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Reptiles and Amphibians

by
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Overview

Amphibians and reptiles are important elements of our national biological heritage and deserve special attention. They are crucial to the natural functioning of many ecological processes and key components of important ecosystems. In some areas certain species are economically consequential; others are aesthetically pleasing to many people, and as a group they represent significant segments of the evolutionary history of North America. Knowledge gained from past study of amphibian development and metamorphosis has contributed immensely to our understanding of basic biological processes and has directly benefited humans.

The native herpetofauna of the continental United States includes about 230 species of amphibians (about 62% of which are salamanders and 38% frogs) and some 277 species of reptiles (about 19% turtles, 35% lizards, 45% snakes, and less than 1% crocodilians). If the list were expanded to include native species from Puerto Rico and the U.S. Virgin Islands in the Caribbean, Hawaii, the Trust Territory of the Pacific Islands, and the U.S. Territories in the Pacific, the amphibian list would increase by about 20 native species (all frogs) and another 5 non-native frog species. If the reptile inventory were expanded similarly, the list would increase by 2 turtles, 83 lizards, 18 snakes, and 1 crocodilian. Another 2 species of turtles, 17 lizards, 2 snakes, and 1 crocodilian have been introduced. An updated summary of this information is scheduled for publication later this year (McDiarmid, unpublished data).

Many U.S. reptile and amphibian checklists and field guides have been written over the past 50 years. The data for such summaries come from researchers working with various aspects of the biology of amphibians and reptiles and are found in many scientific publications. These summary field guides give the impression that the herpetofauna of the United States is well known and well studied. When we realize how little is known of the herpetofauna of comparable areas in South America, such an assumption is valid. A cursory review of U.S. data, however, provides a somewhat different view. Since 1978 the total herpetofaunal diversity of the United States has increased by almost 12%, from 454 to 507 species. Much of that increase, though, has resulted from a new knowledge of complex groups of species (e.g., eastern plethodontid salamanders) through the application of molecular techniques to gain a better understanding of the patterns of species formation and of the phylogenetic (evolutionary) history of certain groups. New species are still being discovered in relatively populated parts of the country (e.g., salamanders from California; D. Wake, Museum of Vertebrate Zoology, University of California, Berkeley, personal communication).

Baseline information of the status and health of U.S. populations of amphibians and reptiles is remarkably sparse. No national program of monitoring populations of amphibians and reptiles, comparable to the North American Breeding Bird Survey (now coordinated by the National Biological Service), is operational. Programs in some states (e.g., Kansas, Illinois, Maryland, Wisconsin) have been moderately successful in monitoring amphibians, but clearly a national program is needed. Long-term data (more than 10 years) from specific sites in many habitats in different parts of the country were and are essential to detect continental or global patterns of change in the distribution and abundance of species' populations. A recent publication (Heyer et al. 1994) recommended standard guidelines and techniques for monitoring amphibian populations and habitats; a similar volume on reptiles is planned. What remains is to establish a national program for such monitoring studies; the Declining Amphibian Populations Task Force, a part of the Species Survival Commission of the World Conservation Union, together with the National Biological Service, should play major roles in establishing such programs for amphibians. Similarly, organizations that deal with the conservation of turtles and crocodilians need to be expanded to develop an effective national monitoring program for reptiles.

Habitat degradation and loss seem to be the most important factors adversely affecting amphibian and reptile populations in North America. The drainage and loss of small aquatic habitats and their associated wetlands have had a major adverse effect on many amphibian species and some reptiles.

Many other factors in the decline of reptiles and amphibians have been implicated; most, perhaps all, are human-caused. For example, non-native species of gamefish introduced for sport have been implicated in the decline of frog populations in mountainous areas of some western states. Similarly, the introduction, accidental or intentional, of other non-native species (e.g., bullfrogs in western states, anoline lizards in south Florida, and snakes in Guam) has harmed native species in other parts of the country. Although populations of a few species have been severely impacted for diverse reasons (*see* the articles on California native frogs and the Tarahumara frog [*Rana tarahumarae*]), it is not too late to prevent the extirpation of others. Certain management and conservation decisions based on adequate scientific data and careful planning have proven successful (*see* articles on Coachella Valley fringe-toed lizard [*Uma inornata*] and the American alligator [*Alligator mississippiensis*]), but too often these initiatives are reactive and occur only after a species is in trouble.

Clearly, a better coordinated national program that looks at all species of amphibians and reptiles is desirable. Local and state programs to monitor amphibian and reptile populations are beginning; these efforts need to be expanded nationally. It is obvious that early detection of problems is crucial to successful remedial action. In many ways, a national program of monitoring amphibian and reptile populations is like preventive medicine; the earlier a problem is detected, the greater the likelihood of successful treatment and the lower the cost. A proactive national program based on standardized scientific methodology and applied across all species and habitats will go a long way toward ensuring that amphibians and reptiles remain a healthy component of our national biological heritage. They are too important overall to receive anything less.

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Turtles

by
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Turtles have existed virtually unchanged for the last 200 million years. Unfortunately, some of the same traits that allowed them to survive the ages often predispose them to endangerment. Delayed maturity and low and variable annual reproductive success make turtles unusually susceptible to increased mortality through exploitation and habitat modifications (Brooks et al. 1991; Congdon et al. 1993).

In general, turtles are overlooked by wildlife managers in spite of their ecological significance and importance to humans. Turtles are, however, important as scavengers, herbivores, and carnivores, and often contribute significant biomass to ecosystems. In addition, they are an important link in ecosystems, providing dispersal mechanisms for plants, contributing to environmental diversity, and fostering symbiotic associations with a diverse array of organisms. Adults and eggs of many turtles have been used as a food resource by humans for centuries (Brooks et al. 1988; Lovich 1994). As use pressures and habitat destruction increase, management that considers the life-history traits of turtles will be needed.

Documenting Turtle Population Status

I reviewed the population trends of turtles in the United States by examining most references (Ernst et al. 1994) that document the trends of turtle species and populations. Because few long-term studies (lasting more than one generation of the species examined) have focused on turtles, data on population fluctuations over time are generally unavailable (but *see* Gibbons 1990; Congdon et al. 1993). Techniques for conducting population studies of turtles and analyzing the data are summarized in Gibbons (1990).

Although we know less than desired about the actual extent of population fluctuations in most turtle populations, we do know that many turtles in the United States are at great risk of decline and extinction. Of the 55 native turtle species in the United States and its offshore waters, 25 (45%) require conservation, and 21 (38%) are protected or are candidates for protection under the Endangered Species Act. Of the 11 species and subspecies listed as candidates for protection under the ESA, 4 are considered declining, and 7 have unknown population statuses (Table). All tortoises and marine turtles require conservation action. Of the remaining 46 turtle species (aquatic and semi-aquatic forms), 16 (35%) require conservation action. The percentage of U.S. turtles requiring conservation action (45%) is similar to that of the world (41%; IUCN/SSC Tortoise and Freshwater Turtle Specialist Group 1991).

Although no turtles in the United States are known to have become extinct since European colonization (Honegger 1980), many species have experienced significant declines in numbers and distribution during the last 100 years. For example, several bog turtle (*Clemmys muhlenbergii*) populations in western New York, and all populations in western Pennsylvania, are apparently extirpated (Collins 1990; Ernst et al. 1994). Some populations of the spotted turtle (*C. guttata*) have also shown dramatic declines (Lovich 1989). Even wide-ranging, formerly common species such as the common box turtle (*Terrapene carolina*; Ernst et al. 1994), desert tortoise (*Gopherus agassizii;* USFWS 1993), gopher tortoise (*G. polyphemus*; McCoy and Mushinsky 1992), common slider (*Trachemys scripta*; Warwick 1986), and the alligator snapping turtle (*Macroclemys temminckii*; Pritchard 1989) have declined significantly, underscoring the importance of monitoring "common"

Table. U.S. turtle species in need of conservation.

Family and species	Common name	Status*
Cheloniidae	Sea turtles	
Caretta caretta	Loggerhead	Threatened
Chelonia mydas	Green sea turtle	Endangered or threatened according to population or geographic area
Eretmochelys imbricata	Hawksbill	Endangered
Lepidochelys kempii	Kemp's ridley	Endangered
L. olivacea	Olive ridley	Endangered or threatened according to population or geographic area
Chelydridae	Snapping turtles	
Macroclemys temminckii	Alligator snapping turtle	Unknown but vulnerable; C2 candidate
Dermochelyidae	Leatherback sea turtles	
Dermochelys coriacea	Leatherback	Endangered
Emydidae	Semi-aquatic pond turtles	
Clemmys insculpta	Wood turtle	May become threatened if trade not brought under control
C. marmorata	Western pond turtle	Declining; C2 candidate
C. muhlenbergii	Bog turtle	Unknown; are or may be threatened by international trade; C2 candidate
Emydoidea blandingii	Blanding's turtle	Declining; C2 candidate
Graptemys barbouri	Barbour's map turtle	Unknown; C2 candidate
G. caglei	Cagle's map turtle	Unknown
G. flavimaculata	Yellow-blotched map turtle	Threatened, but insufficiently known; may be threatened by international trade
G. oculifera	Ringed map turtle	Threatened; restricted distribution
Malaclemys terrapin	Diamondback terrapin	Some populations unknown, others declining; C2 candidate; listing applies to population or geographic area
Pseudemys alabamensis	Alabama red-bellied turtle	Endangered; restricted distribution
P. rubriventris	Red-bellied turtle	Endangered, according to population or geographic area
Kinosternidae	Mud and musk turtles	
Kinosternon flavescens	Yellow mud turtle	Unknown; C2 candidate; listing applies to population or geographic area
K. hirtipes	Mexican rough-footed mud turtle	Unknown; C2 candidate; listing applies to population or geographic area
Sternotherus depressus	Flattened musk turtle	Threatened
Testudinidae	Tortoises	
Gopherus agassizii	Desert tortoise	Some populations threatened, others are C2 candidates; may become threatened if trade not brought under control. Status of Sonoran Desert population unknown
G. berlandieri	Texas tortoise	May become threatened if trade not brought under control Receiving some conservation action
G. polyphemus	Gopher tortoise	Declining. Some populations threatened, others are C2 candidates; may become threatened if trade not controlled. Receiving some conservation action
Trionychidae	Softshell turtles	
Apalone spinifera	Spiny softshell turtle	Are or may be affected by international trade

*C2 — Possibly qualifying for threatened or endangered status, but more information is needed for determination.

Barbour's map turtle (*Graptemys barbouri*) is restricted to the Apalachicola River system of Alabama, Florida, and Georgia. The species is a candidate for listing under the Endangered Species Act.

species (Dodd and Franz 1993). The alarming decline of marine turtle populations is discussed later in this section.

Perhaps the best data on long-term population changes in turtles are for the diamondback terrapin (*Malaclemys terrapin*), a species exploited heavily during the 19th century as a gourmet food (McCauley 1945; Carr 1952). Terrapin populations declined rapidly, causing some states to set seasons and limits for their protection as early as 1878. The market for terrapin meat eventually waned, and terrapin populations recovered somewhat because their habitat remained largely intact. Unfortunately, some terrapin populations may be declining again because of renewed regional harvesting (Garber 1988), increased habitat destruction, mortality from vehicles, and drowning in crab traps (Ernst et al. 1994).

Some turtle species, such as members of the map turtle genus *Graptemys*, have restricted ranges (Lovich and McCoy 1992) that place them at extreme risk of extinction. In addition, the popularity of many species, particularly tortoises, as pets, contributes to the decline of wild populations (IUCN/SSC 1989; Ernst et al. 1994). Disease also appears to contribute to population declines in some turtles (Balazs 1986; Dodd 1988; Jacobson et al. 1991) and even seems a major challenge to the recovery of the federally threatened desert tortoise (USFWS 1993).

Because of individual longevity, delayed maturity, and long generation times of turtles, long-term studies are required to monitor the dynamics of turtle populations (Gibbons 1990); recovery of most threatened species will be slow. Programs in which hatchlings are propagated in captivity and later released into the wild will do little to assist the recovery of turtles until the ultimate causes of decline are corrected (Frazer 1992).

Efforts to conserve turtles in the United States should be concentrated in areas of high species diversity, where many species have limited distributions, and where populations are at great risk. Notable high-risk areas include shallow wetlands inhabited by freshwater turtles and coastal zones occupied by sea turtles. The most significant area of turtle endemism in the United States is along the Coastal Plain of the Gulf of Mexico (Lovich and McCoy 1992). Eleven species of turtles in the southeastern United States, where diversity is high (Iverson and Etchberger 1989; Iverson 1992), require conservation action, adding to the importance of implementing immediate conservation programs in that region.

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Marine Turtles in the Southeast

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Five species of marine turtles frequent the beaches and offshore waters of the southeastern United States: loggerhead (*Caretta caretta*), green (*Chelonia mydas*), Kemp's ridley (*Lepidochelys kempii*), leatherback (*Dermochelys coriacea*), and hawksbill (*Eretmochelys imbricata*). All five are reported to nest, but only the loggerhead and green turtle do so in substantial numbers. Most nesting occurs from southern North Carolina to the middle west coast of Florida, but scattered nesting occurs from Virginia through southern Texas. The beaches of Florida, particularly in Brevard and Indian River counties, host what may be the world's largest population of loggerheads.

Marine turtles, especially juveniles and subadults, use lagoons, estuaries, and bays as feeding grounds. Areas of particular importance include Chesapeake Bay, Virginia (for loggerheads and Kemp's ridleys); Pamlico Sound, North Carolina (for loggerheads); and Mosquito Lagoon, Florida, and Laguna Madre, Texas (for greens). Offshore waters also support important feeding grounds such as Florida Bay and the Cedar Keys, Florida (for green turtles), and the mouth of the Mississippi River and the northeast Gulf of Mexico (for Kemp's ridleys). Offshore reefs provide feeding and resting habitat (for loggerheads, greens, and hawksbills), and offshore currents, especially the Gulf Stream, are important migratory corridors (for all species, but especially leatherbacks).

Most marine turtles spend only part of their lives in U.S. waters. For example, hatchling loggerheads ride oceanic currents and gyres (giant circular oceanic surface currents) for many years before returning to feed as subadults in southeastern lagoons. They travel as far as Europe and the Azores, and even enter the Mediterranean Sea, where they are susceptible to longline fishing mortality. Adult loggerheads may leave U.S. waters after nesting and spend years in feeding grounds in the Bahamas and Cuba before returning. Nearly the entire world population of Kemp's ridleys uses a single Mexican beach for nesting, although juveniles and subadults, in particular, spend much time in U.S. offshore waters.

The biological characteristics that make sea turtles difficult to conserve and manage include a long life span, delayed sexual maturity, differential use of habitats both among species and life stages, adult migratory travel, high egg and juvenile mortality, concentrated nesting, and vast areal dispersal of young and subadults. Genetic analyses have confirmed that females of most species return to their natal beaches to nest (Bowen et al. 1992; Bowen et al. 1993). Nesting assemblages contain unique genetic markers showing a tendency toward isolation from other assemblages (Bowen et al. 1993); thus, Florida green turtles are genetically different from green turtles nesting in Costa Rica and Brazil (Bowen et al. 1992). Nesting on warm sandy beaches puts the turtles in direct conflict with human beach use, and their use of rich offshore waters subjects them to mortality from commercial fisheries (National Research Council 1990).

Marine turtles have suffered catastrophic declines since European discovery of the New World (National Research Council 1990). In a relatively short time, the huge nesting assemblages in the Cayman Islands, Jamaica, and Bermuda were decimated. In the United States, commercial turtle fisheries once operated in south Texas (Doughty 1984), Cedar Keys, Florida Keys, and Mosquito Lagoon; these fisheries collapsed from overexploitation of the mostly juvenile green turtle populations. Today, marine turtle populations are threatened worldwide and are under intense pressure in the Caribbean basin and Gulf of Mexico, including Cuba, Mexico, Hispaniola, the Bahamas, and Nicaragua. Subadult loggerheads are captured extensively in the eastern Atlantic Ocean and Mediterranean Sea. Thus, marine turtles that hatch or nest on U.S. beaches or migrate to U.S. waters are under threats far from U.S. jurisdiction. Marine turtles can be conserved only through international efforts and cooperation.

Information on the status and trends of southeastern marine turtle populations comes from a variety of sources, including old fishery records, anecdotal accounts of abundance, beach surveys for nests and females, and trawl and aerial surveys for turtles offshore. Surveys for marine turtles are particularly difficult because most of their lives are spent in habitats that are not easily surveyed. Hence, most status and trends information comes from counting females and nests. Few systematic long-term (more than 10-20 years) surveys have been conducted; the most notable are the nesting surveys at Cumberland Island and adjacent barrier islands in Georgia (T.H. Richardson, University of Georgia, unpublished data), and beaches south of Melbourne in Brevard County, Florida (Ehrhart et al. 1993). Beach monitoring is fairly widespread in many areas of the Southeast, but coverage varies considerably among beaches and field crews. The only long-term sampling of lagoonal or bay populations occurs at Mosquito Lagoon and Chesapeake Bay, although short-duration surveys have sampled Florida Bay, Pamlico Sound, and Laguna Madre. Trawl surveys of inlets and ship channels and aerial surveys of offshore waters have been undertaken periodically.

Loggerhead and Green Turtles

The number of turtles nesting fluctuates substantially from one year to the next, making interpretation of beach counts difficult. The Florida nesting populations of loggerheads and green turtles appear stable based on 12 years of data from east-central Florida (Ehrhart et al. 1993; Fig. 1). The green turtle nesting population may be increasing because of protective measures over the last 20 years or so, although the number of nesting females is still low (assuming 3-5 nests per female). North of Florida, nesting loggerhead numbers are declining 3%-9% a year in Georgia and South Carolina (National Research Council 1990). The main cause of mortality is drowning in shrimp and fish nets (National Research Council 1990), although turtle excluder devices (TEDs; Fig. 2a) have helped reduce mortality (Fig 2b; Henwood et al. 1992). Large juveniles are most susceptible to drowning, and this is a critical life stage in the population dynamics of sea turtles (Crouse et al. 1987).

Few data are available for lagoonal turtles, although similar numbers have been captured in Mosquito Lagoon and Chesapeake Bay from one year to the next. Loggerhead and green turtle populations, both adult and subadult, have undoubtedly declined from historical levels because of beach development and disturbance, the collection of eggs, and destructive fishing

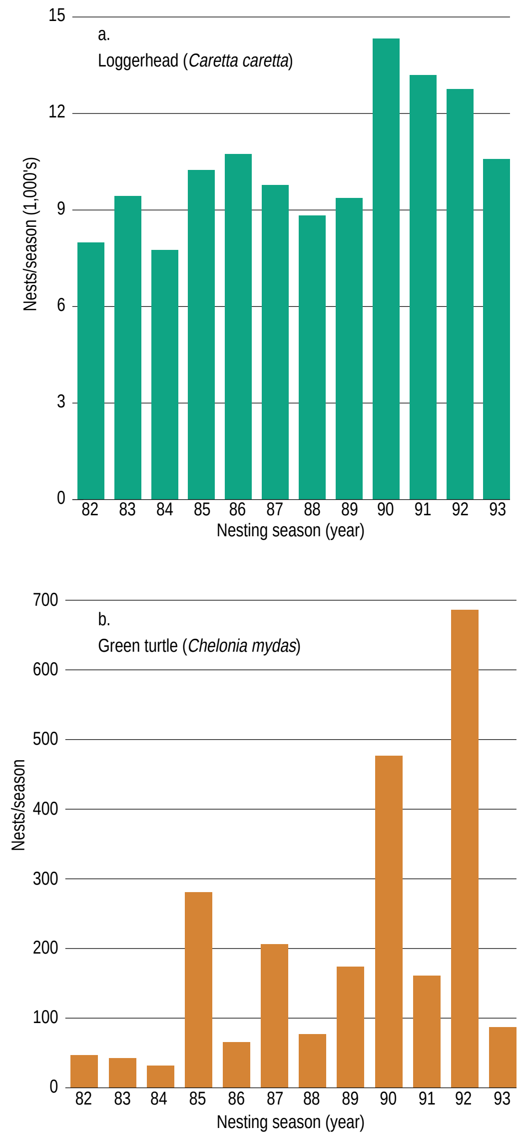

Fig. 1 a. Loggerhead nest totals in south Brevard County, Florida, 1982-93. **b.** Green turtle nest totals in south Brevard County, Florida, 1982-93. From Ehrhart et al. (1993).

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practices. Most high-level nesting occurs on the remaining undeveloped or lightly developed beaches. Even there, plans for development and disorientation from lights pose serious and continuing problems.

Kemp's Ridley

At one time, more than 40,000 females nested in a single mass nesting (termed "arribada") in Tamaulipas, Mexico. Several arribadas probably occurred each year. Since 1947 a drastic reduction in the number of nesting females caused the near extinction of this species (Ross et al. 1989). Today only about 400-500 turtles nest each year despite stringent protection of the nesting beach. The principal threat to this species is incidental take during shrimp fishing.

Leatherback and Hawksbill

The leatherback and hawksbill are rare nesters in the southeastern United States, but offshore waters are important for feeding, resting, and as migratory corridors. The status and trends of these species in U.S. offshore waters are unknown, although they are severely threatened throughout the Caribbean. Leatherbacks are taken by trawlers or are otherwise entangled in nets. Hawksbills are sought, especially in Cuba, for their shell, which is used for jewelry and similar items. The solitary nesting habits of hawksbills make them particularly difficult to monitor.

Summary

Sea turtles are threatened by beach development, light pollution, ocean dumping, incidental take in trawl and longline fisheries, disease (especially fibropapillomas), and many other variables. Because sea turtles are long-lived species, trends are difficult to monitor. Present methods of beach monitoring are extremely labor-intensive, expensive, and biased toward one segment of the population. Very little is known about marine turtle life-history and habitat requirements away from nesting beaches, and virtually nothing is known about male turtles. Because the effectiveness of measures aimed at protecting turtles may not be seen for decades, known conservation strategies should be favored over unproven mitigation schemes. Acquiring nesting habitat should be encouraged. One of the most important management measures to protect sea turtles, especially of the juvenile and subadult size class, in the southeastern United States, Caribbean, and western Atlantic Ocean is the use of TEDs to minimize drowning in commercial fisheries. Mature females should also be protected because of their importance to future reproduction. Researchers need to identify migratory routes, feeding and developmental habitat, and ways to minimize adverse impacts during all life-history stages.

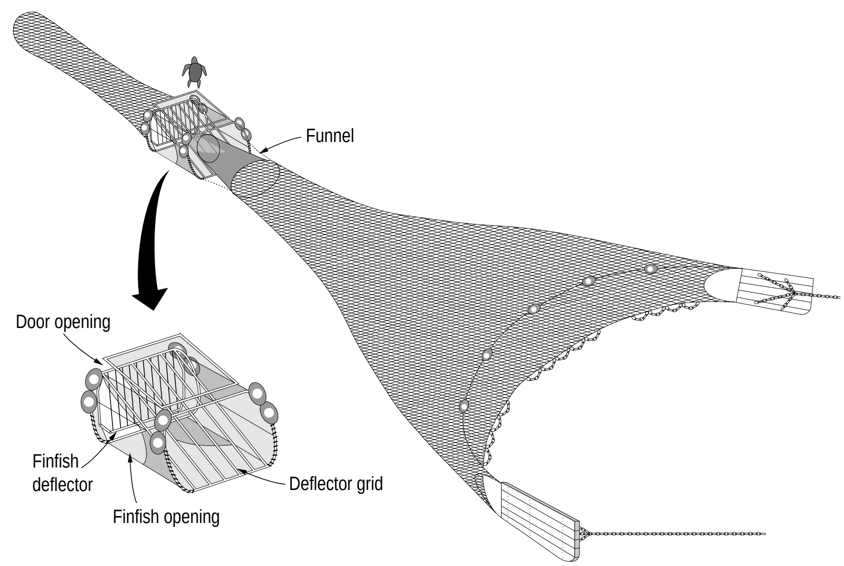

Fig. 2a. Schematic of a turtle excluder device (TED). From Watson et al. (1986).

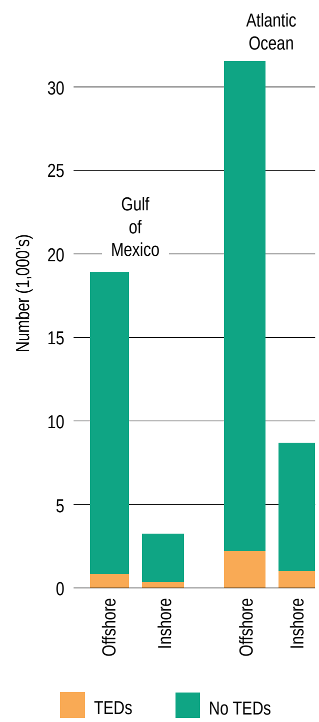

Fig. 2b. Incidental capture of sea turtles in inshore and offshore waters of the United States before and after regulations requiring the use of TEDs on the U.S. shrimp fleet. From Henwood et al. (1992).

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Amphibians are ecologically important in most freshwater and terrestrial habitats in the United States: they can be numerous, function as both predators and prey, and constitute great biomass. Amphibians have certain physiological (e.g., permeable skin) and ecological (e.g., complex life cycle) traits that could justify their use as bioindicators of environmental health. For example, local declines in adult amphibians may indicate losses of nearby wetlands. The aquatic breeding habits of many terrestrial species result in direct exposure of egg, larval, and adult stages to toxic pesticides, herbicides, acidification, and other human-induced stresses in both aquatic and terrestrial habitats. Reported declines of amphibian populations globally have drawn considerable attention (Bury et al. 1980; Bishop and Petit 1992; Richards et al. 1993; Blaustein 1994; Pechmann and Wilbur 1994).

Approximately 230 species of amphibians, including about 140 salamanders and 90 anurans (frogs and toads) occur in the continental United States. Because of their functional importance in most ecosystems, declines of amphibians are of considerable conservation interest. If these declines are real, the number of listed or candidate species at federal, state, and local levels could increase significantly. Unfortunately, because much of the existing information on status and trends of amphibians is anecdotal, coordinated monitoring programs are greatly needed.

Faunal Comparisons

North American amphibian species exhibit two major distributional patterns, endemic and widespread. Endemic species (Figure) tend to have small ranges or are restricted to specific habitats (e.g., species that occur only in one cave or in rock talus on a single mountainside). Declines are documented best for endemic species, partly because their smaller ranges make monitoring easier. Populations of endemics are most susceptible to loss or depletions because of localized activities (Bury et al. 1980; Dodd 1991). Examples of endemic species affected by different local impacts include the Santa Cruz long-toed salamander (*Ambystoma macrodactylum croceum*) in California, the Texas blind salamander (*Typhlomolge rathbuni*) in Texas, and the Red Hills salamander (*Phaeognathus hubrichti*) in Alabama; these three species are listed as federally threatened or endangered.

The number of endemic species that have suffered losses or are suspected of having severe threats to their continued existence has increased in the last 15 years (Table). In part, the increase reflects descriptions of new species with restricted ranges, but the accelerating pace of habitat alteration is the primary threat.

Table. The number of amphibian species showing documented or perceived declines in 1980 (Bury et al. 1980) and 1994.

Distribution pattern	Number of species	
	1980	1994
Endemic or relict	33	52
Widespread	5	33

The ranges of most endemics in the western states (26 species) are widely dispersed across the landscape. In contrast, endemics in the eastern and southeastern states (25 species) tend to be clustered in centers of endemism, such as in the Edwards Plateau (Texas), Interior (Ozark) Highlands (Arkansas, Oklahoma), Atlantic Coastal Plain (Texas to Virginia), and uplands or mountaintops in the Appalachian Mountains (West Virginia to Georgia).

Widespread species often are habitat generalists. Many were previously common, but have shown regional or rangewide declines (Hine et al. 1981; Corn and Fogelman 1984; Hayes and Jennings 1986; Table). Reported declines of widespread species often lack explanation, perhaps because these observations have only recently received general attention or because temporal and spatial variations in population sizes of many amphibians are not well understood. Some reports are for amphibians in relatively pristine habitats where human impacts are not apparent.

A few examples of declines in widespread species illustrate the threats they face across the country:

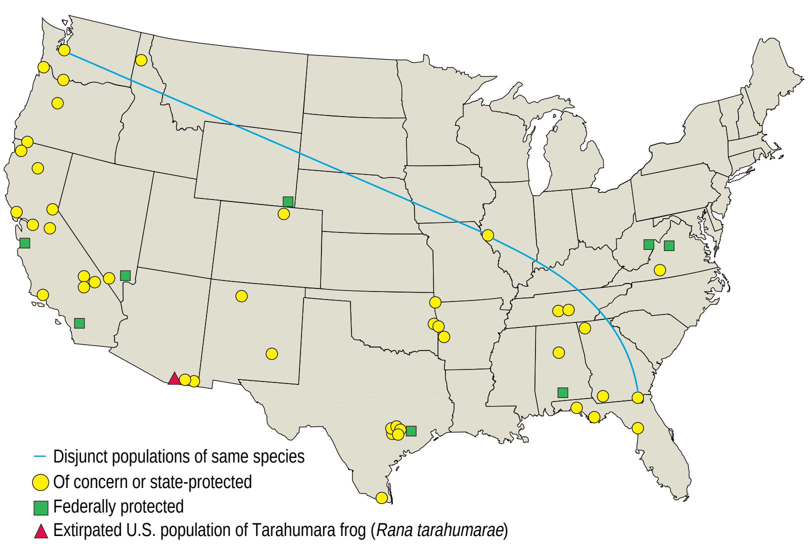

Figure. Distribution of U.S. endemic amphibian species; those west of the 100th meridian tend to be more broadly dispersed.

- Amphibians predominate in small forest streams of the Pacific Northwest. Because timber is harvested without adequate streamside protection, many populations of the tailed frog (*Ascaphus truei*) and torrent salamanders *(Rhyacotriton* spp.) have been severely affected; some populations soon will warrant consideration for listing.
- The western toad (*Bufo boreas*) once was common in the Rocky Mountains, but now occurs at fewer than 20% of known localities from southern Wyoming to northern New Mexico.
- Many salamander and frog populations in the southeastern United States have been negatively affected, some severely, because of degradation of stream habitats (e.g., the hellbender, *Cryptobranchus alleganiensis*) and conversion of natural pinewood and hardwood forests and associated wetlands (e.g., gopher frog, *Rana capito*) to plantation forestry, agriculture, and urban uses.
- Leopard frogs (*Rana* spp.), which are used in teaching and research institutions, were once abundant in most of the United States. Populations in this diverse group have declined, sometimes significantly, in midwestern, Rocky Mountain, and southwestern states.

Courtesy P.S. Corn, NBS

Western toad (*Bufo boreas*).

Causes of Declines

No single factor has been identified as the cause of amphibian declines, and many unexplained declines likely result from multiple causes. Human-caused factors may intensify natural factors (Blaustein et al. 1994b) and produce declines from which local populations cannot recover and thus they go extinct. Known or suspected factors in those declines include destruction and loss of wetlands (Bury et al. 1980); habitat alteration, such as impacts from timber harvest and forest management (Corn and Bury 1989; Dodd 1991; Petranka et al. 1993); introduction of non-native predators, such as sportfish and bullfrogs, especially in western states (Hayes and Jennings 1986; Bradford 1989); increased variety and use of pesticides and herbicides (Hine et al. 1981); effects of acid precipitation, especially in eastern North America and Europe (Freda 1986; Beebee et al. 1990; Dunson et al. 1992); increased ultraviolet radiation reaching the ground (Blaustein et al. 1994a); and diseases resulting from decreased immune system function (Bradford 1991; Carey 1993; Pounds and Crump 1994).

A Success Story:
The Barton Springs Salamander

A success story from the Edwards Plateau in Texas illustrates the importance of baseline ecological data, current science, and the types of partnerships essential for conservation of amphibians. The recently described Barton Springs salamander (*Eurycea sosorum*) occurs only in three springs within about 300 m (984 ft) of each other within the city limits of Austin. This salamander has one of the smallest known distributions of any North American vertebrate.

Pools associated with the two primary springs had been developed as municipal swimming and wading pools, and standard cleaning procedures had eliminated most salamanders. With cooperation of city authorities and local volunteers, pool maintenance practices detrimental to the salamander were modified, and populations of the salamander seem to be increasing and expanding their ranges within the spring system.

Courtesy W. Meinzer, USFWS

Barton Springs salamander (*Eurycea sosorum*).

Amphibian populations also may vary in size because of natural factors, particularly extremes in the weather (Bradford 1983; Corn and Fogelman 1984). The size of amphibian populations may vary, sometimes dramatically, from year to year, so what is perceived as a decline may be part of long-term fluctuations (Pechmann et al. 1991). The effect of global climate change on amphibians is speculative, but it has the potential for causing the loss of many species.

Monitoring Needs

A profound need exists for national coordination of regional inventories and population studies, including a national effort to monitor amphibians on parks, forests, wildlife refuges, and other public lands. Only through long-term studies will better data on population changes through time and between sites become available. Such data are essential to evaluating the status and trends of amphibian species in the United States. Some regional surveys and inventories exist but only for a few species; these studies should be expanded into a coordinated effort with long-term monitoring of populations at many sites across the country as the goal.

In addition, more research is needed to determine the impact of natural and human-caused factors on the different life-history stages and environments of amphibians. Also, the assumption that amphibians are good indicators needs to be tested rigorously (Pechmann and Wilbur 1994). Likewise, understanding the dynamics of populations between habitats and regions, and the roles amphibians play in aquatic and terrestrial ecosystems is essential. Detailed work on the ecology of species and the factors implicated in declines needs to continue.

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American Alligators in Florida

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The American alligator (*Alligator mississippiensis*) is an integral component of wetland ecosystems in Florida. Alligators also provide aesthetic, educational, recreational, and economic benefits to humans. Because of the commercial value of alligator hides for making high-quality leather products, alligator hunting was a major economic and recreational pursuit of many Floridians from the mid-1800's to 1970. The Florida alligator population varied considerably during the 1900's in response to fluctuating hunting pressure caused by unstable markets for luxury leather products.

The declining abundance of alligators during the late 1950's and early 1960's led to the 1967 classification of the Florida alligator population as endangered throughout its range. Federal and international regulations imposed during the 1970's and 1980's helped control trade of alligator hides, and illegal hunting of alligators was checked. The Florida alligator population responded immediately to protection and was reclassified as threatened in 1977 and as threatened because of its similarity in appearance to the American crocodile (*Crocodylus acutus*) in 1985 (Neal 1985).

Assessments of Florida's alligator population were based on sporadic surveys before 1974 (Wood et al. 1985). The Florida Game and Fresh Water Fish Commission implemented annual night-light surveys that used spotlights to detect alligator eyeshine in 1974 to provide a more objective basis for assessing population trends (Wood et al. 1985). Although all areas were not sampled every year, these data are the best available for alligator populations in Florida and are useful for estimating population trends (Woodward and Moore 1990). Because survey areas were not a random sample of all alligator habitat in Florida, trend results are applicable only to deepwater habitats and navigable wetlands.

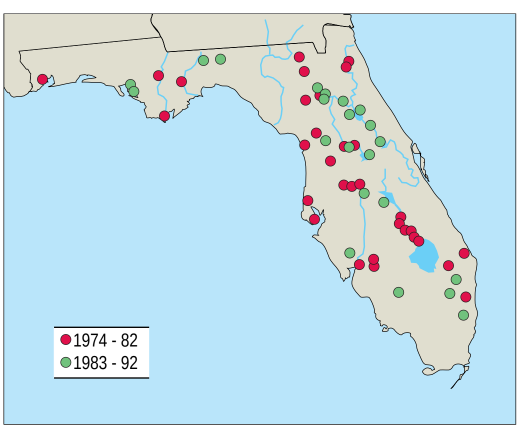
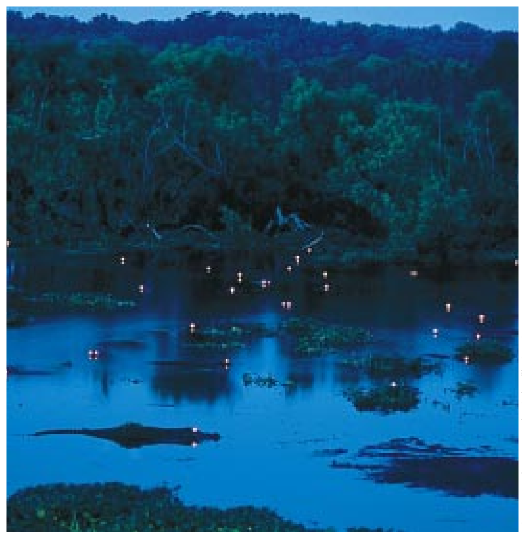


Fig. 1. Locations of survey areas for night-light counts of alligators in Florida, 1974-92.

Design of Alligator Surveys, 1974-92

We conducted night-light counts (Woodward and Marion 1978) with high-intensity spotlights from boats on 54 areas throughout Florida (Fig. 1) during 1974-92 (Woodward and Moore 1990). The number of areas surveyed in any year ranged from 7 in 1974 to 43 in 1980. In 1983 the number of areas surveyed was reduced to 22 to allow observers to conduct replicate counts on areas each year (Fig. 1). Eighteen of the 22 areas were subjected to alligator harvests of some type.



Courtesy John Moran, The Gainsville Sun ©

Alligators at dusk, Payne's Prairie State Preserve, Florida.

We analyzed observed densities of alligators per kilometer (0.62 mile) of shoreline to estimate trends for each area during the periods 1974-92 and 1983-92. Size classes corresponded to the overall population, juveniles (0.3-1.2 m [1-4 ft]), harvestable sizes (1.2 m or longer [4 ft or longer]), and adults (1.8 m or longer [6 ft or longer]; hatchlings less than 0.3 m long [1 ft] were excluded from trend analysis).

Count densities represent only alligators observed during the survey. Most (more than 65%) alligators were submerged during surveys and not detected (Murphy 1977; Brandt 1989; Woodward and Linda 1993). Alligators in wetlands adjacent to surveyed areas may have been undetected (Woodward and Linda 1993). Counts, however, do provide a relative measure of alligator abundance that is useful for estimating population trends, provided that rates of detection do not vary annually.

Status and Trends

From 1974 to 1992, the density of alligators on surveyed wetlands increased an average 41%

or 1.9% annually. Average annual densities of harvestable alligators increased 2.7%, while average annual densities of adults increased 2.5%. The 0.5% average annual increase in counts of juvenile alligators during 1974-92 was not significant. These trends confirm that the Florida alligator population increased during the apparent recovery of the 1970's and 1980's (Neal 1985). We observed cyclic patterns in abundance over time for all size classes (Fig. 2). Cyclic population levels may represent varying availability of counted alligators due to fluctuations in water level not fully accounted for in our analyses. They may also reflect population changes brought about by periodic droughts or, to a lesser extent, severe winters.

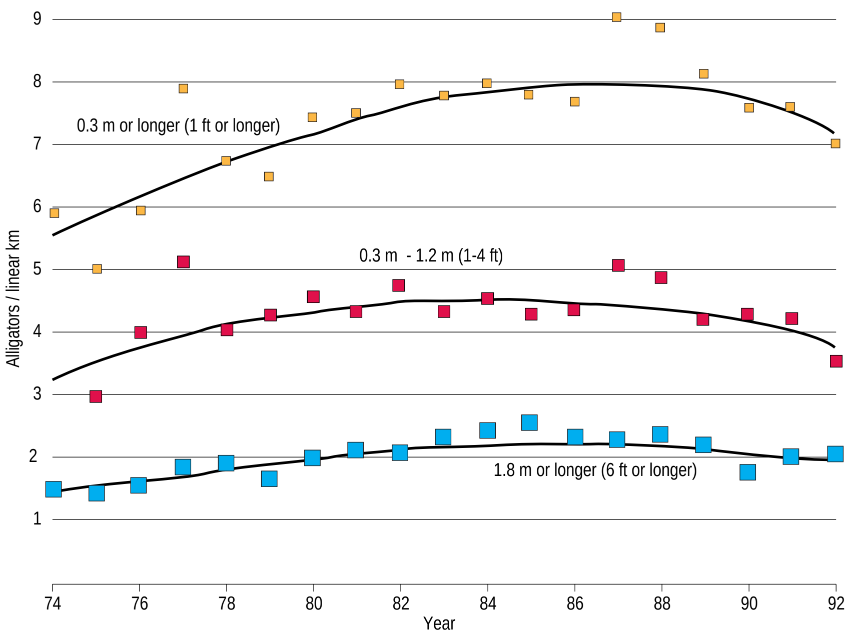


Fig. 2. Annual indices (mean number of alligators detected per linear kilometer [0.62 mi] of survey route) and smoothed trend estimates (Cleveland 1979) for three size classes of the statewide alligator population in Florida, 1974-92.

From 1983 to 1992, observed densities of adult alligators declined 3.2% per year, but we did not detect such trends in other size classes (Fig. 2). It is too early to draw conclusions concerning the influence of harvests on alligator populations since legal harvesting began in 1987 because of the variable nature of night-light alligator counts and the uncertain effects of wariness. Relatively stable populations of juveniles and harvestable alligators indicate that hatchling recruitment (replenishment) is sufficient to replace alligators lost through harvest. Consequently, alligator harvests do not seem to have negatively affected the Florida alligator population as a whole.

Historically, the Florida alligator population was threatened by habitat loss and excessive illegal hunting (Hines 1979), but recently environmental contamination has been associated with population declines. Wetland drainage and alteration during the 1900's destroyed alligator habitat and permanently reduced alligator populations in some wetlands, particularly in freshwater marshes (Neal 1985). State legislation, most recently the Wetlands Protection Act of 1984 (Florida Statutes 403.91), has significantly protected remaining wetlands, but alteration and loss of wetlands persist. Between the mid-1970's and mid-1980's, 10,542 ha (26,030 acres) of wetlands per year were lost to agriculture and other development (Frayer and Hefner 1991). Thus, habitat loss remains a threat to alligator populations.

Illegal hunting is now negligible and has been replaced by regulated, managed harvests. Florida implemented a nuisance alligator control program in 1978 in response to increasing problem alligators during the 1970's (Hines and Woodward 1980). Because the nuisance alligator program targets individual alligators, the removal of these animals is unlikely to measurably affect alligator populations (Hines and Woodward 1980; Jennings et al. 1989). The state game commission introduced managed harvests of alligators and their eggs in 1987 to create conservation incentives by enhancing economic value of wild alligators (Wiley and Jennings 1990). Studies of the effects of harvest on alligator populations demonstrated that harvests are sustainable at certain rates (Jennings et al. 1988; Woodward et al. 1992). Annual monitoring and effective control of harvest rates ensure that populations will not suffer long-term depletion.

More recently, environmental toxins have been implicated in the sharp decline of the alligator population on Lake Apopka, Florida's third-largest lake (Woodward et al. 1993; Guillette et al. 1994). Widespread pollution of wetlands by potentially toxic petrochemicals and metals may threaten the long-term viability of other alligator populations within Florida. For the present, the status of the Florida alligator population is secure; however, continued habitat loss and toxic contamination will negatively affect alligator populations and may eventually compromise their conservation.

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Reptiles and Amphibians in the Endangered Longleaf Pine Ecosystem

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The Coastal Plain of the southeastern United States contains a rich diversity of reptiles and amphibians (herpetofauna). Of the 290 species native to the Southeast, 170 (74 amphibians, 96 reptiles) are found within the range of the remnant longleaf pine (*Pinus palustris*) ecosystem (Fig. 1). Many of these species are not found elsewhere, particularly those amphibians that require temporary ponds for reproduction. Many Coastal Plain species are listed federally or by states as endangered or threatened or are candidates for listing (Fig. 1). Examples include the flatwoods salamander (*Ambystoma cingulatum*), striped newt (*Notophthalmus perstriatus*), Carolina and dusky gopher frogs (*Rana capito capito* and *R.c. sevosa*), eastern indigo snake (*Drymarchon corais couperi*), gopher tortoise (*Gopherus polyphemus*), eastern diamondback rattlesnake (*Crotalus adamanteus*), and Florida pine snake (*Pituophis melanoleucus mugitus*).

Studies in the Southeast

Information on the status and trends of the Coastal Plain herpetofauna comes from limited studies of selected species or populations, mostly within the last decade. The only intensive long-term quantitative and community-based studies have been at the Savannah River Site on the upper Coastal Plain of South Carolina. Most other studies have been distributional surveys for species such as Red Hills salamanders (*Phaeognathus hubrichti*), gopher frogs, striped newts, flatwoods salamanders, gopher tortoises, and Florida scrub lizards (*Sceloporus woodi*). Few studies have reported detailed habitat requirements for suspected declining species throughout their range. Surveys generally range 1-2 years in duration. Other trend information is derived from studies conducted by university scientists, private organizations, or state resource agencies. Concern for the future of the entire herpetofaunal community in the Southeast rests mostly on the well-documented loss of the old-growth longleaf pine ecosystem, although few community-based herpetofaunal surveys have been undertaken in this habitat.

Status

The fire-adapted longleaf pine community once stretched from southeastern Virginia to eastern Texas (Fig. 2). At present, less than 14% of the historical 282,283 km^2 (70 million acres) longleaf pine forest remains (Means and Grow 1985; Noss 1989), and most of it is on private land. Less than 1% is old-growth forest. Conversion of longleaf pine forests for agriculture, timber plantations, and urban needs (Ware et al. 1993) is accelerating (Fig. 3) and probably threatens the continued existence of many amphibian and reptile species, particularly in southern Georgia and Florida. For example, longleaf pine forests in Florida declined from 30,756 km^2 (7.6 million acres) in 1936 to only 3,845 km^2 (0.95 million acres) in 1989, an 88% decrease (Cerulean 1991). In southeastern Georgia the longleaf pine forest declined 36% (to 931 km^2 [230,000 acres]) between 1981 and 1988 (Johnson 1988). Most of this conversion has been from second- or third-growth longleaf pine stands to slash or loblolly pine plantation forestry.

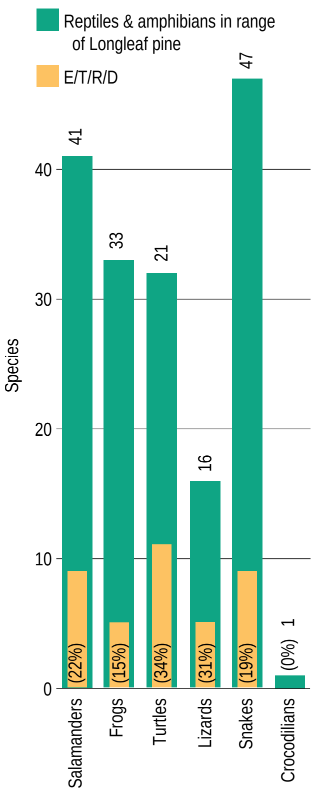

Fig. 1. Reptiles and amphibians within the southeastern Coastal Plain. Green bars = total number; Gold bars = number of species in need of conservation and management. E = endangered, T = threatened, R = rare, D = declining.

The effects of the loss of the longleaf pine ecosystem on the herpetofaunal community have never been assessed directly, but several species are known to have been affected. For example, the number of gopher tortoises, a key species within the longleaf pine ecosystem, has declined by an estimated 80% during the last 100 years (Auffenberg and Franz 1982). More than 300 invertebrates and 65 vertebrates use gopher tortoise burrows (Jackson and Milstrey 1989; Fig. 4), so an 80% reduction in gopher tortoises could represent a substantial reduction in the biodiversity of the longleaf pine ecosystem.

Amphibians that breed in temporary ponds have been particularly affected both because of direct habitat destruction and the slower loss of wetland breeding sites by ditching. Breeding, foraging, and overwintering sites are also affected by certain types of forest plantation site preparation. Only five populations of striped newts remain in Georgia (Dodd 1993; L. LaClaire, USFWS, personal communication); the flatwoods salamander has disappeared from the eastern section of its range; gopher frogs are nearly extirpated in North Carolina, Alabama, and Mississippi; and dusky salamanders (*Desmognathus* spp.) appear to have declined or disappeared in coastal South Carolina and peninsular Florida.

On the other hand, the long-term community studies at the Savanna River Site, where the destructive effects of plantation forestry are not prevalent, do not reveal declining trends, although some amphibian populations there fluctuate widely from one year to the next in both numbers and reproductive output (Pechmann et al. 1991). A 5-year study on a north Florida biological preserve disclosed declining amphibian numbers, but the study coincided with a severe regional drought (Dodd 1992). In west-central Florida, amphibian communities have changed composition because of

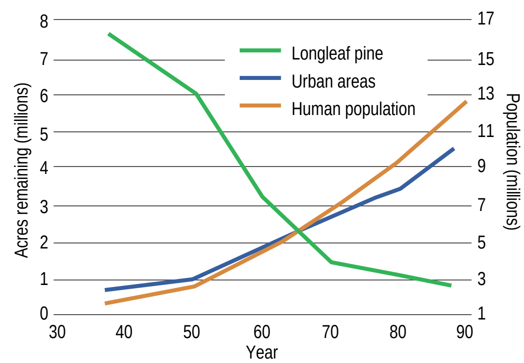

Fig. 3. Trend in loss of longleaf pine forest in relation to urban development and increases in human population in Florida, 1930-90 (Cerulean 1991; used with permission from The Nature Conservancy).

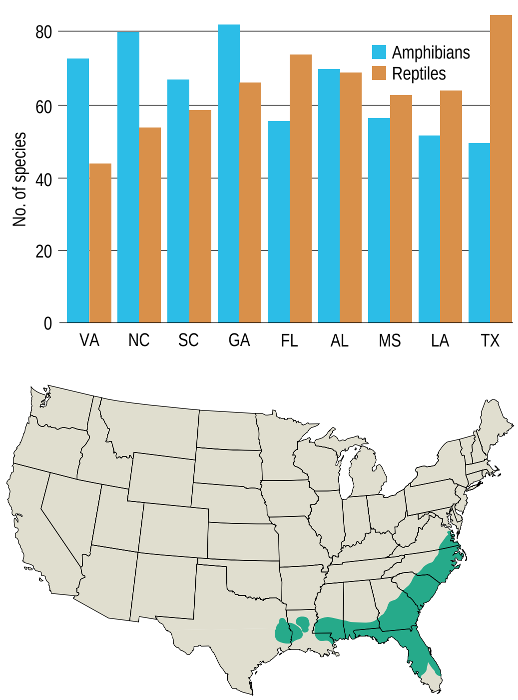

Fig. 2. Historical distribution of the longleaf pine ecosystem in the southeastern Coastal Plain. Chart shows the present total number of species of amphibians and reptiles in various southeastern states.

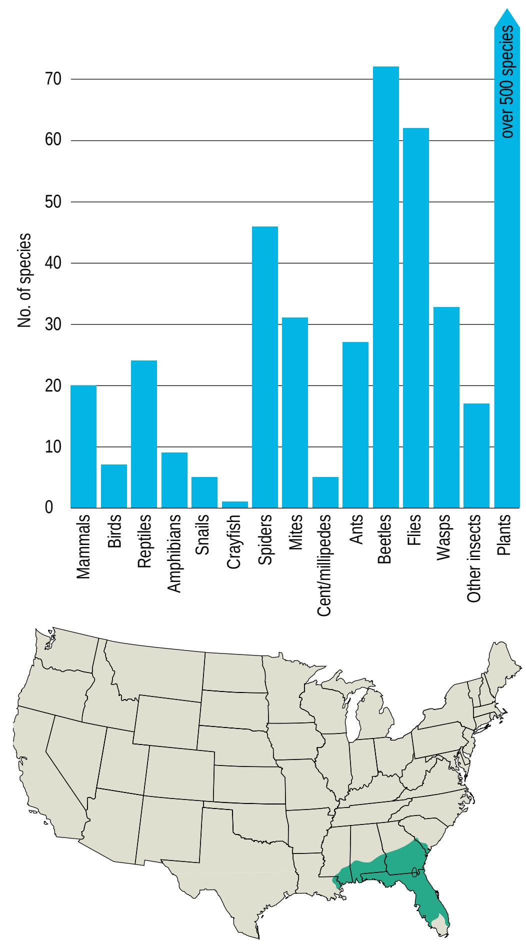

Fig. 4. The distribution of the gopher tortoise (*Gopherus polyphemus*) in the southeastern United States. The chart shows the number of species of various taxa known to use its burrow and the number of plant taxa described from the longleaf pine-wiregrass ecosystem.

urbanization (Delis 1993), but the long-term effects of the change are unknown. The overall status of the Red Hills salamander (federal threatened list) remained the same from 1976 to 1988 (Dodd 1991), although habitat loss continued from plantation forestry. Virtually no data exist for terrestrial reptile populations or communities except for the gopher tortoise. Anecdotal information for all terrestrial reptiles suggests population declines, particularly in areas affected by imported red fire ants (*Solenopsis invicta*).

Local centers of amphibian and reptile diversity need to be identified within the remaining longleaf pine community. Surveys, basic life-history studies of sensitive species, and long-term monitoring of amphibian and reptile populations need to be initiated. Many species that are restricted to wetland and upland habitats appear to be declining, but precise baseline data are lacking. Factors impeding the identification of population trends include the longevity of many species, the effects of periodic natural events such as drought, and what appear to be random population fluctuations. At the same time, when the known extent of habitat loss is coupled with declining trends elsewhere (Blaustein and Wake 1990; Wyman 1990) that result from unknown or hypothesized causes (UVB light, acidity, heavy metals, estrogen-mimicking compounds, roads, habitat fragmentation), the study and monitoring of amphibian and reptile populations in remnant southeastern longleaf pine forests will become especially imperative.

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Native Ranid Frogs in California

by

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Many recent declines and extinctions of native amphibians have occurred in certain parts of the world (Wake 1991; Wake and Morowitz 1991). All species of native true frogs have declined in the western United States over the past decade (Hayes and Jennings 1986). Most of these native amphibian declines can be directly attributed to habitat loss or modification, which is often exacerbated by natural events such as droughts or floods (Wake 1991). A growing body of research, however, indicates that certain native frogs are particularly susceptible to population declines and extinctions in habitats that are relatively unmodified by humans (e.g., wilderness areas and national parks in California; Bradford 1991; Fellers and Drost 1993; Kagarise Sherman and Morton 1993). To understand these declines, we must document the current distribution of these species over their entire historical range to learn where they have disappeared.

In 1988 the California Department of Fish and Game commissioned the California Academy of Sciences to conduct a 6-year study on the status of the state's amphibians and reptiles not currently protected by the Endangered Species Act. The study's purpose was to determine amphibians and reptiles most vulnerable to extinction and provide suggestions for future research, management, and protection by state, federal, and local agencies (Jennings and Hayes 1993). This article describes the distribution and status of all native true frogs in California as determined by the California Fish and Game study.

Status

All species studied have suffered declines in distribution and abundance, largely because of habitat loss or modification from farming, grazing, logging, urban development, suppression of brush fires, and flood-control or water-development projects. The species have also been affected by the widespread introduction of vertebrate and invertebrate aquatic predators.

Northern Red-legged Frog (*Rana aurora aurora*)

This frog, restricted to lower elevations (300 m [984 ft]) of the north coast region of California (Fig. 1), has disappeared from about 15% of its historical range in California. It is not in danger of extinction in the state.

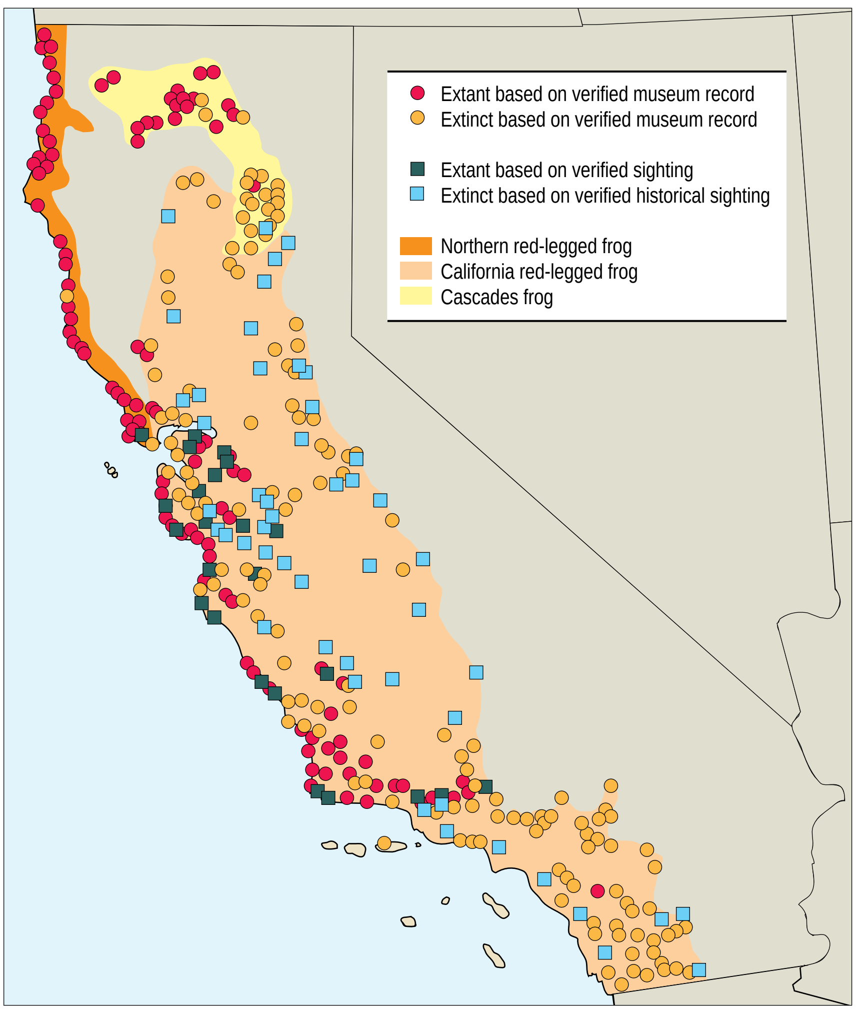

Fig. 1. Historical and current distribution of the northern red-legged frog, California red-legged frog, and Cascades frog in California based on 2,068 museum records and 302 records from other sources. Dots indicate locality records based on verified museum specimens. Squares indicate locality records based on verified sightings (e.g., field notes, photographs, published papers). Red dots and green squares denote localities where native frogs are extant. Gold dots and blue squares indicate where native frogs are presumed extinct. Figure modified from Jennings and Hayes (1993).

California Red-legged Frog (*R.a. draytonii*)

This frog was originally found over most of California below 1,524 m (500 ft) and west of the deserts and the Sierra Nevada crest (Fig. 1). Although the California red-legged frog has now disappeared from about 75% of its historical range in the state, around the turn of the century it was abundant enough to support an important commercial fishery in the San Francisco fish markets (Jennings and Hayes 1984). California red-legged frogs have almost completely disappeared from the Central Valley and southern California since 1970 and are currently proposed for listing as endangered by the U.S. Fish and Wildlife Service (Federal Register 1994).

Cascades Frog (*R. cascadae*)

The Cascades frog was originally found in northern California above 230 m (755 ft; Fig. 1), where it was historically very abundant. Since the mid-1970's, the species extensively declined, disappearing from about 50% of its range in the state. No habitat loss hypothesis adequately explains why this frog survived with current land-use practices for over 50 years before its decline. It is still abundant in California only in the northern third of its range on lands under federal ownership.

Foothill Yellow-legged Frog (*R. boylii*)

This frog was originally found over most of California below 1,829 m (6,000 ft), west of the deserts and the Sierra-Cascade crest (Fig. 2). In many locations before 1970, populations contained hundreds of individuals (Zweifel 1955), but the frog has now completely disappeared from southern California and from about 45% of its historical range over the entire state. Most populations were apparently healthy until the mid-1970's, when a population crash occurred in southern California and the Sierra Nevada foothills after several years of severe floods and drought, which may have been responsible for the declines, although it is not certain. Because this species was an important component of the food web in many streamside ecosystems, its loss has probably negatively affected several organisms, such as garter snakes (*Thamnophis* spp.), which historically relied upon it as a major food source.

Spotted Frog (*R. pretiosa*)

The spotted frog was historically recorded only from scattered localities in the extreme northeastern part of California below 1,372 m (4,500 ft), where it was apparently restricted to large marshy areas filled by warmwater (more than 20°C [68°F]) springs (Fig. 2). It has now

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disappeared from about 99% of its range, and is only known from one location in the state. It appears to be on the verge of extinction in California.

Yavapai Leopard Frog (*R. yavapaiensis*)

This frog was originally found along the Colorado River and in the Coachella Valley of southeastern California (Fig. 2). It has not been seen in the state since the mid-1960's and now seems to be extinct at all sites examined. This leopard frog has been replaced in California by the introduced bullfrog (*R. catesbeiana*) and the Rio Grande leopard frog (*R. berlandieri*), which are able to thrive in human-modified reservoirs and canals in the Yavapai leopard frog's original range (Jennings and Hayes 1994).

Mountain Yellow-legged Frog (*R. muscosa*)

This species was historically abundant in the Sierra Nevada at elevations largely above 1,829 m (6,000 ft), and also in the San Gabriel, San Bernardino, and San Jacinto mountains of southern California above 369 m (1,210 ft; Fig. 3). The mountain yellow-legged frog has disappeared from about 50% of its historical range in the Sierra Nevada and about 99% of its historical range in southern California. Some researchers believe that the widespread introduction of non-native trout into high-elevation lakes is the major reason for the decline of this species in the Sierra Nevada (Bradford 1989; Bradford et al. 1993). The species, however, experienced massive die-offs in many parts of its range during the 1970's (Bradford 1991) after several years of severe floods and drought, and continues to decline in relatively pristine areas such as wilderness areas and national parks.

Such observations indicate that present land-management practices of setting aside large tracts of land for the "protection of biodiversity" may not be adequate for ensuring the continued survival of this species. Already, the loss of this frog over large areas has negatively affected organisms such as the western terrestrial garter snake (*Thamnophis elegans*), which relied upon it as a major food source (Jennings et al. 1992). To keep these populations from extinction, resource managers may need to initiate active management efforts for mountain yellow-legged frogs (such as fish eradication programs in selected high-elevation lakes, fencing of riparian zones to exclude livestock grazing, and relocating hiking trails and campgrounds away from sensitive riparian habitats).

Northern Leopard Frog (*R. pipiens*)

This frog was historically recorded from scattered localities below 1,981 m (6,500 ft) in

Courtesy M.R. Jennings, NBS

Northern red-legged frog (*Rana aurora aurora*).

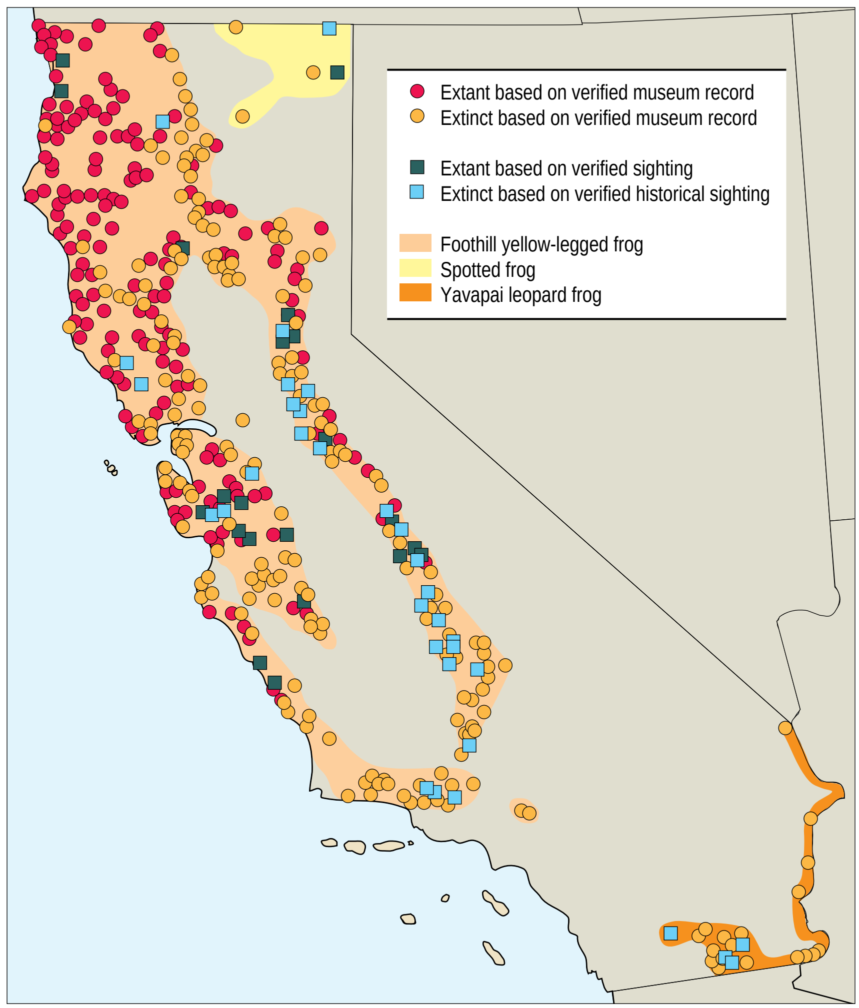

Fig. 2. Historical and current distribution of the foothill yellow-legged frog, spotted frog, and Yavapai leopard frog in California based on 3,316 museum records and 171 records from other sources. Dots indicate locality records based on verified museum specimens. Squares indicate locality records based on verified sightings (e.g., field notes, photographs, published papers). Red dots and green squares denote localities where native frogs are extant. Gold dots and blue squares indicate where native frogs are presumed extinct. Figure modified from Jennings and Hayes (1993).

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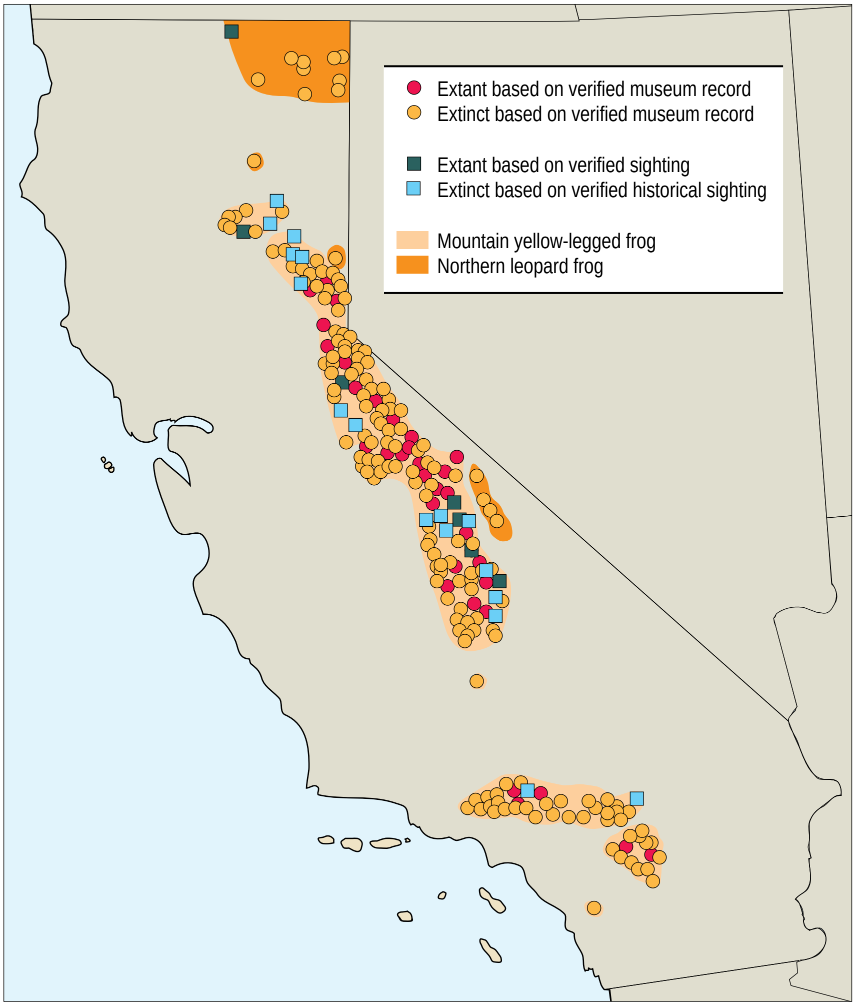

Fig. 3. Historical and current distribution of the mountain yellow-legged frog, and presumed native populations of the northern leopard frog in California based on 2,565 museum records and 673 records from other sources. Dots indicate locality records based on verified museum specimens. Squares indicate locality records based on verified sightings (e.g., field notes, photographs, published papers). Red dots and green squares denote localities where native frogs are extant. Gold dots and blue squares indicate where native frogs are presumed extinct. Figure modified from Jennings and Hayes (1993).

the eastern part of California (Fig. 3). Some populations were introduced into the state within the past 100 years (Jennings and Hayes 1993), most around the turn of the century (Storer 1925). This species has disappeared from about 95% of its range in California and is now found only in one national wildlife refuge near the Oregon border. Most localities where this frog was historically found have not changed appreciatively during the past 50 years, so the reasons for the species' decline and disappearance remain a mystery.

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Desert Tortoises in the Mojave and Colorado Deserts

by
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The desert tortoise (*Gopherus agassizii*) is a widespread species of the southwestern United States and Mexico. Within the United States, desert tortoises live in the Mojave, Colorado, and Sonoran deserts of southeastern California, southern Nevada, southwestern Utah, and western Arizona (Fig. 1). A substantial portion of the habitat is on lands administered by the U.S. Department of the Interior.

The U.S. government treats the desert tortoise as an indicator or umbrella species to measure the health and well-being of the ecosystems it inhabits. The tortoise functions well as an indicator because it is long-lived, takes 12-20 years to reach reproductive maturity, and is sensitive to changes in the environment. In 1990 the U.S. Fish and Wildlife Service listed the species as threatened in the northern and western parts of its geographic range (Fig. 1) because of widespread population declines and overall habitat loss, deterioration, and fragmentation.

Because some populations exhibit significant genetic, morphologic (*see* glossary), and behavioral differences, the Desert Tortoise Recovery Team identified six distinctive population segments (Fig. 1) for critical habitat protection and long-term conservation within the Mojave and Colorado deserts (e.g., Lamb et al. 1989; USFWS 1994). The population segments are representative of distinctive climatic, floristic, and geographic regions.

Surveys

The primary sources of information on status and trends of desert tortoise populations are from study plots established by the U.S. Bureau of Land Management and state fish and game agencies. More than 30 permanent study plots, each of which is 2.6 km^2 or larger (1 mi^2 or more), are surveyed at intervals ranging from 2 to 10 years. Study plots provide data on population characteristics, including density, size-age class structure, sex ratios and numbers of breeding females, recruitment of juveniles into the adult population, causes of death, and mortality rates (Berry 1990). Researchers use mark-recapture techniques to conduct 60-day surveys in spring for live and dead tortoises.

Trends for habitat condition on study plots are measured by using quantitative data on native and exotic annual and perennial vegetation (Berry 1990). Associated data on past and recent human activities or influences include numbers of visitors per season; density of dirt roads, trails, and vehicle tracks; levels and types of livestock grazing; and acreage disturbed by mining and mineral development and utility corridors.

The data base for the six population segments varies considerably; some segments contain several plots that have been sampled for 11-17 years, whereas others have few plots that have been sampled only 1 or 2 years (Berry 1990; USFWS 1994).

Trends

Condition and trends in tortoise populations vary within and between population segments. One measure of population condition is change in density. Examples of changes in density for nine study plots in California and Nevada are shown in Fig. 2 (Berry 1990; D.B. Hardenbrook, Nevada Division of Wildlife, and S. Slone, Bureau of Land Management, personal communication). The greatest declines in densities, for all size classes and for breeding females (up to 90%), occurred in the western Mojave segment between the 1970's and 1990's. Similar declines (30%-60%) also occurred in the eastern Colorado Desert segment between 1979 and 1992, with the greatest declines registered at the Chuckwalla Bench plot (Fig. 2). Moderate declines of 20%-25% were reported from some sites in the eastern Mojave Desert segment (Piute Valley and Goffs). The northeastern Mojave also exhibited declines on some plots (e.g., Ivanpah Valley and Gold Butte). In contrast, the northern Colorado Desert population segment showed indications of growth in the breeding adults at one plot (Ward Valley), and the upper Virgin River segment appears stable (USFWS 1994).

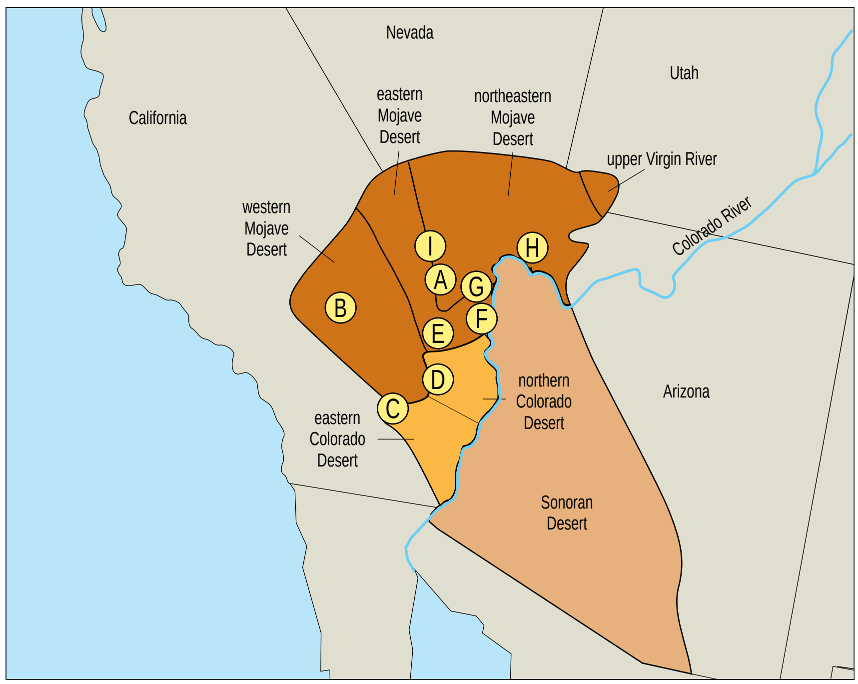

Fig. 1. U.S. range of the desert tortoise (*Gopherus agassizii*). The six population segments for desert tortoises federally listed as threatened occur in parts of the Mojave and Colorado deserts that lie north and west of the Colorado River.

Desert tortoise (*Gopherus agassizii*).

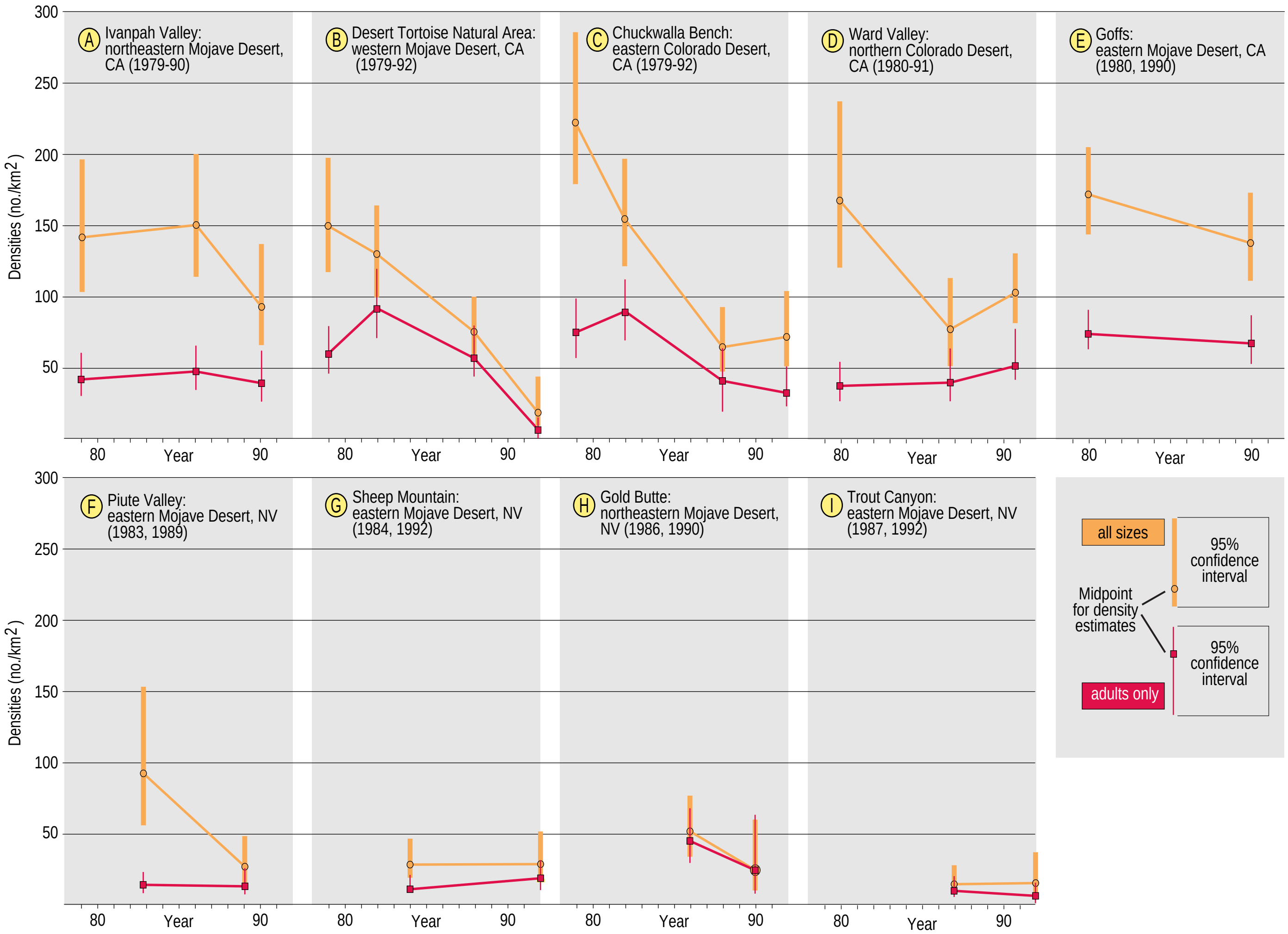

Fig. 2. Examples of changes in desert tortoise population densities at nine study sites in California and Nevada. The midpoint for density estimates of all sizes of tortoises (orange line) is shown by a dot on a bar representing the 95% confidence interval (CI); the midpoint for density estimates for adult tortoises only (red lines) is depicted by a square on a bar representing the 95% CI. Causes of declines vary by site.

Causes of population declines differed somewhat within and between population segments, but were primarily related to human activities. Higher than normal losses or mortality rates were attributed to many causes, such as illegal collecting, vandalism, upper respiratory tract disease or shell disease, predation by common ravens, crushing by vehicles both on and off roads, and trampling by livestock (BLM 1988; USFWS 1994). For example, 14.6%-28.9% of desert tortoise carcasses collected from western Mojave plots in the 1970's and early 1980's showed signs of gunshots (tortoises were shot while still alive), but only 0%-3.1% of carcasses from the less-visited eastern Mojave and northern Colorado deserts showed such signs (Berry 1986). Deaths from vehicles on paved roads were also highest in the western Mojave, where densities of dirt roads and vehicle trails are higher than elsewhere.

Of particular concern is the recent appearance of a highly infectious and usually fatal upper respiratory tract disease caused by the bacterium *Mycoplasma agassizii*. The disease, apparently introduced through the release of captive tortoises (Jacobson 1993), has caused the deaths of thousands of wild tortoises in the Mojave Desert during the last few years (K.H. Berry, unpublished data).

Fragmented and deteriorated habitats also affect population vitality. Populations in areas with high levels of exotic annual plants are declining at substantially higher rates than those in less disturbed areas.

In summary, tortoise populations occurring in relatively undisturbed and remote areas with little vehicular access and low human visitation generally were stable, or exhibited lower rates of decline than tortoise populations in areas with high levels of disturbance, high vehicular access, and high human visitation.

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Coachella Valley Fringe-toed Lizards

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Fringe-toed lizards (*Uma* spp.) inhabit many of the scattered windblown sand deposits of southeastern California, southwestern Arizona, and northwestern Mexico. These lizards have several specialized adaptations: elongated scales on their hind feet ("fringes") for added traction in loose sand, a shovel-shaped head and a lower jaw adapted to aid diving into and moving short distances beneath the sand, elongated scales covering their ears to keep sand out, and unique morphology (form or structure) of internal nostrils that allows them to breathe below the sand without inhaling sand particles.

While these adaptations enable fringe-toed lizards to successfully occupy sand dune habitats, the same characteristics have restricted them to isolated sand "islands." Three fringe-toed lizard species live in the United States: the Mojave (*U. scoparia*), the Colorado Desert (*U. notata*), and the Coachella Valley (*U. inornata*). Of the three, the Coachella Valley fringe-toed lizard has the most restricted range and has been most affected by human activities. In 1980 this lizard was listed as a threatened species by the federal government.

In 1986 the Coachella Valley Preserve system was established to protect habitat for the Coachella Valley fringe-toed lizard. This action set several precedents: it was the first Habitat Conservation Plan established under the revised (1982) Endangered Species Act and the newly adopted Section 10 of the act, it established perhaps the only protected area in the world set aside for a lizard, and its design was based on a model of sand dune ecosystem processes, the sole habitat for this lizard. Three disjunct sites in California, each with a discrete source of windblown sand, were set aside to protect fringe-toed lizard populations: Thousand Palms, Willow Hole, and Whitewater River. Collectively, the preserves protect about 2% of the lizards' original range.

Eight years after the establishment of the preserve system, few Coachella Valley fringe-toed lizards exist outside the boundaries of the three protected sites. Barrows (author, unpublished data) recently identified scattered pockets of windblown sand occupied by fringe-toed lizards in the hills along the northern fringe of the valley, but only at low densities. Fringe-toed lizard populations within the protected sites have been monitored yearly since 1986. During this period, California experienced one of its most severe droughts, which ended in spring 1991. Numbers of fringe-toed lizards within the Thousand Palms and Willow Hole sites declined during the drought, but rebounded after 1991 (Fig. 1). By 1993, after three wet springs, lizard numbers had increased substantially.

Lizards at the Whitewater River site were intensively monitored since 1985 by using mark-recapture methods to count the population on a 2.25-ha (5.56-acre) plot. In 1986 this site

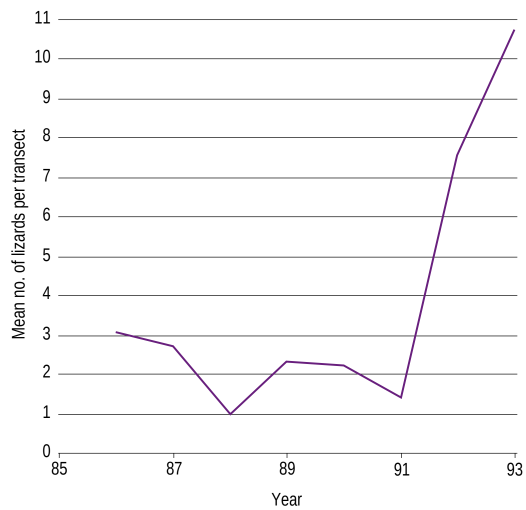

Fig. 1. The mean number of lizards per transect at the Thousand Palms and Willow Hole sites, 1986-93. Data were pooled from five 10 x 1,000 m (32.8 x 3,281 ft) transects. All transects were sampled six times each year, and all sampling was conducted within a 6-week span in the late spring of each year.

had the highest population density of the three protected sites. As with the other two sites, the Whitewater River population declined throughout the drought, but only increased slightly after the drought broke in 1991 (Fig. 2). Compounding the drought effect, much of the fine sand preferred by fringe-toed lizards was blown off the site during the dry years. This condition was unique to the Whitewater River

Coachella Valley fringe-toed lizard (*Uma inornata*).

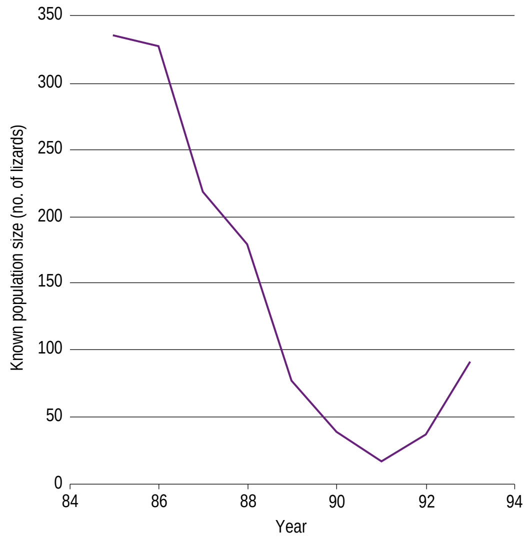

Fig. 2. The known population size of a Coachella Valley fringe-toed lizard population on a 2.25-ha (5.56-acre) study plot on the Whitewater River preserve.

site; the other two protected sites have much deeper sand deposits and are less susceptible to wind erosion. New windblown sand was deposited on the Whitewater River site in 1993 after a period of high rainfall. The population appears to be increasing in response to these favorable conditions.

The decline in fringe-toed lizards during the monitoring period appears to be the result of responses to natural fluctuations in habitat. The dynamic nature of sand dune systems, coupled with the lizards' apparent sensitivity to drought, underlines the importance of preserve design. Appropriate designs anticipate the effect of natural habitat fluctuation.

The ecological model that governed the design of the Coachella Valley Preserve system was reevaluated in 1993 with one disturbing result. A primary sand source was identified that supplies the sand dunes at the Thousand Palms site, but was not emphasized sufficiently in the original model and design. Fortunately, the sand source and its path to the existing preserve have not been affected severely by human development at this time, so options for correcting the design's shortcomings are still available. The fringe-toed lizard population sustained by this sand source has been the largest of the three sites for the past few years. Monitoring the lizards without investigating ecosystem processes would not have identified the design error until it was too late to correct.

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Disappearance of the Tarahumara Frog

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In the spring of 1983 the last known Tarahumara frog in the United States was found dead. Overall, the species seems to be doing well in Mexico, although the decline of more northern populations are of concern. The Tarahumara frog (*Rana tarahumarae*) inhabits seasonal and permanent bedrock and bouldery streams in the foothills and main mountain mass of the Sierra Madre Occidental of northwestern Mexico. It ranges from northern Sinaloa, through western Chihuahua and eastern and northern Sonora, and until recently into extreme south-central Arizona (Fig. 1). Arizona localities, all in Santa Cruz County, include three drainages in the Atascosa-Pajarito Mountains (Campbell 1931; Little 1940; Williams 1960) and three in the Santa Rita Mountains (Hale et al. 1977).

Population Estimates, 1975-93

We have drawn our review from museum records, the published literature, and reports, journal entries, and personal observations by the authors, other biologists, and knowledgeable persons. From May 1975 through June 1977, we conducted an ecological, demographic, and life-history study of the population at Big Casa Blanca Canyon (Santa Rita Mountains).

Between 1980 and 1993, we visited 22 of 30 historical Tarahumara frog localities. We surveyed 43 additional streams with potential habitat and found Tarahumara frogs at 25 new localities in Mexico. Localities were extensively searched, often both day and night, sometimes repeatedly. Frogs and tadpoles were counted, size-classed, and sexed when possible. Time, streamwater pH, air, substrate and water temperatures, habitat description and condition, and relative abundances of other aquatic vertebrates were noted.

During the summers of 1982-83, rain samples were collected at The Nature Conservancy's Sonoita Creek and Canelo Hills preserves for pH determination and heavy metal analysis. Both sites are within 22-56 km (14-35 mi) of declining frog populations and 64-129 km (40-80 mi) north and northwest of copper smelters. Streamwater samples from sites of declining populations in Sycamore and Big Casa Blanca canyons in Arizona and Carabinas Canyon in northeastern Sonora were also collected for pH and heavy metal analyses.

Decline of Populations

In April 1974, 27 dead and dying Tarahumara and leopard frogs were observed at Sycamore Canyon, Atascosa-Pajarito Mountains, the best-known and most frequently visited Tarahumara frog population. The last sightings of Tarahumara frogs in that range were in the summer of 1974.

The decline of the Santa Rita Mountains population began in 1977 (Fig. 2). Total numbers of frogs (adults and juveniles) captured plummeted from 252 in 1976 to 46 in 1977; estimated total population size fell from a maximum 1,020 frogs to 625 (Hale and May 1983). In June 1977 some captured frogs became unresponsive and often died, apparently from the stress of capture, a response not previously observed. In 1978 no frogs marked in prior years, nor tiny larvae attributable to that year's breeding, were found. Larger tadpoles from 1976-77 persisted. Twenty newly metamorphosed frogs were observed in 1978 and 40 in 1979; from 1980 to 1982 we saw one to three frogs attributable to those frogs. In spring 1983 the last known Tarahumara frog in the United States was found dead. Repeated visits (some times yearly) to all former Arizona localities have yielded no additional sightings.

Three of seven populations studied from 1981 to 1986 in northern Sonora appeared healthy, with adult and juvenile frogs as well as both small and large larvae, suggesting a stable, reproductive population. Frogs were not seen at three other sites where they had been found in the 1970's and early 1980's. The last population, in Carabinas Canyon, Sierra El Tigre, which contained numerous frogs and tadpoles, was in the initial stages of a major decline when first observed in fall 1981. Within a year all frogs had disappeared from the downstream end of this population, but frogs in the upper portion of the drainage appeared to have suffered no decline in numbers through our most recent visit to the site in 1986.

Carabinas Canyon frogs displayed clinical signs suggestive of heavy metal poisoning, including irregular muscular activity and failure of muscular coordination (ataxia), partial paralysis of the hind legs, dilated pupils unresponsive to light, and a loss of the righting response. The skin was often dry on the head and back. Symptoms were amplified by the stress of capture and handling. Frogs displaying obvious signs of heavy metal poisoning were already dying.

Field examinations of dead frogs showed no evidence of gross pathological disorders. Skin cultures showed no common pathogens; species representing probable normal skin flora and opportunistic secondary pathogens attacking a debilitated host were present. Histopathological examinations of five dying frogs (E. Jacobson, J. Hillis Miller Health Center, College of Veterinary Medicine, Gainesville, Florida) revealed no gross pathologies (Hale and May 1983; Hale and Jarchow 1988).

Populations of Chiricahua and Yavapai leopard frogs (*Rana chiricahuensis* and *R. yavapaiensis*) declined with the Tarahumara frog where they occurred together, although leopard frogs were not eliminated from most Tarahumara frog sites. In Sycamore Canyon, Chiricahua leopard frogs have managed to maintain a small but viable population near Yank Spring, but numbers decrease downstream in previously favorable leopard frog habitat. The Chiricahua leopard frog has experienced catastrophic declines elsewhere, and is in danger of disappearing from most of its range (Clarkson and Rorabaugh 1989).

Rain collected at the Sonoita Creek and Canelo Hills preserves in the summers of 1982

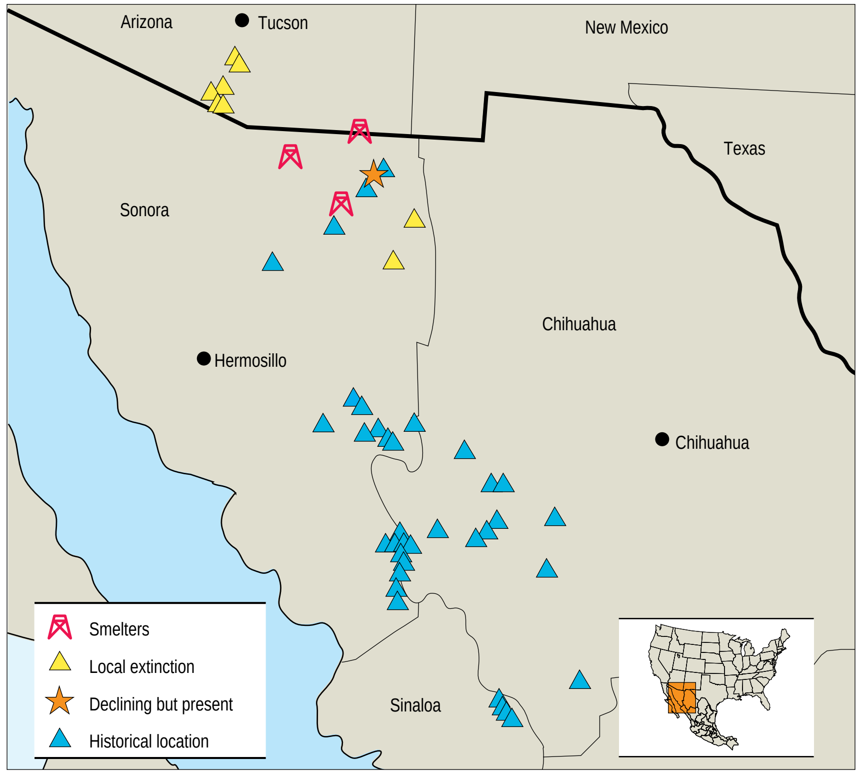

Fig. 1. Range of the Tarahumara frog, *Rana tarahumarae*. Copper smelters are at Douglas, AZ (now closed), and Cananea and Nacozari, Sonora. Historical locations include both surveyed populations that appeared stable, and unvisited historical localities (Campbell 1931; Little 1940; Williams 1960; Hale et al. 1977; Hale and May 1983; Hale and Jarchow 1988).

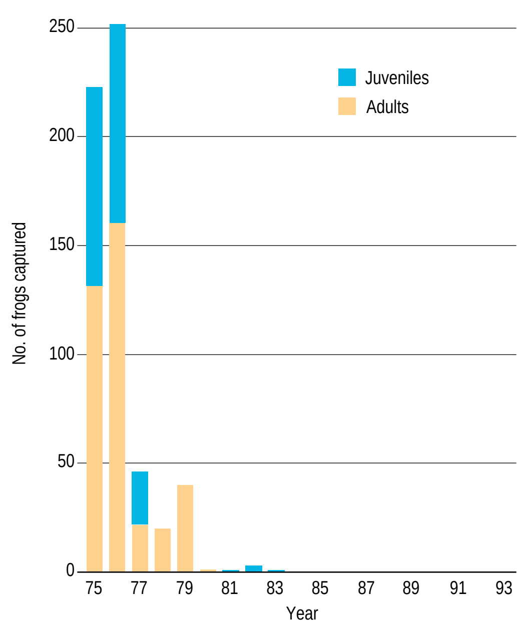

Fig. 2. Number of Tarahumara frogs captured 1975-93, Big Casa Blanca Canyon, Santa Rita Mountains, Santa Cruz County, AZ (Hale and May 1983).

Courtesy C. R. Schwalbe

Tarahumara frog (*Rana tarahumarae*) in Mexico.

and 1983 was consistently very acidic, attributed primarily to particulates produced by a copper smelter in Douglas, Arizona (Blanchard and Stromberg 1987), which has since been shut down. The alkaline soils in the area may buffer the streams from the acid rain; stream pH values were always slightly basic.

Analyses of water from affected streams showed consistently elevated levels of cadmium, a toxic metal, especially in relation to levels of the essential metal, zinc. In several species of vertebrates, sensitivity to cadmium toxicity is reduced with zinc supplementation (Supplee 1963; Webb 1972). At Sycamore Canyon and Big Casa Blanca Canyon localities, frogs survived longest near springs where zinc concentrations were highest. Levels of arsenic in streamwater were occasionally elevated (Hale and Jarchow 1988).

Although the proximity of operating copper smelters is correlated with population declines in Tarahumara and leopard frogs, exact causes of declines are not clear. No declines in frogs were noted until the 1970's, yet copper smelter emissions were much higher in the areas of declines in the early 1900's than recently. One of our hypotheses that accounts for the timing of the declines relates them to a long-term leaching of acid-soluble zinc from canyon walls, accumulation of insoluble cadmium in stream sediment, and sediment accumulation in stream pools from infrequent heavy rains before declines.

In southern and central Sonora, ranid frog populations appeared stable and reproductive at least through 1986; no population declines or extirpations were noted, either of Tarahumara or leopard frogs. Populations visited since 1986 do not appear to be declining.

Conclusions

We are confident that the Tarahumara frog no longer occurs in the United States, based upon repeated surveys of historical and potential habitat in southern Arizona. Although repeated surveys since 1983 in Mexico have not been as extensive as in the United States, sites visited in central and southern Sonora apparently continue to support healthy frog populations.

We conclude that the Tarahumara frog is not threatened with extinction throughout its range at this time, although the sudden declines and local extirpations in northern populations, coincident with declines of leopard frogs, are a serious concern.

State and federal resource management agencies in both Arizona and Sonora, Mexico, with independent biologists and the Arizona-Sonora Desert Museum (ASDM) and Centro Ecologico de Sonora have formed the Tarahumara Frog Reestablishment Oversight Group. This group proposes to reestablish the Tarahumara frog in selected historical sites and maintain captive frog populations at ASDM and elsewhere to provide stock for additional reintroduction. By intensively monitoring reintroduced populations and measuring important environmental variables we hope to determine the cause of declines in native ranid frogs in this area. Rain, streamwater, and air quality will be assessed continuously at each site, including pH, heavy metals, solar radiation (especially ultraviolet), and air particulates. Stream bottom substrate and tissue samples from frogs and frog prey and predator species will be sampled for heavy metals. Only after the causes of the declines have been identified and corrected can we expect long-term reestablishment of Tarahumara frogs and recovery of leopard frogs.

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