CATASTROPHIC WILDFIRE AND NUMBER OF POPULATIONS AS FACTORS INFLUENCING RISK OF EXTINCTION FOR GILA TROUT (*ONCORHYNCHUS GILAE*)

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ABSTRACT.—We used the computer program RAMAS to explore the sensitivity of an extinction-risk model for the Gila trout (*Oncorhynchus gilae*) to management of wildfires and number of populations of the species. The Gila trout is an endangered salmonid presently restricted to very few headwaters of the Gila and San Francisco river tributaries in southwestern New Mexico. Life history data for 10 extant populations were used to examine sensitivity of the species' viability to changes in a variety of factors including population size, fecundity, life stage structure, number of populations, severity and probability of forest fires, and a regulated fishery. The probability and severity of forest fires and number of populations had the greatest effect on viability. Results indicate that successful conservation of Gila trout requires establishment of additional populations and reduction of the severity of forest fires through a program incorporating more frequent, but less severe, fires.

Key words: *Gila trout,* Oncorhynchus gilae, *endangered species, southwestern New Mexico, management.*

In this paper we use an extinction-risk model to explore management strategies for a federally listed endangered fish, the Gila trout (Salmonidae: *Oncorhynchus gilae*). This species is endemic to 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 central Arizona (Miller 1950, Behnke 1992). The Verde River population has been extirpated, and the species has declined by >95% in the remainder of its range as a result of overexploitation, interactions with stocked, nonnative trouts, loss of habitat, and habitat degradation (Sublette et al. 1990, Dowling and Childs 1992, Propst et al. 1992, Propst 1994). Currently, Gila trout are restricted to headwater reaches of a few small 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 when a captive stock was established at the Jenks Cabin Hatchery by the New Mexico Department of Game and Fish (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 policy of not stocking nonnative salmonids into areas occupied by Gila trout (Propst et al. 1992). Conservation efforts for the species also included creation of more pool habitats by using log structures installed by the Civilian Conservation Corps during the 1930s and repatriation of populations to several streams (Rinne 1982, Propst et al. 1992). In the 1970s five relict populations were replicated into reclaimed stream reaches treated with fish toxicant to remove nonnative salmonids. The reclaimed reaches were insulated from upstream movement by nonnative trout species by natural and artificial barriers (Propst and Stefferud 1997). Each of the 5 relict populations known at the time was believed genetically distinct (David 1976, Loudenslager et al. 1986), and each was stocked into separate reclaimed stream reaches. A 6th relict population was discovered in 1992 in Whiskey Creek, a small tributary of the West Fork of the Gila River.

We used population viability analysis (PVA) to evaluate population sensitivity to changes in

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variables that affect risk of extinction. Results from such analyses can suggest hypotheses for conservation management (Reed et al. 1998). To gain such insight, we developed a viability model for the species and one for each of its 2 major lineages, one comprising populations in the Gila River drainage and the other comprising populations in the San Francisco River drainage (Riddle et al. 1998, R. Leary and F. Allendorf, University of Montana, personal communication). We were specifically interested in modeling sensitivity of extinction risk to population size, number of populations, and effects of forest fires. We also assessed sensitivity of the base model to errors in estimates of fecundity and life stage structure.

We chose to focus on environmental stochasticity as the primary controlling factor in the viability models of Gila trout, an approach that avoids 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 tools to guide management options than they are as predictors of the fate of a species (Akcakaya et al. 1995). Gila trout is the only trout species listed as endangered under the Endangered Species Act of the United States and it is listed as threatened by the State of New Mexico (Behnke 1992). Our objective was to use PVA to evaluate management options that might contribute toward conservation of the species.

STUDY AREA AND HISTORY OF GILA TROUT CONSERVATION

Gila trout are now restricted to streams in narrow, steep-gradient canyons and small, moderate-gradient valleys at elevations of 1650–2820 m in the Black and Mogollon mountain ranges of southwestern New Mexico (Fig. 1; Propst and Stefferud 1997). Canyon reaches have swift-running waters with numerous cascades and plunge pools. Valley reaches have meandering channels, cobble riffles, and fewer pools, most of which are formed around log-debris piles and boulders. Base summer flows range from <0.05 m^3 s^{-1} to 0.65 m^3 s^{-1} (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).

The relict and replicated populations of Gila trout are located primarily in federally designated wilderness areas, where most human activities are minimized or controlled. Consequently, wildfire and interactions with introduced trout seem to be the most important factors affecting survival of existing Gila trout populations (Propst and Stefferud 1997). Originally, wildfires were primarily lightning-caused understory fires occurring in spring and early summer and ceasing with the rainy (monsoon) season in July–August (Rinne 1996). Within the range of Gila trout, historic wildfires consisted primarily of cool-burning understory fires with return intervals of 3–7 years in ponderosa pine forests and 75–100 years in spruce forests at higher elevations (Swetnam and Dieterich 1985). 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 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 and diminished herbaceous cover caused by grazing reduced the frequency of wildfires. Lack of periodic fires resulted in more woody debris on the forest floor, increased sapling densities, and establishment of brush. These changes have increased the potential for catastrophic crown fires (Rieman and Clayton 1997).

By the early 1900s, populations of Gila trout were restricted to the upper reaches of a few headwater streams primarily because of habitat modifications, overfishing, and introductions of nonnative trouts (Miller 1950, Propst et al. 1992). In these small, isolated systems, refugia from ash flow are limited and opportunities for recolonization often are nonexistent. Consequently, in the past decade 6 populations of Gila trout have been extirpated by extreme fire events followed by intense summer
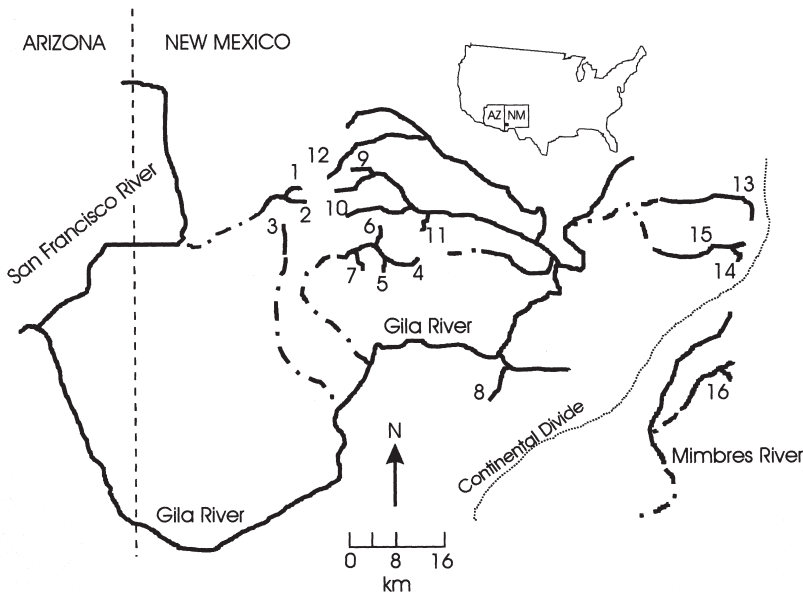
Fig. 1. Map of upper Gila River drainage showing populations used in PVA. Names of numbered streams are given in Table 2.

(July–August) rains that washed ash and debris into the stream (Table 1). The Divide fire in 1989 resulted in extirpation of the Main Diamond Creek population (Propst et al. 1992). The Bonner fire in 1995 extirpated 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 (JEB, DKB, DLP personal observation).

In addition to catastrophic loss following fires, Gila trout populations in many streams within the historic range of the species have been eliminated through hybridization with nonnative rainbow trout (*Oncorhynchus mykiss;* Loudenslager et al. 1986, Riddle et al. 1998, R. Leary and F. Allendorf personal communication). We did not model effects of hybridization and genetic introgression because management options associated with such factors (e.g., treat trout populations as "pure" because of low introgression, or renovate the stream and restock because of higher levels) are influenced by level of genetic introgression, and the degrees appropriate for different options are debatable and somewhat arbitrary (Campton 1987, Allendorf and Leary 1988, Dowling and Childs 1992).

METHODS

Sampling and Population Estimates

Ten Gila trout populations considered free of genetic contamination by nonnative congeners in 1996 (Table 2, Fig. 1) were used to develop a baseline PVA model. Subsequently, 3 of these were found to be genetically introgressed by rainbow trout (R. Leary and F. Allendorf personal communication). One of these was subsequently eliminated by chemical renovation and the stream was restocked with pure Gila trout in 1997. The other 2 represent relict populations, 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, partly because each may harbor locally adaptive genetic material. For this study we retained the original 10 populations included in the PVA.

Life history data were primarily taken or estimated from the literature (Nankervis 1988, Propst et al. 1992, Propst and Stefferud 1997), but population size estimates (*N*) for 6 streams were based on field data gathered during May through September 1996 and 1997. These 6 streams were Iron, McKnight, McKenna, and

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 U.S. Forest Service records and the Gila Trout Recovery Team of the U.S. Fish and Wildlife Service.

Drainage
    Relict 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

Mogollon creeks in the Gila drainage and Spruce and Dry creeks in the San Francisco drainage. For each stream we used a battery-powered, backpack electroshocker (24 V, DC) to sample one to three 200-m sites, with number of sites dependent on stream length. Prior to sampling we blocked each site at the upper and lower ends with fine-mesh nets. Depletion sampling was conducted by making 3–4 passes upstream through each site, capturing stunned fish with dip nets. To minimize injury and ensure equal capture effort between passes, we made no effort to "hunt" individuals. All Gila trout captured were weighed to nearest 0.1 g and measured to nearest millimeter 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 site 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). Estimates of population sizes are somewhat questionable because they assume that observed local densities can be extrapolated to the entire reach of stream occupied by Gila trout; however, results reported below indicate that the model is robust to this source of error.

Depletion-shocking efforts consistently captured about 60% ($n = 20$, $\bar{x} = 0.57$, $s_{\bar{x}} = 0.05$) of the population estimate in the 1st pass at each sample site. This percentage was used in estimating population size for 4 Gila trout streams (Main Diamond, Sheep Corral, Whiskey, and White creeks) for which only single-pass electrofishing data were available (Propst and Stefferud 1997).

Stage-specific Structure, Survivorship, and Fecundity

We estimated stage-specific structure (proportionate abundance of different life stages) from published length-frequency data for each Gila trout population (Propst and Stefferud 1997). Life stages were defined as follows (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).

Fecundity was estimated from the overall mean count of ova (98.6) from 25 field-stripped females from Main Diamond and McKnight creeks (Nankervis 1988, DLP unpublished data). Because RAMAS models each individual as being capable of asexual reproduction, we divided this mean value by 2 to arrive at individual fecundity (Table 3). Dividing the mean

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 asterisks are those used in the base model. Those with an asterisk are streams that either are presently devoid of Gila trout but targeted for restocking, or were restocked subsequent to the viability analysis; these 6 were used in the analysis 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 nonnative rainbow trout population.

TABLE 3. Life history variables and mean values (± *s*) used in the base model of Gila trout viability.

Variable	Mean
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.

fecundity of females by 2 assumed a 1:1 sex ratio and successful reproduction by all adult females every year. Nankervis (1988) found that a small proportion (13%) of large subadult females was reproductive, with a minimum size at reproduction of 130 mm. We estimated "subadult" fecundity by multiplying 0.13 by the mean proportion of our 130–150 mm individuals (0.47) and then multiplying this constant by 1/2 of the mean ova count (30.8) for subadult females.

Population Viability Analysis

We used the computer package RAMAS/ GIS (Akcakaya 1994) to model population viability. This package appears to be the best of those available for PVAs on organisms such as Gila trout, which has relatively large population sizes and high rates of reproduction compared with most other species for which PVAs have been done. The RAMAS algorithm uses a

Monte Carlo simulation of age- or life stage–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 et al. 1999), striped bass (*Morone saxatilis;* Ginzberg et al. 1990), and bluegill sunfish (*Lepomis macrochirus;* Ferson et al. 1991).

Extinction risk for Gila trout in the PVA models was expressed as the percentage of 1000 replicate simulations in which extinction of the species occurred within 100 years. In the base model we 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 6 populations were eliminated by forest fires and resultant habitat degradation (U.S. Forest Service unpublished data; JEB, DKB, DLP unpublished data). We arrived at probability of catastrophe for the base model (2% per population per 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 estimates (Table 3) were used to develop a base model for viability of the Gila

trout. We 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, we increased the fire-flood return interval from the base model of once every 27 years (2% per population per year) to once every 7 (14.3%), 5 (20%), and 3 years (33%). Those rates bracket the range of the pre-1900 fire-return interval for the ponderosa pine forest habitat that predominates in watersheds supporting 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 the 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 assess effect of fecundity upon viability, fecundity estimates were doubled in one model and halved in another. To assess sensitivity to life stage structure, we modeled the mean plus or minus 1 standard deviation for the proportionate abundance of each life stage separately; for each of these models, proportional abundances of the other 2 life stages were adjusted by addition or subtraction, with the amount of adjustment depending on the relative contribution to stage structure in the base model.

Sensitivity to number of populations was examined by considering 4 models in which populations were added to the base model of 10 streams. First, we added 6 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), which were calculated using the previously defined method for expanding single-pass catches. For the other 3 models we added 5, 10, or 15 populations, each having a population size equal to the average of the original 10 streams in the base model.

To assess effects of catch-and-release, artificial-lure, or fly-only angling on local populations, we examined viability of the McKenna and McKnight Creek populations with an annual catastrophe that reduced abundances of life stages 2 (subadult) and 3 (adult) by 5%, 10%, and 15%, respectively, in 3 separate models for each population. These reductions seem reasonable because studies indicate that 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, Schill 1996, Schisler and Bergersen 1996).

RESULTS

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

The base model was relatively insensitive to population size. Doubling and halving population sizes had no significant effect on extinction rate (Table 4). Correspondingly, simulating a catch-and-release fishery causing annual mortality of 5–15% of subadults and adults had no significant effect on viability of either the McKenna or McKnight Creek populations.

Viability of the species was insensitive to changes within 1 standard deviation of the mean in proportional abundances of the 3 life stages. However, the model was sensitive to large changes in fecundity estimates (F). Doubling and halving fecundity produced significant ($P < 0.001$) differences from the base model in probability of extinction (1/2F, 47%; F, 36%; 2F, 31%).

PVA was highly sensitive to number of populations. The model incorporating the planned restocking of 6 additional streams with Gila trout indicated a reduction of extinction risk from 36% to 21%. Adding another 5, 10, and 15 "average" populations lowered the modeled risk to 12%, 7%, and 5%, respectively (Table 4). Probabilities of extinction from each of these models were significantly different from those of the others.

Comparing modeled 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 each
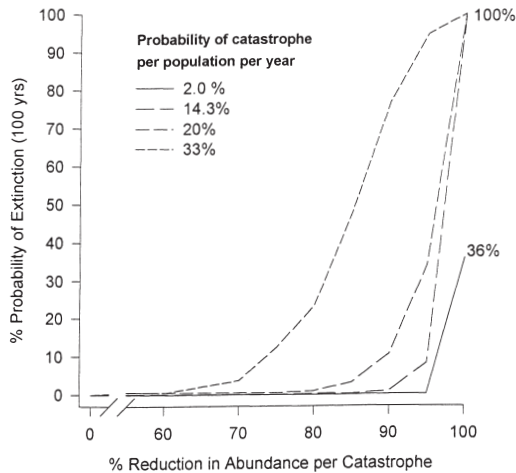
Fig. 2. Effect of catastrophe probability and severity on probability of extinction of Gila trout.

lineage has a significantly higher probability of extinction than does the species as a whole (Table 4). The model incorporating the planned restocking of 6 streams (all in the Gila River drainage) gave a significantly lower risk of extinction for the Gila River lineage (26%; Table 4). Adding those 6 populations plus 5 or 10 average populations also resulted in significantly lower risks (15% and 8%, Table 4). Adding 2 average stream populations to the San Francisco River lineage significantly decreased extinction risk from 81% to 67% (Table 4). Adding 6 average populations significantly reduced the chance of extinction from 67% to 44% (Table 4).

DISCUSSION

The probability of extinction of Gila trout within 100 years (36%), as computed under 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). The weakest aspects of our modeling effort probably are the estimates of population sizes, which assume uniform densities for each stream, and the estimates of fecundity, which had large variance (Table 3) and assumed no

local population variation. However, the base model was robust to population size estimates, and the questionable fecundity estimates do not invalidate use of the base model for insight into management strategies. Even with more precise models, recommendations from sensitivity analyses generally should be treated as hypotheses to be empirically evaluated before implementation by management agencies (Reed et al. 1998).

If Gila trout are left unmanaged, as assumed by our base model, 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 nonnative trout species (hybridization, competition, and predation). Such interactions are important in the decline and current status of Gila trout (Miller 1950, Propst and Stefferud 1997). However, they were not included in the PVA because they allow few management options beyond stream renovation and restocking or strategies to prevent introductions of nonnative trouts.

Altering the historic fire regime in ponderosa pine forests of southwestern New Mexico from cool-burning understory 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, our models suggest that viability of Gila trout is especially sensitive to effects of forest fires. Ignoring other factors of catastrophic loss, such as effects of nonnative trouts and drought, the models suggest that risk of extinction would be extremely low if effects (percentage of population reduction) of potentially catastrophic fires were reduced by a proactive fire management program. In our analyses even small decreases in catastrophe led to substantial reductions in extinction risk (Fig. 2).

Much of the area occupied by Gila trout 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 fuel moisture

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. Asterisks signify significant difference from all other models in the subset (*P* < 0.01).

	Probability of extinction (%)		
Populations	1/2 N	N	2N
Gila and San Francisco lineages			
Existing	40.0	36.0	34.0
Projected[a]		21.0 *	
Projected + 5[b]		12.0 *	
Projected + 10[b]		7.0 *	
Projected + 15[b]		5.0 *	
Gila River lineage			
Existing	48.0	45.0	44.0
Projected[a]		26.0 *	
Projected + 5[b]		15.0 *	
Projected + 10[b]		8.0 *	
San Francisco lineage			
Existing	83.0	81.0	80.0
Existing + 2[c]		67.0 *	
Existing + 6[c]		44.0 *	

[a]Ten populations in the base model (= existing) plus 6 additional populations in streams designated for restocking with Gila trout.
[b]Projected populations plus 5, 10, or 15 populations with the average size of all existing populations of Gila trout.
[c]Existing populations in Spruce and Dry creeks plus 2 or 6 populations with the average size of all existing populations of Gila trout.

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. Reduction of fuel loads by more frequent low-intensity fires should contribute to a more natural forest structure, thereby reducing the frequency of catastrophic fires (Pyne et al. 1996).

Our results indicate that prescribed fires with a return interval as short as 3 years would not increase extinction risk for Gila trout, even if as much as 50–60% of the local population is lost with each event. Such losses should, however, 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. As emphasized by Rieman and Clayton (1997), prescribed fires can lead to inadvertent population losses because of the large fuel loads that have developed during the past 75+ years of fire suppression in the region. Initially, prescribed fires could be viewed as experimental and restricted to watersheds projected to be renovated for Gila trout. Extension to Gila trout streams might be implemented once the methodology has been perfected, and perhaps only after establishment of more populations of the species.

The model of Gila trout viability was insensitive to size of individual populations, but it 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 refugia (i.e., tributaries) not directly affected by the catastrophe. 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 refuge 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 such a reduction can have a significant effect on risk of extinction (Fig. 2). Furthermore, a marked increase in amount of habitat (length of stream) occupied would reestablish natural connectivity among a number of now-isolated local populations of Gila trout. Increased connectivity would heighten the rate of recolonization following catastrophic losses in local areas, thereby improving viability of the species. In New Mexico alone existing populations occupy <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.

Like many other conservation efforts for endangered and threatened species, recovery of Gila trout is a complicated and politically controversial issue. Some public opposition to Gila trout recovery efforts 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 might 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. Additionally, the 2 populations of the lineage are geographically very close (Spruce Creek is a Dry Creek tributary), and the past history of Gila trout demonstrates a high probability for eliminating both populations by a single catastrophic wildfire.

Ongoing efforts to conserve Gila trout emphasize 3 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). Our results suggest that a 4th approach is central to the success of this effort, namely, an effort to reduce catastrophic effects of wildfires. Besides reducing the expense and effort involved in restocking 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). Our models of Gila trout viability were highly sensitive to the effect of forest fires and indicate that 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 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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