Status of Native and Introduced Species of Aquatic Herpetofauna at San Bernardino National Wildlife Refuge

Philip C. Rosen

Department of Ecology and Evolutionary Biology
University of Arizona, Tucson, AZ 85721

and

Cecil R. Schwalbe

U.S.G.S., Biological Resources Division, Cooperative Park Studies Unit
University of Arizona, Tucson, AZ 85721

FINAL REPORT

to

Arizona Game & Fish Department
Heritage Program
2221 West Greenway Road
Phoenix, AZ 85034

U.S. Fish and Wildlife Service
San Bernardino National Wildlife Refuge
7628 N. Highway 191, P.O. Box 3509
Douglas, AZ 85607

on the challenge cost share project

BULLFROG IMPACTS ON NATIVE WETLAND HERPETOFAUNA

Arizona Game and Fish Department Heritage Grant
IIPAM I96056

August 28, 1998

ARIZONA GAME & FISH
HERITAGE FUND
ARIZONA LOTTERY
LOTTERY DOLLARS WORKING FOR WILDLIFE

DISCLAIMER

The findings, opinions, and recommendations in this report are those of the investigators who have received partial or full funding from the Arizona Game and Fish Department Heritage Fund. The findings, opinions, and recommendations do not necessarily reflect those of the Arizona Game and Fish Commission or the Department, or necessarily represent official Department policy or management practice. For further information, please contact the Arizona Game and Fish Department.

STATUS OF NATIVE AND INTRODUCED HERPETOFAUNA AT SAN BERNARDINO NATIONAL WILDLIFE REFUGE 1998

FINAL REPORT

TABLE OF CONTENTS

EXECUTIVE SUMMARY

This report includes new results on population size and ecology of aquatic reptiles and amphibians affected by the introduced bullfrog. New ecological data and population estimates are provided for the Mexican garter snake, Sonoran mud turtle, and checkered garter snake at San Bernardino National Wildlife Refuge (SBNWR), Arizona, through 1997. Quantitative data on the life history, ecology, and natural history of the checkered garter snake are presented here for the first time.

A complete account of leopard frog conservation and monitoring results through 1997 was presented in Rosen and Schwalbe (1997). Here we report the finding of pathogenic chytridiomycete fungal infection in dead and moribund Chiricahua leopard frogs from the refuge. Although poorly understood, and not necessarily the ultimate cause for sickness or unexplained population declines in Arizona frogs, this offers a promising lead in our attempt to understand unexplained mortality in Southwestern ranid frogs. This fungus has been recently discovered and reported in captive anurans and among frog populations that inexplicably collapsed in certain far-flung and remote parts of the globe.

This report also contains further analysis of the effectiveness of bullfrog removals, as well as new data on removals of adults, tadpoles, and egg masses. These data demonstrate a decrease in size of bullfrogs at SBNWR, and indicate that standing crop is markedly reduced by the removals, while biomass productivity is relatively constant. Benefits of the bullfrog control are described for the Mexican garter snake, and are suggested for the Sonoran mud turtle.

Removal of egg masses and small juvenile bullfrogs is being testing during 1997 and 1998, and both are very useful additions to the existing bullfrog removal protocol. Gigging is also effective, but the analysis presented here indicates that trapping more cost-efficient, and has the added benefit of nearly unlimited potential for intensification.

A key to continued usefulness of the bullfrog removal program is implementation of an experimental design with different levels of removal in different ponds on site, with subsequent monitoring of bullfrog size and numbers, and populations of other aquatic herpetofauna. Preliminary data in 1998 suggest that a strong effect of the removals is demonstrable.

The refuge population of Mexican garter snakes is estimated at 30 - 60 individuals, and a steady decline is documented during the 1985-1997 study; and evidence is presented to support the hypothesis that a decline has been under way for much longer than that--possibly for a full century. A preliminary estimate for the checkered garter snake is nearly 700 individuals on the refuge, or more, and an analysis of capture rate per unit effort suggest that this population has remained remarkably stable during 1985-1997. The best estimate of population size for the Sonoran mud turtle is 72, although we suspect that nearly 100 may be present; however, the sex ratio is strongly male-biased. The checkered garter snake population size may be considered normal, while the other two species clearly have abnormally low population sizes at SBNWR.

For garter snakes, habitat utilization analysis demonstrates that a key factor in permitting checkered garter snakes, but not Mexican garter snakes, to coexist successfully with bullfrogs is strongly developed juvenile terrestriality in checkered garter snakes from birth

to about a year of age. Numerous other differences and similarities in life history and ecology between these species are presented and discussed.

Additional information summarizing this project is available in the published literature (Schwalbe and Rosen, 1988; Rosen and Schwalbe, 1995, 1998) and in previous reports (Rosen and Schwalbe, 1988, 1996a,b, 1997). Some of that is repeated here:

The focus of the work has been the effect of introduced bullfrogs (Rana catesbeiana) on native Mexican garter snakes (Thamnophis eques) and on Chiricahua and lowland leopard frogs (R. chiricahuensis and R. yavapaiensis). Leopard frogs disappeared from the refuge early in this project (1988), persisting in outlying ranch ponds without exotic species. They have been managed with success at and near SBNWR since 1994.

We report a sudden decline in four of the seven managed leopard frog populations, including the possible disappearance of a key population. These declines may share a common cause: all are within 2 miles of the international border, and sick, moribund, or dead frogs were observed in each case before the decline was confirmed. It seems possible that air pollution, a microbial epidemic, or nutritional shortfall could be involved. We are extremely concerned at this time, although frogs apparently remain at 5 or 6 of the 7 localities.

Mexican garter snakes have been declining since before 1985, when we first visited the refuge, but are approaching local extirpation in the late 1990's, as previously predicted. We are collaborating with the refuge to re-establish a viable population in a fenced, native fish and frog refugium at SBNWR, and to this end have established a trial breeding colony in captivity in Tucson. The first breeding attempt is to be fall 1997 and spring 1998. Without this effort, we would anticipate possible extirpation of the Mexican garter snake from Cochise County before active management options are underway.

Bullfrog removal efforts have yielded mixed results. Declines in harvested totals have been seen during 1995 - 1997, but they correlate with relatively small changes in trapping intensity. Our initial efforts in the 1980's were not effective. Intensified trapping in the 1990's produced a decline in bullfrog average size (and presumably age), and a decline in the number of very large bullfrogs. Removals may be responsible for recent recruitment of juvenile Sonoran mud turtles. Temporary successes in bullfrog control in at least one pond system (Twin Pond) produced a small wave of recruitment of Mexican garter snakes, suggesting that intensive local control efforts produce the best results.

More promising for medium-term species preservation is the refurbishment of various wetlands at SBNWR that was initiated in 1998 by U.S. Fish and Wildlife Service (USFWS). The main purpose is to eliminate an exotic parasite in the natives fishes on the refuge, the Asian tapeworm Bothriocephalus achelognathi. We are assisting Refuge Manager Kevin Cobble in the planning and design of the refurbished wetlands. This will allow us to capture bullfrogs at temporary fences around ponds as they are dried, and prevent bullfrog colonization of new wetlands using permanent fences. Fenced wetlands with native fishes, leopard frogs, and garter snakes are the best current option for sustaining large populations of native aquatic amphibians and reptiles. These may serve as sources for repopulation of surrounding ranch and forest areas with Chiricahua leopard frogs.

Captive propagation of garter snakes is not feasible on a large scale. They are very active snakes requiring frequent feeding with frogs, toads, and fishes. Even our small laboratory colony of 12 - 16 snakes in the genus Thamnophis requires considerable attention. A fenced semi-natural wetland at SBNWR would support one or two orders of magnitude more individuals. The fenced populations would be the most reasonable source for re-introduction of Mexican garter snakes in restored habitats.

Leopard frogs at the nearby Magoffin Ranch, at Rosewood Tank and Belency Tank, have been *a focus of conservation work by private landowners, federal and state wildlife agencies, and university scientists. Frogs from Rosewood Tank have been established and have bred successfully in a bullfrog exclosure at SBNWR, and at newly constructed Choate Tank on Magoffin Ranch. Progeny have been released at a newly created pond at Douglas High School and one at Magoffin Ranch headquarters, and in a screened-in ranarium (frog breeding facility) at SBNWR. With funding and assistance from Arizona Game and Fish Department (AGFD), Malpai Borderlands Group (MBG), and The Arizona Nature Conservancy (TANC)--not to mention the great efforts of the Magoffins--wells, pumps, water lines, sediment traps at main cattle ponds, and a small cement pond, have been installed or repaired on Magoffin Ranch.*

The work on Magoffin Ranch, SBNWR, and Douglas High School is the nucleus for a potentially more encompassing conservation project. It has led toward development of a plan for repatriation of frogs to a number of ranches in the Malpai Borderlands area. Under guidance from AGFD Nongame Branch and USFWS Ecological Services, we are surveying ponds in this area, and speaking with local landowners and land managers to clarify "on-the-ground" realities concerning a conservation plan for the Chiricahua leopard frog in this area. We anticipate that this will assist the development of the apparently imminent proposal to list the Chiricahua leopard frog as a federally threatened species. We also view this plan as a step toward a recovery program for the species based on public-private partnerships and on using our understanding of causes of frog declines to manage habitat for recovery. Development of this conservation plan is becoming an important focus for the ongoing work under this project.

INTRODUCTION AND OVERVIEW

This report presents results of ongoing garter snake and mud turtle monitoring, leopard frog conservation, and bullfrog removal efforts at San Bernardino National Wildlife Refuge through 1997. Project objectives are to (1) evaluate the feasibility of bullfrog removal as a means of control, (2) determine whether control of bullfrogs can assist population recovery of affected species of reptiles and amphibians, and (3) develop strategies for management of native wetland herpetofauna (see Schwalbe and Rosen, 1988; Rosen and Schwalbe, 1988, 1995, 1996a & b). In particular, the project aims to (4) assist recovery of Chiricahua leopard frog and Mexican garter snake populations in southern Arizona.

The first phase of the project (1985-9) primarily involved two or three trips a year of 3-4 days each to remove bullfrogs from SBNWR. Results suggested weak positive effects of bullfrog removals on garter snakes (below, and Schwalbe and Rosen 1988); however, bullfrog populations recovered from the removals within less than a year. The second project phase was initiated in 1992 after a two-year hiatus. Effort was intensified beginning May 1993. This involved manual removals of bullfrogs by approximately 14-18 persons from University of Arizona, Arizona Game & Fish Department, U.S. Fish & Wildlife Service, and others, plus extended trapping of bullfrogs during much of their active season. Trapping was conducted jointly by USFWS and University of Arizona personnel during June-November 1993, March-October 1994, and May-October of 1995, 1996, and 1997.

In 1994 we initiated a third phase of the project involving active conservation and management of Chiricahua leopard frog populations at and near the refuge. This was in collaboration with SBNWR, AGFD Nongame Branch, and the MBG including especially Magoffin Ranch. We assisted in preserving leopard frog populations at Rosewood Tank, 7 mi east of the refuge, and Belency Tank, along the border at 5 mi east of the refuge, both on the Magoffin Ranch. Rosen, Magoffin, and Cobble conducted regular and frequent visual surveys of the leopard frogs, and the Magoffins managed waters to forestall habitat drying. Rosen and other UA personnel additionally surveyed the surrounding region, and monitored Chiricahua leopard frogs in the next nearest known population site, Guadalupe Canyon, as well as at Leslie Canyon.

Principal investigators, refuge managers, and AGFD Nongame Branch personnel collaborated in a frog re-establishment project during 1994, in which a portion of the tadpoles and metamorphs from Rosewood Tank were removed before the tank dried, and relocated (total 188) to a pond-enclosure system created for them on-refuge at SBNWR (Rosen and Schwalbe, 1996a). The rest were maintained by the Magoffins in a pool dug in the dry floor of Rosewood and supplied by hauled water (about 1000 gallons/week). Rosewood has received scant runoff and has not filled completely since it dried in 1994. The refurbished tank bed has received very scant runoff in the locally extreme drought years since 1994. A strong population at Belency Tank, on the Mexican border, has been left undisturbed except for installation of a windmill-powered well, cleaning of the sediment trap with a bulldozer, and periodic monitoring.

This project has been expanded in cooperation with Douglas High School, University of Arizona, USFWS, and AGFD. In 1995, a small pond was constructed at the Choate Well site on Magoffin Ranch; and stunted tadpoles from Rosewood were established there in August-September 1995. In fall 1996, an outdoor, screened, frog room (a "ranarium") was completed at SBNWR and stocked with wild Chiricahua leopard frogs from Hay Hollow at Guadalupe Canyon Road. Also in fall 1996, eggs collected at Choate were hatched and reared by students at Douglas High School. Tadpoles reared Douglas High School were released in May 1997 in a new pool constructed by SBNWR personnel at Douglas High School (n = 27 tadpoles), in a new small pond at Magoffin Ranch headquarters (n = 41 tadpoles), and at the SBNWR ranarium (n = 6 tadpoles). In 1997-8, a portion of an additional egg mass from the refuge enclosure was reared at Douglas High School.

BULLFROG REMOVAL AND ABUNDANCE

Review of Removal Program. Ongoing removal methods (detailed in Schwalbe and Rosen, 1988; Rosen and Schwalbe, 1995) consist of hand capture, spearing with a frog gig (as in sport hunting), and trapping using turtle traps (hoop nets) set at the ends of seine drift fences.

Bullfrog Egg Mass, Tadpole, and Juvenile Removals. Bullfrog egg mass removals were conducted at SBNWR during 1994-1997. Based on available records, we removed the following: (May 20-29, 1994, Mesquite Pond, 26 removed, 6 hatched masses observed); (May-June, 1994, House Pond, ca. 6 egg masses removed); (May-June 1995, House Pond, ca. 10 egg masses removed; Mesquite Pond, 1 egg mass removed; Little Mesquite, 1 egg mass removed; Double PhD, 1 egg mass removed); (July 27, 1995, Little Mesquite, 1 egg mass removed); (August 3, 1995, House Pond, 2 egg masses removed); (May 25, 1996, Astin Spring, 1 mostly hatched egg mass removed); (May 27, 1996, House Pond, 11 egg masses removed; Double PhD., 1 egg mass removed; Twin Pond, no egg masses found); (June 1996, House Pond, ca. 6 egg masses removed); (July 1996, Leopard Frog Enclosure, 1 egg mass transferred to garbage can, eggs infertile); May 24-26, 1997, House Pond, 7 egg masses removed; Twin Pond, 1 egg mass removed); (June-July 1997, Twin Pond, ca. 5 egg masses removed; Oasis Pond, 2 or more egg masses removed). All personnel removing egg masses did not invariably record removals, although we believe that less than 10 additional (to those enumerated here) egg masses were removed. The annual totals for recorded egg mass removals were approximately as follows: 1994, 32; 1995, 16; 1996, 20; 1997, 14; total 1994-1997, 82+ egg masses removed.

Tadpole removals were attempted in 1995 and 1996 using submerged minnow traps and larger wire tadpole traps. Submerged minnow traps, set primarily for fish sampling, captured large numbers of tadpoles, but unfortunately also inadvertently drowned garter snakes, including at least one large adult female Mexican garter snake. On our recommendation, the refuge discontinued minnow trapping in the warm season. An experimental tadpole trap designed and constructed by Matt Magoffin captured 140 bullfrog tadpoles in Little Mesquite Pond, June 4 - August 14, 1995. We will re-use this trap in 1998 on a further trial basis; the effort of construction may not justify pursuing this method, since dipnetting for tadpoles (frequently carried out to obtain live food for captive garter snakes) yielded 1-4 large tadpoles per sweep in Twin Pond during 1996 and 1997.

Hand capture of juvenile bullfrogs at night from a canoe proved to be highly effective; it is easiest with two persons, and thus somewhat labor intensive, but is a most promising additional removal technique. Capture success is about 50-80%. Under proper conditions, it is also possible to capture approximately 30% of observed juveniles while wading. Two persons can easily remove 50-100 or more juveniles from certain areas, such as Twin Pond and

Double Phd., in a single hour of work.

Refuge Manger Kevin Cobble attempted periodic removals of egg masses in Twin and Oasis Ponds in 1997, and suggested that tadpole densities were reduced as a result. There appear to be fewer bullfrogs present in 1998 than previously at these sites, and we have intensified and systematized the egg mass removals in 1998 (as of 8-28-98, 27 egg masses have been removed in 1998, primarily from Twin Pond and Oasis Pond, where intensified trapping, collection of juveniles, and removal of all egg masses were attempted throughout the active season). Preliminary results thus far in 1998 suggest that the intensified protocol is promising, although it requires rigorous diligence to schedule. In particular, fresh egg masses collected at Twin Pond began hatching at 3 days post-deposition, and would be difficult or impossible to remove after 4 days total.

Bullfrog Removal Methods Efficiency. Trapping throughout the active season (April or May to September or October) was done 1993-7, with the longest trapping period and most extensive manual removal occurring in 1994 and 1995 (Table 1A-C). Table 1 is the best available index of removal effort, which varied from year to year due to fluctuations in weather, number of skilled frog collectors during the manual removal efforts, and number of trap-days.

Table 1C was constructed as a model of the effectiveness of gigging ("manual effort") versus trapping ("trap effort"). Each person-day of manual effort yielded about 4 frogs, although an employee dedicated solely to collecting the frogs (as opposed to the numerous other activities also carried out during the two major manual removal trips per year) could probably remove at 4-5X that number. Efficiency thus might be as high as 20 frogs removed/person day of effort, although this could not be maintained through a season of collecting, with steadily declining frog numbers.

Trapping yielded about 1 bullfrog per 10 trap-days; however, a single person can check 25 bullfrog traps twice per week in roughly one person-day of work. With 25 traps operating, at 4-day trap check intervals, one person would capture roughly 40 bullfrogs in about 1 day/week total effort (including setup and maintenance), or about twice the maximum efficiency of gigging.

Both methods of removing adults and subadults appear to be effective and efficient. Manual removal generally calls for 2 persons working together from a canoe, decreasing its efficiency. On the other hand, only about 3-4 hours of actual work would be required to accomplish the captures, leaving additional time for other activities. Both methods should be used, although trapping has advantages, including (1) higher efficiency, (2) almost infinite expandability, and (3) repeatability and consistency. However, trapping requires consistent presence on-site, with checks of the traps once to twice per week.

Trends in Total Bullfrog Removals Per Unit Effort. Effort is correlated with total numbers and mass of frogs removed from the refuge (Tables 2-10, 1C, Fig. 1), making it difficult to statistically test the significance of changes of the magnitude we have seen in the bullfrog population of the refuge.

Comparing years (Table 1C) suggests that removal of numbers and biomass of bullfrogs has remained a relatively constant function of removal effort during 1993-1997. Captures per unit effort index averaged 0.114 $\pm$ 0.0296 s.d. (range among years 0.084 -

0.156), with no clear trend evident. The tendency to capture mostly smaller adult and subadult-sized frogs (below) contributed to the high rates of capture/unit effort in 1996 and 1997.

Biomass removal per unit effort (Table 1C, Fig. 1) is probably the best index available to search for a trend toward persistent reduction of the bullfrog population at SBNWR. This index averaged 0.0214 $\pm$ 0.0030 s.d (with a narrow range of 0.018 - 0.025). Again, we have no evidence for a persistent decline in removals per unit effort.

Trends In Bullfrog Size on the Refuge. Tables 6 and 7 indicate a trend for consistent annual reduction of bullfrog size on the refuge during the 1992-1997 period. Table 8 shows that the number of adult-sized bullfrogs (i.e., excluding subadults) followed the trends shown in Table 1C; however, Tables 9 and 10 show that the number of small adults has increased, whereas large adults have decreased dramatically.

We cannot necessarily ascribe the progressive 1992-1997 decline of bullfrog size at SBNWR to our removal efforts, although that appears as the likely explanation. It is possible that unobserved die-offs of large adults occur, or that growth has been markedly and steadily reduced during the 1990's. However, neither of these possibilities are likely, as explained below, and it appears that our removals have markedly lowered adult survival and altered population structure.

Unobserved mortality of large adult bullfrogs is unlikely, since the observed mortality, during the cool season, has involved juvenile frogs (winter 1996-7 and 1997-8; primarily small juveniles) or tadpoles (winter ca. 1994-5, 1996-7, and 1997-8). Modest numbers of dead froglets have been observed (perhaps 20-25 at a time, Kevin Cobble, personal communications). Significant tadpole die-offs were observed in North Pond in 1994-5 (Kevin Cobble and Matt Magoffin, personal communications) and in Twin Pond in 1996-7 and 1997-8 (Rosen, personal observations), and at Twin, Double Phd., and Oasis Ponds during spring and summer 1998 (Rosen, David H. Hall, personal observations). The tadpole die-offs, though in some cases dramatic, still left the affected areas with conspicuous, dense tadpole populations. A few dead adults have been, found, but only one (seen in 1998, David Hall, personal communication) did not have apparently lethal gig wounds.

Steadily declining bullfrog growth during the 1990's also appears unlikely: there is no apparent reason for such a steady trend. The constant, or increasingly frequent, appearance of small, presumably young, subadult and adult bullfrogs suggests that rapid growth is occurring. Without control populations, being implemented in 1998, we cannot be certain of the cause for the observed size decline. Tentatively, we suggest that it is most likely that our removal efforts are responsible.

Conclusions Suggested by Trends in the Bullfrog Data. The findings described in the previous section are consistent with previous suggestions (Rosen and Schwalbe, 1997) that the existing protocol results in removal of a high proportion of the adult bullfrogs. The effects of the bullfrog removal protocol used through 1997 are (1) a decrease in mean age and size among the subadult plus adult portion of the population, (2) an increase in the number of small, and decrease in the number of large, bullfrogs removed, (3) markedly lowered standing crop, but (4) consistent level of biomass removal (and thus productivity) on an annualized basis. While these changes are probably inadequate for leopard frog recovery, less sensitive organisms such as Mexican garter snakes and Sonoran mud turtles would be more likely to benefit.

Protocol Modifications for 1998 and Future Work. The bullfrog removal protocol as employed through 1997 appears clearly to be insufficient for elimination of bullfrogs. It would not likely lead to population reductions sufficient for leopard frog restoration. Further, without any control populations, in which no removals can be contrasted to various removal intensities, it will be difficult to clearly define causation. The changes in protocol recommended by Rosen and Schwalbe (1997) are consistent with the problems identified by this most recent data analysis.

The use of drift fences to encircle new wetlands at SBNWR appears to hold strong promise for recovery of native frogs, based on leopard frog results discussed below and previously. We expect that Mexican garter snake populations may also benefit from such fencing operations. Completion of the new stream wetland near the site of North Pond is occurring as of this writing (August 25, 1998), with closure of the encircling fence now complete. During fall 1998, we anticipate drying of North Pond, treatment of native fishes to kill parasites, and introduction of fishes into the stream system. Wetland herpetofauna will be placed into the stream enclosure based on availability.

It bears re-stating here that there appear to have been some successes as a result of the bullfrog removal protocol: (1) successful bullfrog control in 1993 at Twin Pond (shortly after it was re-established after being dry for some years) was associated with a significant recruitment event in the Mexican garter snake population (Rosen and Schwalbe, 1996a), (2) an apparently similar phenomenon in North Pond during the 1980's (below), (3) recruitment of Sonoran mud turtles into the refuge population since bullfrog removals began in 1985 (Rosen and Schwalbe, 1996a; below).

Leopard Frogs

All available monitoring data through for the Chiricahua leopard frog October 1997 at Leslie Canyon NWR (LCNWR), SBNWR, Magoffin Ranch, and Douglas High School, are presented in a previously-submitted final report to Heritage (Rosen and Schwalbe, 1997). In addition, the material is included in a useful compendium in a recently published article (Rosen and Schwalbe, 1998), which is appended to this report.

Possible Fungal Connection in Area Frog Declines

The leopard frog die-offs at Belency Tank, Choate, the refuge Leopard Frog Enclosure, and the refuge ranarium remain mysterious. We are pursuing the matter through an examination of air quality data that may be locally available. However, we have discovered that dead Chiricahua leopard frogs (n=2) from the SBNWR Ranarium and dying lowland leopard frogs (*R. yavapaiensis*; n=2) from Ciénega Creek Preserve (near Marsh Station) suffered from a chytridiomycetes fungal infection of the dermis, which appeared to be the proximate cause of morbidity (Dr. Greg Bradley, University of Arizona Veterinary Diagnostics Laboratory, and Dr. Donald Nichols, National Zoological Park, personal communications).

The Chytridiomycota is a somewhat obscure (but ubiquitous and worldwide) group of fungi known from decaying vegetation and as a plant and insect parasite; yet it has appeared in amphibians moribund in zoos and captive breeding programs, in non-lethal (and perhaps non-pathogenic) occurrence in cricket frogs in the American Midwest (Donald Nichols, personal communication, in press) and lowland leopard frogs in Blue River (Blue River, Arizona, fall 1997; Mike Sredl and Greg Bradley, personal communications), and most importantly, in amphibians from pristine sites with unexplained population declines in Australia and Panama (Berger et al., 1998). Why such a fungus would suddenly appear to be attacking amphibians is not clear, and even if we track the proximate cause of mortality to it, we must consider the possibility that the pathogenic effect is dependant on some ultimate stressor. We will pursue this matter further as the opportunity presents itself at any of the sites that we visit to observe leopard frogs. These findings, including ours specifically, have been widely reported in the state and national media.

SONORAN MUD TURTLES

Overview and Sex Ratio. The intensive bullfrog trapping operation incidentally yielded many captures of Sonoran mud turtles (*Kinosternon sonoriense*) during 1994-1997 (Table 11). Adults of both sexes have been found moving between ponds separated by 0.3 - 1.6 km ($\bar{X}$ = 0.81 km ± 0.52 s.d., n = 8) of terrestrial habitat. Two adult females and six adult males were recorded making these overland movements, although the two females moved only 0.3 km (from Tule to 2PhD Ponds). Among a total of 85 individuals marked, the sex ratio of known-sex individuals was 20 females:58 males; we had averages of 2.7 captures/male and 4.6 captures/female for the 1985-1997 data set.

The observed overland movement of adults was unexpected based on studies in Sonoran Desert populations (Rosen, 1987 and personal observations). In the wetter summer climate of southeastern Arizona, this species may have a greater tendency to move overland.

The sex ratio at SBNWR is strongly skewed in favor of males, as Gibbons (1990 and therein) and others have noted in many other turtle populations. However, at SBNWR, the difference is not a data or age-at-maturity artifact. The fast growth makes the onset of male secondary sex character development quite early, and thus sexing males directly, and females by default, is reliable at a young age (Rosen, personal observations). There is a strongly skewed sex ratio at SBNWR, which is markedly similar to that seen at Quitobaquito (2 males per female; Rosen and Lowe, 1996).

Turtles are subject to temperature-dependant sex determination during the egg stage, and Sonoran mud turtles might be expected to have an excess of males produced at intermediate incubation temperatures (ca. 27-31°C), with females produced at higher temperatures and at lower ones (Ewert and Nelson, 1991; Ewert, personal communication; see discussion in Rosen and Lowe, 1996, p23). Perhaps this pattern does not apply for *Kinosternon sonoriense*, or perhaps high environmental temperatures in Arizona during 1985 - 1997 have put Sonoran mud turtle egg incubation into male-inducing temperature ranges. Speculatively, it could be that the normal (usual) incubation temperature range is lower (i.e., 25-28°C), such that elevated environmental temperatures move the incubation process into the male range.

The cause or causes for the sex ratio anomalies at SBNWR and Quitobaquito are unclear and deserve further attention. It is possible that climate change could be their ultimate cause.

Estimation of Population Size. From the standpoint of population estimation, our finding of inter-pond movement means we cannot properly apply estimation functions or concepts within local ponds, but should apply them to the refuge area as a whole. This approach is relatively justifiable for our data because we trapped extensively in all of the water bodies except House Pond. The latter may contribute some migration interchange with the refuge bottom lands, but we cannot estimate this now, and we therefore ignore it for these computations; we have now recorded turtle movement between House Pond and Double Phd. Ponds. Similarly, there is undoubtedly some emigration and immigration from ponds across the border on the Mexican portion of the San Bernardino bottomland. We have not attempted to correct for this, and it represents a small potential error.

The number of captures is not sufficient to test for unequal catchabilities of individuals. Previous work (Rosen, 1987; Rosen and Lowe, 1996 indicates that this problem is most severe among juveniles; we thus excluded juveniles under three years of age (i.e., 2 full years post-hatching) from all computations. This is acceptable here because rapid growth makes juveniles at 3+ years large enough to be relatively mobile and catchable; and furthermore, there are not many juveniles present in the refuge population. We have also compensated for unequal catchabilities by collapsing the data set to at most one capture per individual per year. Finally, the relatively high proportion of marked individuals minimizes both the size of the biases and the inherent variance of the estimates. Nonetheless, the issue of unequal catchabilities among individuals implies that our population estimates are biased toward low values.

Using the assumptions and utilization of the data set described above, we were able to generate 6 separate Lincoln-Petersen Index estimates of the population size of Sonoran mud turtles on the refuge bottomlands. These were using the 1995 data to compute an index for 1994 abundance; doing the same with the 1996 and 1997 data; using the 1996 and 1997 data to compute estimates for abundance in 1995; and finally, using the 1997 data to compute an estimate for 1996. As a result, mortality becomes irrelevant to the computation. By eliminating the young turtles, whose age can be determined accurately, permitting appropriately keeping them in or out of the computations, natality can also be eliminated as a source of bias. Only undocumented immigration from Mexico remains a problem, and for the turtles, this is a smaller problem than for other taxa. Such immigration would inflate our computed estimate of population size slightly, tending to offset the bias associated with unequal catchabilities of individuals.

Our population estimate for the refuge bottomland ponds and springs is 63.8 individuals, with a 95% confidence interval computed from the variance among the 6 estimates of 55.4 - 72.3. This is not too different from a previous estimate for the refuge of 34 turtles in 1985-1994, based on smaller samples and spatially less extensive trapping (Rosen and Schwalbe, 1996a). Our previous estimates did not include turtles at Robertson Ciénega, or from the Mesquite, Little Mesquite, and Oasis Ponds areas, which are included in the present estimate.

We can adjust our estimates by weighting each of the 6 estimates (Table 12) by the number of recaptures used to compute it: this yields an estimate of 66.6 turtles (retaining the variance from above to compute the confidence interval, which is a conservative procedure, gives a confidence interval of 58.2 - 75.0). Finally, we can restrict our estimates to those computed from the recapture rates in 1997, when we had the most uniformly intensive sampling of ponds: the three estimates are 71, 72, and 73, and we think that 72 (64 - 80) is the best available estimate of the population of 3 year and older Sonoran mud turtles on the refuge bottomlands for about 1995. Baited hoop nets in House Pond in 1996 yielded a respectable 7 individuals in 4 trap-days, suggesting that another 20 or more turtles are present there.

In conclusion, the refuge population of Sonoran mud turtles remains small at approximately 100. It seems possible that our bullfrog removal efforts may benefit this turtle population by reducing the average size of the bullfrogs, and hence allowing turtles to reach an unpalatable or inedible size sooner. This remains conjectural, but we now have an accurate population estimate on which to base any future results. We have continued this aspect of the study in 1998, using baited hoop traps and the bullfrog traps, including at House Pond. Similarly, another round of trapping turtles at Arivaca Ciénega, where a substantial sample was marked in 1993, seems warranted.

GARTER SNAKES

Application of PIT (Passive Integrated Transponder)-tagging in addition to scale clipping has confirmed that our mark-recapture results are valid. Results indicate excellent retention of scale clips and PIT tags (but difficulty accurately reading some scale clips, as well as loss of some PIT tags). The combination of the two methods ensures a high degree of certainty and precision in the results discussed below.

Overview of Trends. Intensive aquatic trapping yielded sizable annual samples of garter snakes in 1993-7 (Tables 13, 14). Trapping data (Table 13) shows a decline of garter snake capture rate per capture effort of about 39% between the 1980's and 1990's. Although this indicates a decline, there is surprising stability, which, however, masks a critical, rapid decline in one of the species, the Mexican garter snake (*T. eques*).

This project began with observations in 1985-6 that the Mexican garter snake population was top-heavy, lacked successful recruitment, and was composed of individuals that had suffered tail damage ascribed to repeated attacks by large bullfrogs (Rosen and Schwalbe, 1988). At that time, we observed that Mexican garter snakes were unusually uncommon at SBNWR, even though they were a majority of the garter snake sample we captured then (during most of that period, our sampling was directed specifically toward the Mexican garter snake). We predicted that the Mexican garter snake population would continue to decline, under the influence of bullfrog predation, while the checkered garter snake population would probably persist, as has been the case.

Mexican garter snakes were the majority species in garter snake samples at San Bernardino Ranch from 1950 to about the 1970's (16 Mexican, 9 checkered garter snakes in museum records, (Fig. 2). These early samples probably do not contain a significant bias for capturing Mexican as opposed to checkered garter snakes.

Our data for the 1980's are 43 captures of Mexican and 41 for checkered garter snakes (Table 13), at a time when our efforts were made primarily at North Pond, Black Draw, and Twin Pond, where Mexican garter snakes were most abundant and capturable: the Mexican garter snake population on the refuge was thus over-sampled, relative to the foregoing museum record and our subsequent sampling during the 1990's.

Although Fig. 2 could be interpreted to mean that the decline of the Mexican garter snake at SBNWR occurred mainly during the 1990's, this discussion of sample biases indicates that the decline was already ongoing by the 1980's, and presumably much earlier.

In 1992-1997, we conducted garter snake trapping in most wetlands at SBNWR (North Pond, Twin Ponds, Bathhouse Well, Robertson Ciénega, Double Ph.D Ponds, and briefly in Little Mesquite Pond. The results comprise a relatively unbiased sample of garter snakes in wetland habitats on the refuge; they include trapping and hand collecting in sites with checkered but not Mexican garter snakes, which we previously did not sample intensively.

The Mexican garter snake is approaching extinction at SBNWR (Table 13, Fig. 2). Only 9 different individuals were observed in each year, 1996 and 1997, although trapping into

October 1997 revealed the occurrence of relatively high activity levels in males and well as females of this species, at a season when checkered garter snake activity in the ponds had declined markedly. The 1997 sample contained only large, old snakes. We are holding two of these (2 large females) plus three (2 females, 1 male) that have been reared as lab-born young from a North Pond female since 1995. We will attempt to breed these successfully in 1998 and 1999 to obtain neonates to repopulate protected sites at the refuge. It will be necessary to utilize a wild-caught male to mate with the siblings of the lab-reared male.

The checkered garter snake population has continued to flourish at SBNWR. Although our samples contain a reduced frequency of smaller, younger individuals in 1995-1997 compared to earlier samples (Fig. 3), the capture rate per unit effort (Table 13) has remained remarkable constant, at 0.328 in the 1980's and 0.348 in the 1990's.

Population Size Estimates and Life History.

Methods (General). The accumulation of mark-recapture data over a period of 12 years permits us to begin to examine the population biology and life history of garter snakes at SBNWR. Consistent aquatic trapping has been done each year 1986-1989 and 1992-1997 inclusive, along with on-foot directed searches for garter snakes in each year. Those data (Table 13) show a moderate decline in garter snake numbers observed per unit effort, and a sharp decline for the Mexican garter snake. In 1993 and 1994, more intensive trapping was begun, and in 1995-1997 this effort was increased further. PIT-tagging of all snakes large enough for the tag was initiated in 1996.

The available data set is suitable for analysis using a Lincoln-Petersen method modified to the data that could be obtained. For the present analysis, we have performed modified Lincoln Index computations using years as sampling intervals, as was done for the turtles (above).

A preliminary look at growth and movement was made based on data through 1997 for both species of garter snakes. A formal presentation of this will be given following the 1998 season, as the additional data will greatly enhance the results. Here, we detail key elements of the natural history and ecology of the checkered garter snake, contrast these with new data as well as earlier analysis for the Mexican garter snake (see Rosen and Schwalbe, 1988), and discuss key trends for each species. Further detailed analysis on Mexican garter snake life history and ecology will be forthcoming.

Mexican Garter Snake. We have previously (Rosen and Schwalbe, 1996a & b, 1997) described movement ecology of this species, but we re-iterate it here for completeness. We have a total of 144 records for this species at SBNWR, including 76 individuals marked and 57 recaptures; recapture rates are quite good for the 1990's samples especially, with many individuals seen repeatedly.

Ecology. Mexican garter snakes have not been recorded moving between wetlands except in the singular case of disappearance of one of the Central Draw Spring in Black Draw. That site was a stronghold for native lowland leopard frogs, Mexican garter snakes, and longfin dace and Yaqui topminnows during the mid-1980's. It was a series of spring-fed pools which did not have subadult and adult-sized bullfrogs, but it dried during the late-1980's, and has not provided a significant wetland habitat area since 1992. Seventeen individual Mexican garter snakes were marked in that area, including a substantial number of small juveniles. Of the 17, five were recaptured during 1993-1997 in other wetlands--3 at Twin Ponds and 2 at North Pond. It appears that this species usually avoids overland migration. However, movement between Twin Pond and adjacent Evil Twin (40 m overland) was frequent.

Birth occurred early in this species at SBNWR, as expected based on our earlier findings (Rosen and Schwalbe, 1988), with a confirmed record of a neonate (taken from a bullfrog stomach) as early as May 31. Clutch sizes were large, young were small, and about half the females failed to reproduce in a given year.

Growth of Mexican garter snakes at SBNWR was rapid, with individual females reaching 700-850 mm snout-vent length (SVL) at ages of 3 years, with continuing growth to sizes well in excess of 900 mm SVL. There were 29 captures of females 900-1018 mm SVL, with the largest snake measuring 4 feet 4 inches (1321 mm) total length. Individuals aged 7-10 yr old were confirmed in several cases. At ages of 9 or 10, the individuals (all of the known age ones at these ages were females) appeared less robust than large adults 3-6 yr old, and may have been experiencing senescence. Some older individuals developed grotesquely swollen tails associated with injuries we presume are bullfrog bite squeezes (Schwalbe and Rosen, 1988; Rosen and Schwalbe, 1995).

<u>Sampling and Computational Procedures</u>. We were fortunate to obtain a few recaptures of young juvenile snakes that confirm the high growth rate inferred from the appearance of unmarked snakes and the growth of young adults. It was thus possible to separate age classes with considerable accuracy up to 3 years of age.

In assigning ages to snakes without known growth histories (for the purpose of deciding whether they belonged in population size computations for a year based on re-samples in subsequent years) we relied on (1) the growth rate observed as described above, (2) growth rates of recaptured individuals of similar size seen during the same year and season as the unknown-aged snake, and (3) subsequent growth trajectory of the unknown-aged individual. Most snakes involved were clearly older individuals that could unquestionably be utilized in computations up to three years back in time; and little ambiguity was apparent in assigning unknown-age individuals to be included or excluded from the computations of population size.

Sample sizes were too small to compute population sizes separately for each pond throughout the course of the study. Since we are particularly interested in trends starting in the 1980's or before, we combined all captures on the refuge for analysis. The captures all came from Black Draw (Central and Lower, intensive on-foot searches in each year of study 1985-1992), North Pond and Twin Ponds (trapping and hand collecting 1985-1997), and Robertson Ciénega (trapping 1995-1997). The Robertson Ciénega area (including the previous wetland there at Border Well) was periodically searched for garter snakes throughout the study period.

Mexican garter snakes also occur in an East Wetland Complex on the refuge

(Mesquite and Oasis Ponds and the marshy wetlands associated with them--Cottonwood Spring, Middle Spring, and the former Ciénega Spring), but we have not marked any there nor trapped or hand-collected intensively. This wetland complex is isolated from the rest of the refuge. Although it is presumed to have received some immigrants when the Central Draw Spring population dispersed, we assume that it has not otherwise mingled with the portion of the refuge population we sampled.

Our computations therefore apply to the central and western portion of the refuge, where we sampled throughout the course of our study. Sampling at Robertson Ciénega was weak until 1995, but prior to its construction in 1989 Mexican garter snake habitat there was marginal. The Mexican garter snake population there has been and remains small, and the bias introduced into the computations by uneven sampling intensity there should be minimal.

Population Size, Trend, and Significance. Sixteen estimates of population size (Fig. 4) of the Mexican garter snake at SBNWR (excluding the East Wetland Complex) averaged 30.4 $\pm$ 8.85 s.d. (95% confidence interval 26.1 - 34.8). The linear regression of year versus population size estimate was significant ($r = -0.49$, $p = 0.02$) with a negative slope and a value of 47.3 for the year 1980 (Fig. 3), an extrapolated value of 22.1 in 1998, and predicted extinction in the year 2013. This, combined with the evidence showing marked reduction of trapping success for this species over the course of this study (above), demonstrates a significant population reduction since 1985.

Actual population size on the refuge is larger than that indicated by the computations for several reasons. First, only adults were included in the computation, probably excluding 10-25% of the actual population from consideration. Second, the East Wetland Complex was not sampled, and this presumably contained not more than a fourth of the Mexican garter snakes on the refuge. Finally, males were under-sampled by our sampling focus during late spring and summer, and possibly by greater female catchability overall.

Among 48 individual females marked, 18 were recaptured (38%) a total of 50 times (recaptures were 51% of total sample); 28 males were marked, of which only 5 were recaptured (18%), each of them just once (recaptures were 15% of total captures). Males were thus under-sampled, producing a low bias in the population estimate; but they were also included as 25% of the total sample, and were mostly unmarked in the re-samples, tending to produce a compensatory high bias. Based on the available data, it would appear the sex ratio may not be diverge from unity, and that the male catchability bias probably produced a 10-25% underestimate of total population size.

A more accurate idea of the average population of the Mexican garter snake at San Bernardino National Wildlife Refuge, during 1985-1996 would be 30 (the computed estimate) + 10 (for the unsampled East Wetland Complex) + 10 (to correct for sampling bias between the sexes) + 10 (non-neonatal juveniles during late summer, an arbitrary number based on the general rarity of this age class), yielding a total population of about 60 individuals. Based on our 1985-1986 findings, we previously suggested that about 100 Mexican garter snakes were at SBNWR (Rosen and Schwalbe, 1988). With our estimate of 60 for the entire period, and given the marked decline during 1985-1997, the population in 1985-1986 was surprisingly close to the 100 we originally estimated based on encounter rates.

For 1997 or 1998, the total population size of the Mexican garter snake at SBNWR is probably closer to 25-40 non-newborns. However, the true number could be as low as 20, or less.

It is not surprising that population reduction has been verified for this species, but it is remarkable that the decline has not been more acute. This decline has probably been ongoing since at least the 1940's, when bullfrogs were introduced, and quite possibly from earlier still, when fishes were introduced (Hendrickson et al., 1980), or even as early as the turn of the century, when severe ciénega habitat degradation began (Hendrickson and Minckley, 1985). Back-calculating with the regression equation (Fig. 3) puts the population size at about 96 in 1945, 167 (the intercept of the equation) in 1900, and 187 in 1880: these appear to be rather low values, even if we double them as suggested to compensate for various sampling biases. The refuge area prior to human disturbance was most probably a major, productive wetland complex, and highly suited to garter snakes. It should then have supported several hundred to nearly 1000 garter snakes, primarily Mexican garter snakes, much as it today support large numbers of checkered garter snakes (below).

We have previously demonstrated that Mexican garter snake recruitment occurred successfully at the Central Draw Spring and at Twin Pond when bullfrogs were scarce at those sites (Rosen and Schwalbe, 1996a). Review of the data suggests that our bullfrog removal efforts at North Pond during the 1985-1989 period may also have produced some recruitment. The remarkably gradual species decline on-site is therefore a function of these two or three waves of recruitment, which were produced by (1) the bullfrog removal effort, and (2) natural recruitment in the Central Draw Spring following the high rainfall years that ended in 1985.

Checkered Garter Snake. We have recorded 887 observations of the checkered garter snake (*Thamnophis marcianus*) at and near the refuge, 1985-1997. Of these, 664, including 272 recaptures, were on the refuge and are pertinent to mark-recapture computations. Recapture rates were low until the last three years of study. Recent results from the intensive trapping, with PIT-tagging in addition to scale clipping, have demonstrated that the low recapture rates reflected large population sizes and modest catchability, rather than mark loss, very large and shifting home ranges, or rapid population turnover rates. The ecology of this species appears to be quite different from that of the Mexican garter snake, and is detailed here for the first time in print, with a focus on potential mechanisms of coexistence with the introduced bullfrog.

Habitat Use. Checkered garter snakes occur in all wetlands on the refuge, as well as at water sources of highly varied description across the floors of the San Bernardino and Sulphur Springs Valleys. It is a relatively abundant species at many sites; on the refuge, it appears not to be very abundant in Black Draw, and, remarkably, was not found at the Central Draw Spring during the time Mexican garter snakes were thriving there.

Like the Mexican garter snake, individuals of this species are seen swimming in ponds and are frequently caught in aquatic traps where water depths are 1 m or more. This species is found on or near the bank of ponds and pools, rather than in the water, more frequently than the Mexican garter snake (Figs. 5, 6; Tables 15, 16), and is regularly observed on land

and on roads within about 150 m of aquatic habitat on the refuge (Table 17). These data confirm broader observations in southern Arizona for these two species (Woodin, 1950; Fouquette, 1954; Rosen and Schwalbe, 1988 and unpublished observations; Rossman et al., 1996).

Capture results for this species show the steady appearance of young, unmarked snakes into the sampled population, but also demonstrate that the snakes usually are not captured until they reach about 425 mm SVL (Fig. 3, Table 15). The 1/4-inch mesh on our snake traps permits capture of garter snakes at least as small as 282 mm SVL; thus, it would appear that fully aquatic foraging is infrequent in the small juveniles of this species. In contrast, 3 trappable-sized small juvenile Mexican garter snakes (250 - 400 mm SVL) were trapped and 2 hand-captured in the water; and in other populations, small juvenile Mexican garter snakes have also been readily trapped in the water with adults (Rosen and Schwalbe, 1988).

It appears that a non-aquatic juvenile stage may be the pivotal aspect of life history differentiating the checkered from Mexican garter snakes and permitting its co-existence with the bullfrog. We have no evidence for extensive terrestrial activity among juvenile Mexican garter snakes. Checkered garter snakes, especially when young, are frequently seen on boggy or wet ground or even away from wetlands (Table 17, Fig. 5). Adults of both species at SBNWR regularly and extensively utilize water of all depths (Figs. 5, 6). The most striking differences between the species in syntopic sympatry at SBNWR are (1) the significantly greater utilization of water for activity in juvenile Mexican garter snakes compared to juvenile checkered garter snakes (for snakes $\leq$ 500 mm SVL, the G-likelihood test gives $G = 8.84$, $G_{adj} = 8.31$, $p = 0.004$), and (2) terrestrial activity for the checkered garter snake (Table 17).

The less aquatic juvenile stage of the checkered garter snake, compared to that of the Mexican garter snake, is, on empirical and theoretical grounds, the key aspect of its life history allowing it to coexist successfully with the introduced bullfrog *Rana catesbeiana.* This may reflect an evolutionary history of sympatry with bullfrogs and other large aquatic predators (such as catfishes, centrarchid fishes, and snapping turtles) and competitors (especially water snakes of the genus *Nerodia*) east of the Continental Divide, or its position along a partitioned niche axis with other species of garter snakes (such as the Mexican garter snake west of the Divide), or very likely, a combination of these processes.

The Mexican garter snake is highly aquatic throughout its life in southeastern Arizona, behaving much like a water snake of the genus *Nerodia* (Rosen and Schwalbe, 1988 and herein): like the even more aquatic narrow-headed garter snake *T. rufipunctatus*, it is allopatric with *Nerodia* and the rest of the characteristic aquatic fauna of eastern North America. The records we have on land at SBNWR (Fig. 6) are, with few exceptions, of snakes basking on banks or sloping embankments close to perennial water pools and riffles. We have one record of an individual moving along an embankment away from the direction of nearby water, and three records of neonates (shown in Fig. 6) that were all in vegetation from which they could not flee directly to water.

Confinement of our records for juvenile Mexican garter snakes to shallower waters (Fig. 6) might be an artifact of their virtual absence from deepwater habitats where large bullfrogs were prominent, and we cannot be certain that Fig. 6 illustrates a real, behavioral, ontogenetic change in habitat selection according to water depth by this species. In the checkered garter snake, by contrast, ontogenetic changes in both macrohabitat and water depth selection appear to be real, since juveniles are abundant around, but not frequently seen in, water, especially deep water.

Our data for SBNWR also show that larger, adult checkered garter snakes are significantly more terrestrial than similar-sized and larger Mexican garter snakes (SVL $\geq$ 500 mm; Figs. 5 & 6, Table 17; G-test, $p = 0.025$), although this is less pronounced than for juveniles. Excluding data more than 10 m from standing or running surface water, our data are not sufficient to show a significant difference between the species ($0.10 < p < 0.20$) in utilization of wetland habitat (59% of Mexican versus 41% of checkered garter snakes were found from the water's edge out rather than on land).

Movements. Recapture data confirm that movement of checkered garter snakes between wetlands at SBNWR occurs at a low but consistent rate. Movement between Robertson Ciénega and the Twin Pond area probably involves 5-15% of the snakes of trappable size per year (of 16 recaptures at Robertson Ciénega in 1996-7, 3 were records of snakes that had moved from Twin-Evil Twin [0.6 km movement). We also have two records of movement from North Pond to Twin (1.7 km), 2 movements from Lower Black Draw to Twin (0.2 and 0.6 km), 2 movements from Bathhouse Well to Double PhD Ponds (ca. 0.2 km). Thirty-five of 184 (19.0%) recaptures within the Twin-Evil Twin complex were movements between the two ponds, which are separated by approximately 40 m. We expect that 1998 data will allow a refined estimate of inter-wetland movement rates.

Female Reproduction. Mean litter size among 26 litters we observed through 1997 in southern Arizona was 13.5 young, and increased with female size (Fig. 7). This is intermediate between values for southern Texas (9.5; Ford and Kargas, 1987), the Tamaulipan Biotic province of Texas and northeastern Mexico (13; Ford and Kargas, 1987; Ford and Seigel, 1989), and east-central Arizona (15.3, n=14; Rossman et al., 1996). The smallest gravid female was 501 mm SVL, slightly smaller than the minimum of 515 mm SVL reported for Arizona by Rossman et al. (1996). Along with a number of other small gravid individuals (Fig. 7), this small adult was apparently two years old. This species is mature at much smaller sizes in the Tamaulipan Province (345 mm SVL; Ford and Kargas, 1987).

Through 1997, 5 wild-caught female *T. marcianus* have been held in the laboratory to obtain data on their litters (Table 18). Neonatal SVL (175.2 mm) and mass (2.67 g) were significantly greater than for southern Texas (150 mm and 2.0 - 2.3 g; Ford and Kargas, 1987; Ford and Seigel, 1989), but similar to Rossman et al.'s (1996) value for Arizona (2.8 g). These values are significantly smaller than for young of *T. eques* in southern Arizona (personal observations). Relative clutch mass (RCM) in southern Arizona averaged 21.6% of female gravid mass, similar to values reported for southern Texas (20%; Ford and Kargas, 1987), and for the species (22.4%, no locality given; Seigel et al., 1986). These values are quite low for garter snakes, as well as for viviparous snakes in general (Seigel and Fitch, 1984; Seigel et al., 1986), but similar to that for *T. eques* (mean = 23.0% $\pm$ 2.07% SE, n=5; personal observations) in southern Arizona.

Ford and Kargas (1987), studying samples taken over about 7° of latitude, from the Balcones Escarpment of Texas through San Luis Potosí, Mexico, argued that individual female checkered garter snakes had a lengthy annual reproductive period, were potentially able to produce two litters per year, and probably did so on occasion. We have no evidence for a similar phenomenon in Arizona, and considerable evidence that such "double-clutching" does not occur.

May-June samples from SBNWR contained numerous adult-sized, non-reproductive snakes, at a time when all snakes that reproduced in a given year would have been detectably gravid or obviously post-partem (see Table 19). A total of 58% at SBNWR were reproductive in any given year (29 of 50) compared to 67% of dissected museum specimens from southern Arizona (8 of 12). Overall, reproductive frequency indicated by these data was 59.7%, not too different for that estimated for the Tamaulipan Biotic Province (71.4%; Ford and Kargas, 1987). Neither value encourages confidence that resources and reproductive rates in this species might be adequate for production of two annual litters, even where the season is long enough. We previously noted that approximately 50% of Mexican garter snake females in Arizona were reproductive in any given year of adult life (Rosen and Schwalbe, 1988).

In southern Arizona, our dissection, palpation, and laboratory birth results for the checkered garter snake all consistently demonstrate a narrowly timed annual reproductive cycle (Table 19) that can involve no more than a single annual litter of young. Ovulation occurs during April-May, gravid females are observed during May-August, and birth occurs June-August. The observed variation in timing reflects elevational effects (Table 19), as well as differences in annual temperature patterns. These factors may help explain Rossman et al.'s (1996) report of May parturition in Arizona. The reproductive cycle is earlier in the Mexican garter snake, with birth occurring by late May at SBNWR and during mid-July at 1500 m (personal observations).

Growth, Size, and Activity. Other attributes of the life history of the checkered garter snake are emerging from our study, and one or more of these also appear to contribute to successful coexistence with bullfrogs. Although this species also grows to a large size (23 captures of individuals in the largest size class (800-870 mm SVL), we have found that growth is normally remarkably slow. While we await further data, especially on the previously under-sampled juvenile life stages, our tentative age-size assignments based on birth size in the laboratory, on size-frequency histograms (Fig. 8A-D), and on inspection of recapture data pertinent to growth, are as follows: 0 yr [birth], 175 mm SVL (see Table 18); 1 yr (July, Fig. 8C), 425 mm SVL; 2 yr, 550; "older", for males $\geq$ 510 mm SVL, females $\geq$ 600 mm SVL. It appears that a full year of semi-terrestrial life may precede the more semi-aquatic adult lifestyle.

Growth is slow, and it appears to be relatively variable. Further, total activity indicated by our trapping is lower than that for Mexican garter snakes. The ratio of captures of checkered to Mexican garter snakes for the SBNWR mark-recapture program was 5:1, while population estimates (below) indicate the disparity in numbers approaches 13:1.

Although this differential catchability between the species is due at least partly to the aquatic focus of our trapping method, the slow growth of checkered garter snakes suggests that behavioral limits on habitat selection and intensity of foraging activity are life history tactics. This slower growth may reflect an adaptation to avoid selection of the richest (i.e., the aquatic) habitat type for foraging, as an evolutionary response to sympatry with aggressive aquatic predators.

Mexican garter snakes grow much more rapidly than checkered garter snakes at all ages, a striking and possibly key life history difference from the checkered garter snake. Radiotelemetry and energetics studies comparing these species would be of great interest, but will be impossible until Mexican garter snakes are brought from the verge of extirpation at SBNWR, or the work will have to be done at a site where Mexican garter snakes remain abundant in sympatry with checkered garter snakes.

Sampling and Computational Procedures. Population estimates were computed using modified Lincoln-Petersen Index methods, similar to those used for mud turtles and Mexican garter snakes (above). Age determination was more difficult for the checkered garter snakes and will remain so until growth rates are clarified. We included individuals estimated to be in their second year of life (ages 9-14 months, assuming July birth) or older in the pre-sample, and included individuals in their third year or older (males > ca. 475 mm SVL; females > ca. 535 mm SVL; depending on month of capture) for the recapture sample. Although these computations are thus a first approximation and require refinement, we think they are sufficient for a preliminary evaluation of population size.

Sufficient data were available to permit computation of population sizes within several wetland areas at SBNWR for 1995 and 1996. Immigration and emigration from these wetlands has been demonstrated (above), introducing a further bias into the computations, although not a gross bias. We have not attempted to adjust our computed values because we anticipate being able to refine the computational procedure in the future.

Population Size. Computed population sizes for 1 yr old and older checkered garter snake at SBNWR are shown in Table 20. The preliminary estimate of 687 snakes on the refuge clearly demonstrates that this species is an order of magnitude (or more) more abundant at the refuge than the Mexican garter snake. It may be the most numerous snake on the refuge, with the possible exception of the secretive Southwestern blackheaded snake.

The number of checkered garter snakes, including juveniles, is probably at least 800, and may approach 1000. The ratio of total records for checkered to Mexican garter snakes (664:133, or 5 to 1) is much less than the disparity in estimated population sizes (800:60, or 407:30, which are 13 to 1). This finding is consistent with the higher recapture rates for Mexican than checkered garter snakes.

The abundance of checkered garter snakes on the refuge, while not remarkably high, is what one might expect for garter snakes in such productive and auspicious habitat.

RECOMMENDATIONS

Recommendations from the recently-submitted report (Rosen and Schwalbe, 1997) are being implemented now, and will not be reiterated in detail here. The following recommendations are intended to complement or expand upon those:

1. Modify the bullfrog removal and garter snake study protocols at SBNWR using the experimental design suggested by Rosen and Schwalbe (1997) with modifications that may be indicated by 1998 field data.

2. Expand the sampling program for garter snakes and continue PIT-tagging, to ensure adequate sampling of both species of garter snakes.

3. Intensify the turtle study program to ensure that proper age-determinations are made for each individual, and accurate population estimates and growth data are obtained from the strong basis established during 1985-1997.

4. Contrast the SBNWR turtle data with that from Arivaca, which has bullfrogs, centrarchid fishes, and crayfish, Sycamore Creek at Sunflower, which has crayfish in great abundance, and other sites without these introduced turtle predators.

5. Initiate radio-telemetric study of garter snake ecology in southern Arizona.

6. Maintain leopard frogs, Mexican garter snakes, and Sonoran mud turtles in refuge enclosures, and continue to support and collaborate with refuge staff in developing conservation plans for, and managed populations of, declining aquatic herpetofauna.

7. Assist development of a plan to restructure and refurbish the Robertson Ciénega wetland and Black Draw. The effectiveness and desirability of the fencing approach should be evaluated using the experiences of North Pond and the Leopard Frog Enclosure.

8. Conduct an aquatic survey of the Mexican portion of the original San Bernardino land grant lowlands, and of surrounding country, which is now very timely and is sorely needed to properly understand wetland vertebrate status and landscape dynamics at and near SBNWR.

9. Monitor managed leopard frog populations, assist development of a valley-wide conservation plan, and investigate new, reported population sites for the Chiricahua leopard frog.

ACKNOWLEDGEMENTS

We are most grateful to the many people who contributed to the field work throughout this project. We have acknowledged most of them in Rosen and Schwalbe (1996a & b). Here we specifically acknowledge contributions to this project in 1996 and 1997. For assistance in the field we thank Hans Bodenhamer; Bruce Christman; Kevin Cobble; Andrew, Cindy and Ted Cordery; Amanda Gaide; Steve Hale; Trevor, Troy, Janet, Cassidy, and Delany Hare; Jeff Howland; Robin Llewellyn; Matt, Anna, Chris and Mike Magoffin; Verma Miera; Mike Pruss; Julia Rosen; Shawn Sartorius; Adam Schwalbe; Mike Sredl; Dennis Suhre; Marty Teugel; Travis Tuchak; Eric Wallace; and Bridget Watts. Again, we apologize profusely to those whose names we may have inadvertently omitted.

Arizona Game and Fish Department has also been instrumental in all phases of the work, including consistently funding the work through the Heritage Program. USFWS has been equally instrumental, both through funding and through the longtime collaboration of Refuge Manager Kevin Cobble and Officer Matt Magoffin. Their direct participation, and that of Tony Velasco and Myles Traphagen under refuge direction has been an integral and essential part of this project. The continuing advice, participation, and collaboration by AGFD Nongame personnel, particularly Jeff Howland, Mike Sredl, and Eric Wallace, has been instrumental to the guidance and conduct of the work. Residents of the lower San Bernardino Valley area who have assisted importantly and generously with information and access to lands are Warner and Wendy Glenn, Drummond Hadley, Seth Hadley, Anna and Matt Magoffin, and Bill and Mary McDonald. Their assistance, in part through the Malpai Borderlands Group, is gratefully acknowledged. We are grateful to Joan Ford, Mary Greene, Richard Maze, Guillermina Sumpter, and Carol Wakely for administrative and clerical assistance on the project.

LITERATURE CITED

Berger, L., R. Speare, P. Daszek, D. E. Green, A.A. Cunningham, C.L. Goggin, R. Slocombe, M.A. Ragan, A.D. Hyatt, K.R. McDonald, H.B. Hines, K.R. Lips, G. Marantelli, and H. Parkes. 1998, *in press.* Chytridiomycosis causes amphibian mortality associated with population declines in the rain forests of Australia and Central America. Proceedings of the National Academy of Science 95:000-000.

Ewert, M.A. and C.E. Nelson. 1991. Sex determination in turtles: diverse patterns and some possible adaptive values. Copeia 1991:50-69.

Ford, N.B. and J.P. Kargas. 1987. Reproduction in the checkered garter snake, *Thamnophis marcianus*, from southern Texas and northeastern Mexico: seasonality and evidence for multiple clutches. Southwestern Naturalist 32:93-101.

Ford, N.B. and R.A. Seigel. 1989. Phenotypic plasticity in reproductive traits: evidence from a viviparous snake. Ecology 70:1768-1774.

Fouquette, M.J. 1954. Food competition among four sympatric species of garter snakes, genus *Thamnophis*. Texas Journal of Science VI:172:188.

Gibbons, J.W. 1990. Sex ratios and their significance among turtle populations. Pp. 171- 182 *in* J.W. Gibbons (*ed.*), Life History and Ecology of the Slider Turtle. Smithsonian Institution Press, Washington, D.C.. xiv + 368 pp.

Hendrickson, D.A., and W.L. Minckley. 1985. Cienegas--vanishing climax communities of the American Southwest. Desert Plants 6:131-175.

Hendrickson, D.A, W.L. Minckley, R.R. Miller, D.J. Siebert, and P.H. Minckley. 1980. Fishes of the Rio Yaqui Basin, Mexico and United States. J. Ariz,-Nev. Acad. Sci. 15:65-106.

Rosen, P.C. 1987. Female reproductive variation among populations of the Sonoran mud turtle, *Kinosternon sonoriense*. Masters Thesis, Arizona State University, Tempe, Arizona.

Rosen, P.C. and C.H. Lowe. 1996. Status and ecology of the Sonoran mud turtle at Quitobaquito Springs, Arizona. Unpubl. report from Arizona Game and Fish Dept. (Phoenix, Arizona).

Rosen, P.C. and C. R. Schwalbe. 1988. Status of the Mexican and narrow-headed gartersnake (*Thamnophis eques megalops* and *Thamnophis rufipunctatus rufipunctatus*) in Arizona. Unpubl. report from Arizona Game and Fish Dept. (Phoenix, Arizona) to U.S. Fish and Wildlife Service, Albuquerque, New Mexico. iv + 50 pp + appendices.

Rosen, P.C. and C.R. Schwalbe. 1995. Bullfrogs: introduced predators in southwestern wetlands. Pp. 542-454 *in* Laroe, E.T., G.S. Farris, C.E. Puckett, P.D. Doran, and M.J.

Mac (*eds.*), Our Living Resources: A Report on the Distribution, Abundance, and Health of U.S. Plants, Animals, and Ecosystems. U.S. Dept. of Interior, National Biological Service, Washington, D.C. 530 pp.

Rosen, P.C. and C.R. Schwalbe. 1996a. Bullfrog Impacts on Sensitive Wetland Herpetofauna, and Herpetology of the San Bernardino National Wildlife Refuge. Final Report to Arizona Game and Fish Department Heritage Fund and U.S. Fish and Wildlife Service, June 6, 1996. 47 pp. + Appendices.

Rosen, P.C. and C.R. Schwalbe. 1996b. A critical interim evaluation of the effectiveness of bullfrog removal methods at San Bernardino National Wildlife Refuge. Final Report to Arizona Game and Fish Department Heritage Fund and U.S. Fish and Wildlife Service, October 30, 1996. 21 pp.

Rosen, P.C. and C.R. Schwalbe. 1997. Status of native and introduced species of aquatic herpetofauna at San Bernardino National Wildlife Refuge. Final Report to Arizona Game and Fish Department Heritage Fund and U.S. Fish and Wildlife Service, December 31, 1997. 44 pp.

Rosen, P.C. and C.R. Schwalbe. 1998. Using managed waters for conservation of threatened frogs. Pp. 180-202 *in* Anonymous (*compiler*), Environmental, Economic, and Legal Issues Related to Rangeland Water Developments; Proceedings of a Symposium, November 13-15, 1997 at Arizona State University, Tempe (available from Rosalind Pearlman, Arizona State University, College of Law, P.O. Box 877906, Tempe, AZ 85287-7906, (602) 965-2124, rosalind.pearlman@asu.edu).

Rossman, D.A., N.B. Ford, and R.A. Seigel. 1996. The Garter Snakes: Evolution and Ecology. University of Oklahoma Press, Norman, Oklahoma. xx + 332 pp.

Schwalbe, C.R. and P.C. Rosen. 1988. Preliminary report on effects of bullfrogs on wetland herpetofauna in southeastern in Arizona. Pp. 166-173 *in* R.C. Szaro, K.E. Severson, and D.R. Patton (editors), Management of Amphibians, Reptiles, and Small Mammals in North America. U.S. Forest Service, General Technical Report RM-166, Fort Collins, CO.

Seigel, R.A. and H.S. Fitch. 1984. Ecological patterns of relative clutch mass in snakes. Oecologia 61:293-301.

Seigel, R.A., H.S. Fitch, and N.B. Ford. 1986. Variation in relative clutch mass in snakes among and within species. Herpetologica 42:179-185.

Woodin, W.H. III. 1950. Notes on Arizona species of *Thamnophis*. Herpetologica 6:39-40.

Table 1A. Manual bullfrog (*Rana catesbeiana*) removal effort at San Bernardino National Wildlife Refuge, Cochise County, Arizona. Removal trips were Memorial and Labor day weekends, beginning with Labor Day in 1986. An additional intensive manual removal was conducted on Independence Day weekend, 1988.

Year	Days Intensive Manual Removal	Mean Number of Personnel/Trip
1985	0	2
1986	2	7
1987	6	10
1988	10	10
1989	7	11
1990	0	0
1991	0	0
1992	7	18
1993	6	15
1994	7	16
1995	6	25
1996	6	13
1997	6	12

Table 1B. Trapping effort for bullfrogs (*Rana catesbeiana*) at San Bernardino National Wildlife Refuge, Cochise County, Arizona. The number of trap days is followed by (number of traps set). The entry "ne" indicates a wetland did not exist in the year indicated.

	YEAR											Site
LOCALITY	85	86	87	88	89	92	93	94	95	96	97	Total
Astin Spring												
Bathhouse Well							67 (1)	224 (1)	153 (1)	134 (1)	119 (1)	563
Cottonwood Well												0
Black Draw				30 (10)	54 (9)	44 (11)		120 (4)				248
Evil Twin	ne	ne	ne	ne			270 (3)	224 (1)	298 (2)	134 (1)	141 (1)	933
House Pond												0
Mesquite Pond	ne	ne	ne	ne	ne			674 (5)	918 (6)	134 (1)	612 (4)	2204
North Pond	2 (1)			6 (2)	6 (1)		627 (4)	896 (4)	745 (5)	629 (5)	564 (4)	2846
Oasis Pond	ne	ne	ne	ne	ne			76 (1)	298 (2)	278 (2)	119 (1)	493
Double PhD	ne	ne	ne	ne				118 (1)	270 (2)		357 (3)	745
Robertson Cienega	ne	ne	ne	ne			264 (2)	278 (2)	149 (1)	278 (2)	504 (3)	1195
Tule Pond							134 (2)	374 (2)	306 (2)	134 (1)	238 (2)	1052
Twin Pond	2 (1)		ne	ne			552 (5)	832 (4)	596 (4)	402 (3)	423 (3)	2405
ANNUAL TOTAL	4 (2)	0 (0)	0 (0)	36 (12)	60 (10)	44 (11)	1914 (17)	3816 (25)	3733 (26)	2103 (15)	3077(22)	12684

Table 1C. Estimated total removal effort and efficiency/effort for bullfrog removal at San Bernardino National Wildlife Refuge, Cochise County, Arizona, 1993-1997, comparing trapping, manual removals, and total removals. Computational methods are described in the text.

	YEAR							
LOCALITY	93	94	95	96	97	**ave**	**no./unit effort**	**rel. efficiency**
trap effort (trap-days)	1914	3816	3733	2103	3077	**2928.6**	**0.110**	**33.788**
traps captures	121	302	367	259	555	**320.8**		
manual effort (person-days)*	90	112	150	78	72	**100.4**	**3.701**	
manual captures*	435	333	427	358	305	**371.6**		
adjusted effort**	4955	7600	8801	4738	5510			
total captures (>90 mm SVL)	556	635	794	617	860			
biomass removal (kg)	165.7	195	210	128	171			
captures / effort* **	**0.112**	**0.084**	**0.090**	**0.130**	**0.156**			
biomass removal / effort* **	**0.024**	**0.020**	**0.018**	**0.020**	**0.025**			

**during two large trips (Memorial Day and Labor Day).*

***taken as [(trap effort) + (33.788 X manual effort)]*

****based on adjusted effort*

Table 2. Number of bullfrogs removed, SBNWR, Arizona.

LOCALITY	YEAR 85	86	87	88	89	92	93	94	95	96	97	SITE TOTAL
Astin Spring	0	0	0	0	0	0	30	4	14	3	33	84
Bathhouse Well	0	0	2	0	0	0	7	15	11	6	1	42
Cottonwood Well	0	2	0	0	0	0	0	0	0	0	0	2
Black Draw	0	0	71	102	45	11	0	0	93	0	0	322
Evil Twin	0	0	0	0	0	0	27	5	17	18	14	81
House Pond	0	0	110	73	175	143	189	234	269	313	294	1800
Mesquite Pond	0	0	0	0	0	0	0	261	186	57	133	637
North Pond	0	85	139	168	290	140	146	247	181	176	95	1667
Oasis Pond	0	0	0	0	0	0	0	9	35	44	23	111
Double PhD	0	0	0	4	0	0	1	6	30	9	85	135
Robertson Cienega	0	0	0	0	0	16	72	78	132	74	232	604
Tule Pond	0	1	6	2	0	1	19	15	23	21	3	91
Twin Pond	4	5	0	3	0	43	111	114	491	78	195	1044
ANNUAL TOTAL	4	93	328	352	510	354	602	988	1482	799	1108	6620

Table 3. Number of adult and subadult bullfrogs (>90 mm SVL) removed, SBNWR, Arizona.

LOCALITY	YEAR 85	86	87	88	89	92	93	94	95	96	97	SITE TOTAL
Astin Spring	0	0	0	0	0	0	26	3	4	1	21	55
Bathhouse Well	0	0	0	0	0	0	6	15	11	4	1	37
Cottonwood Well	0	2	0	0	0	0	0	0	0	0	0	2
Black Draw	0	0	64	102	44	11	0	0	0	0	0	221
Evil Twin	0	0	0	0	0	0	27	5	16	17	14	79
House Pond	0	0	106	53	144	111	186	161	196	248	165	1370
Mesquite Pond	0	0	0	0	0	0	0	250	184	41	132	607
North Pond	0	80	123	117	198	135	141	213	152	107	92	1358
Oasis Pond	0	0	0	0	0	0	0	9	35	43	23	110
Double PhD	0	0	0	2	0	0	0	6	28	5	85	126
Robertson Cienega	0	0	0	0	0	15	72	75	109	68	229	568
Tule Pond	0	1	5	1	0	1	19	14	23	21	3	88
Twin Pond	4	5	0	2	0	39	107	107	132	47	82	525
ANNUAL TOTAL	4	88	298	277	386	312	584	858	890	602	847	5146

Table 4. Mass (kg) of bullfrogs removed, SBNWR, Arizona.

LOCALITY	YEAR 85	86	87	88	89	92	93	94	95	96	97	SITE TOTAL
Astin Spring	0.0	0.0	0.0	0.0	0.0	0.0	4.4	0.5	1.3	0.2	4.0	10.3
Bathhouse Well	0.0	0.0	0.0	0.0	0.0	0.0	1.2	3.0	1.6	0.9	0.2	7.0
Cottonwood Well	0.0	0.4	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.4
Black Draw	0.0	0.0	15.4	19.7	7.5	2.3	0.0	0.0	0.7	0.0	0.0	45.7
Evil Twin	0.0	0.0	0.0	0.0	0.0	0.0	5.2	1.0	3.2	3.4	2.9	15.8
House Pond	0.0	0.0	26.1	13.2	37.8	47.1	79.7	64.3	63.9	53.6	30.4	416.1
Mesquite Pond	0.0	0.0	0.0	0.0	0.0	0.0	0.0	59.3	38.8	8.9	31.3	138.3
North Pond	0.0	18.5	30.3	28.3	45.8	36.4	31.9	29.0	28.0	21.8	15.8	285.5
Oasis Pond	0.0	0.0	0.0	0.0	0.0	0.0	0.0	1.6	6.1	7.5	5.3	20.5
Double PhD	0.0	0.0	0.0	0.6	0.0	0.0	0.1	1.4	6.0	1.6	18.5	28.2
Robertson Cienega	0.0	0.0	0.0	0.0	0.0	3.4	13.4	15.9	22.4	14.6	41.3	111.0
Tule Pond	0.0	0.2	1.2	0.3	0.0	0.2	5.1	2.4	4.5	4.4	0.5	18.8
Twin Pond	0.8	1.4	0.0	0.4	0.0	9.6	24.6	16.4	33.6	11.5	20.6	119.0
ANNUAL TOTAL	0.8	20.4	73.0	62.4	91.1	98.9	165.7	194.8	210.1	128.5	170.8	1216.6

Table 5.Mass (kg) of adult and subadult bullfrogs (>90 mm SVL) removed, SBNWR, Arizona.

LOCALITY	YEAR 85	86	87	88	89	92	93	94	95	96	97	SITE TOTAL
Astin Spring	0.0	0.0	0.0	0.0	0.0	0.0	4.2	0.5	1.0	0.1	3.7	9.4
Bathhouse Well	0.0	0.0	0.0	0.0	0.0	0.0	1.2	3.0	1.6	0.9	0.2	6.9
Cottonwood Well	0.0	0.4	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.4
Black Draw	0.0	0.0	15.2	19.7	7.5	2.3	0.0	0.0	0.0	0.0	0.0	44.6
Evil Twin	0.0	0.0	0.0	0.0	0.0	0.0	5.2	1.0	3.1	3.4	2.9	15.7
House Pond	0.0	0.0	25.9	12.5	36.9	46.4	79.6	61.9	61.2	50.3	24.6	399.4
Mesquite Pond	0.0	0.0	0.0	0.0	0.0	0.0	0.0	59.0	38.7	8.5	31.3	137.5
North Pond	0.0	18.3	29.4	26.2	42.9	36.2	31.7	27.7	27.1	19.8	15.6	274.8
Oasis Pond	0.0	0.0	0.0	0.0	0.0	0.0	0.0	1.6	6.1	7.5	5.3	20.5
Double PhD	0.0	0.0	0.0	0.5	0.0	0.0	0.0	1.4	5.9	1.6	18.5	28.0
Robertson Cienega	0.0	0.0	0.0	0.0	0.0	3.3	13.4	15.7	21.8	14.5	41.1	110.0
Tule Pond	0.0	0.2	1.2	0.2	0.0	0.2	5.1	2.3	4.5	4.4	0.5	18.7
Twin Pond	0.8	1.4	0.0	0.4	0.0	9.5	24.4	16.3	30.5	11.0	19.8	114.1
ANNUAL TOTAL	0.8	20.3	71.7	59.5	87.2	97.9	164.9	190.5	201.5	122.0	163.5	1179.9

Table 6. Mean snout-vent length (mm) of adult and subadult bullfrogs (>90 mm SVL) captured at SBNWR, Arizona.

LOCALITY	YEAR 85	86	87	88	89	92	93	94	95	96	97	SITE MEAN
Astin Spring							118.2	120.3		102.0	131.0	123.3
Black Draw			135.4	131.0	124.8	129.4						131.0
Evil Twin							137.6	137.5	145.0	138.0	113.0	136.0
House Pond			132.0	136.1	140.8	168.7	164.5	164.8	152.6	134.3	124.4	146.7
Mesquite Pond								146.9	146.3	141.1	143.6	145.9
North Pond		130.5	133.4	133.6	136.3	148.4	139.1	112.8	127.0	123.0	121.0	133.1
Double PhD				98.0				192.0	138.5	170.0	108.0	139.3
Robertson Cienega						142.4	133.6	137.1	137.1	113.0	115.6	134.1
Twin Pond	135.0	133.6		133.5		145.4	143.0	125.4	141.2	146.6	144.7	140.6
ANNUAL TOTAL	135.0	130.7	133.4	133.0	136.7	154.3	148.1	143.4	142.2	133.9	127.6	139.7

Table 7. Mean mass (kg) of adult and subadult bullfrogs (>90 mm SVL) removed from SBNWR, Arizona.

LOCALITY	YEAR 85	86	87	88	89	92	93	94	95	96	97	SITE TOTAL
Astin Spring							161.5	154.7	245.0	75.0	174.3	170.5
Bathhouse Well							200.3	202.6	147.1	217.0	155.0	186.0
Evil Twin							193.5	206.0	195.4	200.9	204.7	198.3
House Pond			244.4	236.3	256.2	418.1	428.0	384.6	312.4	202.8	149.0	291.5
Mesquite Pond								236.0	210.4	208.1	237.1	226.6
North Pond		228.2	239.0	223.7	216.5	268.3	224.6	129.9	178.1	185.3	170.0	202.4
Oasis Pond								182.2	174.3	173.5	230.5	186.4
Double PhD				254.5				226.7	212.5	320.2	218.2	221.9
Robertson Cienega						222.4	186.7	210.0	199.7	213.8	179.7	193.6
Tule Pond		180.0	241.2	207.0		219.0	270.3	167.3	194.8	211.4	164.0	212.5
Twin Pond	200.0	283.2		198.5		242.9	228.1	152.4	230.9	233.7	242.1	217.4
ANNUAL TOTAL	200.0	230.1	240.6	214.9	225.9	313.8	282.3	222.0	226.4	202.7	193.1	229.3

Table 8. Number of adult-sized bullfrogs (>119 gram MASS) removed, SBNWR, Arizona.

LOCALITY	YEAR 85	86	87	88	89	92	93	94	95	96	97	SITE TOTAL
Astin Spring	0	0	0	0	0	0	16	2	4	0	15	37
Bathhouse Well	0	0	0	0	0	0	6	11	6	3	1	27
Cottonwood Well	0	2	0	0	0	0	0	0	0	0	0	2
Black Draw	0	0	57	90	37	10	0	0	0	0	0	194
Evil Twin	0	0	0	0	0	0	20	5	12	16	11	64
House Pond	0	0	95	39	107	108	175	150	164	152	84	1074
Mesquite Pond	0	0	0	0	0	0	0	226	152	33	115	526
North Pond	0	72	105	87	139	124	101	93	110	78	68	977
Oasis Pond	0	0	0	0	0	0	0	8	21	36	20	85
Double PhD	0	0	0	2	0	0	0	5	23	5	78	113
Robertson Cienega	0	0	0	0	0	12	69	74	93	61	184	493
Tule Pond	0	1	5	1	0	1	16	9	17	19	3	72
Twin Pond	4	5	0	2	0	36	87	76	111	42	76	439
ANNUAL TOTAL	4	80	262	221	283	291	490	659	713	445	655	4103

Table 9. Number of young adult-sized bullfrogs (120-350 gram MASS) removed, SBNWR, Arizona.

LOCALITY	YEAR 85	86	87	88	89	92	93	94	95	96	97	SITE TOTAL
Astin Spring	0	0	0	0	0	0	16	2	4	0	14	36
Bathhouse Well	0	0	0	0	0	0	6	10	6	3	1	26
Cottonwood Well	0	2	0	0	0	0	0	0	0	0	0	2
Black Draw	0	0	52	90	37	10	0	0	0	0	0	189
Evil Twin	0	0	0	0	0	0	20	5	12	15	9	61
House Pond	0	0	74	29	68	22	34	37	69	111	80	524
Mesquite Pond	0	0	0	0	0	0	0	208	139	28	96	471
North Pond	0	69	88	70	109	94	74	92	109	71	66	842
Oasis Pond	0	0	0	0	0	0	0	8	20	35	20	83
Double PhD	0	0	0	2	0	0	0	4	20	4	74	104
Robertson Cienega	0	0	0	0	0	9	69	74	92	61	182	487
Tule Pond	0	1	5	1	0	1	12	8	14	19	3	64
Twin Pond	4	5	0	2	0	34	80	76	91	38	60	390
ANNUAL TOTAL	4	77	219	194	214	170	311	524	576	385	605	3279

Table 10. Number of large adult-sized bullfrogs (>350 gram MASS) removed, SBNWR, Arizona.

LOCALITY	YEAR 85	86	87	88	89	92	93	94	95	96	97	SITE TOTAL
Astin Spring	0	0	0	0	0	0	0	0	0	0	1	1
Bathhouse Well	0	0	0	0	0	0	0	1	0	0	0	1
Cottonwood Well	0	0	0	0	0	0	0	0	0	0	0	0
Black Draw	0	0	5	0	0	0	0	0	0	0	0	5
Evil Twin	0	0	0	0	0	0	0	0	0	1	2	3
House Pond	0	0	21	10	39	86	141	113	95	42	4	551
Mesquite Pond	0	0	0	0	0	0	0	18	13	5	19	55
North Pond	0	3	18	18	30	30	27	1	1	7	2	137
Oasis Pond	0	0	0	0	0	0	0	0	1	1	0	2
Double PhD	0	0	0	0	0	0	0	1	3	1	4	9
Robertson Cienega	0	0	0	0	0	3	0	0	1	0	2	6
Tule Pond	0	0	0	0	0	0	4	1	3	0	0	8
Twin Pond	0	0	0	0	0	2	7	0	20	4	16	49
ANNUAL TOTAL	0	3	44	28	69	121	179	135	137	61	50	827

Table 11. Captures of the Sonoran Mud Turtle, *Kinosternon sonoriense*, SBNWR, 1985-1997

	YEAR:											
SITE	85	86	87	88	89	92	93	94	95	96	97	site total
Double PhD Ponds	0	0	0	0	0	0	0	15	38	0	94	147
Bathhouse Pool	0	0	0	0	0	0	0	0	1	3	10	14
Evil Twin	0	0	0	0	0	0	0	0	1	0	0	1
Guadalupe Canyon	0	0	0	0	0	0	0	1	0	0	0	1
House Pond	0	0	0	0	0	1	0	0	0	7	0	8
Lower Draw	0	0	1	4	1	1	0	1	0	0	0	8
Mesquite Pond	0	0	0	0	0	0	0	0	2	0	5	7
North Pond	2	0	0	0	0	1	7	18	12	5	9	54
Oasis Pond	0	0	0	0	0	0	0	0	0	0	1	1
Robertson Cienega	0	0	0	0	0	0	0	0	1	0	5	6
Tule Pond	0	0	0	0	0	0	0	27	14	5	2	48
Twin Pond	0	3	0	0	0	0	0	3	4	0	0	10
year total	2	3	1	4	1	3	7	65	73	20	126	305

Table 12.Computed population size estimates for the Sonoran Mud Turtle, SBNWR, 1994-1996. Biasing factors and corrections are discussed in text. House Pond not included in estimates.

YEAR ESTIMATED	DATA YEAR USED FOR ESTIMATE	NO. OF RECAPTURES FOR ESTIMATE	COMPUTED POPULATION SIZE (UNCORRRECTED)
1994	1995	8	63
1994	1996	4	58
1994	1997	12	73
1995	1996	5	46
1995	1997	13	71
1996	1997	6	72
			mean = 63.8
			95% C.I. = 55.4 - 72.3

Table 13. Abundance and species composition indicated by garter snake samples at SBNWR, Arizona.

Year:	85	86	87	88	89	92	93	94	95	96	97	total / mean	
No. *T. eques* captures	2	8	9	12	12	10	20	14	31	9	16	143	
% T. eques	40.0	66.7	56.3	66.7	36.4	50.0	24.1	23.0	12.4	5.3	10.5	35.6	
No. *T. marcianus* captures	3	4	7	6	21	10	63	47	219	160	137	677	
% *T. marcianus*	60.0	33.3	43.8	33.3	63.6	50.0	75.9	77.0	87.6	94.7	89.5	64.4	
Total garter snake sample	5	12	16	18	33	20	83	61	250	169	153	820	
Days on site	4	8	6	12	7	7	29	24	42	32	36	207	
Trap-days	0	30	16	26	16	25	204	150	505	390	295	1657	
Effort (=days + trap-days)	4	38	22	38	23	32	233	174	547	422	331	1864	
Captures/Trap-day	undef.	0.40	1.00	0.69	2.06	0.80	0.41	0.41	0.50	0.43	0.52	0.49	
Captures/Effort	1.25	0.32	0.73	0.47	1.43	0.63	0.36	0.35	0.46	0.40	0.46	0.44	(1980's=0.67; 1990's=0.42)

Table 14. Percentage composition of terrestrial snake sample at SBNWR, Arizona, 1985-1997.

	Year:											
Species	85	86	87	88	89	92	93	94	95	96	97	species totals
Crotalus atrox	5	2	1	5	2	3	5	5	7	3	10	48
Crotalus scutulatus	0	0	0	0	0	0	1	0	0	0	0	1
Diadophis punctatus	0	0	0	0	0	1	1	1	0	0	0	3
Hypsiglena torquata	0	0	1	0	0	2	0	1	1	0	3	8
Lampropeltis getula	2	2	1	4	1	1	3	2	9	3	5	33
Masticophis flagellum	3	3	5	2	5	2	10	9	19	11	5	74
Pituophis melanoleucus	0	1	2	5	0	2	4	1	10	8	5	38
Rhinocheilus lecontei	0	0	0	1	0	1	1	1	2	1	4	11
Salvadora hexalepis	0	0	0	0	0	2	0	2	3	8	1	16
Tantilla hobartsmithi	0	1	1	0	0	0	0	0	0	4	3	9
year totals	10	9	11	17	8	14	25	22	51	38	36	241

TABLE 15. Capture type versus size, *T. marcianus*, SBNWR, 1985-1997

	Capture Type:					
SVL (mm)	hand	road	cover	aquatic trap	% hand	TOTAL
100-199	11	1	1	2	0.73	15
200-299	11	2	0	1	0.79	14
300-399	9	1	1	12	0.39	23
400-499	14	4	2	66	0.16	86
500-599	19	1	3	149	0.11	172
600-699	21	0	0	138	0.13	159
700-799	24	0	1	54	0.30	79
800-899	10	2	1	8	0.48	21
ALL	119	11	9	430	0.21	569

TABLE 16. Capture type as a function of size, *T. eques*, SBNWR, 1985-1997

	Capture Type:				
SVL (mm)	hand	cover	aquatic trap	% hand	TOTAL
100-199	2	0	0	1.00	2
200-299	3	1	2	0.50	6
300-399	2	0	1	0.67	3
400-499	2	0	0	1.00	2
500-599	5	0	4	0.56	9
600-699	6	0	10	0.38	16
700-799	11	0	10	0.52	21
800-899	9	0	25	0.26	34
900-999	4	0	21	0.16	25
1000-1099	0	0	2	0.00	2
ALL	44	1	75	0.37	120

Table 17. T. marcianus records far from water, SBNWR, 1985-97

SVL (mm)	Distance from water (m)
246	75
269	25
351	75
453	50
465	150
480	11
588	150
605	100
634	50
644	35
728	20
817	45
833	15
844	75

Table 18. Statistics for laboratory-born young from wild-caught pregnant females, *T. marcianus*, 1987-1997 Variances for offspring parameters are pooled (mean of within-litter variances).

year:	locality:	Maternal SVL	# young	sex ratio	SVL	tail length	Mass	RCM1	RCM2	RCM3
1987	Nogales	821	36	20F:16M	178.5	50.1	2.62	0.224	0.324	0.244
1987	Nogales	717	23	10F:13M	176.4	47.7	2.62	0.266	0.552	0.356
1997	SBNWR	770	18	7F:11M	157.2	44.2	2.20	0.181	0.247	0.193
1987	Nogales	590	10	5F:5M	186.3	49.5	3.16	0.208	0.328	0.247
1987	Nogales	557	5	4F:1M	194.0	51.6	3.93	0.203	0.260	0.206
mean		691.0	18.4	46F:46M	175.18	48.20	2.670	0.2163	0.3420	0.2493
s.d.					5.15	3.17	0.246	0.0319	0.1230	0.0639
SE					1.30	0.88	0.073	0.0270	0.0550	0.0286
min					149	42	2.07			
max					197	57	4.22			

RCM1 = clutch mass / pregnant mass

RCM2 = clutch mass / post-partem female mass

RCM3 = clutch mass / (clutch mass + post-partme female mass)

Table 19. Reproductive tract dissections, lab births, and palpation results (in parentheses [sample size in brackets]) for female reproductive cycle, *T. marcianus*, southern Arizona. *SB* = San Bernardino National Wildlife Refuge, el. 1159m.

Reproductive Status:	March	April	May	June	July	August	September	October
non-reproductive		Apr 29	(*SB*:May [11 of 26])	(*SB*:Jun [10 of 24])	Jul 9 Jul 18 (*SB*:Jul [29 of 39])	Aug 14 (*SB*:Aug [57 of 57])	Sep 6 Sep 14 (*SB*:Sep [23 of 23])	(*SB*:Oct [7 of 7])
follicles enlarging		Apr 11, 6-9 mm Apr 19, 5-10mm (May 1)						
follicles immediately pre-ovulatory	Mar 30, 8x17mm		(May 1 [4]) May 21, 10x16mm					
recent ovulation			May 7, 13x18mm =ovulating May 11, ovulated May 15, ovulated					
(gravid)			(*SB*:May [15 of 26])	(*SB*:Jun [4 of 24])	(*SB*:Jul [6 of 39])			
advanced embryos				Jun 6, 1/2 devel. (Jun 7 [2]) (Jun 13)				
term embryos						Aug 2		
birth				Jun 20	Jul 8 Jul 12* Jul 14 Jul 15* Jul 22 Jul 23			
recently post-partem				(Jun 16, el. 450m) (*SB*:Jun [10 of 24])	(*SB*:Jul [4 of 39])	Aug 14, el. 1525m		

Table 20. Population size estimates (modified Lincoln-Petersen) for adult and subadult-sized *T. marcianus*, SBNWR. See text for discussion of estimates.

	Computed Population	Estimated Population
Computed Estimates:		
Twin Ponds	137.45	
North Pond	53.60	
Robertson Cienega	95.00	
Double Phd, including Bathhouse	121.30	
subtotal	**407.35**	
Other Wetland Sites:		
House Pond		100
East Wetland complex		100
Tule Pond		20
Black Draw		40
Astin Spring		20
subtotal		**280**
combined total	**687.35**	

Fig. 1. Bullfrog removal effort and total removals.

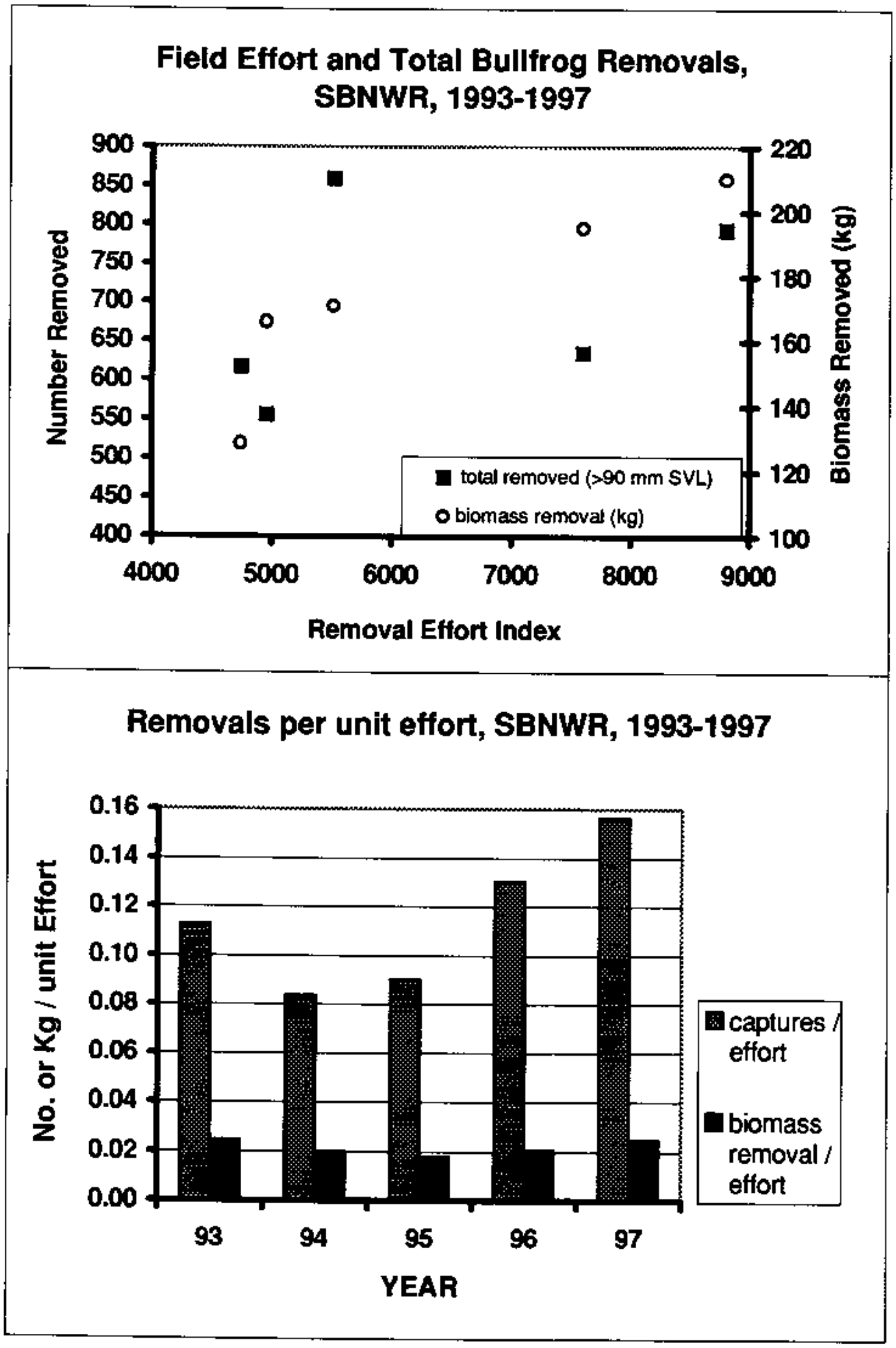
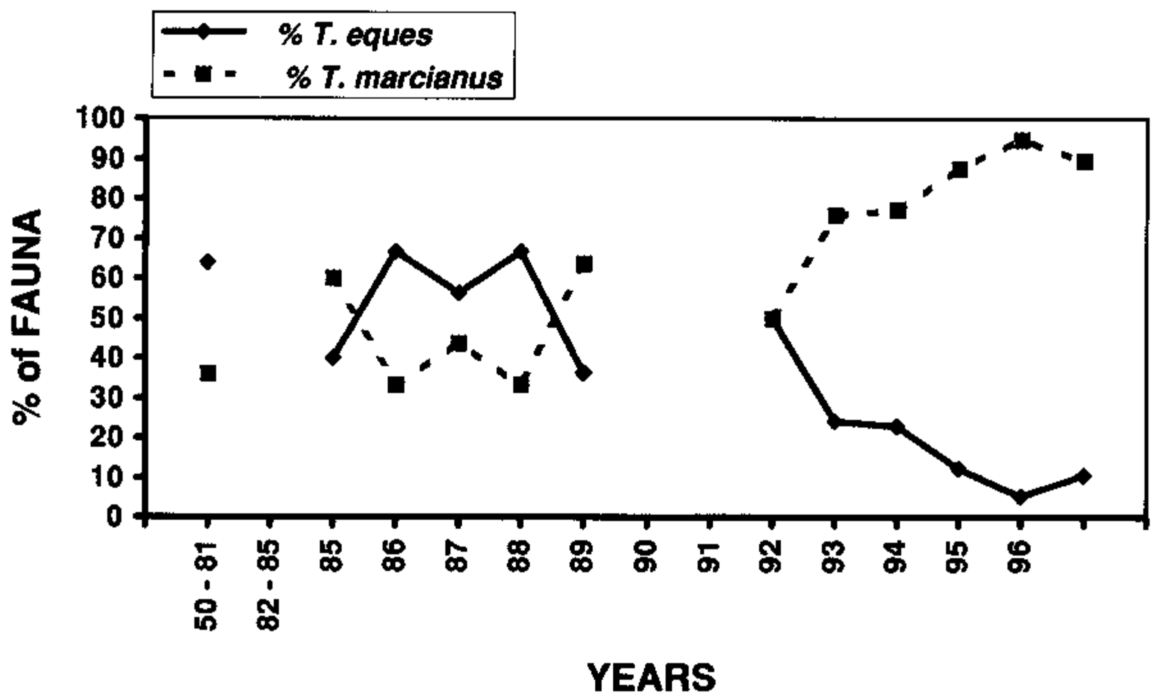
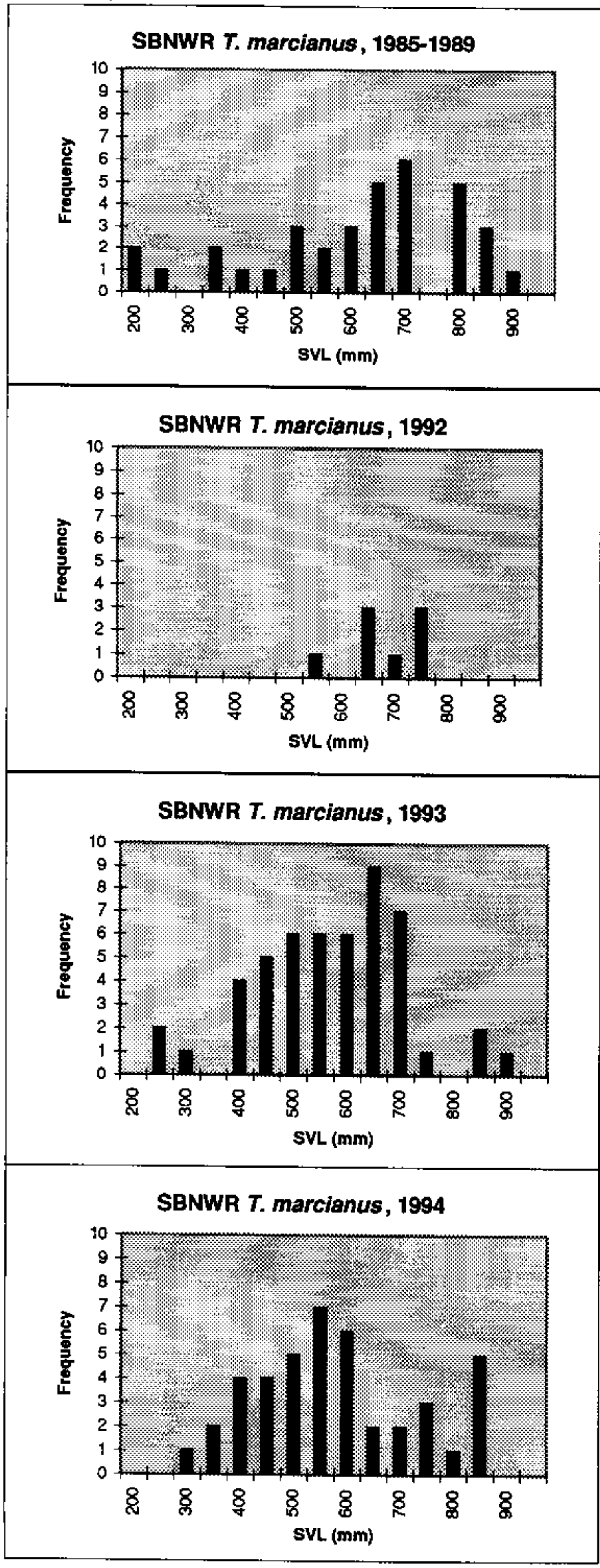


Fig. 2. Change in composition of the garter snake fauna at SBNWR, Cochise Co., Arizona, 1950 - 1997





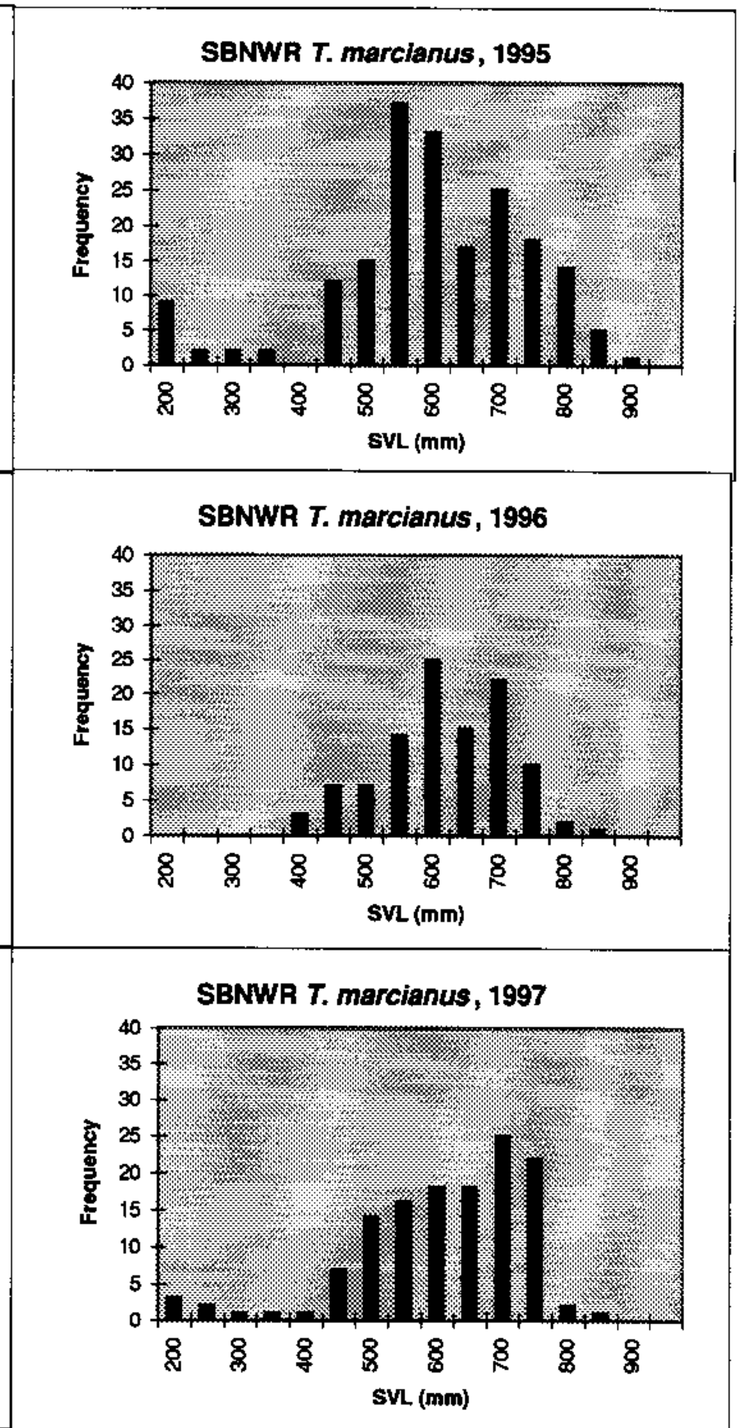


Fig. 3. Body size distributions of checkered garter snake samples at SBNWR over the course of study.

Fig. 4. Computed population size, adult *T. eques*, Black Draw, North Pond, Twin Ponds, Robertson Cienega, SBNWR, 1985 - 1997.

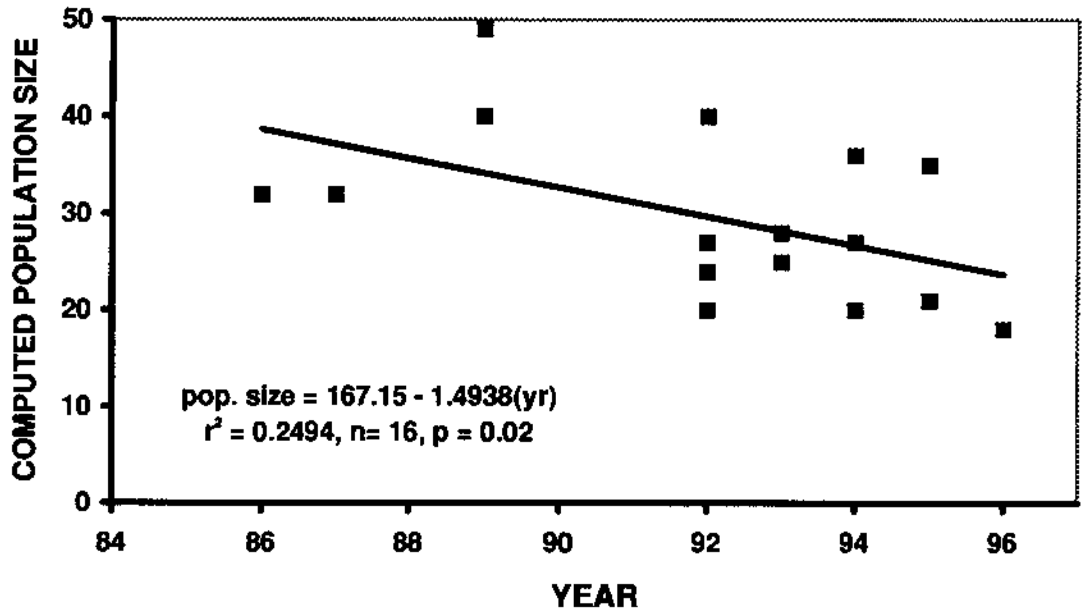
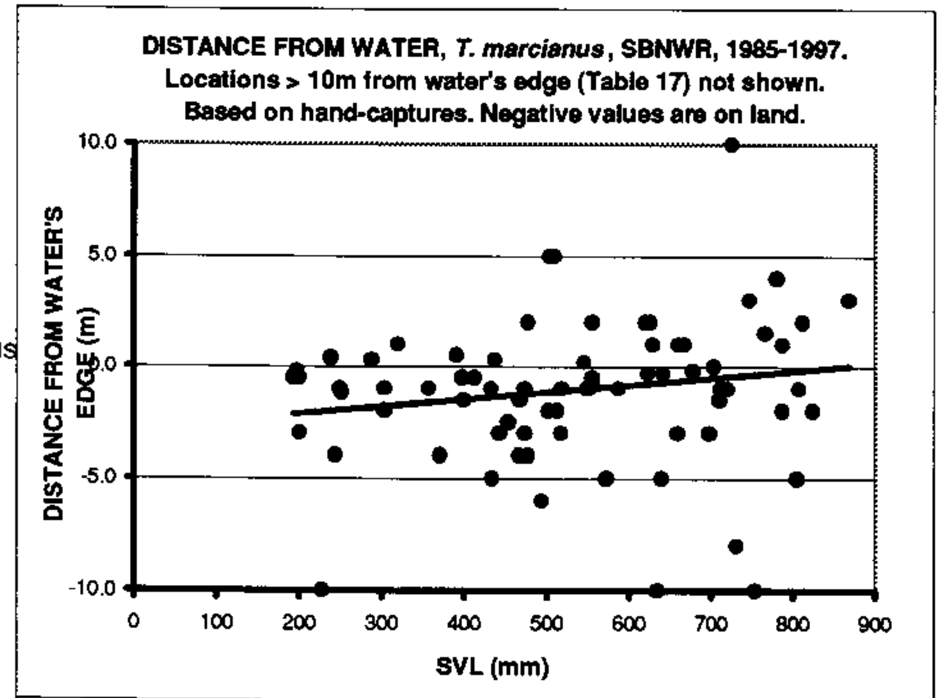


Fig. 5.


DISTANCE FROM WATER, *T. marcianus*, SBNWR, 1985-1997.
Locations > 10m from water's edge (Table 17) not shown.
Based on hand-captures. Negative values are on land.
DISTANCE FROM WATER'S EDGE (m)
10.0
5.0
0.0
-5.0
-10.0
0
100
200
300
400
500
600
700
800
900
SVL (mm)

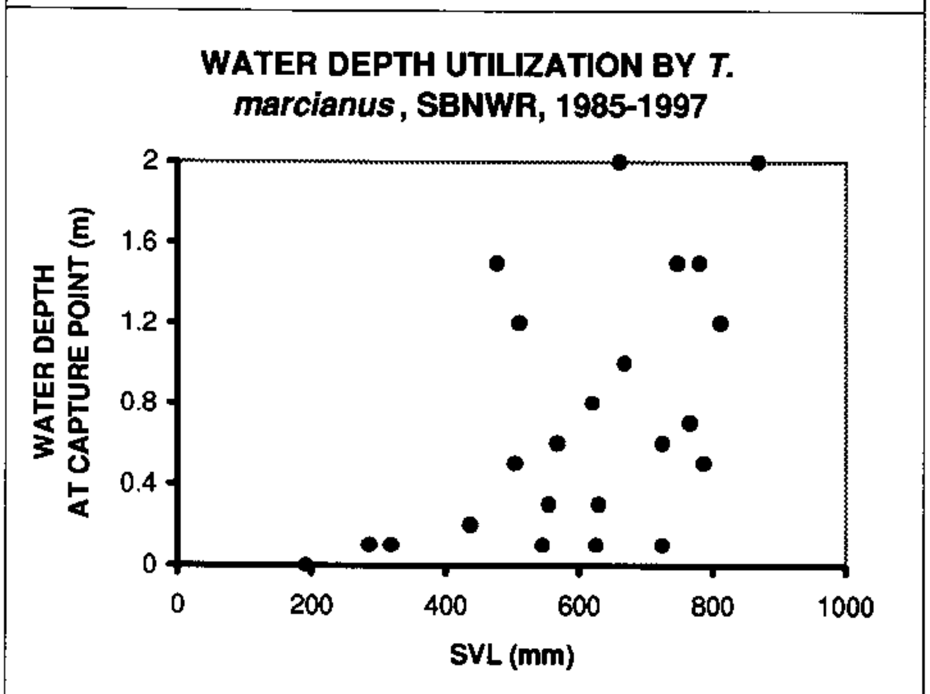
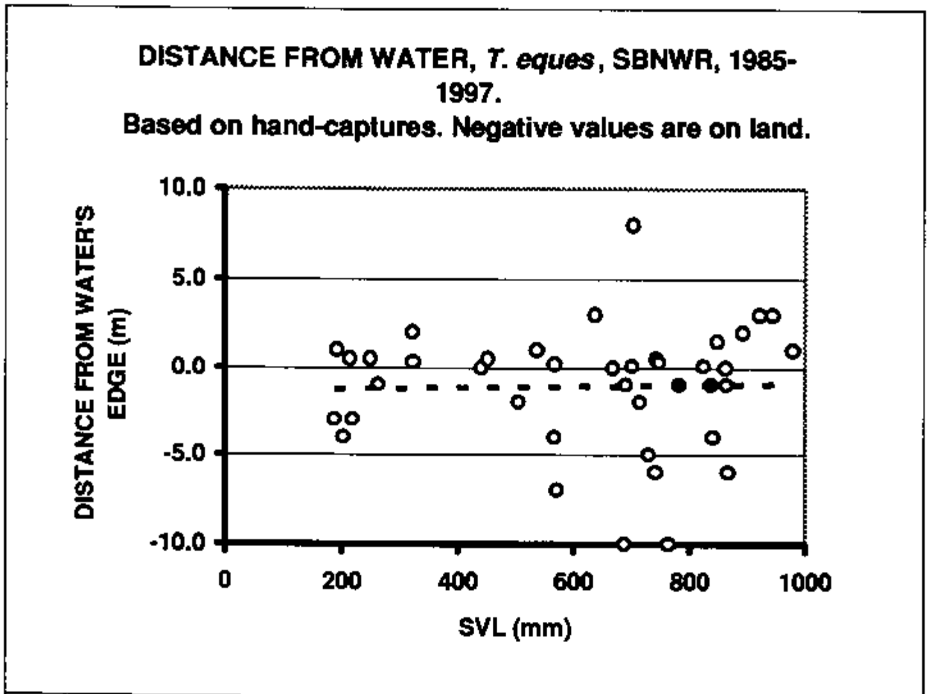

WATER DEPTH UTILIZATION BY *T. marcianus*, SBNWR, 1985-1997
WATER DEPTH AT CAPTURE POINT (m)
2
1.6
1.2
0.8
0.4
0
0
200
400
600
800
1000
SVL (mm)

Fig. 6.


DISTANCE FROM WATER, *T. eques*, SBNWR, 1985-1997.
Based on hand-captures. Negative values are on land.
DISTANCE FROM WATER'S EDGE (m)
10.0
5.0
0.0
-5.0
-10.0
0
200
400
600
800
1000
SVL (mm)

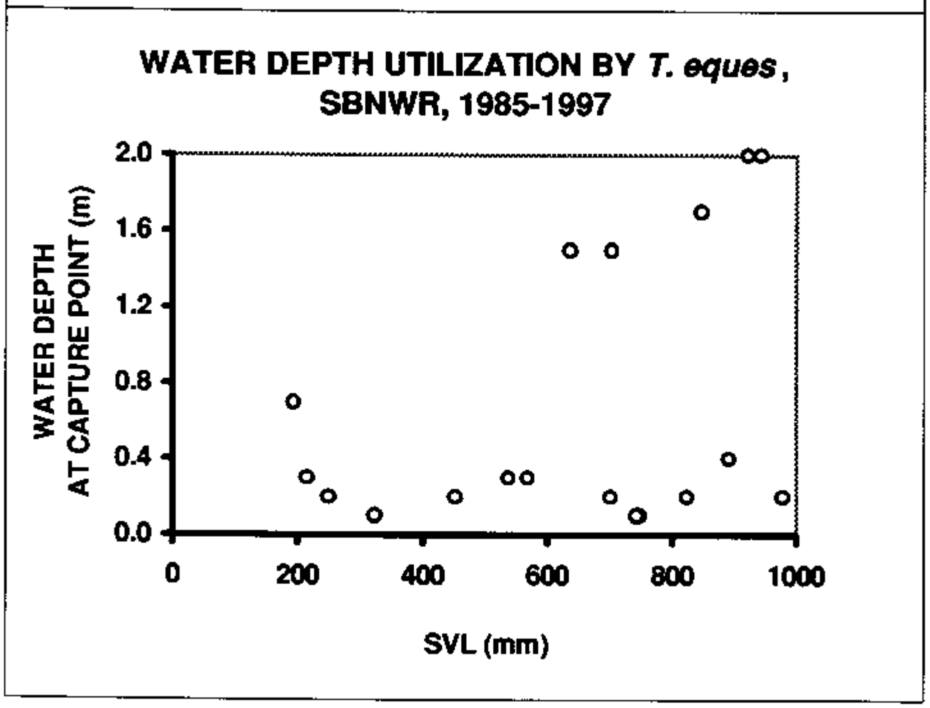

WATER DEPTH UTILIZATION BY *T. eques*, SBNWR, 1985-1997
WATER DEPTH AT CAPTURE POINT (m)
2.0
1.6
1.2
0.8
0.4
0.0
0
200
400
600
800
1000
SVL (mm)

Fig. 7. Relationship between clutch size and body size in *Thamnophis marcianus*, SE Arizona, 1985-1997.

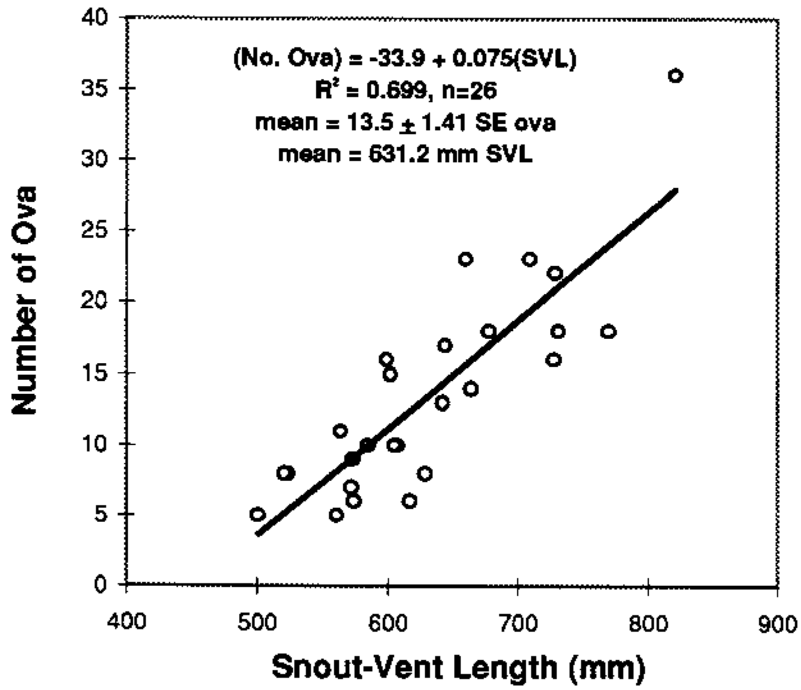


Fig. 8A. Size-frequecny diagrams for male and female *T. marcianus*.

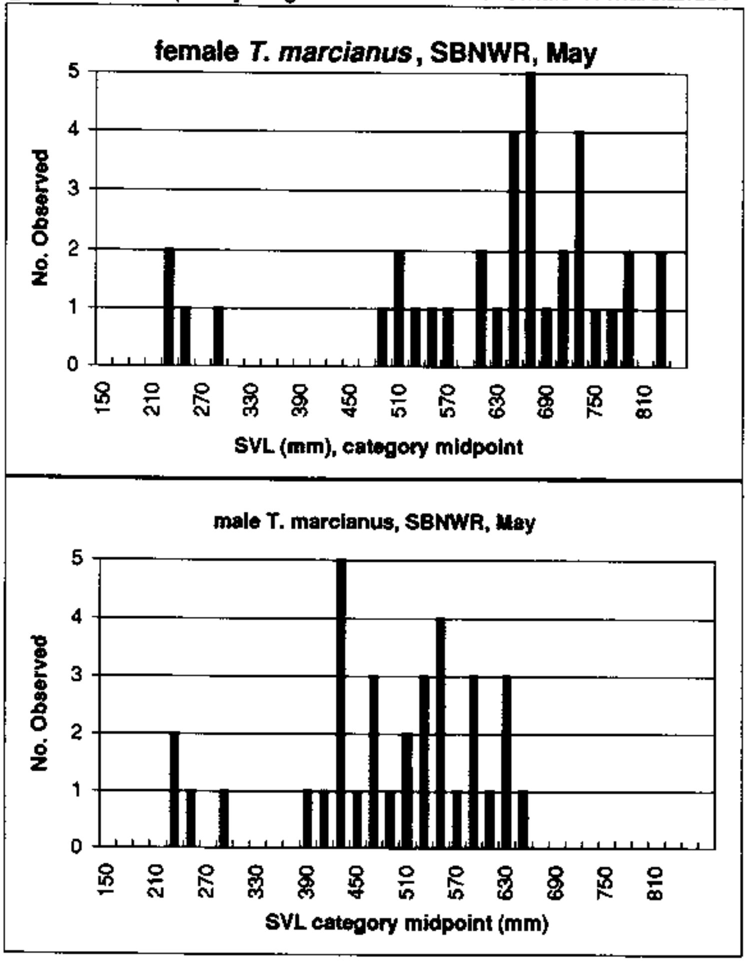


Fig. 8B. Size-frequecny diagrams for male and female *T. marcianus*

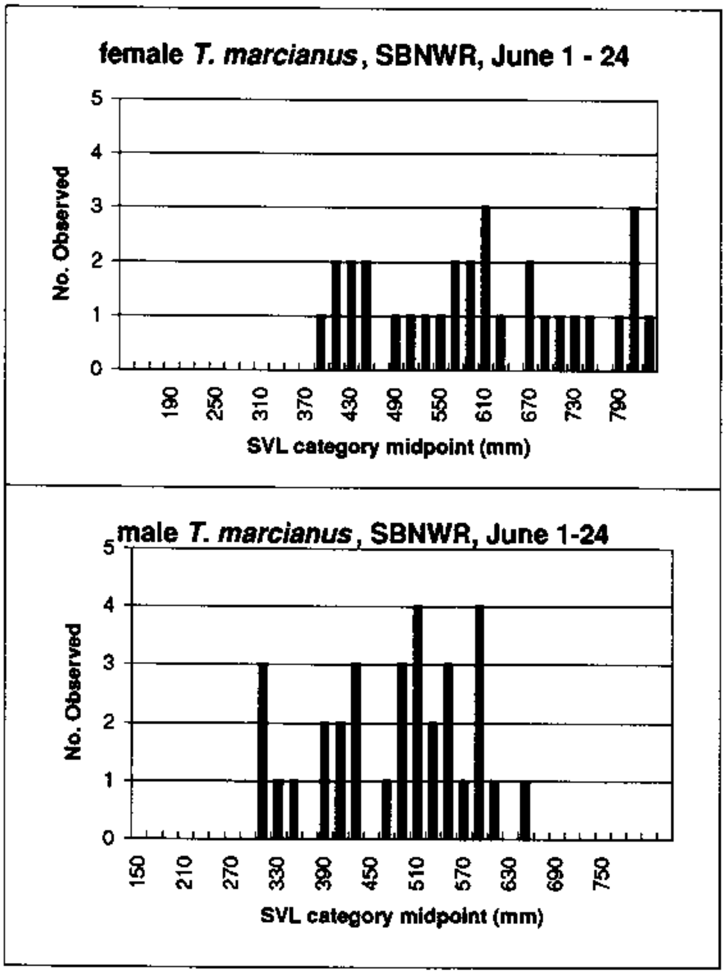


Fig. 8C. Size-frequecny diagrams for male and female *T. marcianus*

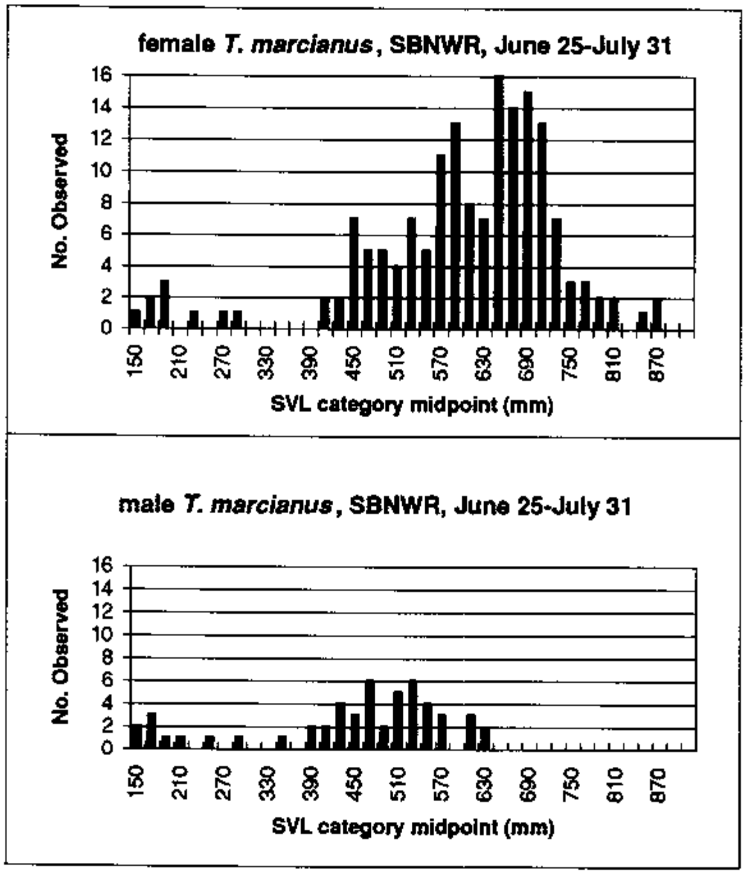
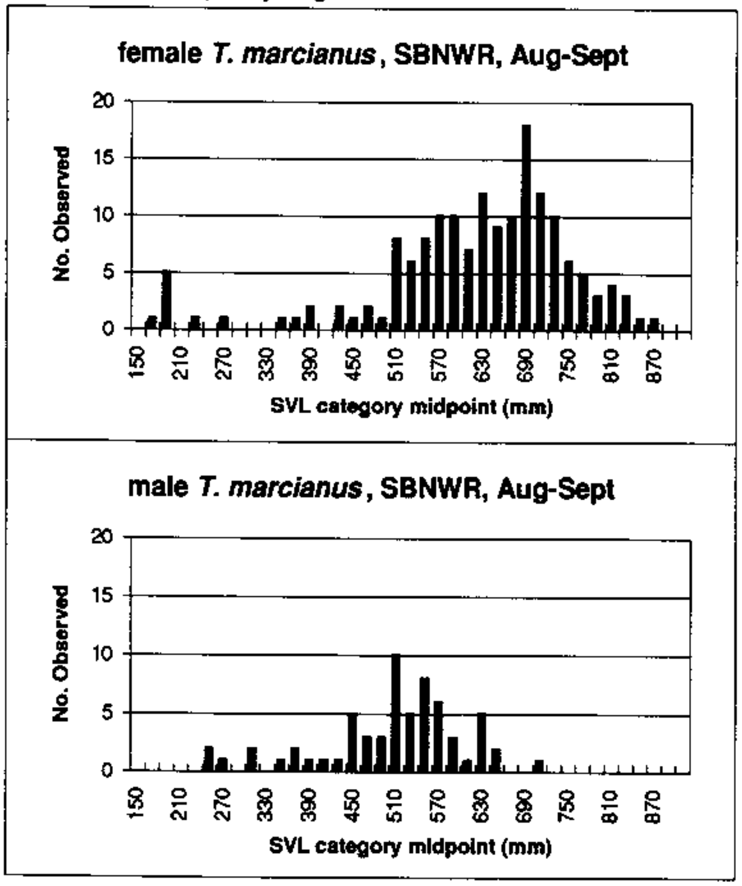


Fig. 8D. Size-frequecny diagrams for male and female *T. marcianus*



Proceedings of
A Symposium on Environmental, Economic, and Legal Issues Related to Rangeland Water Developments

November 13-15, 1997
Tempe, Arizona

Sponsored by:

The Center for the Study of Law, Science and Technology
Arizona State University

Arizona Game and Fish Department
Arizona Wildlife Federation
ASU School of Planning and Landscape Architecture
Sierra Club, Grand Canyon Chapter
The Wildlife Society, Arizona State Chapter
U.S.Bureau of Land Management
U.S. Fish and Wildlife Service
U.S. Forest Service

Published by: The Center for the Study of Law, Science and Technology, Arizona State University, 1998.

To order additional copies contact: Rosalind Pearlman, Arizona State University, College of Law, P.O.Box 877906, Tempe, AZ 85287-7906, (602) 965-2124, rosalind.pearlman@asu.edu.

USING MANAGED WATERS FOR CONSERVATION OF THREATENED FROGS

Philip C. Rosen[1] and Cecil R. Schwalbe[2]

[1]Department of Ecology and Evolutionary Biology, University of Arizona, Tucson, Arizona 85721. [2]U.S. Geological Survey, Cooperative Park Studies Unit, University of Arizona, Tucson, Arizona 85721

ABSTRACT.

Native aquatic animals, especially frogs and fishes, have declined dramatically in the American West. Many are already threatened or endangered, and many continue to decline. One cause of the declines is predation and competition by introduced (non-native) species. Such exotics are difficult to control in natural waters, which tend to be large and complex. Man-made ranch ponds offer alternatives that are typically small, isolated from other waters, and manageable by drying or supplementation with wells. Wells may create the stable water level that some species need; conversely, some natives thrive under the fluctuating water conditions seen in most cattle tanks. Finally, introduced species may be controllable in tanks by removal, chemical treatment, or by drying the tank.

Ranch ponds can be refugia for native species that thrive and reproduce in still or slow-moving waters, including some ranid frogs (the "true frogs"), native fishes, and even invertebrates. It is possible to form public-private partnerships to manage native species in conjunction with livestock operations. Public partners (university and government biologists and managers) can provide expertise and resources. Private partners can provide knowledge of landscapes and existing populations, and can also function as on-site wardens. Both wildlife and livestock may benefit from enhanced quality and landscape distribution of waters. Thus, funding may be available for ranch-based biodiversity conservation.

We describe work with the Chiricahua leopard frog in the San Bernardino Valley of southeastern Arizona. We established a public-private partnership of university, state, and federal personnel, a ranching organization, a local high school, and an individual ranching family with a strong interest in conservation. Over 5 years we enhanced two existing frog populations, and founded new populations at 5 sites. Winter 1996-7 saw an unexplained decline at several sites, and our collective efforts appear to have prevented the disappearance of this species in the local management region.

This has led to a broader attempt to enhance Chiricahua leopard frog populations in the region. The objective is to secure enough populations that the species would be "recovered" (legally and ecologically) in the area. The declines we observed underscore the need to restore metapopulations, and, further, point to the importance of regular monitoring of populations. Such monitoring will assist in understanding and resolving causes of decline. While efforts like ours ought not substitute for restoration of natural waters, they may be the best hope for some species while we seek solutions to problems in the broader environment and in natural habitats.

I. Introduction.

One of the most pressing conservation problems in the American West, and particularly in the Southwest, is the decline of native fishes (see Minckley and Deacon, 1991) and amphibians (Vial and Saylor, 1993; M. Jennings, 1995; Sredl and Howland, 1995; Sredl et al., 1997a). Both habitat degradation and loss are occurring, but within existing habitat the presence of non-native predators and competitors appears to be a major factor affecting natives (Moyle et al., 1986; Minckley and Deacon, 1991; Rosen et al., *submitted*). The aquatic vertebrate fauna of the Southwest in particular appears to be in a broad and serious decline.

Declines of native "true", or ranid (genus *Rana*), frogs in the American West are now known to be caused by a variety of introduced species, including predatory fishes of many kinds (Hayes and Jennings, 1986; Bradford, 1989; Rosen et al., *submitted*), bullfrogs (Hammerson, 1982; Schwalbe and Rosen, 1988; Rosen and Schwalbe, 1995), and crayfishes (Fernandez and Rosen, 1996). These exotic species are now very widespread in Southwestern landscapes (e.g., Rosen et al., 1995), as well as elsewhere. It may be generally difficult to eliminate exotic species from complex natural systems (Rosen and Schwalbe, 1996a, 1997) because these systems have spatial and structural diversity that makes complete removal difficult, and because it can be difficult or impossible to de-water them as a means of exotic species removal.

The Chiricahua leopard frog (*Rana chiricahuensis*) is a native southwestern species that is in decline (Clarkson and Rorabaugh, 1989; Sredl et al., 1997a) and is likely to be listed as federally threatened in the near future (J. Rorabaugh, personal communications). Like some other native ranids, it is often abundant in cattle watering ponds in Arizona (Sredl and Saylor, this volume). It may persist in stock ponds that are isolated from sources of exotic species introductions or, in lower numbers, in ponds where exotics are eliminated by infrequent and temporary drying (Rosen et al., 1995; personal observations). Leopard frogs do not appear to tolerate prolonged habitat drying (personal observations), although there has been no systematic attempt to document or quantify this. The Chiricahua leopard frog has been eliminated from most natural waters by exotic species, habitat drying, and possibly by air pollution (Hale et al., 1995). In the current situation, metapopulations (Sjögren, 1991; Bradford et al., 1993) of frogs are disrupted, and recolonization of drought-impacted sites may now be infrequent or impossible.

One solution to the problem of conserving native wetland species that are affected by exotic species may be management in waters where the exotics are controllable (Rosen, 1997). The ideal situation would be one including the option of controlling the water, including drying as well as using well or spring water to stabilize water level, and one in which the habitat is not so complex that removal of unwanted exotics is difficult. The possibility that cattle ponds may meet these criteria, and may have other advantages for native frog conservation, is explored in this paper. We present the example of our attempt to implement this idea, and conclude with a general description of how such projects might best be carried out.

II. Methods.

The Study System and Human Resources.

The lower San Bernardino Valley is a Chihuahuan desert grassland region in extreme southeastern Arizona that has historically supported Chiricahua and lowland (*R. yavapaiensis*) leopard frogs at several localities and in substantial numbers (Rosen and Schwalbe, 1988, 1996; Clarkson and Rorabaugh, 1989). San Bernardino National Wildlife Refuge (SBNWR) is on the valley floor at a rich riparian site with many springs and wells that support re-introduced native fishes, and where leopard frogs were formerly abundant (Lanning, 1981) but were apparently driven to extinction in the mid 1980's by the introduced bullfrog (Rosen and Schwalbe, 1995, 1996a). Other sites in the area have been increasingly affected by drought, in some cases also leading to recent elimination of populations (Rosen et al., *submitted*, and personal observations).

The refuge has served as a catalyst and was the early focal point of management efforts, primarily involving attempts to eradicate or control the bullfrog. These efforts have proven difficult (Rosen and Schwalbe, 1996b), in part due to the bullfrog's ecological resilience, and in part because of habitat complexity. Further, the primary mission of the refuge is native fish conservation, and this makes it difficult to use pond desiccation as a method to control bullfrogs. As a result, leopard frogs have not re-invaded the refuge successfully, and other species of wetland herpetofauna have continued to decline (Rosen and Schwalbe, 1997). The refuge has nonetheless been the focal point for the evolution of a coalition of university and agency researchers, land managers, and private ranching interests that has become active in leopard frog conservation and management in the study system.

Another important human factor contributing to the possibility of pursuing active management in the study system is the presence of a number of progressive ranchers who are organized into the Malpai Borderlands Group (McDonald, 1995). The objectives of this group are sustainable ranching within an ecosystem management framework that includes biological conservation and preservation of open space. The Magoffin Ranch, just east of SBNWR has a strong interest in conservation of endangered species of plants and animals on their active cattle ranch. Matt Magoffin is also a refuge officer at SBNWR, and he and his family have played a leading role in leopard frog conservation efforts on private lands in the area.

The research and management efforts leading to the conservation work described here originated with status surveys on public and private lands in the mid-1980's (Rosen and Schwalbe, 1988) supported by Arizona Game and Fish Department (AGFD) and U.S. Fish and Wildlife Service (USFWS). The AGFD then continued to play a key role in organizing, and, along with USFWS, subsequently funding experiments on bullfrog control at SBNWR. This early work attracted the interest and participation of personnel from other agencies as well as private individuals, a number of whom have played key roles as the broader conservation project developed. This broader, funded project expanded to include the leopard frog conservation efforts described in this paper.

Early Development of the Conservation Program.

The conservation program was not pre-designed or envisioned originally as a grand scheme. Rather, it developed from the bullfrog management work as opportunities and conservation crises arose. The bullfrog control program was initially established to attempt recovery of impacted populations of the Mexican garter snake (*Thamnophis eques*) at SBNWR (Rosen and Schwalbe, 1995), which is also affected by bullfrog predation on the smaller age classes of snakes. We recognized that eliminating bullfrogs as predators of the snake would also eliminate young bullfrogs, a primary food source for the snake population. This would leave the diet deficient in ranid frogs, its key component, unless native frogs could also be recovered. During this phase of the work, we observed the disappearance of leopard frogs from the refuge, and recognized the role bullfrogs were playing in the range-wide decline of the Chiricahua leopard frog and other native ranid species.

We confirmed that leopard frogs were, in fact, extirpated at SBNWR, but remained present at Magoffin Ranch. A drought that began in the mid-1980's and intensified in 1993 presented the opportunity to rescue tadpoles from a site on Magoffin Ranch that was drying, and to attempt to establish them at SBNWR under our regime of bullfrog removal. This effort mobilized the coalition of Magoffin Ranch, University of Arizona researchers (the authors), AGFD Nongame Branch, and USFWS to develop formal protocols and funding sources for active management of leopard frogs in the area.

The year 1994 saw a further intensification of drought in southeastern Arizona. At that time we began active management that has been ongoing since. It has included hauling water, adding of wells, digging of new and refurbishment of existing ponds, and construction of a concrete drought-refugium pool at a tank, all at Magoffin Ranch; establishment of a bullfrog exclosure for re-introduced leopard frogs, and erection of a small screened-in facility (a "ranarium") for frogs at SBNWR; and extension of leopard frog tadpole husbandry to a classroom and outside pond at nearby Douglas High School. In so doing, frogs were established at five new sites, and conservation efforts were also undertaken at the two remaining natural population sites in the area, both on Magoffin Ranch.

Specific Aspects of the Conservation Program.

From 1994, we developed the program in a relatively organized and planned fashion, although we also proceeded along serendipitous or necessary paths as these presented themselves.

Sources of tadpoles for reintroduction. Any frog conservation program must attempt to minimize its impact on existing populations. It is imperative that source populations not be seriously harmed by removal of large numbers of adults or tadpoles. We have attempted to solve this problem by (1) using tadpoles that were about to perish by desiccation at Magoffin Ranch in the 1994 drought, (2) harvesting partial egg masses from managed populations for rearing at the Douglas High School, and (3) utilizing tadpoles that were overcrowded in the wild managed population sites. In general, captive rearing and propagation are by far the most desirable approaches unless circumstances by chance afford an abundance of harvestable tadpoles.

Tadpoles or newly metamorphosed frogs have been exclusively selected for translocation. Although egg masses can be safely transported with a modicum of care (Fernandez, personal communication), hatchling and very small tadpoles are likely to suffer high natural mortality. Rearing tadpoles is quite feasible, transporting them in aerated water by vehicle is uncomplicated as long as temperature extremes are avoided, and we believe growth to a total length of 25 - 50 mm (1-2 inches) greatly improves survival probability. Tadpoles are vulnerable at metamorphosis, and thus should be released well before that, or afterwards. We suspect that translocated older frogs may be likely to leave an introduction site, or may lose the advantages they have acquired through learning at their original locality.

Initial translocation of tadpoles. On June 10, 1993, 789 large *R. chiricahuensis* tadpoles were harvested by seine from the smaller of two drying pools in Rosewood Tank, Magoffin Ranch. They were immediately transported 11 km east to SBNWR and released in seven wetland sites, where bullfrog removal was underway, representing the full range of habitat conditions on the refuge. Trained personnel periodically searched the refuge for survivors of this experiment, beginning in September 1993.

Rearing of tadpoles. Tadpoles have been reared at Douglas High School as part of the science education program, and at SBNWR headquarters on a temporary basis. The tadpoles were collected as portions of three egg masses from Choate Tank (see below) in fall 1996, and as one entire egg mass from the SBNWR re-introduced population enclosure (below) in fall 1997. The have generally thrived on a diet of thawed frozen spinach leaves, with approximately 26% survival from hatching to release prior to metamorphosis. Rearing was in 15 gallon aquaria with aged (de-chlorinated) water with filtration. Better results may be obtained by feeding a combination of spinach, with finely sliced fish meat, and sliced cucumber or zucchini (M. Demlong, personal communication).

Use of outdoor enclosure. The active involvement of SBNWR has permitted the successful development of a leopard frog population in the bullfrog exclosure (Fig. 1). The exclosure consists of four small (4 X 6 m, 1-2 m deep), artesian well-fed pools. The water level of the pools is regulated by float valves, and they are surrounded by a 1 m tall hardware screen ("hardware cloth") fence (1/8 inch mesh) extending 15 cm into the ground and enclosing a terrestrial area of nearly one hectare. It was constructed in late winter 1994, and 188 tadpole stage and small juvenile Chiricahua leopard frogs were introduced in May-June 1994.

We were quickly confronted by several unexpected complications in the management of this enclosed leopard frog population. The shallow open habitat populated by a high density of relatively naive frogs immediately attracted green-backed herons (*Butarides striatus*) and great blue herons (*Ardea herodias*) in numbers. The pools were therefore covered with 10 cm mesh predator netting, which effectively eliminated avian predation on frogs in the enclosure. Then, following the first rains in July 1994, scores of small juvenile bullfrogs entered the exclosure by climbing the hardware screen; presumably, numerous leopard frogs also departed, and these were probably consumed by larger bullfrogs that remained abundant elsewhere at SBNWR.

The immigrant bullfrogs were removed with some effort, and hardware screen eaves (Fig. 1) were installed on the enclosure fence. Nonetheless, a small number of juvenile bullfrogs continued to enter the enclosure, requiring regular monitoring and removal to prevent them from maturing and reproducing. Juvenile checkered garter snakes (*T. marcianus*, all removed) also penetrated the fence, as did kingsnakes (*Lampropeltis getula*). The snakes probably gained access through tunnels constructed by rodents, but the bullfrog entry points are not apparent. They may be able to squeeze through occasional irregularities between the eaves and the top of the fence, suggesting that eaves should be constructed by folding the top of the fence outward.

Use of ranarium. The ranarium (Fig. 2) was built in 1996 as a holding and breeding facility at SBNWR, based upon the design of Fernandez (1996). It differs from the original design as follows: (1) it is covered with hardware screen rather than fiberglass, (2) does not require artificial cooling, and (3) has recirculating water treated by ozonation. Ten adult Chiricahua leopard frogs from a drying puddle at a road crossing in Hay Hollow in August 1996 were released in the ranarium, and 6 tadpoles from Douglas High School were added in May 1997.

Management of cattle pond environments. *I. Hauling water.* Because of the critical situation during the 1994 drought when this project began, it was desirable to maintain some frogs at the originally strongest population locality, Rosewood Tank on Magoffin Ranch. This was a hedge against failure of tadpoles moved to the new enclosure at SBNWR. Immediately prior to drying of the tank in early April 1994, a small pool (3 X 3 m, 1.8 m deep) was excavated on an elevated bench of the tank bottom and filled with well water. Tadpoles (n = 400) were harvested from puddles in the tank bed and split evenly between SBNWR and the Rosewood refugium pool. The pool was fertilized lightly with cow manure and inoculated with pondweed (*Potamageton* sp.) from the original tank. Cattle were denied access to this pool throughout its existence.

It was necessary to haul approximately 1000 gal of water per week to sustain water levels in this pool. Poor rains over this portion of the Hay Hollow drainage during the remainder of this study necessitated continued water hauling until 1997, a task enjoyed by the Magoffin family at considerable expense.

II. Refurbishing stock ponds. Rosewood Tank, like all simple stock ponds, had over the years filled with in-washing sediment. All runoff-fed stock ponds can be expected eventually to fill with sediment and become increasingly less likely to hold surface water permanently. Rosewood Tank dried briefly in 1989, and nearly dried in 1993, prior to the early and prolonged drying in 1994. It is presumed that the 1994 drying would have killed all leopard frogs in the population, as no runoff occurred in the drainage until September 1994.

Matt Magoffin repaired and re-designed Rosewood Tank from July 1994 - March 1995, first deepening it, and then adding a sediment trap and new overflow system. The sediment trap system he recommended (Fig. 3) is a key feature allowing the use of runoff ponds for conservation. This design permits sediment to accumulate in the trap, which also serves as a pond, while smaller quantities of finer particles are deposited in the main pond. As the sediment trap fills, it will hold less water but deliver more of the runoff to the main tank. The trap can be re-excavated with a bulldozer when it is thoroughly dry, on the presumption that aquatic amphibians and reptiles will be out of harm's way in the main pond. Alternately, the sediment trap may serve as a temporary pond or holding site for aquatic organisms during re-excavation of the main pond or some other management action requiring drying of the main pond (such as removal of an exotic species).

Rosewood Tank was excavated in two stages. First, two deep lobes were excavated around the perched refugium pool. Eventually, the refugium pool began to leak, and a concrete pool was installed just above the head of the tank (see below), and then the excavation was competed. A sediment trap was also added at Belency Tank, Magoffin Ranch, in 1994, at the other existing population site for *R. chiricahuensis* in the study system.

III. Creation of new pond population sites. Using existing wells, the Magoffins constructed small, non-runoff filled ponds at sites above Belency (Choate Tank, 1995) and below Rosewood (Headquarters Pond, 1997). These ponds were approximately 10 m in diameter and 1.5 m deep, perched on self-sealing clay substrata, and inoculated with pondweed. Choate was stocked with 66 under-sized tadpoles from the Rosewood refugium pool in late summer 1995. Headquarters Pond was stocked with 41 tadpoles from Douglas High School in May 1997; these were expected to be supplemented by immigration from Rosewood Tank at 3 km distance, as *R. chiricahuensis* normally appear at headquarters during very wet episodes (M. Magoffin, personal communications).

IV. Establishment of new wells at existing ponds. In order to guarantee a minimum of water for the native ranid frogs on-site, it was considered desirable to have wells to supplement runoff. Wells were installed with outside funding at Belency Tank (1995; funding from Malpai Borderlands Group) and near Rosewood Tank (1997; funding from AGFD Stewardship Program). The procedure was (1) conceptual review by university and agency herpetologists, (2) preparation of proposed plans for the wells by Magoffin Ranch, (3) evaluation of hydrology and engineering feasibility by funding agency, and (4) construction and contractor supervision by Magoffin Ranch. The Belency Well was placed within 100 m of the pond. The Rosewood Well was 3 km from the tank, where groundwater was accessible and storage and delivery to Rosewood and other sites was feasible.

V. Construction of concrete drought-refugium pool. The persistent drought at Rosewood Hollow and difficulties associated with long term water hauling encouraged the building of a 3 X 4 m, 1.3 m deep concrete pool (Fig. 4) as part of the AGFD Stewardship Program project. This pool was installed just above the upper end of the inundation area of Rosewood Tank, and fenced to exclude cattle. The new well supplied water to the pool as well as to a separate steel tank serving as a livestock drinker, both of which were regulated by float valves. Five adult *R. chiricahuensis* were removed from the small puddle remaining in Rosewood Tank, and released in the pool in March 1997.

Predator removal. In addition to removal of bullfrogs (n=43) and checkered garter snakes (n=9) from the refuge leopard frog enclosure, we removed checkered garter snake from the Rosewood refugium pool (n=4), Belency Tank (n=25), Choate (n=2), and Headquarters (n=2). With the exception of the constrained situation at the Rosewood refugium pool (one large individual removed from this site consumed 180 ranid frogs in two years in captivity), we now regard these removals as questionable. Predator-prey dynamics are not understood in this system, and the effect of removing all or part of one class of predators (i.e., the snakes) is not clear.

Monitoring. Monitoring was accomplished by counting frogs during careful examinations of wetland sites. Additionally, data was recorded on egg masses, tadpoles, and unusual events including predator presence and abundance and appearance of dead frogs. The leopard frog enclosure at SBNWR was monitored regularly (9 - 28 times per year) by refuge and university personnel. The ranarium was monitored regularly by refuge personnel. Magoffin Ranch wetland sites were observed frequently by Matt Magoffin, with reporting to university personnel, and were monitored 1-3 times per year by university and AGFD herpetologists. Observation and monitoring were thorough searches using binoculars during the day and bright lights at night; attempts were made to locate and count every visible individual. All leopard frog egg masses observed were reported to university monitoring personnel.

II. Results.

Population status in the study region.

SBNWR. The initial attempt to re-establish leopard frogs on the refuge by releasing at sites where bullfrogs were being removed yielded no evidence of success. Ongoing monitoring from 1993-1997 never revealed the presence of any leopard frogs on any of the unfenced sites on the refuge.

Chiricahua leopard frog tadpoles and small juveniles released into the SBNWR bullfrog exclosure in spring 1994 grew quickly to adult size and were observed breeding in September 1994. Breeding (calling males and subsequent egg masses and tadpoles) was observed during each fall 1994-1997. The enclosure population increased rapidly in 1995 and 1996 (Fig. 5). In fall 1996, a dead frog was found in the enclosure, and the population declined sharply between fall 1996 and early spring 1997. Two egg masses were observed at the site in September 1997, one of which was removed to Douglas high school for rearing.

The ranarium population was largely unsuccessful. Two of the original 10 adults died without evident cause in November 1996, and six more died in March 1997. In October 1997, the last two died, along with all but one of the juveniles derived from releases of reared tadpoles. These frogs had acquired a severe epidermal fungal mold disease (Chytridiomycota; G. Bradley, University of Arizona Veterinary Diagnostic Laboratory, personal communication) which appeared to be the proximate cause of death. Whether this disease was triggered by some other, ultimate factor or cause remains unknown.

Magoffin Ranch. Southeastern Arizona has experienced generally poor summer rainfall, generally essential for filling runoff-fed ponds, since 1988. Summer 1996 was an exception, but the Magoffin Ranch area generally remained extremely dry during all years 1994-1997. Rosewood Tank and all of Rosewood Hollow had poor rains and meager runoff throughout the period. Belency Tank and surrounding Belency Hollow flooded in summer 1996, and received moderate runoff in 1997. Other tanks on the ranch, which hold water for generally shorter time periods and support immigrant individual leopard frogs, never filled during 1994-1997, and puddles in lower Rosewood Hollow and Hay Hollow, which also are occupied by leopard frogs, filled only once, in summer 1996.

Rosewood Tank. The large population at Rosewood (with hundreds, and probably thousands of tadpoles in 1993) was rapidly reduced from spring to summer in 1994, but remained remarkably stable during 1995-1997 (Table 1). By 1997, almost all frogs seen at Rosewood were very large adults, well over 100 mm snout-vent length.

Table 1. Trends in adult leopard frogs observed per monitoring count, and total egg masses observed, at Rosewood Tank, Magoffin Ranch.

Time Period	Adults (mean ± SE)	Egg Masses (total observed)
Spring 1994	11.6 ± 2.38	0
Summer 1994	4.3 ± 0.64	1
Sp-Sum 1995	2.2 ± 0.71	2
Sp-Sum 1996	1.8 ± 0.75	0
Sp-Sum 1997	2.9 ± 0.78	5
March 20, 1998	3	4

Egg masses were observed in each year except 1996 (Table 1), and we assume that other egg masses were present but unobserved. Tadpoles were metamorphosing in the Rosewood refugium pool in substantial numbers in 1994. In 1995, 66 tadpoles from spring egg masses were removed and transferred to newly established Choate Tank, and in 1996 very young tadpoles were observed in May at Rosewood, and a few small juvenile froglets were observed. During August 1996, up to 25 young adult and subadult Chiricahua leopard frogs were found at a roadside puddle 3.5 mi (5.5 km) below Rosewood Tank; there is apparently no possible source for these frogs except Rosewood and Belency Tanks on Magoffin Ranch, and it thus appears that some of the 1994-1996 breeding at the Rosewood refugium pool was successful.

In March 1997, five large adults were transferred from the drying bottom of Rosewood Tank to the new concrete refugium pool at the upper margin of the tank, and two egg masses were immediately deposited there. No tadpoles were observed. In late summer, 2 fertilized egg masses completely failed in the pool; pH was 10.8, although the well water filling the pool was pH 8 at the inlet. The pool was drained and re-filled, revealing 2 adult frogs and a third egg mass, which also failed. Alternative concrete and sealing processes, or a longer curing period prior to adding water, may have prevented the egg mortality. Eggs appear to be hatching successfully in spring 1998, as of this writing.

Belency Tank. Belency Tank was constructed in approximately 1970, and had very rarely gone dry in the succeeding two decades, including briefly in 1989 (M. Magoffin, personal communications). The water level declined to less than 0.5 m maximum depth in spring 1995, and the tank would apparently have dried without the newly installed well. The site appeared to support the most robust and secure population of Chiricahua leopard frogs in the region in the early 1990's. Yaqui chubs (*Gila purpurea*) were introduced to the tank in the early 1990's; no evidence of fish reproduction was found, and only a small number of larger adults were detected by seining during low water times in 1994 and 1995.

Table 2. Monitoring results for the Chiricahua leopard frog at Belency Tank, Magoffin Ranch.

Time Period	Individuals / census (mean +/- SE)	Egg Masses (total observed)
1994	23.5 ± 6.17	3
1995	13.7 ± 2.46	6
1996	14.2 ± 5.40	0
1997	0	0

It was initially common to observe numerous frogs at Belency Tank (Table 2), with counts ranging to 41 or more adults. Frogs were infrequently seen in spring-summer 1996, although large numbers of toads bred there in July. In August, as many as 50 adult leopard frogs were seen at Belency (M. Magoffin, personal communications), during and after the observation of what we presumed were numerous Rosewood Tank frogs 3.5 mi below Rosewood Tank (see above). In November 1996 a dead adult was found in the sediment trap; no frogs have been seen at the site since, despite repeated searches. In 1997, the sediment trap and main tank filled with water on July 31, but very few toads and no leopard frogs were observed at the site through the remainder of the year.

During 1994 and 1995, egg masses were seen in the tank, and in each case appeared to hatch normally. Despite the substantial number of frogs, we were never able to capture or observe many tadpoles, and we never observed a flush of metamorphosing juveniles, as expected in times of successful reproduction. It appears that this population has been extirpated. We can not explain the population failure at Belency Tank.

Other ponds on Magoffin Ranch. Choate Tank was constructed in spring 1995, and stocked with 66 tadpoles from Rosewood Tank in August-September 1995. In May 1996, 25 subadults were observed, but 15 were found dead when the pump failed and the water appeared to become anoxic. Seven to twelve frogs per monitoring census were observed thereafter, and 3 egg masses were found in fall 1996. Each of the egg masses was divided in half, with half left at Choate and the other half brought to Douglas High School for rearing. All three partial egg masses removed hatched successfully, yielding approximately 290 tadpoles. In December 1996, a moribund frog was seen at Choate. No more frogs were observed there until late summer and fall 1997, when two small adults were seen.

Magoffin Headquarters Pond was constructed in winter 1997, and stocked in May with 41 large tadpoles from Choate that were reared at Douglas High School. Two juveniles were observed in the pond in early August, and one small adult was regularly observed during early fall; a leopard frog egg mass was found hatching at this pond on March 12, 1998 (M. Magoffin, personal communications).

The Headquarters Pond was constructed in Rosewood Hollow at Magoffin Ranch headquarters, 2 mi (3 km) below Rosewood Tank. Leopard frogs were often seen in small pools at the site during wet years, including 3 in 1996, and large numbers of toads (*Bufo* spp.) and spadefoot toads (*Scaphiopus* spp.) normally bred there (M. Magoffin, personal communications). In 1997, only 4 spadefoot toads (*S. couchii*) were heard at the site (M. Magoffin, personal communication), but we found up to 7 juvenile checkered garter snakes at once, foraging in the new pond. The fate of the leopard frog populations at Choate and Headquarters remain uncertain.

Douglas High School. A small fenced pond was constructed for frogs on the high school property in winter 1997, and 27 of the reared tadpoles were released there in late May 1997. During August and September 1997, at least one frog plus several large tadpoles persisted at the pond (H. Bodenhamer, personal communication), and the site remained under observation.

III. Discussion

The leopard frog management program appeared to be an un-mitigated success during summer and early fall of 1996, but by mid-1997 it was evident that some unexpected effect or set of causes had produced population declines at almost every site (Fig. 6). What had seemed to be the strongest remaining naturally occurring population, at Belency Tank, disappeared; other populations also declined sharply, although they persisted. We have been unable to locate the underlying causes of these declines (Rosen and Schwalbe, 1997), although they resemble unexplained declines seen at certain other sites in the region (Scott, 1992; R. Jennings, 1995; Sredl et al., 1997b; Rosen and Schwalbe, 1998 and personal observations).

While these unexplained declines are discouraging, they do not indicate that our conservation efforts have been a failure. The opposite is true. In 1993, the Chiricahua leopard frog existed at three sites in the region: Rosewood Tank, Belency Tank, and Guadalupe Canyon. The 1994 drought eliminated the frogs in Guadalupe Canyon, and would have eliminated the Rosewood population without our efforts. The unexplained die-offs in 1996-7 eliminated the last natural population, at Belency Tank. The nearest Chiricahua leopard frogs would apparently be at Leslie Canyon and Limestone Mountain, 35 km distant, and probably beyond the range of natural recolonization under current ecological conditions. The latter populations are isolated and potentially at risk, as are most, if not all remaining populations of this species in the United States.

Without our active efforts, the Chiricahua leopard frog metapopulation in the study region would be gone. As of this writing, the species is known to have reproduced successfully at one site in the study region in fall 1997, tadpoles are present at least two places, and the species remains at five sites. Additionally, tadpoles from one clutch of eggs are being reared in captivity.

It remains to be seen whether our efforts ultimately succeed. Diversifying the population base within the metapopulation was initially a strategy to hedge against extinction, and one we thought, apparently incorrectly, was more than what was necessary. Our results thus far have supported the idea that introduced species, in this particular case the bullfrog, prevent re-establishment of native leopard frogs. They also tend to validate such general axioms of conservation biology (e.g., Meffe and Carroll, 1997) as the desirability or need to work with local people and to make conservation compatible with sustainable local economies. Our experience further clearly underlines the importance of careful monitoring. The more we know about the managed populations in these early efforts, the better. It will most likely be populations under such intense scrutiny that will finally provide the insight needed to decipher the unexplained declines now apparently plaguing Southwestern leopard frogs.

ACKNOWLEDGEMENTS

The support, advice, and direct participation of SBNWR Refuge Manager Kevin Cobble, the Magoffin family (Matt, Anna, Chris, and Mike), and Hans Bodenhamer have been central to the functioning and success of the project. The AGFD Nongame Branch contributed both field work and consultative advisement, and we especially thank Pat Collins, Jeff Howland, Mike Sredl, and Eric Wallace for their efforts.

Many have assisted importantly with the field work and advisement throughout this project, and we especially acknowledge the extensive contributions of Andrew, Cindy and Ted Cordery; Steve Hale; Paul Langdale; Robin Llewellyn; Verma Miera; Shawna Nelson; David Parizek; Shawn Sartorius; Myles Traphagen; and Tony Velasco.

Our work on this project has been supported by funding from AGFD Heritage Grants Program, USFWS, and USDA Forest Service. Additional funding for the leopard frog management activities by SBNWR was provided by USFWS. Funding for wells and pond management on Magoffin Ranch was supplied by the Malpai Borderlands Group and by AGFD through a Stewardship Program grant.

Literature Cited

Bradford, D. 1989. Allotopic distribution of native frogs and introduced fishes in high Sierra Nevada lakes of California: implication of the negative effect of fish introduction. Copeia 989:775-778.

Bradford, D.F., F. Tabatabai, and D.M. Graber. 1993. Isolation of remaining populations of the native frog, Rana muscosa, by introduced fishes in Sequoia and Kings Canyon national parks, California. Conservation Biology 7:882-888.

Clarkson, R.W., and J.C. Rorabaugh. 1989. Status of leopard frogs Rana pipiens complex: Ranidae) in Arizona and southeastern California. SW Nat. 34:531-538.

Fernandez, P.J. and P.C. Rosen. 1996. Effects of the introduced crayfish *Orconectes virilis* on native aquatic herpetofauna in Arizona. Final Report to Arizona Game & Fish Dept. Heritage Program. 56 pp + appendix.

Hammerson, GA. 1982. Bullfrogs eliminating leopard frogs in Colorado? Herp. Review 13:115-116.

Hale, S.F., C.R. Schwalbe, J.L. Jarchow, C. May, C.H. Lowe, and T.B. Johnson. 1995. Disappearance of the Tarahumara frog. Pp. 138-140 *in* Laroe, E.T., G.S. Farris, C.E. Puckett, P.D. Doran, and M.J. Mac (*eds.*), Our Living Resources: A Report on the Distribution, Abundance, and Health of U.S. Plants, Animals, and Ecosystems. U.S. Dept. of Interior, National Biological Service, Washington, D.C. 530 pp.

Hayes, M.P. and M.R. Jennings. 1986. Decline of ranid frog species in western North America: are bullfrogs (*Rana catesbeiana*) responsible? J. Herpetol. 20:490-509.

Jennings, M. 1995. Native ranid frogs in California. Pp. 131-134 *in* Laroe, E.T., G.S. Farris, C.E. Puckett, P.D. Doran, and M.J. Mac (*eds.*), Our Living Resources: A Report on the Distribution, Abundance, and Health of U.S. Plants, Animals, and Ecosystems. U.S. Dept. of Interior, National Biological Service, Washington, D.C. 530 pp.

Jennings, R.D. 1995. Investigations of recently viable leopard frog populations in New Mexico: *Rana chiricahuensis* and *Rana yavapaiensis*. Unpublished report to New Mexico Department of Game & Fish, Endangered Species Program, Santa Fe, NM. 36 pp.

Lanning, D.V. 1981. The vertebrates of San Bernardino Ranch, Cochise County, Arizona. Unpublished report to The Nature Conservancy and U.S. Fish & Wildlife Service. ii + 88 pp.

McDonald, B. 1995. The formation and history of the Malpai Borderlands Group. Pp. 483-486 *in* L.F. DeBano, P.F. Ffolliott, A. Ortega-Rubio, G.J. Gottfried, R.H. Hamre, and

C.B. Edminster (*tech. coords.*), Biodiversity and Management of the Madrean Archipelago: the sky islands of southwestern United States and northwestern Mexico. Gen. Tech. Rep. RM-GTR-264. Fort Collins, Colorado. Dept. Agr., U.S. Forest Serv., Rocky Mountain Forest and Range Experiment Station. 669 pp.

Meffe, G.K. and C.R. Carroll, and contributors. 1997. Principles of Conservation Biology, 2nd edition. Sinaur Associates, Inc., Sunderland, Massachusetts. 729 pp.

Minckley, W.L., and J.E. Deacon, editors. 1991. Battle against extinction: native fish management in the American West. University of Arizona Press, Tucson. 517 pages.

Moyle, P.B., H.W. Li, and B.A. Barton. 1986. The Frankenstein Effect: Impact of introduced fishes on native fishes in North America. Pp. 415-426 *in* R.H. Staub (*ed.*), Fish Culture in Fisheries Management. American Fisheries Society, Bethesda, Maryland.

Rosen, P.C. 1997. Causes of declines--opportunities for restoration. Bajada 5:12.

Rosen, P.C. and C. R. Schwalbe. 1988. Status of the Mexican and narrow-headed gartersnake (*Thamnophis eques megalops* and *Thamnophis rufipunctatus rufipunctatus*) in Arizona. Unpubl. report from Arizona Game and Fish Dept. (Phoenix, Arizona) to U.S. Fish and Wildlife Service, Albuquerque, New Mexico.

Rosen, P.C. and C.R. Schwalbe. 1995. Bullfrogs: introduced predators in southwestern wetlands. Pp. 542-454 *in* Laroe, E.T., G.S. Farris, C.E. Puckett, P.D. Doran, and M.J. Mac (*eds.*), Our Living Resources: A Report on the Distribution, Abundance, and Health of U.S. Plants, Animals, and Ecosystems. U.S. Dept. of Interior, National Biological Service, Washington, D.C. 530 pp.

Rosen, P.C. and C.R. Schwalbe. 1996a. Bullfrog Impacts on Sensitive Wetland Herpetofauna, and Herpetology of the San Bernardino National Wildlife Refuge. Final Report to Arizona Game and Fish Department Heritage Fund and U.S. Fish and Wildlife Service, June 6, 1996. 47 pp.

Rosen, P.C. and C.R. Schwalbe. 1996b. A critical interim evaluation of the effectiveness of bullfrog removal methods at San Bernardino National Wildlife Refuge. Final Report to Arizona Game and Fish Department Heritage Fund and U.S. Fish and Wildlife Service, October 30, 1996. 21 pp.

Rosen, P.C. and C.R. Schwalbe. 1997. Bullfrog impacts on sensitive wetland herpetofauna, and Herpetology of the San Bernardino National Wildlife Refuge. Final Report to Arizona Game & Fish Dept. Heritage Program, and USFWS. 30 pp.

Rosen, P.C., C.R. Schwalbe, D.A. Parizek, P.A. Holm, and C.H. Lowe. 1995. Introduced aquatic vertebrates in the Chiricahua region: effects on declining native ranid frogs.

Pp. 251-261 *in* L.F. DeBano, P.F. Ffolliott, A. Ortega-Rubio, G.J. Gottfried, R.H. Hamre, and C.B. Edminster (*tech. coords.*), Biodiversity and Management of the Madrean Archipelago: the sky islands of southwestern United States and northwestern Mexico. Gen. Tech. Rep. RM-GTR-264. Fort Collins, Colorado. Dept. Agr., U.S. Forest Serv., Rocky Mountain Forest and Range Experiment Station. 669 pp.

Rosen, P.C., C.R. Schwalbe, and S.S. Sartorius. *submitted.* Decline of the Chiricahua leopard frog mediated by introduced species in southern Arizona.

Schwalbe, C.R. and P.C. Rosen. 1988. Preliminary report on effects of bullfrogs on wetland herpetofauna in southeastern in Arizona. Pp. 166-173 *in* R.C. Szaro, K.E. Severson, and D.R. Patton (editors), Management of Amphibians, Reptiles, and Small Mammals in North America. U.S. Forest Service, General Technical Report RM-166, Fort Collins, CO.

Scott, N.J. Jr. 1992. Ranid frog survey of the Gray Ranch with recommendations for management of frog habitats August 1990-September 1991. Unpublished report to Gray Ranch, Animas, New Mexico. 12 pp.

Sjögren, P. 1991. Extinction and isolation gradients in metapopulations: the case of the pool frog (*Rana lessonae*). Pages 135-147 *in* M.E. Gilpin and I. Hanski, Metapopulation Dynamics. Academic Press, London. 336p.

Sredl, M.J. and J.M. Howland. 1995. Conservation and management of Madrean populations of the Chiricahua leopard frog. Pp. 379-385 *in* L.F. DeBano, P.F. Ffolliott, A. Ortega-Rubio, G.J. Gottfried, R.H. Hamre, and C.B. Edminster (*tech. coords.*), Biodiversity and Management of the Madrean Archipelago: the sky islands of southwestern United States and northwestern Mexico. Gen. Tech. Rep. RM-GTR-264. Fort Collins, Colorado. Dept. Agr., U.S. Forest Serv., Rocky Mountain Forest and Range Experiment Station. 669 pp.

Sredl, M.J., J.M. Howland, J.E. Wallace, and L.S. Saylor. 1997a. Status and distribution of Arizona's native ranid frogs. P. 37-89 *in* M.J. Sredl (*editor*). Ranid Frog Conservation and Management. Nongame and Endangered Wildlife Program Technical Report 121. Arizona Game and Fish Department, Phoenix, AZ.

Sredl, M.J., E.P. Collins, and J.M. Howland. 1997b. Mark-recapture studies of Arizona leopard frogs. Pp. 1-20 *in* M.J. Sredl (*editor*). Ranid Frog Conservation and Management. Nongame and Endangered Wildlife Program Technical Report 121. Arizona Game and Fish Department, Phoenix, AZ.

Sredl, M.S. and L.S. Saylor. 1998. Conservation and management zones and the role of rangeland waters in conserving Arizona leopard frogs on large landscapes. *This volume*.

Vial, J.L., and L. Saylor. 1993. The status of amphibian populations: a compilation and analysis. IUCN/SSC Declining Amphibian Populations Task Force Working Document No. 1. 98 pp.

Author contact numbers/addresses:

Phil Rosen: telephone 520.621.3187, fax 520.670.5001, email pcrosen@u.arizona.edu

Cecil Schwalbe: telephone 520.621.5508, fax 520.670.5001, email cecils@srnr.arizona.edu

FIGURE CAPTIONS.

Fig. 1. A bullfrog exclosure-leopard frog enclosure at San Bernardino National Wildlife Refuge, Cochise County, Arizona. (a) Enclosure fence with eaves to prevent movement of frogs in or out. (b) One of four small, densely vegetated pools utilized by re-introduced Chiricahua leopard frogs within the enclosure.

Fig. 2. Ranarium for holding and breeding frogs under semi-captive conditions.

Fig. 3. Belency Tank, Cochise County, Arizona, exemplifying a sediment trap-main pond system. Run-off water from Belency Hollow enters the sediment trap from the left side of the photograph, fills the trap until it flows through a culvert in the mesquite-lined levee at center to fill the main pond to the right. Once the sediment trap and main pond are full, any further inflow runs around the system on Belency Hollow's natural floor, just beyond the tank system in the photograph.

Fig. 4. Concrete pool (3 X 3.5 m, 1.2 m deep) at Rosewood Tank, Cochise County, Arizona, serving as a refugium during drought times for a population of Chiricahua leopard frogs. This pool is gravity-fed through a float valve from a storage tank filled by a windmill-powered well. Cattle are excluded from the pool, but are served by a nearby steel tank on the same water system.

Fig. 5. Monitoring results for the Chiricahua leopard frog population introduced into the bullfrog exclosure at San Bernardino National Wildlife Refuge.

Fig. 6. Lineage and population history diagram for all known leopard frogs in the study region during 1994-1997. The populations to the right (Guadalupe Canyon, Belency Tank) became extinct. Arrows among the other populations indicate translocations made by resource managers to establish new population sites.

Fig. 1. A bullfrog exclosure-leopard frog enclosure at San Bernardino National Wildlife Refuge, Cochise County, Arizona. (a) Enclosure fence with eaves to prevent movement of frogs in or out. (b) One of four small, densely vegetated pools utilized by re-introduced Chiricahua leopard frogs within the enclosure.

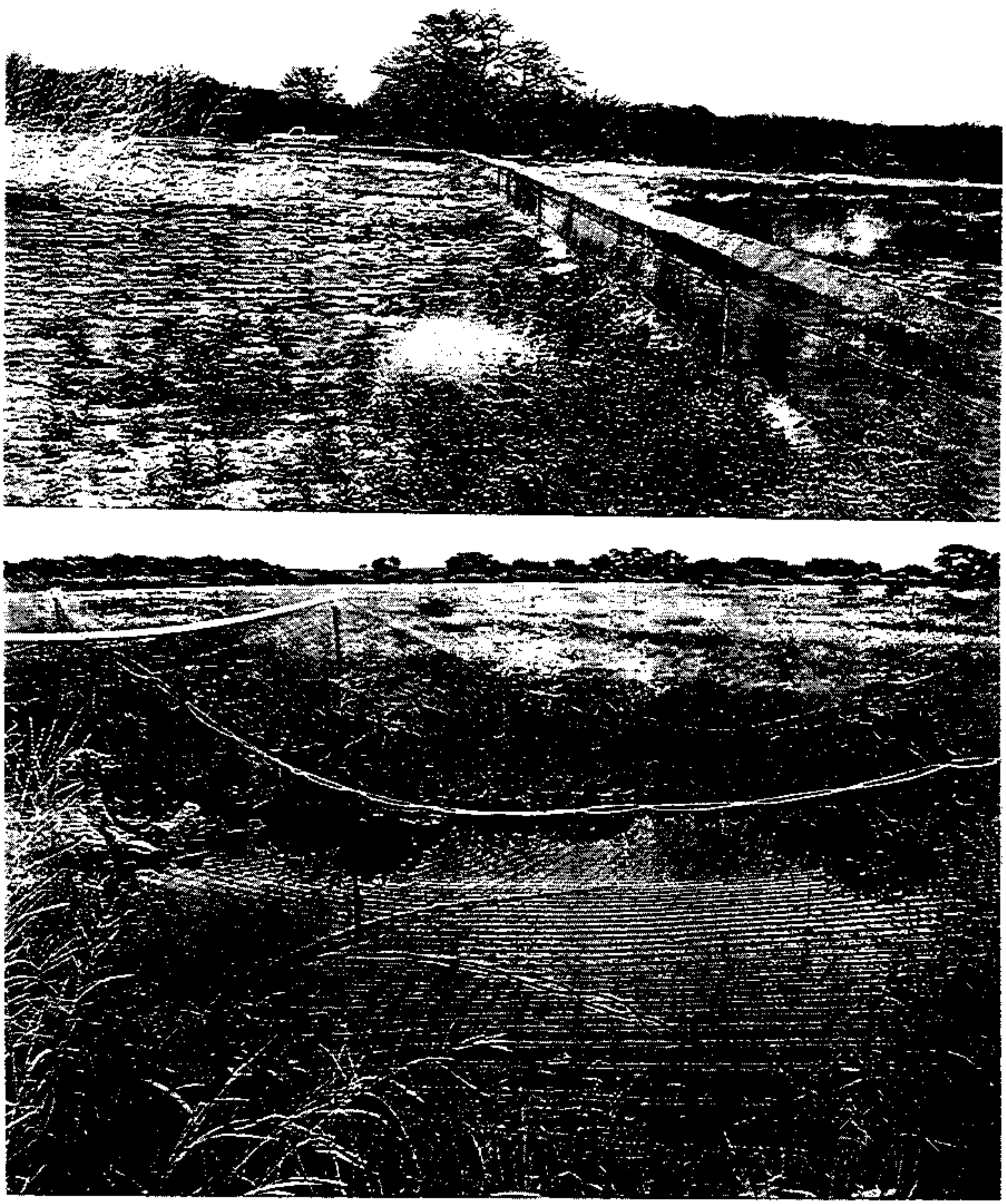


Fig. 2. Ranarium for holding and breeding frogs under semi-captive conditions.

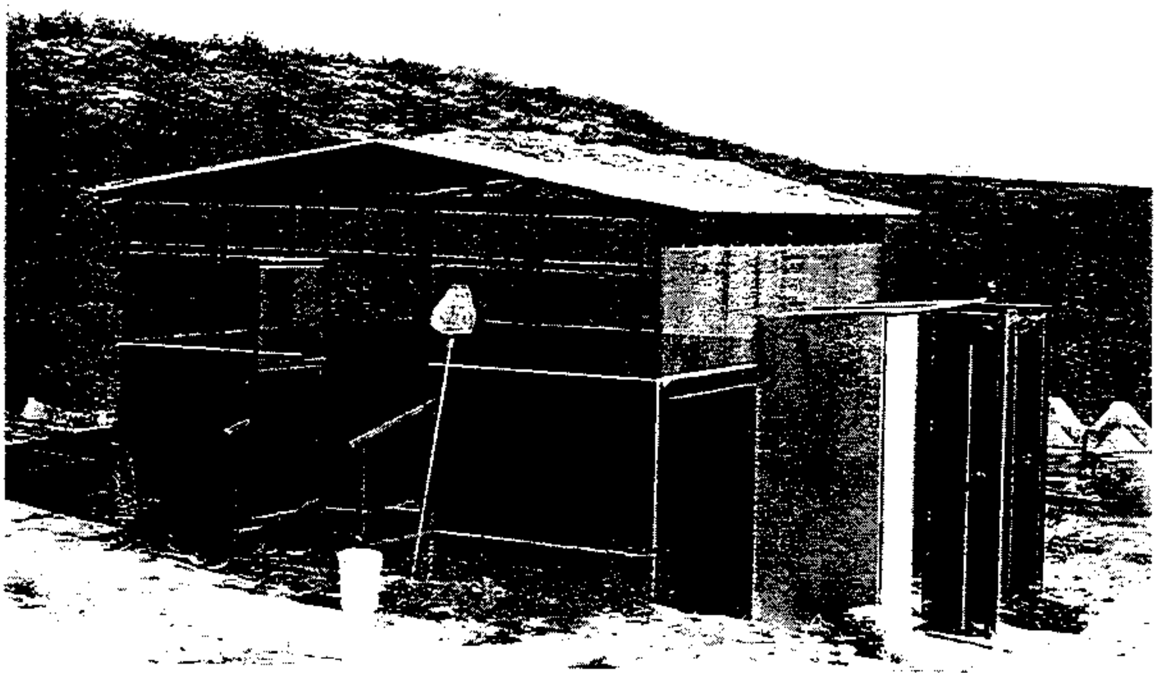


Fig. 3. Belency Tank, Cochise County, Arizona, exemplifying a sediment trap-main pond system. Run-off water from Belency Hollow enters the sediment trap from the left side of the photograph, fills the trap until it flows through a culvert in the mesquite-lined levee at center to fill the main pond to the right. Once the sediment trap and main pond are full, any further inflow runs around the system on Belency Hollow's natural floor, just beyond the tank system in the photograph.

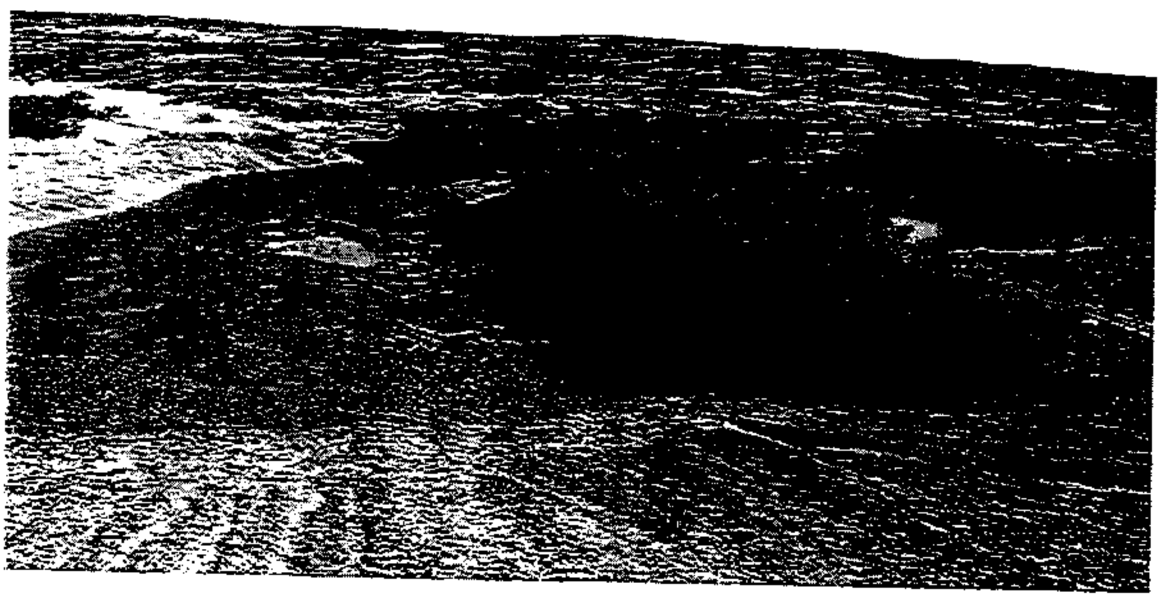


Fig. 4. Concrete pool (3 X 3.5 m, 1.2 m deep) at Rosewood Tank, Cochise County, Arizona, serving as a refugium during drought times for a population of Chiricahua leopard frogs. This pool is gravity-fed through a float valve from a storage tank filled by a windmill-powered well. Cattle are excluded from the pool, but are served by a nearby steel tank on the same water system.

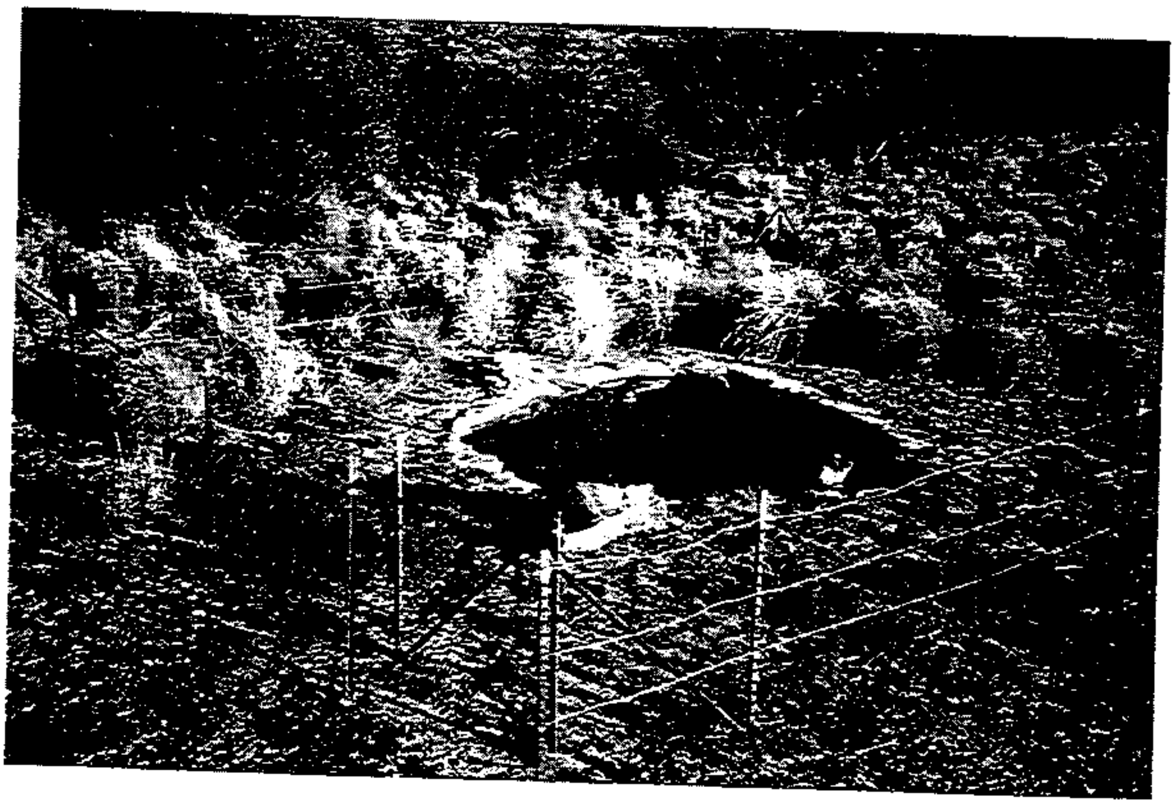


Fig. 5. Monitoring results for the Chiricahua leopard frog population introduced into the bullfrog exclosure at San Bernardino National Wildlife Refuge.

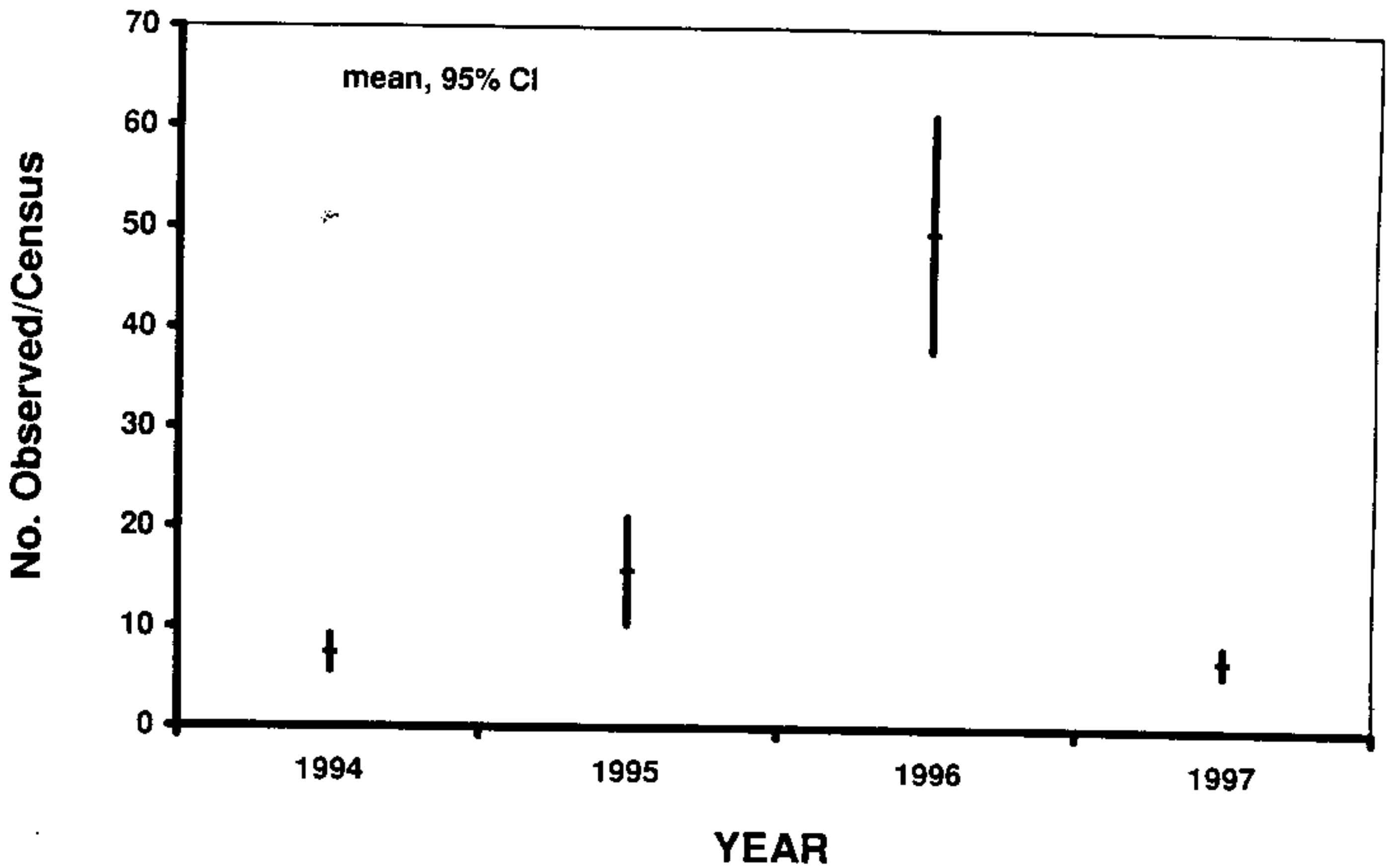


Fig. 6. Lineage and population history diagram for all known leopard frogs in the study region during 1994-1997. The populations to the right (Guadalupe Canyon, Belency Tank) became extinct. Arrows among the other populations indicate translocations made by resource managers to establish new population sites.