POPULATION VIABILITY ANALYSIS FOR GILA

TROUT (SALMONIDAE: *ONCORHYNCHUS*

GILAE), AN ENDANGERED

SOUTHWESTERN

FISH

By

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ABSTRACT

I used the computer program RAMAS to perform a population viability analysis (PVA) for the Gila trout, *Oncorhynchus gilae*, an endangered salmonid with extant populations restricted to headwaters of the Gila and San Francisco river drainages in southwestern New Mexico. An initial PVA model from life-history data for 10 extant populations was used to examine sensitivity of Gila trout viability to changes in a variety of factors, including population size, number of populations, severity and probability of catastrophic events, and a catch-and-release artificial-lure fishery. Catastrophes (modeled as the probability and severity of forest fires) and number of populations had the greatest effect on viability. The results indicate that a central factor in successful conservation of Gila trout is reduction of the severity of forest fires through a proactive program of fire management.

INTRODUCTION

In this paper, I present a Population Viability Analysis (PVA) for a federally listed endangered fish, the Gila trout (Salmonidae: *Oncorhynchus gilae*), an endemic of the Gila River system of the Colorado River drainage in southwestern United States. Historically, the species occurred throughout the upper San Francisco and Gila river drainages in southwestern New Mexico and the Verde River drainage in southcentral Arizona (Miller 1950; Behnke 1992). The Verde River population has been extirpated, and the range of the species has declined by more than 95% as a result of over-exploitation, stocking of non-native trouts, degradation and loss of habitat, and changes in water quality and quantity (Sublette et al. 1990; Propst et al. 1992, Propst 1994; Dowling and Childs 1992). Currently, Gila trout are restricted to a few small, headwater streams subject to catastrophic events such as drought, wildfire, flooding, and anchor ice (Rinne 1990).

Efforts to conserve and propagate Gila trout began in 1923 with establishment of the Jenks Cabin Hatchery by the New Mexico Game and Fish Department (Miller 1950). This hatchery and a similar facility at Glenwood, New Mexico were discontinued in 1939 and 1947, respectively (Propst et al. 1992). Since 1923, the New Mexico Department of Game and Fish has followed a

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policy of not stocking non-native salmonids into areas occupied by Gila trout (Propst et al. 1992). Additional conservation efforts for the species have included placement of stream improvement structures constructed of logs by the Civilian Conservation Corps in the 1930s and repatriation of populations in several additional streams (Rinne 1982; Propst et al. 1992). Current recovery efforts for the species began in the 1970s with relictual headwater populations isolated by natural barriers from upstream movement by non-native trout species (Propst and Stefferud 1997). Each of the five relict populations known at the time was believed genetically distinct (David 1976; Loudenslager et al. 1986) and emphasis was placed on establishing replicates of each population in other streams. This involved removal of non-native salmonids by chemical poisoning and stocking of Gila trout from one of the five relict populations. A sixth relict population was discovered in 1992 in Whiskey Creek, a small tributary of the West Fork of the Gila River (N. Smith, pers. comm.).

A major value of PVA is that it provides a basis for sensitivity analyses aimed at evaluating the robustness of population viability to changes in variables potentially affecting risk of extinction (Akcakaya 1992; Boyce 1992; Reed et al. 1998). These results may then suggest hypotheses for conservation management (Reed et al. 1998). In this study, I examine the sensitivity of the PVA model to variation in several population parameters and the effects and probabilities of catastrophes. Two major lineages of Gila trout have been identified genetically, one comprising populations in the Gila River drainage and

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the other comprising populations in the San Francisco River drainage (Riddle et al. 1998; R. Leary, pers. comm.). Therefore, I developed viability models for the species and for each of the two lineages separately. The results provide insight into the efficacy of various management options for the conservation of the species.

The models presented herein focus on environmental stochasticity as the primary controlling factor in the viability of Gila trout, an approach that avoids the complications and inaccuracies associated with demographic and genetic stochasticity (Akcakaya 1992). As with any model, "relative" effects of varying different parameters are more reliable than "absolute" probabilities of extinction. PVA models are more useful as a tool to guide management options than they are as predictors for the fate of a species (Akcakaya et al. 1995).

I used the program RAMAS/GIS (Akcakaya 1994) to model population viability. RAMAS appears to be the best available PVA program for fishes such as the Gila trout (R. C. Lacy, pers. comm.). Most other programs are designed for species with low population numbers and low rates of reproduction, whereas RAMAS is appropriate for any size of population and level of fecundity. RAMAS uses a Monte Carlo simulation of age- or lifestage-structured population growth based on Leslie matrices (Leslie 1945; Ferson et al.1991) to model extinction risk for metapopulations. The program has been used successfully in PVAs for leopard darter *Percina pantherina* (Williams 1997; Williams et al. in press), striped bass *Morone saxatilis* (Ginzberg et al. 1990), and bluegill sunfish

Lepomis macrochirus (Ferson et al. 1991).

Gila trout is the only trout species listed as endangered under the Endangered Species Act of the United States. Several other trouts have been downlisted from endangered to threatened status as a result of conservation efforts (Behnke 1992). My objective was to use PVA to evaluate management options that might contribute toward conservation of Gila trout.

MATERIALS AND METHODS

Study area and history of Gila trout

Gila trout occupy streams in narrow, steep-gradient canyons and small, moderate-gradient valleys in the Black and Mogollon mountain ranges of southwestern New Mexico (Propst and Stefferud 1997). Flow in canyons is often characterized by swift-running waters with numerous cascades and plunge pools. In the valleys, streams have meandering channels and cobble-riffles with more widely separated pools, many of which are formed around log-debris piles and boulders. Base flows in the summer range from <0.05 cubic meters per second ($m^3 s^{-1}$) in the smallest streams to about 0.65 $m^3 s^{-1}$ in the largest stream (Propst and Stefferud 1997). Riparian vegetation consists of Arizona alder *Alnus oblongifola* and Arizona sycamore *Platanus wrightii* along lower elevation streams; western box elder *Acer negundo*, willow *Salix* spp., New Mexico locust *Robinia neomexicana*, narrowleaf cottonwood *Populus angustifolia*, and

ponderosa pine *Pinus ponderosa* in mid-elevation streams, and blue spruce *Picea pungens*, white fir *Abies concolor*, and quaking aspen *Populus tremuloides* along high-elevation streams (Propst and Stefferud 1997).

Fire plays an integral part in the maintenance and function of the ecosystems occupied by Gila trout. The original fire regime consisted primarily of lightning-caused surface fires occurring in spring and early summer and ceasing with the rainy ("monsoon") season in July-August (Rinne 1996). Swetnam and Dieterich (1985) found historic fire intervals of 3-7 years in the range of Gila trout, and Cooper (1960) concluded that, prior to the 1950s, crown fires were extremely rare or nonexistent in the region.

Starting in the early 1900s, however, fuel loads began to increase, likely as a result of increased livestock grazing and a policy of fire suppression by the newly established U. S. Forest Service (Swetnam and Dieterich 1985; Covington and Moore 1994). Fire suppression activity and diminished herbaceous cover caused by grazing reduced the frequency of wildfire and resulted in increased woody debris and sapling densities and promoted brush invasion. These changes in forest structure have increased the potential for catastrophic crownfires (Rieman and Clayton 1997). By the early 1900s, populations of Gila trout were restricted to the upper reaches of a few headwater streams as a result of habitat modifications, together with introductions of non-native trouts and overfishing by prospectors, ranchers, and others (Miller 1950; Propst et al. 1992). In these small isolated systems, refugia from ash flow are limited and

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opportunities for recolonization often are nonexistent. Consequently, in the past decade, six populations of Gila trout have been extirpated by extreme fire events followed by intense summer (July-August) rains that washed ash and debris into the stream (Table 1). The Divide Fire in 1989 resulted in extirpation of the population in Main Diamond Creek (Propst et al. 1992). The Bonner Fire in 1995 extirpated the populations in South Diamond and Burnt Canyon creeks (Propst and Stefferud 1997). The Lookout fire in 1996 extirpated the populations in Trail Canyon, Woodrow Canyon and Sacaton creeks (J. Brooks pers. comm.; pers. obs.).

Gila trout populations in many streams within the historic range of the species have been eliminated through hybridization with non-native rainbow trout (*Oncorhynchus mykiss*) and resultant genetic introgression (Loudenslager et al., 1986; Riddle et al., 1998; Leary and Allendorf, in preparation). I have not attempted to model effects of hybridization and genetic introgression because these factors provide few management options that can be evaluated by PVA. Management options can be a function of levels of genetic introgression, and the levels associated with different options are debatable and somewhat arbitrary. If the level of genetic introgression is sufficiently low (e.g., 0.01 or less), then the population might be treated as a pure population of Gila trout (Allendorf and Leary, 1988). Even if the level of introgression is higher (e.g., 0.05-0.10), then agencies should consider the possibility (Dowling and Childs, 1992) that, although hybridized, the population still contains locally adaptive mutations of

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Gila trout. In such instances, the population could be managed as pure Gila trout, or genetic swamping with pure Gila trout stock might be advisable. With high levels of introgression, agencies might have no choice but to eliminate the hybrid population and restock with pure Gila trout. All of these possibilities also are affected by a variety of other variables including population size (e.g., genetic swamping is more practical with smaller populations) and whether the population involved is relictually native or a replicate of such in another stream. The PVA models presented here ignore effects of hybridization under the assumption that affected populations will be managed as pure Gila trout or rather quickly restored by appropriate management action.

Sampling and population estimates

The 10 Gila trout populations (Table 2) included in the initial PVA model (= base model) represent those considered free of genetic contamination by non-native congeners in 1996. Subsequently, three of these were found to be genetically introgressed by rainbow trout (Leary and Allendorf, in prep.). One of the three was restocked with pure Gila Trout in 1997. The other two represent relictual populations of the species, and because they exhibit relatively low levels of genetic introgression (Iron Creek, 0.02; McKenna Creek, 0.05), management agencies have decided to manage them as Gila trout, at least in part because each may harbor locally adaptive genetic material. Consequently, I retained the original 10 populations included in the PVA.

Life-history data were compiled primarily from the literature, but

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population size estimates (N) for six streams were based on field data gathered during this study in May through September 1996 and 1997. These six streams were Iron, McKnight, McKenna, and Mogollon creeks in the Gila drainage and Spruce and Dry creeks in the San Francisco drainage. For each stream, a battery-powered, backpack electroshocker (24 V, DC) was used to sample one to three 200-m sites, with number of sites dependent on length of stream. Geographic position was recorded from GPS (Global Positioning System) readings. Prior to sampling, each site was blocked at the upper and lower ends with fine-mesh nets. Depletion sampling was then done by making three to four passes upstream through each site, capturing stunned fish with dipnets. To minimize injury and ensure equal capture effort between passes, no effort was made to "hunt" individuals. All Gila trout captured were weighed to nearest 0.1 g and measured to nearest mm for total and standard lengths. Number of fish at each site was estimated by the depletion method (Zippin 1958). Population sizes within each stream were obtained by multiplying estimated number of fish per meter of stream in the sample area by total length of stream occupied by Gila trout. Length of stream occupied was taken from Propst et al. (1992) and Propst and Stefferud (1997).

Depletion-shocking efforts consistently captured about 60% (N = 20, $\bar{x}$ = 0.57, SE = 0.05) of the population estimate in the first pass at each sample site. This percentage was used in estimating population size for four Gila trout streams (Main Diamond, Sheep Corral, Whiskey, and White Creeks) for which

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only single-pass electrofishing data were available (Propst and Stefferud 1997).

Stage-specific structure, survivorship, and fecundity

I estimated stage-specific (size-class) structure (proportionate abundance of different life stages) from published length-frequency information on Gila trout (Propst and Stefferud 1997). I used three lifestage-size classes as defined by Propst and Stefferud (1997): juveniles (<100 mm TL), subadults (100-150 mm TL), and adults (>150 mm TL). Survivorship estimates (Table 3) were computed from stage-specific abundances as described by Caswell (1989).

The estimate of individual fecundity was based on overall mean count of ova from 25 field-stripped females from Main Diamond and McKnight creeks (Nankervis 1988; Propst unpubl. data); this mean value was divided by two to arrive at individual fecundity (Table 3) for RAMAS, which effectively models the situation where each individual is capable of asexual reproduction. Dividing the mean fecundity of females by two assumed a 1:1 sex ratio and successful reproduction for all adult females every year. Nankervis (1988) found that a small proportion (13%) of females classified here as subadults were reproductive and minimum size of reproduction was 130 mm. I estimated "subadult" fecundity by multiplying 0.13 by the mean proportion of 130-150 mm individuals (0.47) and then multiplying this constant by one-half of the mean ova count for "subadult' females (30.8).

Population Viability Analysis

Extinction risk for Gila trout in the PVA models was expressed as the

percentage of replicate simulations in which extinction of the species occurred within 100 years. All simulations were performed with 1000 replications. In the base model, I used forest fires as the major source of environmental catastrophe, and severity of catastrophe was modeled at 100% population reduction. Probability of such a fire was based upon known effects on populations of Gila trout for the past 27 years (1971-1997), the period of time that the species has been intensively monitored. During that time, six populations were eliminated by forest fires and subsequent ash flow into streams (USFS unpubl. data; D. Propst, pers. comm.; pers. obs.). I arrived at probability of catastrophe for the base model (2%/population/year) by dividing number of extirpations of Gila trout populations (6) resulting from catastrophic fires, by total number of stream-years (288; computed from data in Table 1) for the species during the past 27 years.

Parameter values (Table 3) were used to develop a base model for viability of the Gila trout. I used the statistically conservative Komolgorov-Smirnov D-test (Akcakaya 1994; Sokal and Rohlf 1994) to evaluate significance of differences in extinction rates between the base model and a variety of other models, each differing in a single parameter. Sensitivity to effect of catastrophe (% reduction in N) was modeled by decreasing the effect from extirpation (100% reduction) to no reduction (0%) in increments of 5%. To examine sensitivity to probability of catastrophe, I increased the fire-flood return interval from the base model of once every 27 years (2%/population/year) to once every seven

(14.3%), five (20%), and three years (33%). Those rates bracket the range of the pre-1900 fire-return interval for forest areas in the historical range of Gila trout in New Mexico (Swetnam and Dieterich 1985).

To assess effect of population size, the estimate for each population was doubled in one model and halved in another. This was a crude attempt to model effect of extending or shortening the length of stream occupied by the species in each stream. It also allowed assessment of the robustness of the base model to error in estimating population size.

To access effect of fecundity upon viability, the estimate was doubled in one model and halved in another. To assess sensitivity to life-stage structure, I modeled the mean plus or minus one standard-deviation for the proportionate abundance of each life stage separately; for each of these models, proportionate abundances of the other two life stages were adjusted by addition or subtraction, with the amount of adjustment depending on their relative contributions to stage structure in the base model.

Sensitivity to number of populations was examined by considering four models in which populations were added to the base model of 10 streams. First, I added six streams presently devoid of Gila trout because of hybridization or fire-flood (Table 2). Projected population-size estimates for those were based on past estimates of trout density in those streams (Propst and Stefferud 1997) and calculated using the previously defined method for single-pass collection data. In the other three models, the model just described received an additional 5, 10, or

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15 hypothetical populations having the average population size of the 10 streams in the base model.

To assess effects of a catch-and-release, artificial-lure, or fly-only fishery on local populations, I examined viability of the McKenna and McKnight creek populations with an annual catastrophe that reduced population sizes by 5, 10, and 15%, respectively, in three separate models for each population. These reductions probably were overestimates because studies have indicated that only 3-10% of individual trout die as a result of hooking by artificial-lure or fly fishing (Nuhfer and Alexander 1992; Taylor and White 1992; Schisler and Bergersen 1996; Schill 1996). Additonally, a model simulating annual reduction of 15 and 30%, respectively, of stage 2 and stage 3 individuals was performed to access effect of a catch-per-day limit of two trout over 150 mm. The basis of these analyses was to model effect of a fishery management plan in which one to two streams would be opened for fishing after downlisting of the species and additional streams would be opened for fishing as others are closed for renovation (D. Propst pers. comm.).

RESULTS

Life-history data used in PVA models and lengths of streams occupied by Gila trout are given in Tables 2 and 3. Estimates of the probabilities of extinction over 100 years under base conditions with varying severity and return intervals

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of catastrophe are shown in Figure 2. Under base-model conditions, the estimated probability of Gila trout extinction in 100 years was 36%. As expected, increased severity of catastrophe (measured by reduction in abundance per event) and shorter fire return intervals were associated with increased risk of extinction.

The base model was relatively insensitive to population size. Doubling and halving of population sizes had no significant effect on extinction rate (Table 4; Fig. 3). The estimates of population size used in the base model are somewhat questionable because they assume that observed local densities can be extrapolated to the entire reach of stream occupied by Gila trout. However, the model seems robust to this source of error. Correspondingly, simulating a catch-and-release or two-fish limit fishery causing annual mortality of 5-30% of adults and subadults had no significant effect on viability of either the McKenna or the McKnight creek populations. Viability of the species was also insensitive to changes within one standard deviation of the mean in proportionate abundances of the three life-stages.

The model was sensitive to large changes in fecundity estimates (F). Doubling and halving of fecundity indicated significant ($P < 0.001$) differences from the base model in probability of extinction ($\frac{1}{2}$F, 47%; F, 36%; 2F, 31%). Our fecundity estimates are based on a small sample size (N = 25) with large variance (Table 3) and they assume no local population variation. Thus, a more refined model of extinction risk would require more accurate fecundity estimates.

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This, does not, however, invalidate the attempt herein to use the base model as a basis for insight into different management strategies.

The PVA was, as expected, sensitive to number of populations. The model incorporating the planned restocking of six additional streams with Gila trout indicated a reduction of extinction risk from 36% to 21%. Adding 5, 10 and 15 "average" populations lowered the risk to 12% , 7%, and 5%, respectively (Table 4; Fig. 4). Probability of extinction from each of these models was significantly different from those of the others ($P < 0.01$ in all pairwise tests).

Comparing extinction risks of the Gila River lineage of Gila trout (45%) and the San Francisco River lineage (81%) to that of all drainages combined (36%) indicates that, as expected, both lineages have significantly ($P < 0.001$) higher probabilities of extinction than does the species as a whole (Fig. 7). The model incorporating the planned restocking of six streams (all in the Gila River drainage) gave a significantly lower risk of extinction for the Gila River lineage (26%; Fig. 5). Adding those six populations plus five or 10 "average" populations also resulted in significantly lower risks (15% and 8%, respectively; $P < 0.001$; Table 4, Fig. 5). Adding two "average" stream populations to the San Francisco River lineage significantly decreased extinction risk ($P < 0.001$) from 81% to 67% (Table 4). Adding six "average" populations reduced the chance of extinction to 44%, a significant decrease from 67% ($P < 0.001$; Fig. 6).

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DISCUSSION

The probability of extinction of Gila trout within 100 years (36%), as computed under the conditions of the base model presented herein, is only a benchmark for comparison of the effects of different management strategies. Results from such models should not be treated as realistic assessments of extinction risks (Ackakaya et al. 1995; Reed et al. 1998), nor should they be used to classify species according to endangered status (Taylor 1995). Further, recommendations from sensitivity analyses generally should be treated as hypotheses to be empirically evaluated before implementation by management agencies (Reed et al. 1998).

If the Gila trout were left unmanaged, as assumed by the base model presented here, the risk of extinction would be much higher than indicated, because not all risk factors were included. Most importantly, the models do not include population losses resulting from interactions with non-native trout species (hybridization, competition, and predation). Although such interactions have been (Miller 1950) and continue to be (Propst and Stefferud 1997; R. Leary, pers. comm.) important in the decline of Gila trout, they were not included in the PVA because they allow few management options beyond stream renovation and restocking or strategies to prevent introductions of non-native trouts.

The altering of the historic fire regime in southwestern New Mexico from

cool-burning surface fires with regular return intervals of 3-7 years (Swetnam and Dieterich 1985) to less frequent, but more catastrophic crown fires has frustrated efforts to restore Gila trout to a level where the species can be downlisted from endangered to threatened (Propst et al. 1992). Correspondingly, models presented here suggest that viability of Gila trout is especially sensitive to effects of forest fires. Ignoring other factors of catastrophic loss, primarily effects of non-native trouts, the models suggest that the risk of extinction would be near zero if effects (% population reduction) of potentially catastrophic fires were reduced by a proactive fire management program (Fig. 2).

Much of the Gila National Forest is under prescribed natural fire management that allows naturally occurring fires to burn in certain areas and under certain constraints. These fires, however, may not be adequate to reduce fuel loads to a level sufficient to prevent catastrophic crown fires of the type observed in the recent past. Active prescribed burning may be needed to accomplish this goal. Prescribed burns in autumn, when the fuel moisture levels are high and daily temperatures are low, would allow cool, surface-burning fires to reduce fuel loads while minimizing chance of fire escaping from the prescribed area. The reduction of fuel loads by more frequent fires of this type should contribute a more natural forest structure, thereby reducing the frequency of catastrophic fires (Pyne et al. 1996).

My results indicate that prescribed fires with a return interval as short as three years would not increase the extinction risk, even if as much as 50-60% of

the local population is lost with each event (Fig. 2). Such losses should be minimized to reduce genetic and demographic stochasticity, both of which can negatively affect survival of a population (Boyce 1993). Further, the suggested beneficial effects of more frequent fires of lower intensity should be treated as a hypothesis to be tested prior to full-scale implementation in the management of Gila trout.

The model of Gila trout viability was insensitive to size of individual populations. However, the model does not recognize that increased population size requires a corresponding increase in habitat, which, for Gila trout, is primarily a function of length of stream occupied. Increased stream length generally would increase the probability of trout surviving catastrophic events in refugial areas (i.e., tributaries) not directly affected by the catastrophe. The type of wildfires occurring in the last few years usually have been limited to single or small numbers of watersheds where resident trout populations often have had no refugia from post-wildfire ash-flows associated primarily with mid- to late summer rains. Increasing stream lengths often would increase the number of tributaries occupied by Gila trout, thereby reducing the effect of catastrophe from 100% loss of the population to a loss of lesser magnitude, and the models indicate that this can have a significant effect on risk of extinction (Fig. 2). Further, a marked increase in amount of habitat (length of stream) occupied would re-establish natural connectivity among the now-isolated local populations of Gila trout. Increased connectivity would heighten the rate of recolonization following

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catastrophic losses in local areas, thereby improving the viability of the species. In New Mexico alone, the existing populations occupy less than 20% of the approximately 825 km of stream theoretically available for restoration of the species in the Gila River drainage. Similar opportunities exist within the historic range of the species in Arizona, a possibility being considered by the State of Arizona and the U. S. Fish and Wildlife Service (J. Stefferud, pers. comm.).

Like many other conservation efforts for endangered and threatened species, recovery of Gila trout is a complicated and controversial political issue. Some of the public opposition to recovery efforts for Gila trout has been in response to closures of streams to fishing after they have been restocked with the species. The PVA models incorporating an annual "catastrophe" that reduced adult and subadult abundances by as much as 30% had no significant effect on viability of the affected populations, indicating that a regulated fishery would not increase extinction risk for the species.

Consideration should be given to focusing a high proportion of conservation efforts on the San Francisco River lineage. The PVA indicates that this lineage has a much higher extinction risk than the Gila River lineage (81% vs. 45% in 100 years; Fig. 7). Additionally, the two populations of the lineage are geographically very close (Spruce Creek is tributary to Dry Creek), and the past history of Gila trout demonstrates the possibility that both might be eliminated by a single, catastrophic wildfire.

Ongoing efforts to preserve Gila trout emphasize three general

approaches: 1) reducing opportunities for hybridization and other interactions with congeners, 2) increasing number of streams occupied, and 3) restocking streams from which the species has been extirpated by catastrophes or hybridization with congeners (U. S. Fish and Wildlife Service 1993; D. Propst, pers. comm.). My results suggest that a fourth approach is central to the success of this effort; namely a proactive effort to reduce catastrophic effects of wildfires. Besides reducing the expense and effort involved in restocking local areas of extirpation, such an approach would help preserve genetic variation. Repeated restocking is likely to result in losses of genetic variability as a result of genetic drift. For example, all extant populations of the Main Diamond Creek and South Diamond Creek lineages exist only as populations derived from either captive, hatchery populations or from other transplanted populations. Such a program will almost certainly lead to reduced genetic variation (Stockwell et al. 1996; Dunham and Minckley, 1998). My models of Gila trout viability were highly sensitive to the effect of forest fires and indicate that only a small reduction in the effect of this factor greatly increases the viability of the species. Thus, it seems desirable from the standpoint of both management practicality and the long-term genetic viability of the species to implement an aggressive, proactive program of fire management in watersheds supporting Gila trout.

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TABLE 1. Years of occurrence for stockings/restockings and extirpations of Gila trout by forest fire/flood for 17 streams during the 27-year period from 1971 to 1997. Data on stocking history and extirpation are based on Propst et al. (1992) and on information from the Gila Trout Recovery Team and the U. S. Forest Service records.

Drainage: Relictual population (Replicate population)	Extirpation by fire/flood	Stocking/ restocking
San Francisco River drainage:		
Spruce Creek	No	Relict
(Dry Creek)	No	1985
Gila River drainage:		
Iron Creek	No	Relict
(Sacaton Creek)	1996	1990/97
(White Creek)	No	1994
McKenna Creek	No	Relict
(Little Creek)	No	1982
Main Diamond Creek	1989	Relict/1995
(McKnight Creek)	No	1970
(Sheep Corral Creek)	No	1972
South Diamond Creek	1995	Relict/1997
(Burnt Canyon)	1995	Relict
(Mogollon Creek)	No	1989/1997
(South Fork Mogollon Creek)	No	1997
(Trail Canyon)	1996	1988/1997
(Woodrow Canyon)	1996	1989/1997
Whiskey Creek	No	Relict

TABLE 2. Length of stream occupied and population size estimates (N) for Gila trout populations used in viability analyses. Numbers associated with the name of each population correspond with those in Figure 1. Populations without asterisk are those used in the base model; those with an asterisk are streams that are either presently devoid of Gila trout, but targeted for restocking, or they were restocked subsequent to the viability analysis; these six were used in the anaylsis of the effect of adding populations of Gila trout.

Drainage/Population	Occupied length of stream (km)	N
San Francisco River drainage		
1. Spruce Creek	3.7	2236
2. Dry Creek	1.9	537
Gila River drainage		
3. Sacaton Creek *	1.6	1101
4. Mogollon Creek	14.2	9651
5. Woodrow Canyon *	0.4	188
6. Trail Canyon *	1.8	1211
7. South Fork Mogollon Creek*	1.2	1128
8. Sheep Corral Creek	1.3	149
9. Whiskey Creek	0.2	20
10. White Creek[a]	12.0	8248
11. McKenna Creek	1.2	1038
12. Iron Creek	4.3	1529
13. Main Diamond Creek	6.1	5795
14. Burnt Canyon *	1.5	115
15. South Diamond Creek *	5.2	2080
16. McKnight Creek	8.5	2159
Average (N)		2324

[a] Population size estimate based on a non-native rainbow trout population (D. Propst, pers. comm.).

TABLE 3. Life-history variables and values used in the base model of Gila trout viability.

Variable	Value ± SD
Number of Life-stages	3
Fecundity	
Stage 1 (Juvenile)	0
Stage 2 (Subadult)	1.88 ± 0.97
Stage 3 (Adult)	98.57 ± 66.47
Initial Stage Structure Proportions	
Stage 1 (Juvenile)	0.72 ± 0.13
Stage 2 (Subadult)	0.25 ± 0.03
Stage 3 (Adult)	0.04 ± 0.01
Survivorship	
Stage 1 (Juvenile)	0.491 ± 0.445
Stage 2 (Subadult)	0.128 ± 0.063
Stage 3 (Adult)	0.430 ± 0.068
Catastrophe Probability [a]	2.0%
Catastrophe Effect [b]	100%

[a] Probability of catastrophe in a given year for each population.

[b] Effect is percent reduction of a population for each catastrophe occurrence.

TABLE 4. Effects of population size (N) and number of populations on probability of extinction for Gila trout over its extant range and in the Gila and San Francisco drainages separately. Values shown are percent probability of extinction in 100 years (± SD). Asterisks signify significant difference from all other models in the subset (P < 0.01).

Populations	Probability of extinction (%)		
	½ N	N	2N
Gila and San Francisco drainages			
Existing	40.0 ± 2.8	36.0 ± 2.8	34.0 ± 2.8
Projected [a]		21.0 ± 2.8 *	
Projected + 5 [b]		12.0 ± 2.8 *	
Projected + 10 [b]		7.0 ± 2.8 *	
Projected + 15		5.0 ± 2.8 *	
Gila River lineage			
Existing	48.0 ± 2.8	45.0 ± 2.8	44.0 ± 2.8
Projected [a]		26.0 ± 2.8 *	
Projected + 5 [b]		15.0 ± 2.8 *	
Projected +10 [b]		8.0 ± 2.8 *	
San Francisco lineage			
Existing	83.0 ± 2.8	81.0 ± 2.8	80.0 ± 2.8
Existing + 2 [c]		67.0 ± 2.8 *	
Existing + 6 [c]		44.0 ± 2.8 *	

[a] Ten populations in the base model (= "existing") plus six additional populations in streams designated for restocking with Gila trout.

[b] Projected populations plus five or ten populations with the average size of all existing populations of Gila trout.

[c] Existing populations in Spruce and Dry creeks plus two or six populations with the average size of all existing populations of Gila trout.
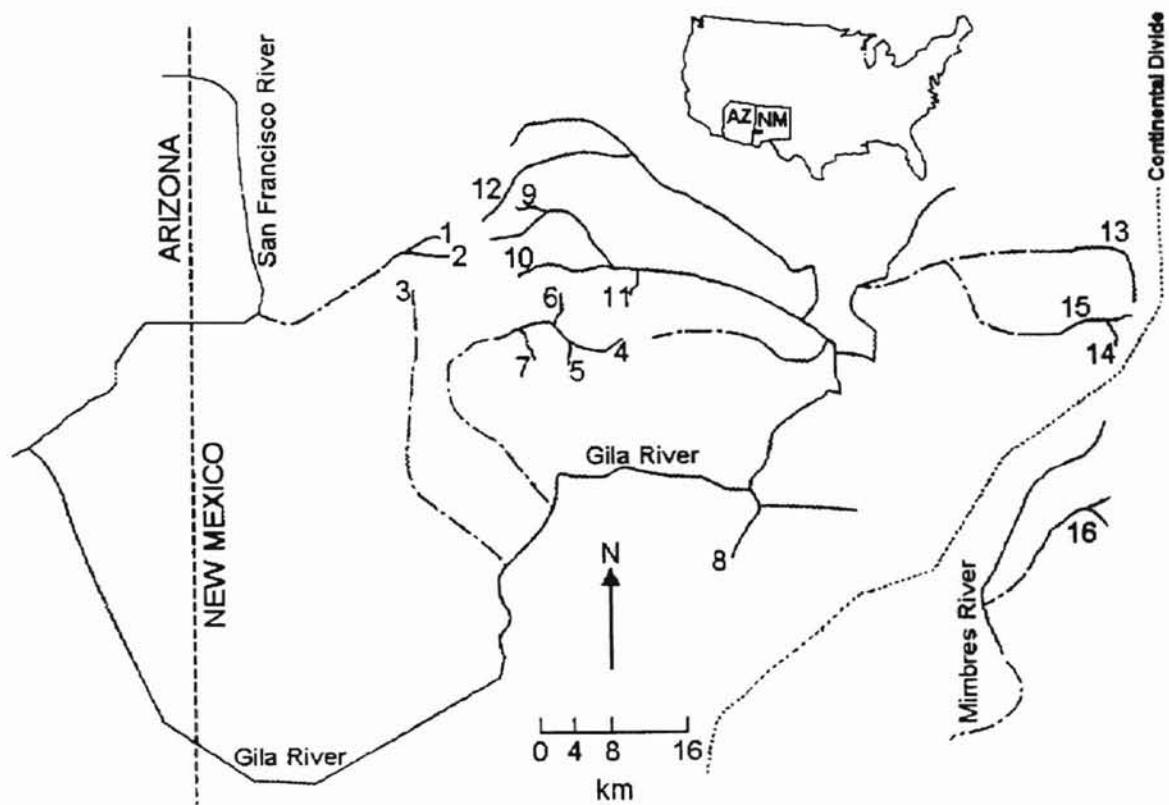
Figure 1. Map of upper Gila River drainage showing populations used in PVA.
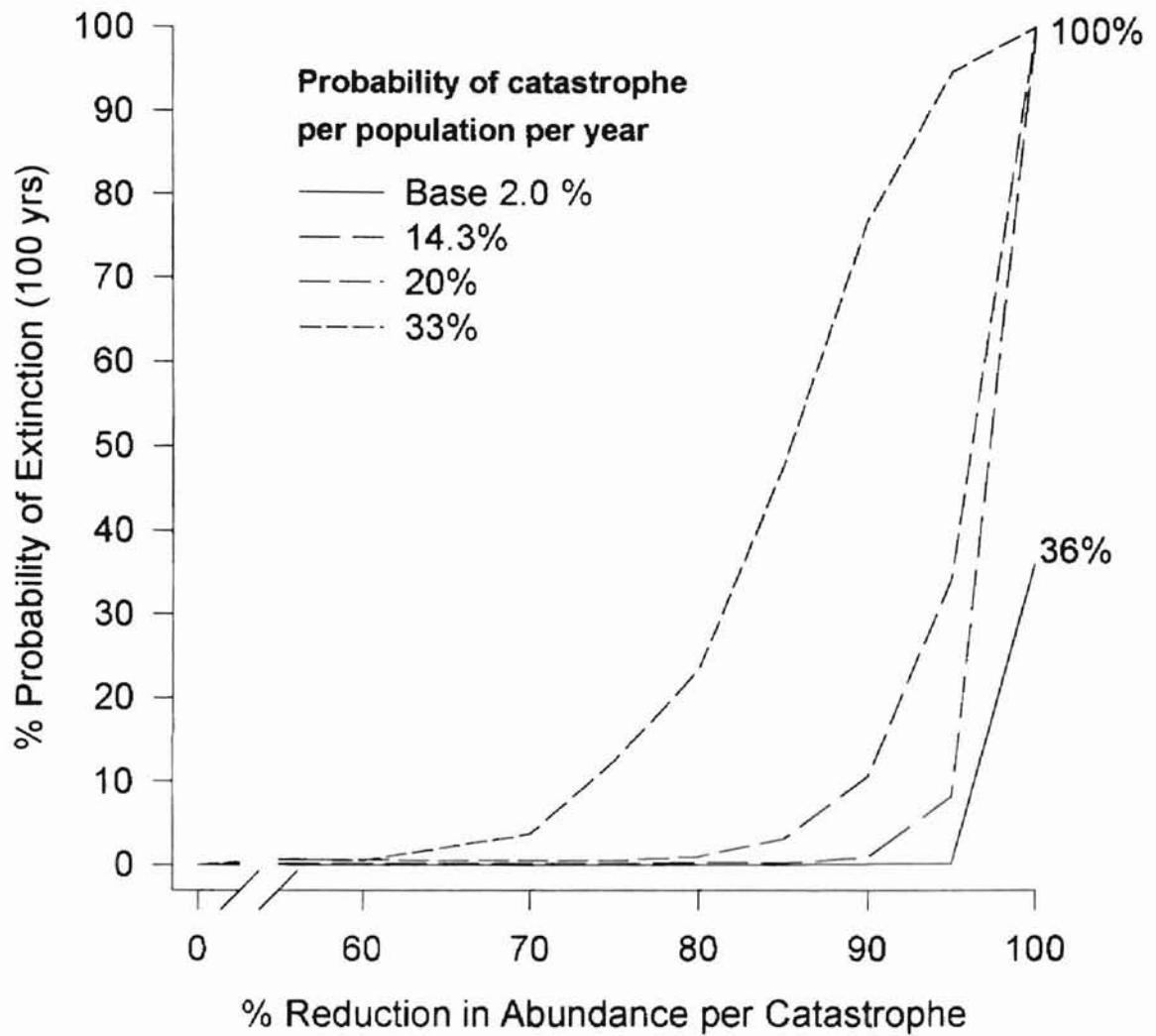Locality number correspond with those given in Table 2.

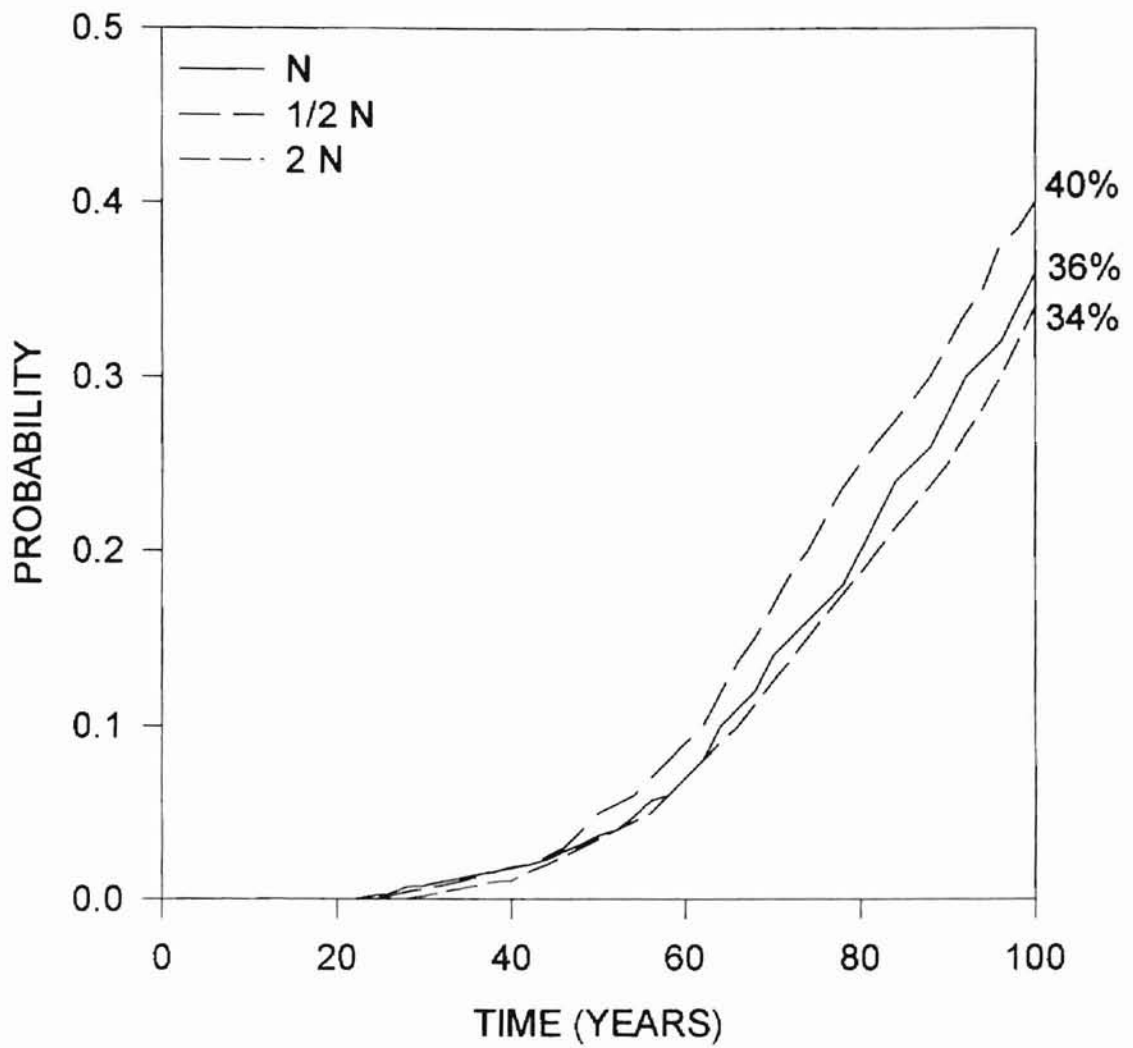Figure 2. Effect of catastrophe probability and severity on probability of extinction of Gila trout.

Figure 3. Effect of varying population size estimates (N) on probability of extinction of Gila trout.
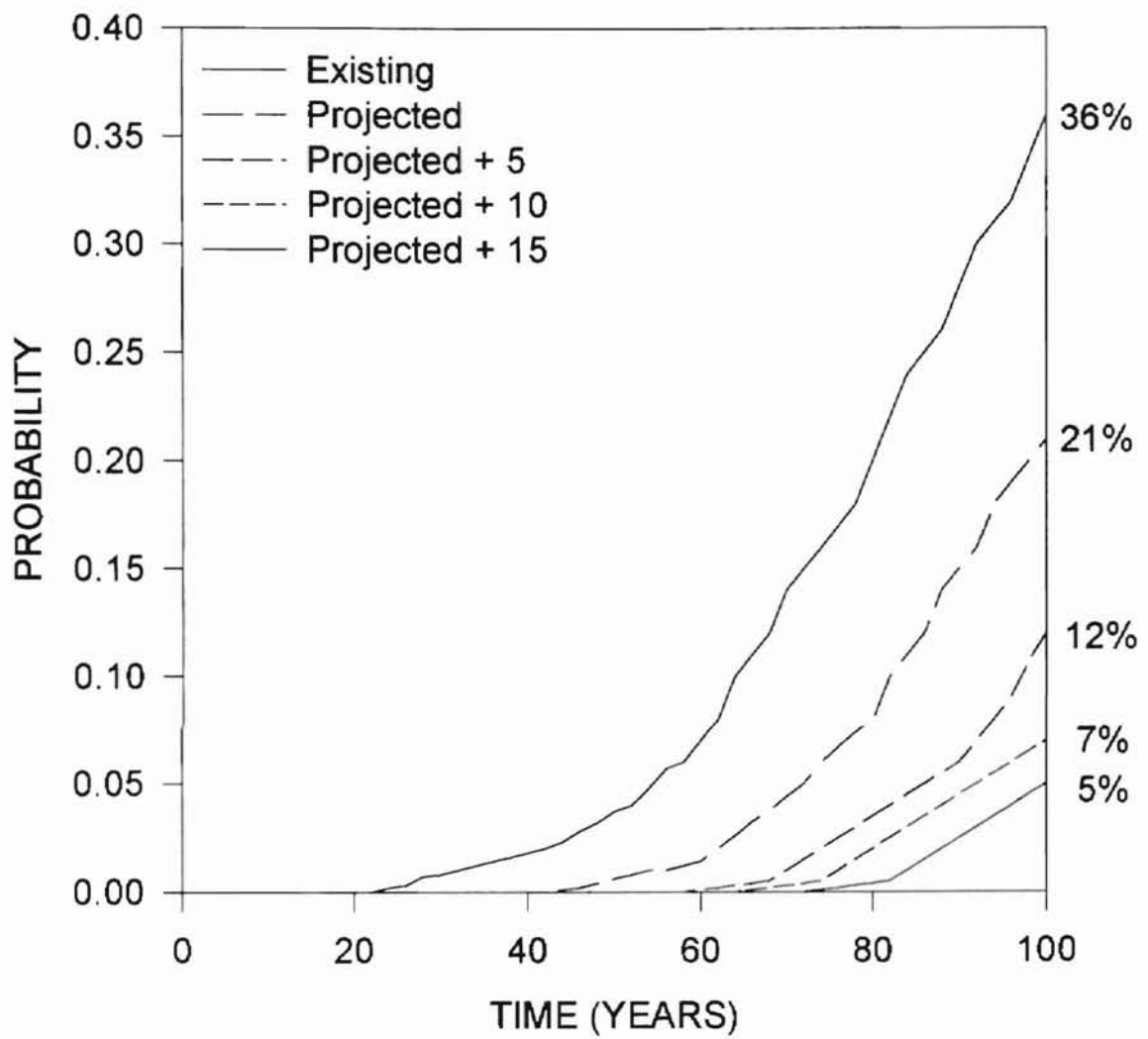
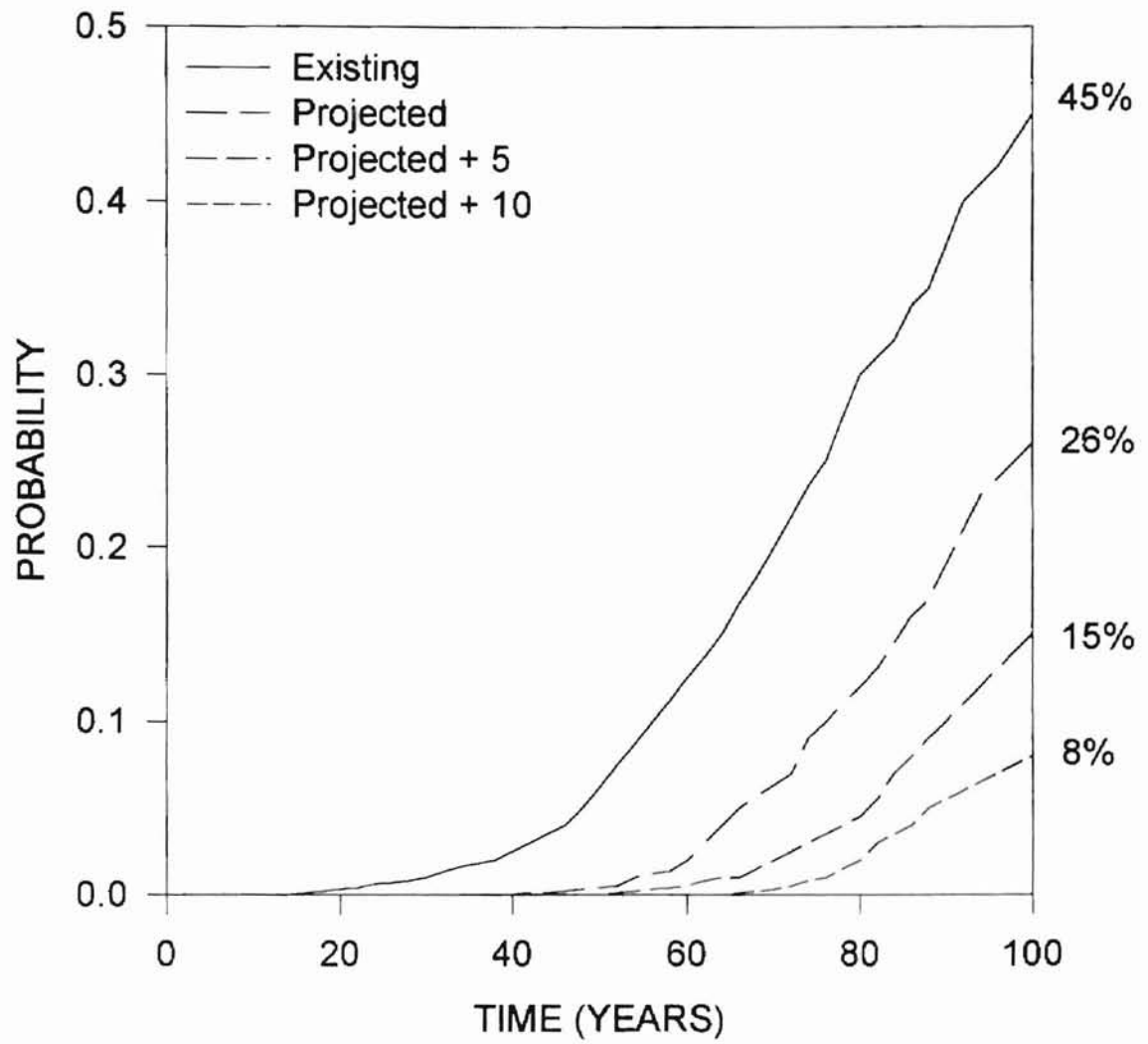Figure 4. Effect of increasing number of populations on probability of extinction of Gila trout.

Figure 5. Effect of number of populations on probability of extinction of the Gila River lineage of Gila trout.
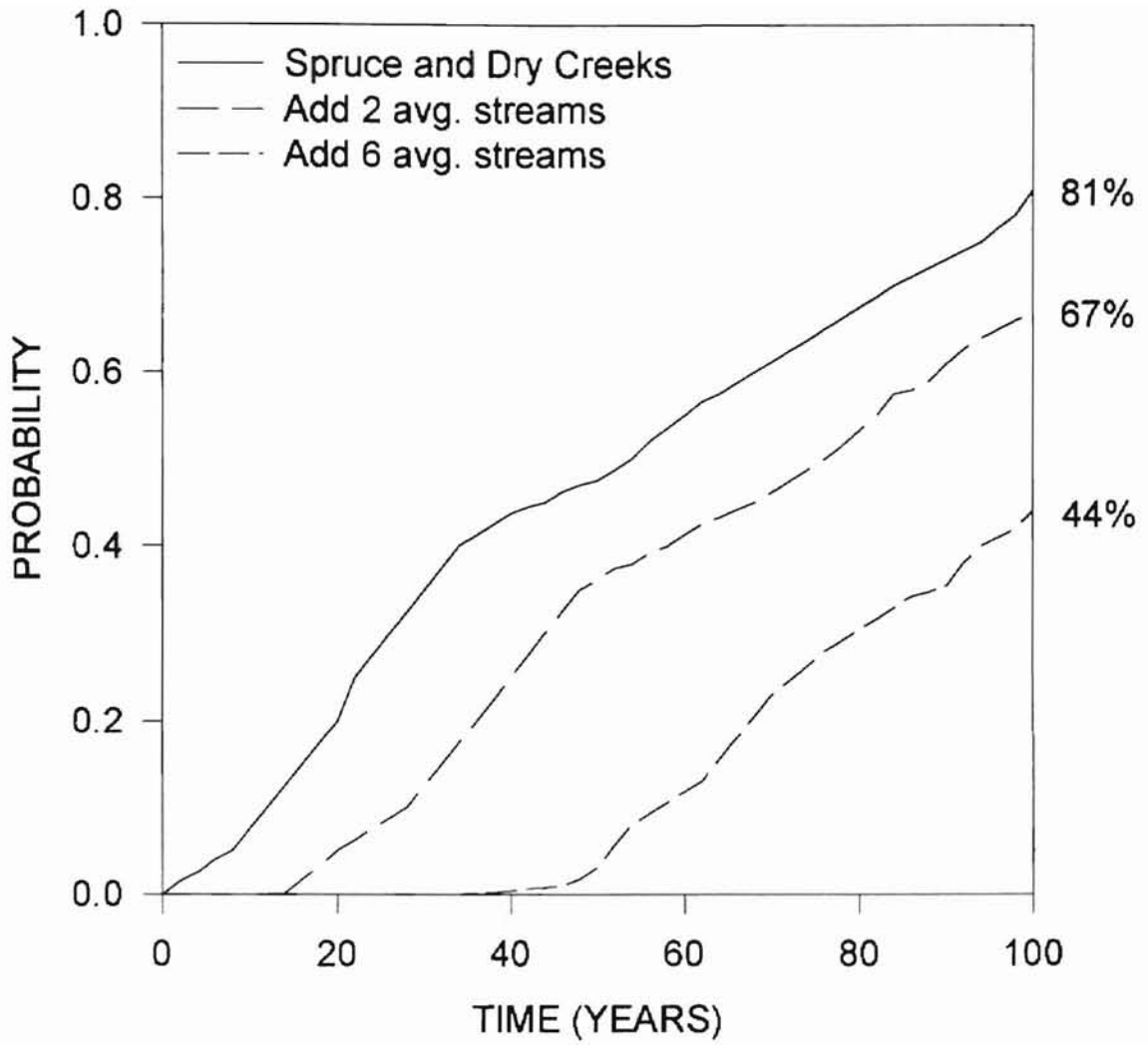
Figure 6. Effect of number of populations on probability of extinction of the San Francisco River lineage of Gila trout.
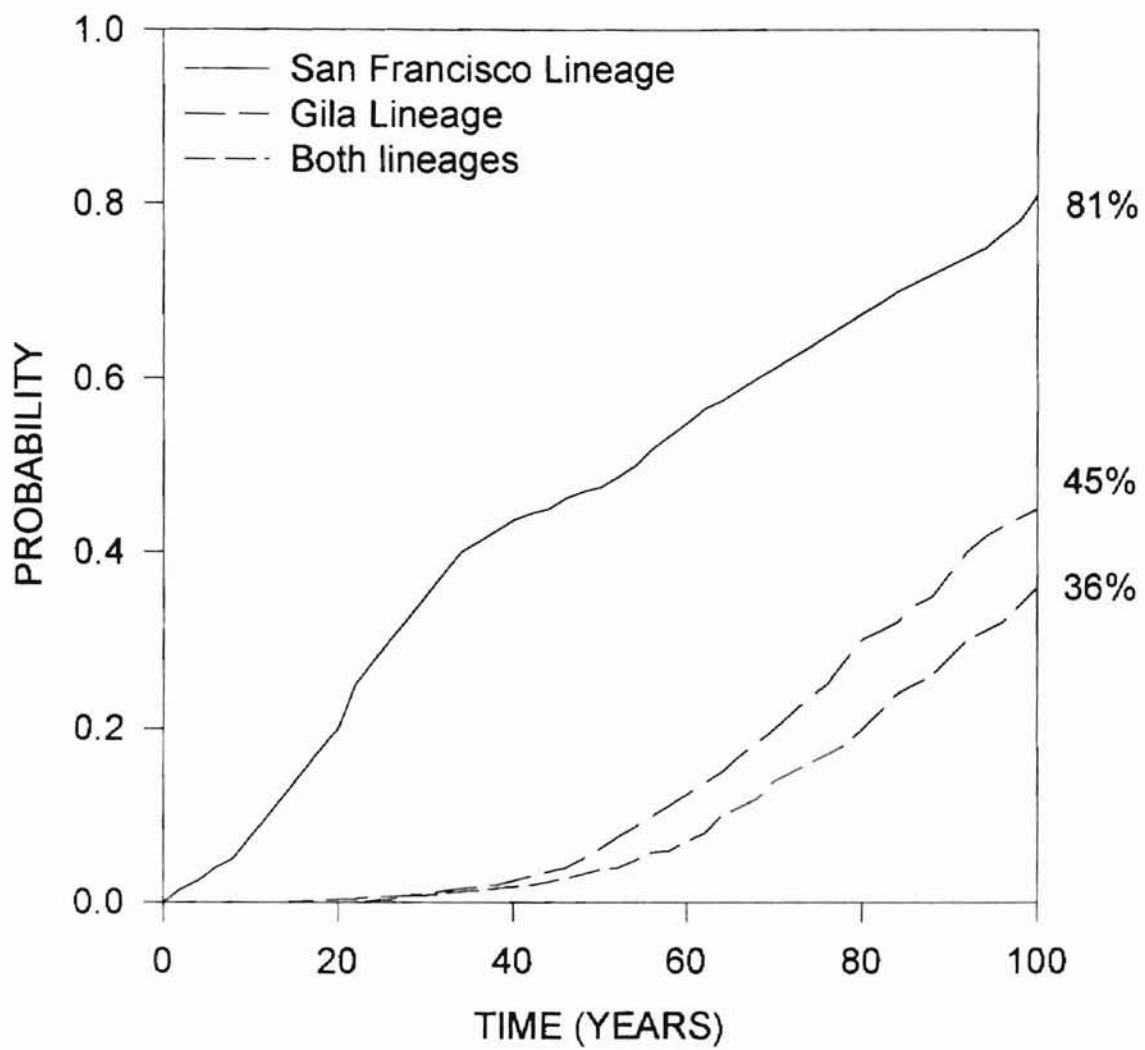
Figure 7. Comparison of extinction probabilities for the San Francisco River and Gila River lineage of Gila trout with that of the species.

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