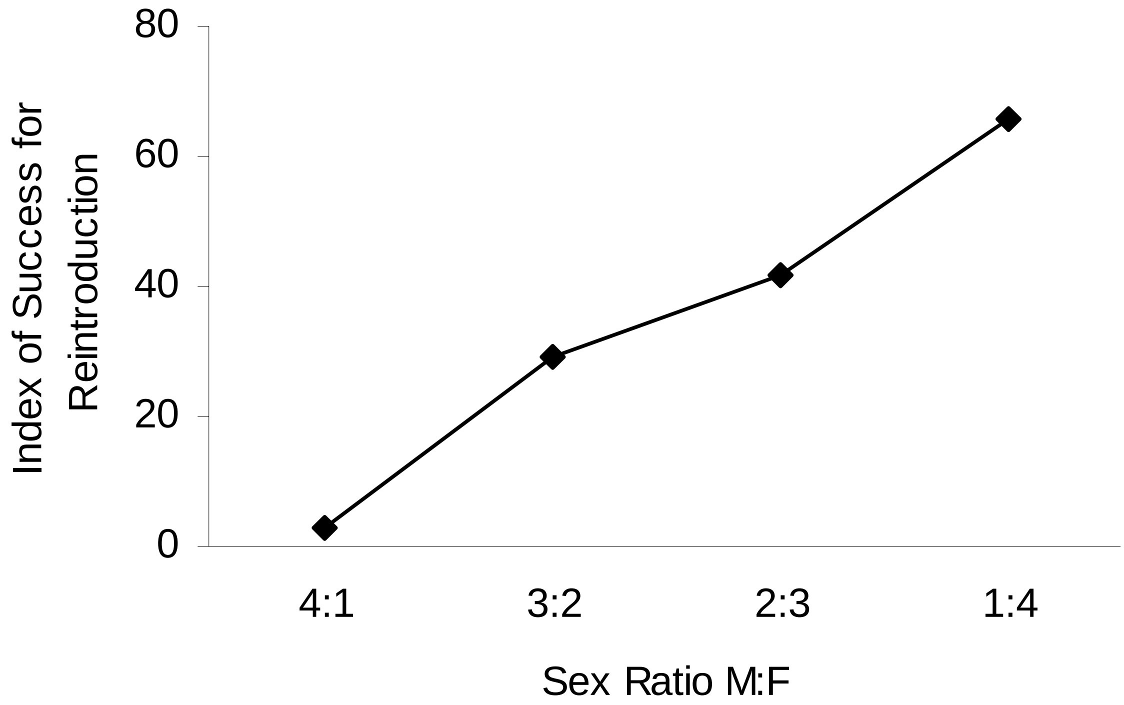

80
60
40
20
0
Index of Success for Reintroduction
4:1
3:2
2:3
1:4
Sex Ratio M:F

Figure 2

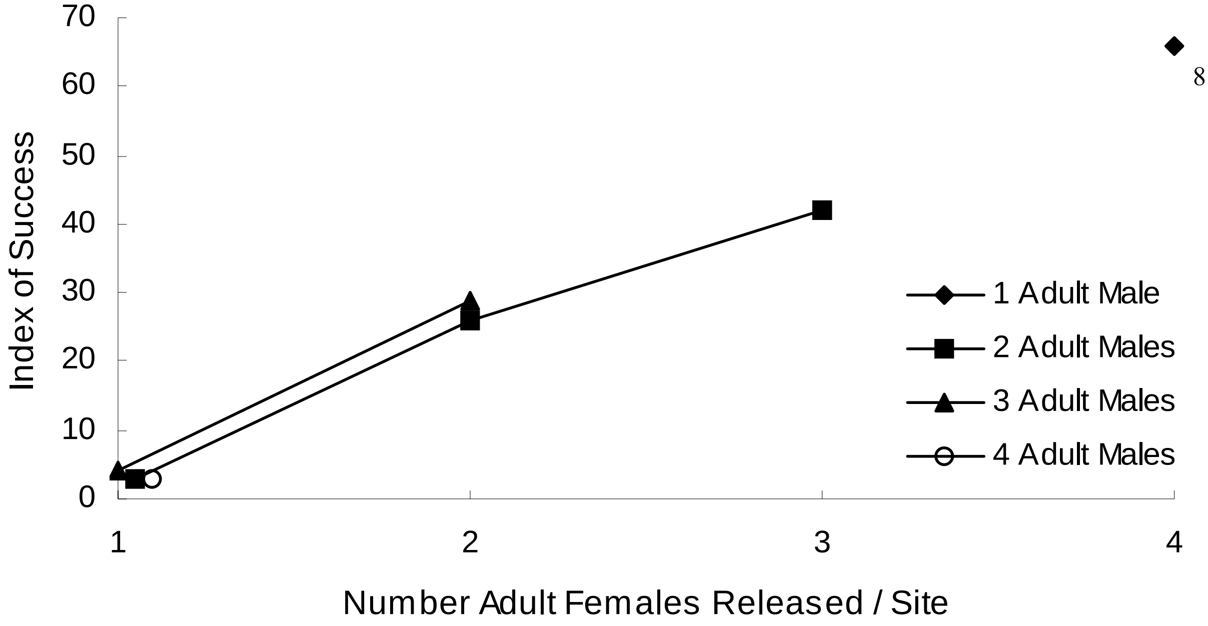

70
60
50
40
30
20
10
0
Index of Success
8
1 Adult Male
2 Adult Males
3 Adult Males
4 Adult Males
1
2
3
4
Number Adult Females Released / Site

Figure 3

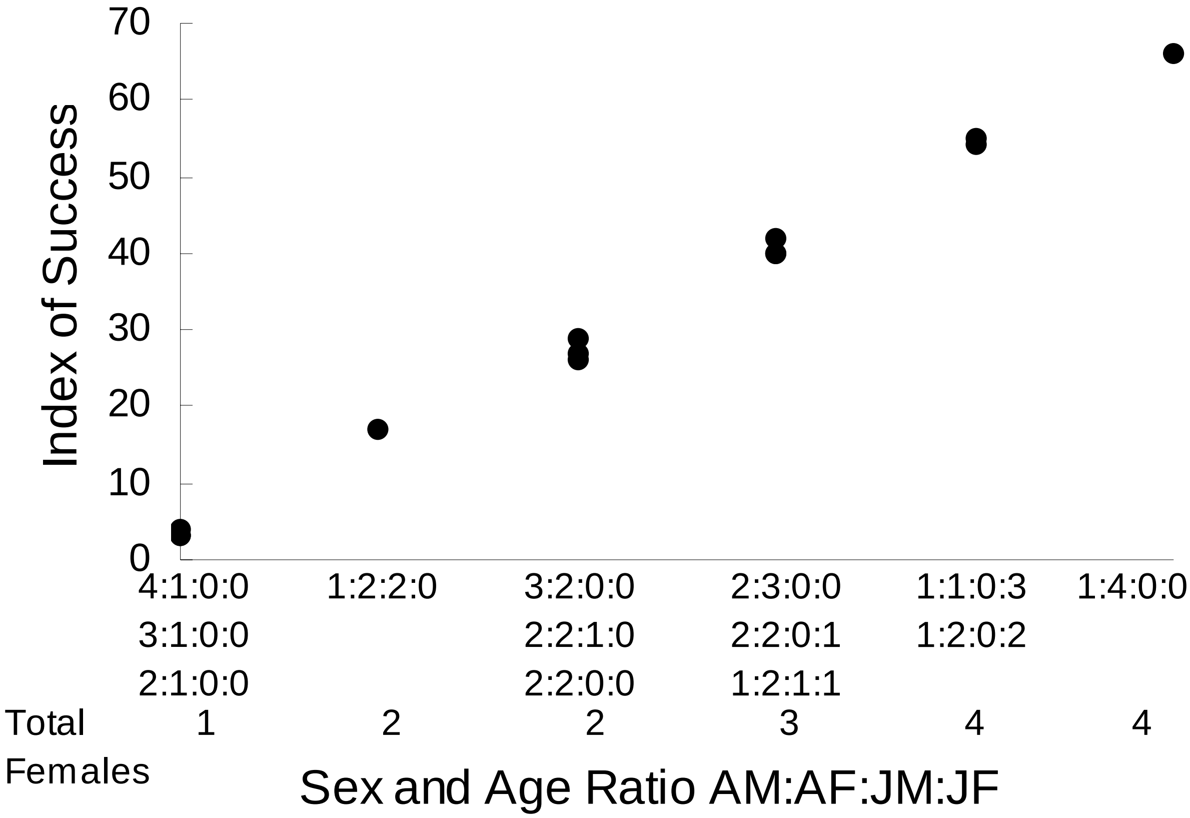
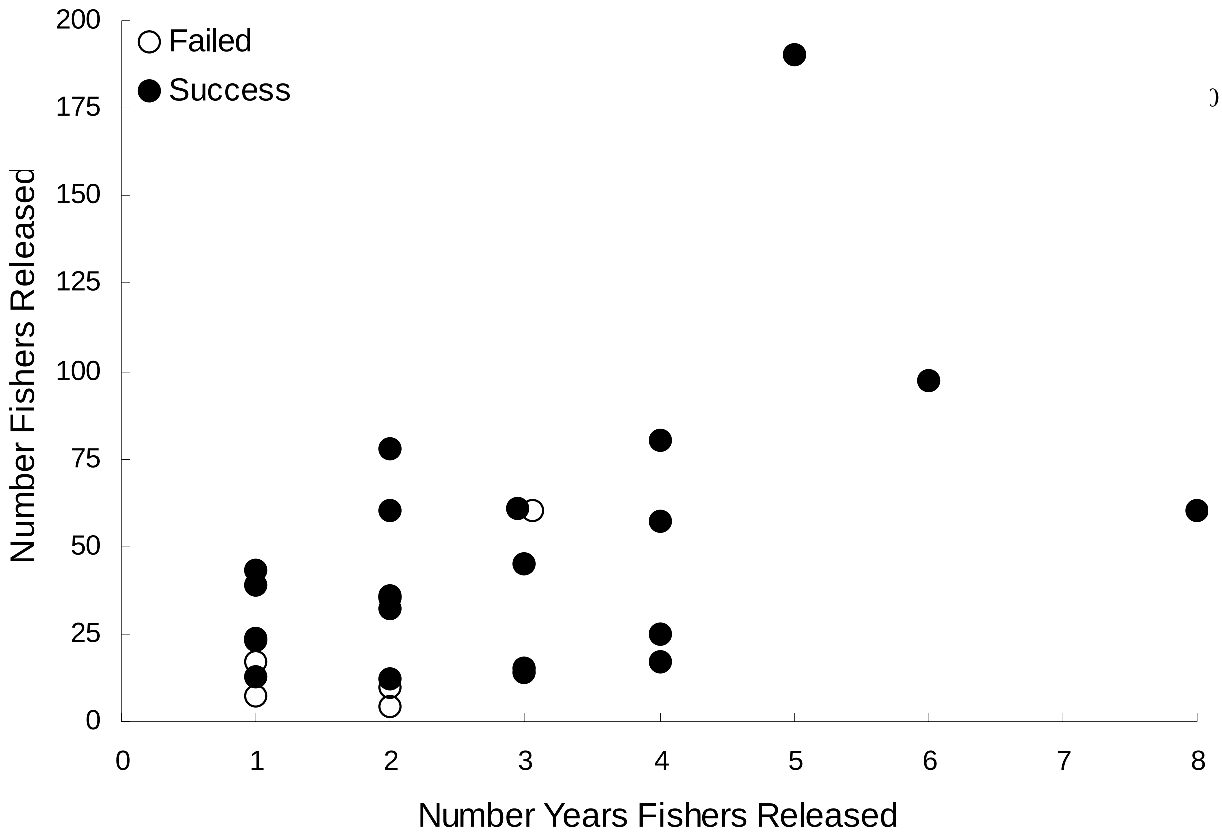

70
60
50
40
30
20
10
0
Index of Success
4:1:0:0
3:1:0:0
2:1:0:0
1:2:2:0
3:2:0:0
2:2:1:0
2:2:0:0
2:3:0:0
2:2:0:1
1:2:1:1
1:1:0:3
1:2:0:2
1:4:0:0
Total Females
1
2
2
3
4
4
Sex and Age Ratio AM:AF:JM:JF

Figure 4



Failed
Success
Number Fishers Released
Number Years Fishers Released
200
175
150
125
100
75
50
25
0
0
1
2
3
4
5
6
7
8

Figure 5