

# 3 The Global Status of Reptiles and Causes of Their Decline

*Brian D. Todd, John D. Willson, and J. Whitfield Gibbons*

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Reptiles have been considered by some to be of “minor importance,” and their disappearance has been suggested to “not make much difference one way or the other” (Zim and Smith 1953). Linnaeus himself described reptiles in his 1758 *Systema Naturae* as “foul and loathsome animals ... abhorrent because of their cold body ... fierce aspect ... and squalid habitation.” Thankfully, such sentiments are increasingly outdated as scientists reveal the significant roles that reptiles play in many ecosystems.

Although reptiles remain among the least studied vertebrate groups and are still frequently considered of less general interest than other fauna (Gibbons 1988; Bonnet et al. 2002), interest in the preservation of biodiversity, and consequently interest in reptile conservation, is growing. Declines of reptile populations, whether unnoticed or widely documented, are troubling not just because of the ecological relevance of reptiles to many habitats, but also because they portend a general decay of environmental health similar to declines of other species. Regardless of the motivation, the desire to conserve reptiles and to better understand their ecology requires knowledge of their status, distribution, and the factors that contribute to their decline.

Our goals in this chapter are to describe the major anthropogenic threats faced by reptiles and to summarize the perceived global conservation status of the traditionally recognized major taxonomic groups of reptiles. Specifically, we discuss the Crocodylia, Squamata, and Testudines, all historically included in the class Reptilia. Unfortunately, because so little is known about the basic ecology, distribution, and status of amphisbaenids, we exclude discussion of them from this chapter. Likewise,



given the low diversity (2 species) and extreme geographical restriction of tuataras (Sphenodontia), we do not separately review this group. Our specific objectives in this chapter are to

- 1) call attention to the lack of data on the status of reptile populations and describe strategies for documenting range-wide and localized declines,
- 2) describe the major threats facing reptile populations globally, and
- 3) describe current patterns of imperilment in reptile populations.

Two major reviews have recently described the main threats to reptile populations, and we expand on these using recent examples from an ever-growing body of literature (Gibbons et al. 2000; Irwin and Irwin 2005).

### 3.1 DETERMINING THE STATUS OF REPTILE SPECIES AND POPULATIONS

As with many fauna in recent years, conservation biologists have raised concerns over reported reptile declines, some of which have garnered widespread recognition (e.g., Asian turtle crisis; Buhlmann et al. 2002). Nevertheless, the considerable and notable lack of information regarding the status of reptiles in most regions has hampered full understanding and appreciation of their current plight. Without question, some reptile populations have been extirpated and numbers of some species have declined with little indication of the underlying cause. In some cases, concerns about declines are based wholly on anecdotal observations or on a growing perception of a species' rarity without accompanying quantitative data. Clearly, a primary goal of herpetologists and wildlife biologists should be to clarify the global status and distribution of reptile populations.

The World Conservation Union (IUCN) has been a global leader in assessing the status of many floral and faunal species as part of its ongoing Red List program (Baillie et al. 2004). Although the IUCN has comprehensively assessed birds, mammals, and amphibians to date, a global reptile assessment was only recently begun. Currently, reptiles remain one of the least known vertebrate taxa, and the conservation status of only about 6% of species has been assessed (Baillie et al. 2004). Some impediments to determining the status of reptiles derive from difficulties in studying or monitoring their populations. For example, many reptiles are characterized by cryptic coloration or behavior, which can impede observation or capture (Zug et al. 2001; Dorcas and Willson 2009). Detectability of reptiles also depends to some degree on the survey technique used, the seasonal or daily timing of surveys, and the environmental conditions during which surveys are conducted (Todd et al. 2007). Additionally, the status of reptiles can be assessed on at least 2 scales: local populations and regional distributions.

Concerns over declines of some reptiles arise from general impressions of range-wide or regional contractions in species' distributions. Obviously, such declines are cause for alarm because they may point to wide-ranging or systemic threats to species. However, adequate quantitative documentation of range-wide or regional declines in reptile species is infrequent, possibly due to the time and collaboration required to collect sufficient data at such large spatial scales. Nevertheless, successful models of range-wide investigations do exist and demonstrate the usefulness of constructing large-scale overviews of changes in species' distributions. In 1 example, investigators compiled distributional information from published scientific records, unpublished reptile surveys, museum databases, state natural heritage programs, and contributions by individual researchers to compare the historical vs. currently known distribution of the southern hognose snake, *Heterodon simus* (Tuberville et al. 2000). This collaborative effort revealed substantial declines of the species in portions of its range and provides a powerful example for other studies of distributional change in reptiles. In some cases, resurveying habitats across a species' range can also reveal that populations may be more widespread than previously suspected, as was the case for the sharp-tailed snake, *Contia tenuis*, in Oregon (Hoyer et al. 2006).



At a finer resolution, studies of individual reptile populations are also useful for revealing the plight of species, albeit at a more limited scope. As pointed out by other authors, studies of individual populations should use standardized methods to monitor changes in abundance or density over long periods of time (Gibbons et al. 2000). As an example, a long-term study of the abundance of eastern kingsnakes, *Lampropeltis getula*, revealed an alarming disappearance of the species from 1 site over 30 years (Winne et al. 2007). Although the decline could not be attributed to any single cause in this case, the examination of local populations is often more likely to reveal specific causes for a species' decline than are range-wide studies. Ultimately, a combination of studies at multiple scales will provide the most comprehensive assessment of the status and distribution of reptiles, as has been the case with the Texas horned lizard, *Phrynosoma cornutum* (Donaldson et al. 1994).

Lastly, as reptile populations fall under increasing scrutiny by scientists and conservation managers, it will be important to distinguish between natural declines and anthropogenic ones, and even to determine whether fluctuations in distribution or abundance represent "declines" per se (Alford and Richards 1999; Pechmann 2003). All animal populations presumably experience some level of normal fluctuation in abundance that will vary depending on the species or population in question. Thus, short-term monitoring that provides limited snapshots of population size may reveal current status but will not expose longer-term population trends or their causes (Gibbons et al. 2000). For this reason, the value of long-term studies and the data they generate cannot be overstated. Nevertheless, accumulation of numerous accounts from short-term studies may reveal a declining trajectory that can lend credence to conservation concerns for a given species or population (Gibbons et al. 2000). The scientific community would do well to take notice of the incredible mobilization of inventory and monitoring programs and other research activities that have followed recognition of the acute imperilment of many amphibians. Proactive recognition of the need to closely monitor reptiles may be instrumental in preventing or mediating their declines and minimizing the economic cost of reactive, and sometimes belated, conservation efforts.

### 3.2 FACTORS CONTRIBUTING TO REPTILE DECLINES

Establishing a causal link between any specific factor and declines of reptile populations can be difficult but is of foremost concern for effective conservation. Although in some cases 1 factor may weigh heavily on a population, multiple interacting factors nearly always affect a species' abundance and distribution. Several factors have been identified as threats to reptile populations and are implicated in declines of at least some reptiles, including habitat loss and fragmentation, unsustainable removal, anthropogenic environmental contamination, climate change, invasive species, disease and parasitism, and trophic cascades, and we discuss these in detail in the following sections. Two other seldom mentioned but nonetheless important factors bearing on the status of reptile populations are social apathy and special or political interests. Indeed, social apathy can be a major obstacle to reptile conservation because many reptiles are subjects of personal derision, a problem that must be overcome before appropriate motivation can spur conservation interest (Gibbons 1988). Similarly, the willingness of nongovernmental organizations and state, provincial, or national governments to recognize the plight of declining species and the need for conservation effort often depends on special or political interests and will undeniably have considerable impact on the persistence of many reptiles.

#### 3.2.1 HABITAT LOSS

Habitat loss, including degradation, fragmentation, or conversion for other use, is typically regarded as the single greatest cause of faunal declines globally (Wilcove et al. 1998; Sala et al. 2000). Thus, the fact that habitat loss is considered to be the leading cause of reptile declines is not surprising (Mittermeier et al. 1992; Gardner et al. 2007). Habitat loss due to conversion of land for human use typically occurs for agriculture, housing or infrastructure development, commercial forestry, and to support recreation, including constructing golf courses or dredging lakes and other aquatic habitats.



Habitat loss from conversion can result in direct animal mortality that is often difficult to quantify. Nevertheless, mortality of turtles and aquatic snakes during lake dredging has been documented (Aresco and Gunzburger 2004), and entombment of live gopher tortoises (*Gopherus polyphemus*) during land development was widespread in Florida prior to a ban on the practice (Cox 2007). Ultimately, however, immediate mortality from habitat alteration likely poses less threat than the subsequent long-term, indirect effects of habitat loss and degradation on survival and reproduction.

Habitat loss can affect reptiles indirectly by limiting their ability to meet ecological needs for survival and reproduction. For example, many reptiles decline in abundance over time following the clearing of primary forest or conversion to plantation forest (Glor et al. 2001; Kanowski et al. 2006). In fact, the decline of multiple reptile species in the southeastern United States has followed widespread and nearly complete loss of native longleaf pine habitat (Ware et al. 1993; Gibbons et al. 2000). At a finer scale, 1 study from the southeastern United States demonstrated that planted pine forests and recent clear-cuts supported reduced abundances of small snakes compared with open-canopied partially harvested forests (Todd and Andrews 2008). The precise mechanisms of decline remain unknown but are presumably related to a general degradation of habitat quality from anthropogenic land conversion. Additional examples of the effects of habitat degradation include the loss of foraging and refuge due to bush-rock collection, which has contributed to the decline of the Australian broad-headed snake (*Hoplocephalus bungaroides*; Shine et al. 1998), and reductions in lizard abundance due to human-induced bush encroachment in Africa (Meik et al. 2002).

Loss and degradation of aquatic habitats also pose a serious threat to reptiles. Notable examples include declines of the crocodylian fauna of the Ganges and Yangtze Rivers (IUCN 2009), which have become increasingly imperiled following damming, flow modification, and general degradation of river habitat (Dudgeon et al. 2006). Similarly, channelization and dam building have been implicated in declines of river-dwelling North American map turtles (*Graptemys* spp.; Kofron 1991). The vulnerability of sea turtles to coastal development that degrades or eliminates nesting habitat has been appreciated for decades (Lutcavage et al. 1997; Spotila 2004). Likewise, the effects of terrestrial habitat alteration that disturbs or eliminates nesting and refuge sites of freshwater turtles can also be severe (Buhlmann 1995; Burke and Gibbons 1995). Many semiaquatic snake species that use wetland habitats share the plight of turtles if wetlands are lost or terrestrial habitat around sensitive aquatic resources is altered (Roe et al. 2003, 2004; Willson et al. 2006).

Habitat fragmentation is the emergence of discontinuities in an organism's preferred environment. Habitat fragmentation may occur due to natural processes but increasingly results from anthropogenic habitat loss or land conversion that isolates remaining patches of suitable habitat. The degree to which habitat fragmentation threatens a species depends on how greatly a species' movements are affected by the interspersed barriers that separate remaining usable habitat. Some lizard, snake, and turtle species are vulnerable to habitat fragmentation, with general declines in abundance being reported (Dodd 1990; Kjos and Litvaitis 2001; Driscoll 2004), whereas others are not (Driscoll 2004). In other cases, habitat fragmentation affects the demography of remaining reptile populations. For example, patch size was positively correlated with abundance, survivorship, and recruitment of Florida scrub lizards, *Sceloporus woodi* (Hokit and Branch 2003). Of particular concern is the possible role that roadways play in fragmenting habitat. Because road mortality of turtles and snakes is often high (Aresco 2003; Gibbs and Steen 2005; Andrews and Gibbons 2008; see the next section), roads effectively become barriers that separate and isolate habitat (Roe et al. 2006). Again, however, species will differ in the extent to which roads act as barriers to movement (Andrews and Gibbons 2005), and therefore fragment populations.

### 3.2.2 UNSUSTAINABLE REMOVAL

Removal of reptiles from wild populations occurs both commercially and noncommercially for food, "traditional" medicine, curios, and the pet trade, as well as unintentionally as by-catch in



other harvesting activities and, increasingly, as a result of road mortality. Although removal per se is not necessarily harmful to population persistence — many reptile populations could presumably sustain some low level of harvest — removal at unsustainable rates is a serious threat that places many reptile populations and species in peril. To date, few studies have demonstrated sustainability of removal activities on reptile populations. In contrast, studies documenting intense levels of reptile harvesting and subsequent declines in wild populations are common.

Perhaps the mostly widely recognized removal-driven peril results from the ongoing use of reptiles for food, skins, or “traditional” medicines. Imperilment of Asian turtles due to unsustainable removal has reached crisis levels and has grim consequences for the persistence of many freshwater and terrestrial turtles if not remedied (Buhlmann et al. 2002). The exploitation of turtles, however, is not restricted to Asia; many Central and South American cultures relish turtles and their eggs, resulting in continued threats to both freshwater and marine turtle populations (Lagueux 1991; Thorbjarnarson et al. 1997). Moreover, the consumptive use of reptiles is not limited to turtles. In Asia, snakes face rapidly growing pressure from exploitive use, with as many as 1 million snakes being harvested in northeast China and nearly 8 million kg traded each year across the country (Zhou and Jiang 2004, 2005). Likewise, in Cambodia, an estimated 6.9 million aquatic snakes are removed annually from Tonle Sap Lake to feed the growing crocodile farms in that region (Brooks et al. 2007). Subsequently, hunters have reported a 74% to 84% decline in snake catch from 2000 to 2005 (Brooks et al. 2007). Pythons (*Python* spp.), too, face significant harvest pressure in many parts of Indonesia (Shine et al. 1999). Monitors (*Varanus* spp.) and tegus (*Tupinambis* spp.) are heavily harvested for their skins at rates of as much as 1 million animals per year in the case of South American tegus (Pianka and Vitt 2003; Mieres and Fitzgerald 2006). In Africa, bushmeat consumption often extends to highly endangered species such as the dwarf crocodile, *Osteolaemus tetraspis* (Willcox and Nambu 2007). The complete list of affected species is lengthy (see also reviews in Gibbons et al. 2000; Irwin and Irwin 2005) and a precautionary policy of preventing massive exploitation until sustainable removal limits are identified appears to be the best method of ensuring the persistence of reptile populations.

The commercial removal of reptiles from wild populations for use as pets is another consumptive use affecting reptiles globally. In many cases, evidence linking collection of animals for the pet trade to declines of wild reptile populations is lacking because long-term studies of the status of reptile populations following targeted collecting have not been conducted. Nevertheless, collection of reptiles for the pet trade may endanger wild populations due to the large scale at which some reptiles are collected. Many species of turtles, snakes, chameleons, and other lizards are under increasing demand, and collection in wild populations continues at high levels (Reed and Gibbons 2003; Carpenter et al. 2004; Schlaepfer et al. 2005) that can threaten population persistence (Webb et al. 2002).

Removal of individual reptiles also occurs as by-catch in fisheries and from intentional and unintentional road mortality. Mortality of long-lived sea turtles in longline fisheries and shrimp nets is likely unsustainable and contributes to the rapid and ongoing declines of many sea turtle species (Lewison et al. 2003, 2004). Similarly, both commercial and recreational crab trapping have been implicated in declines of the North American diamondback terrapin, *Malaclemys terrapin* (Bishop 1983; Dorcas et al. 2007), a species already heavily depleted from commercial exploitation for food in the 1800s to early 1900s (Carr 1952). In developed countries, roads represent an additional, substantial, and often ignored source of mortality in many reptile populations. Snakes and turtles are probably hardest hit by road mortality because they are large, often move slowly, and are sometimes direct targets for persecution by motorists (Andrews and Gibbons 2005; Ashley et al. 2007). Increasingly male-biased sex ratios in turtle populations attest to the potential long-term effects of road mortality on reptiles; female turtles are more likely to cross roads during nesting forays and are enticed to oviposit on road shoulders due to the presence of sunny nesting habitat, leading to disproportionately higher female mortality (Aresco 2005; Steen et al. 2006).



### 3.2.3 ENVIRONMENTAL CONTAMINATION

Release of contaminants — including pesticides, herbicides, heavy metals, and radioactive waste — into the environment has been listed as 1 of the 6 major contributors to the global decline of reptiles (Gibbons et al. 2000). Reptiles exhibit a suite of ecological and life history characteristics that make them particularly vulnerable to contaminants (Hopkins 2000). With the exception of a few lizard and turtle species, reptiles are strictly carnivorous, and many occupy high trophic positions within food webs. Thus, reptiles are at risk from biomagnification of contaminants. Additionally, many reptiles are long-lived and have small home ranges compared to similar-sized endotherms, making them susceptible to long-term contaminant exposure and subsequent bioaccumulation (Hopkins 2000; Shelby and Mendonca 2001; Bergeron et al. 2007). Although reptiles may be at particularly high risk from contaminants, they are currently the least studied vertebrate group in ecotoxicology (Hopkins 2000; Chapter 1, this volume). Within reptiles, ecotoxicological research has primarily been restricted to turtles and crocodylians, and our knowledge of the effects of toxins on squamates is still limited (Campbell and Campbell 2001).

The most overt measurable effect that contaminants can have on reptiles is direct mortality of individuals resulting from exposure. Several studies have reported mortality of reptiles in association with intentional (e.g., pesticide application) or accidental (e.g., spills or contaminant leakage) introduction of toxins into the environment (reviewed in Campbell and Campbell 2000, 2001), but few authors have related such acute mortality events to population declines. However, Romero and Wilkelski (2002) noted population declines of Galapagos marine iguanas (*Amblyrhynchus cristatus*) following a low-level oil spill. Declines were not observed on islands that remained unaffected by the spill. Romero and Wilkelski (2002) speculated that iguanas died of starvation after the digestive bacteria in their guts were killed by oil residues found in their diet of marine algae. In a similar example, Ernst et al. (1994) noted the disappearance of yellow-blotched map turtles (*Graptemys flavimaculata*) from sections of river immediately downstream from a paper mill. However, it is unclear if changes in turtle abundance represented population declines or movement of turtles out of contaminated areas due to lack of prey (aquatic invertebrates) or other factors. To date, reports of population declines associated with environmental contamination are largely restricted to turtles and large lizards. Cryptic behavior and/or low activity levels (which result in low capture rates during surveys; Dorcas and Willson in press) of many snakes and smaller lizards would conceal mortality events and hamper detection of population declines.

Although less obvious than direct mortality, sublethal effects of contaminants may be more detrimental to the long-term persistence of reptile populations. High tissue loads of various contaminants have been documented from reptiles in the field (e.g., lizards and snakes; reviewed in Campbell and Campbell 2000, 2001; Bergeron et al. 2007; this volume), and sublethal effects of contaminants on reptile locomotor performance (e.g., Hopkins et al. 2005; Holem et al. 2006; Hopkins and Winne 2006; DuRant et al. 2007a) and metabolic energy consumption (e.g., Hopkins et al. 2005; DuRant et al. 2007b) have been demonstrated in the laboratory. Although these studies provide insight on the mechanisms linking sublethal contaminant exposure to population dynamics, few studies have attributed declining reptile populations to sublethal contaminant exposure. A well-cited exception is the decline of American alligators (*Alligator mississippiensis*) in a Florida lake contaminated with estrogenic compounds (Guillette et al. 1994; Semenza et al. 1997). Alligator declines were attributed to reproductive failure resulting from reduced testosterone levels and gonadal malformations. Likewise, Shelby and Mendonca (2001) found reduced testosterone levels in some male yellow-blotched map turtles (*Graptemys flavimaculata*) from polluted habitats, suggesting that effects of pollutants on reproduction may have been responsible for population declines previously observed at that site. Although much of the investigation of indirect effects of contaminants on reptiles has focused on the effects of endocrine disrupters on reproduction, exposure to contaminants may also affect energy acquisition and expenditure. For example, Hopkins et al. (1999) found that banded watersnakes (*Nerodia fasciata*) collected from a wetland polluted with coal combustion waste had



elevated tissue concentrations of trace metals and standard metabolic rates that were 32% higher than those of snakes from an unpolluted site. Elevated metabolic rates presumably limit energy availability for growth and reproduction in snakes from contaminated sites.

Finally, consider possible synergistic effects of contaminants with other threats facing reptiles. For example, a plausible scenario is one in which sublethal exposure to contaminants compromises immunocompetence, resulting in outbreaks of opportunistic pathogens that might otherwise be benign or manageable under normal circumstances. Although these types of questions have not been addressed in reptiles, consideration of the effects of multiple stressors on reptiles will undoubtedly become increasingly important as human populations continue to grow and expand around the globe.

### 3.2.4 CLIMATE CHANGE

Global climate change has been an ongoing process throughout the evolutionary history of reptiles and has been nearly continuous in the past 65 million years (Zachos et al. 2001). However, given indications that recent global climate warming is occurring at a pace unprecedented in recent history (IPCC 2007), rapid climate change is particularly relevant in our consideration of threats to reptile populations. Possible effects from climate change fall broadly into categories of direct and indirect effects, both of which have either caused or are expected to cause changes in reptile populations.

Because reptiles are ectothermic, they are highly dependent on suitable external temperatures to regulate their own body temperatures and support metabolic and other functions. Subsequently, direct effects of climate change may manifest as changes in growth rates or the age at onset of reproductive maturity, as shown in painted turtles, *Chrysemys picta* (Frazer et al. 1993). Theoretical models further suggest that changes in global climate can have profound effects on reptile populations. Dunham (1993) used individual-based models to estimate physiological responses of Big Bend Canyon Lizards (*Sceloporus merriami*) to climate change and predicted that increases in air temperature of 2 °C to 5 °C could constrict activity sufficiently to drive populations to extinction. Others have expressed concern that climate change may have a considerable impact on reptiles with temperature-dependent sex determination, such as some turtles, lizards, and crocodylians, by altering sex ratios within populations (Janzen 1994).

Other obvious effects that global climate change may have on reptile populations include direct and indirect influences on habitat suitability. For example, changes in temperature and precipitation may directly affect the habitability of a reptile's environment and could cause shifts in reptile distributions at a large scale. Araújo et al. (2006) explored possible scenarios of habitat change in Europe and concluded that, although suitable habitat for European reptiles is likely to expand under most circumstances, limited dispersal abilities of reptiles may increase their vulnerability to climate change. Dispersal capability may be further constrained by the ever-increasing habitat fragmentation that accompanies human population growth. Furthermore, regional changes in precipitation and temperature regimes are likely to broadly affect community composition, and some landscapes may change dramatically (Guertin et al. 1997; Still et al. 1999). Whitfield et al. (2007) suggested 1 instance of indirect effects of climate change when they documented a steady decline in several lizard species at La Selva, Costa Rica, over a 35-year period. They attributed declines to climate-driven reductions in the quantity of leaf litter. Also, severe climatic events are expected to become more frequent due to the destabilization of regional weather patterns under many global warming scenarios (IPCC 2007). Subsequently, droughts or other meteorological events such as cyclones may negatively affect reptile populations (Seigel et al. 1995; Willson et al. 2006).

### 3.2.5 INVASIVE SPECIES

Recent expansion of human populations and increases in global transportation and trade have resulted in introduction and establishment of many species in areas outside of their native geographic range. Many introduced species have subsequently proliferated, resulting in severe ecological and



economic damage (Pimentel et al. 2000). Consequently, invasive exotic species are currently recognized as one of the foremost threats to global biodiversity (Park 2004), including reptiles (Gibbons et al. 2000).

Invasive exotic species affect native species in a variety of ways. One of the most obvious ways introduced animals can affect reptile populations is by directly preying upon them. Predator introductions that have resulted in the decline or extirpation of many species are especially obvious on island ecosystems that were previously bereft of predators. For example, introduced predators (mongoose, *Herpestes javanicus*, and rats, *Rattus rattus*) have been identified as the single greatest threat to snakes on the Lesser Antilles and have been implicated in at least 6 historical snake extirpations and at least 1 historical extinction in that region (Henderson 2004). Similarly, introduction of brown treesnakes (*Boiga irregularis*) to the island nation of Guam has devastated the vertebrate fauna of that island, including populations of several native lizards (Fritz and Rodda 1998). A final example demonstrates that introduced predators need not be large vertebrates. Since their accidental introduction to Mobile, Alabama, in the 1930s, imported red fire ants (*Solenopsis invicta*) have spread throughout much of the southeastern United States (Wojcik et al. 2001). Mount (1981) expressed concern about the potential impacts of predation by *S. invicta* on vertebrates in the Southeast, noting observations of fire ant predation on eggs and hatchlings of several reptile species. He also made anecdotal observations of declines of many litter-dwelling snakes and lizards, and large terrestrial oviparous snakes in the Alabama coastal plain in conjunction with *S. invicta* invasion. Although quantitative evidence of the effects of *S. invicta* on native reptiles has been slow to emerge, fire ants are documented predators of turtle nests (Buhlmann and Coffman 2001) and have been implicated in the decline of southern hognose snakes (*Heterodon simus*; Tuberville et al. 2000), eastern kingsnakes (*Lampropeltis getula*; Wojcik et al. 2001, Winne et al. 2007), and Texas horned lizards (*Phrynosoma cornutum*; Goin 1992). Importantly, the effects of *S. invicta* on reptiles may be exacerbated by habitat disturbance, possibly leading to synergistic effects of habitat alteration and predation (Todd et al. 2008).

Introduction of exotic prey can also have profound effects on reptile populations. In some cases, exotic prey may possess defenses (e.g., poisons, morphological defenses) to which native predators are unaccustomed, resulting in direct mortality of reptiles that attempt to consume the exotics. For example, exotic cane toads (*Bufo marinus*) possess potent parotoid secretions and have become abundant in many areas of tropical Australia since their introduction in 1929 (Lampo and de Leo 1998). Covacevich and Archer (1975) noted several instances of direct mortality of snakes and monitor lizards (*Varanus* spp.) that attempted to ingest the toads. Other authors have observed declines of several snake and lizard species following arrival of *B. marinus* (Phillips et al. 2003), as well as mortality of Australian freshwater crocodiles. Moreover, laboratory studies have shown that many Australian snake species are sufficiently vulnerable to toad toxins to die after ingesting a single toad, prompting Phillips et al. (2003) to suggest that the toads threaten as much as 30% of Australia's terrestrial snake species. In other cases, exotic prey may be palatable, but of poorer nutritional quality than native prey taxa. For example, exotic Argentine ants (*Linepithema humile*) have been introduced worldwide (Suarez et al. 2001) and eliminate nearly all native ground-dwelling ants when they invade new habitats (Suarez et al. 1998). Suarez and Case (2002) demonstrated that Argentine ants represent an inferior prey resource for coastal horned lizards (*Phrynosoma coronatum*), a species that has declined dramatically in California. They found that hatchling *P. coronatum* fed a diet of introduced *L. humile* exhibited zero or negative growth, but resumed normal growth when switched back to a diet of native ants.

Introduced exotic prey taxa may sometimes be beneficial to native reptiles. For example, round gobies (*Neogobius melanostomus*) introduced into the Great Lakes region of North America have become favored prey of the federally listed Lake Erie watersnake (*Nerodia sipedon insularum*), resulting in increased growth and body sizes of snakes (King et al. 2006a). Introduction of gobies has been implicated as a partial cause for the recovery of this snake in recent years (King et al. 2006a) to levels that warrant delisting under the US Endangered Species Act (King et al. 2006b).

Introduced species may also exert substantial indirect effects on native reptile populations through competition for resources. For example, introduced geckos (*Hemidactylus frenatus*) compete with and



have displaced native geckos (*Lepidodactylus lugubris*) throughout the tropical Pacific (Case et al. 1994). Likewise, competition with *H. frenatus* has caused declines and extirpations of several species of night geckos (*Nactus* spp.) on the Mascarene Islands. Introduced slider turtles (*Trachemys scripta*) compete with native European pond turtles (*Emys orbicularis*) for basking sites (Cadi and Joly 2003), resulting in weight loss and reduced survival of *E. orbicularis* in experimental mixed populations compared with controls (Cadi and Joly 2004). Importantly, in some cases, introduced species may gain a competitive advantage because of release from their native pathogens or parasites (Reed 2005).

Another indirect means by which invasive species can affect native reptiles is through habitat modification. The most obvious examples of this phenomenon occur in cases where invasive plant species displace native vegetation, rendering habitat unsuitable for native species. For example, lush growth of invasive annual plants in the Mojave Desert of the American Southwest negatively affects desert tortoises (*Gopherus agassizii*), primarily through increased fire frequency (Brooks and Pike 2001). Likewise, habitats dominated by exotic rubber vine (*Cryptostegia grandiflora*) are avoided by native Australian lizards (Valentine 2006).

Finally, introduced species may be important vectors for disease and parasites (see Section 3.2.6). Unlike other ways in which exotic species may affect reptiles, exotic species do not need to become established in the wild to serve as disease vectors. In fact, release of a single infected individual, or human contact with a wild reptile after handling an infected captive reptile, may be sufficient to introduce a pathogen to native reptile populations. Reed (2005) cautioned that release of captive boas and pythons could be an important source of disease to native snakes such as the federally threatened eastern indigo snake (*Drymarchon couperi*) or boid species native to the United States (rubber boa, *Charina bottae*, and rosy boa, *Lichanura trivirgata*).

### 3.2.6 DISEASE AND PARASITISM

Pathogens and parasites have long been recognized as potentially important factors regulating natural populations (Anderson and May 1978; Dobson and Hudson 1986). Virtually every species hosts a multitude of parasites and pathogens, some of which can cause dramatic population fluctuations (e.g., Hudson et al. 1998). However, when human activities alter rates of disease transmission or reduce resistance of animals to disease, the results can be catastrophic (Daszak et al. 2000). For example, outbreaks of pathogenic chytrid fungus (*Batrachochytrium dendrobatidis*) have devastated amphibian populations worldwide (Daszak et al. 2003). Moreover, the spread of chytrid appears to have been facilitated by global climate change (Pounds et al. 2006), introduction of exotic species (Mazzoni et al. 2003), and direct spread by humans (Weldon et al. 2004).

Disease outbreaks have been implicated in declines of several reptile taxa and are apparently of particular concern for turtles. For example, incidence of upper respiratory tract disease (URTD) infections of gopher tortoises (*G. polyphemus*) and desert tortoises (*G. agassizii*) has increased in recent years, and URTD has been implicated in declines of some populations (Dodd and Seigel 1991; Seigel et al. 2003). Upper respiratory tract disease may have been introduced into natural populations through release of infected captive individuals (Dodd and Seigel 1991), and authors have expressed concern that spread of this disease could be exacerbated through translocation of infected animals for conservation purposes (Dodd and Seigel 1991). Disease outbreaks have been noted in other protected turtles, such as the flattened musk turtle (*Sternotherus depressus*; Dodd 1988; Fonnesbeck and Dodd 2003) and green sea turtle (*Chelonia mydas*; Chaloupka and Balazs 2005); however, factors underlying these outbreaks are poorly understood.

### 3.2.7 CASCADING DECLINES

An additional consideration seldom explicitly addressed is how reptiles respond to declines in other taxa (but see Irwin and Irwin 2005). Although ecologists have yet to form a consensus about the role of biodiversity in maintaining ecosystem function (Thompson and Starzomski 2007), loss of



important species could result in subsequent extinction of previously unaffected taxa. Living in a time when species extinctions are increasing at an alarming rate, we are probably only beginning to see the long-term effects of extinctions that have already occurred.

An obvious concern regarding cascading declines is the possible response of reptiles to the recent catastrophic declines of many amphibian populations. Many snake species are specialists that feed exclusively on amphibians (Toledo et al. 2007); we might therefore expect that these species would experience declines concomitant with those of amphibians. Whiles et al. (2006) reported the disappearance of several common frog-eating riparian snakes soon after amphibian declines related to chytridiomycosis in Panama. Similarly, in the Sierra Nevada Mountains of the American West, the presence of mountain garter snakes (*Thamnophis elegans*) is strongly associated with the presence of anurans (Matthews et al. 2002). Matthews et al. (2002) suggest that amphibian declines associated with stocking of predatory trout may have a strong adverse effect on populations of *T. elegans*.

Cascading declines may be most frequently associated with the disappearance of keystone species, those species that play a role disproportionate to their abundance in maintaining community composition and ecosystem function (Power et al. 1996). For example, the gopher tortoise (*G. polyphemus*) is considered a keystone species (Eisenberg 1983) because its burrows provide critical refugia for a variety of upland species in the southeastern United States. Gopher tortoises have declined across their range (McCoy et al. 2006), and loss of tortoise burrows is considered a serious threat to persistence of the federally threatened eastern indigo snake (*Drymarchon couperi*; Stevenson et al. 2003) and eastern diamondback rattlesnake (*Crotalus adamanteus*; Timmerman and Martin 2003), among other species. Although such cascading declines affect all species, reptiles, which often have highly specialized food and habitat requirements, may be less able than more generalist taxa to withstand sequential removal of individual species from ecosystems.

### 3.3 GLOBAL STATUS OF REPTILE POPULATIONS

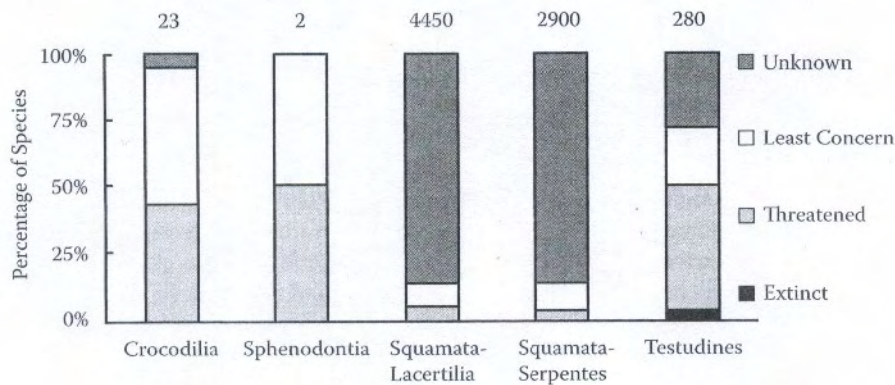
As discussed previously, the status of many reptiles remains unknown and is the subject of ongoing global assessment by the IUCN (Figure 3.1). Here, we describe the current perceived status of reptile populations and provide overviews of reptile declines among turtles, crocodylians, lizards, and snakes. Our reports on the current status of reptile populations represent our best understanding of the available scientific literature and current information from the 2008 IUCN Red List (IUCN 2009).

#### 3.3.1 TESTUDINES

A greater proportion of turtles are recognized as imperiled and in categories of conservation concern than any other group of reptiles excluding the 2 tuatara species, Sphenodontia (Figure 3.1). Overall, 42% of turtle species included in the IUCN 2004 global assessment were classified as threatened (including all IUCN categories of imperilment; Baillie et al. 2004). However, at that time only a portion of all described species were evaluated. Consequently, the actual rate of imperilment of evaluated species is closer to 62% (Baillie et al. 2004). Perhaps most alarming is that the IUCN currently lists a total of 8 turtle species that have gone extinct in the wild in modern times (IUCN 2009). The Turtle Conservation Fund even compiled a list of 25 additional turtle species on "death row," that is, the most endangered species of tortoises and freshwater turtles in the world (Buhlmann et al. 2002). Although not included in the list, all 7 species of sea turtles (families Cheloniidae and Dermochelyidae) are considered imperiled. The taxonomic distribution of endangered species (critically endangered [CE] and endangered [EN]) includes species from each of the 11 families of freshwater turtles, the single family of tortoises, and both families of sea turtles (Baillie et al. 2004).

Turtles are long-lived (Gibbons 1987), and commercial harvesting of wild populations of most species is not sustainable (Reed and Gibbons 2003). Nonetheless, turtles differ from other reptile groups in that human consumption is the documented cause for the majority of declines on a global scale (Buhlmann et al. 2002). Commercial harvesting in Southeast Asia is a major cause





**FIGURE 3.1** Status of the major lineages of reptiles according to the World Conservation Union (IUCN) Red List in 2009. Status categories include “extinct” (including species extinct in the wild but extant in captivity), “threatened” (including IUCN categories “critically endangered,” “endangered,” and “vulnerable”), “least concern” (including IUCN categories “lower risk” and “near threatened”), and “unknown” (including species that have not been evaluated by the IUCN and those that have been evaluated but were deemed “data deficient”). Numbers above bars indicate the approximate number of species within each lineage according to Zug et al. (2001). Data were accessed from the IUCN database on March 18, 2009 (IUCN 2009). Note that although bars are difficult to see, 11 species of lizards (0.25%) and 3 species of snakes (0.10%) are listed as extinct by the IUCN as of 2009.

for concern, with some turtle species clearly on a trajectory toward extinction at current rates of removal from the wild (Baillie et al. 2004). Indeed, of 73 species of tortoises and freshwater turtles classified as endangered and critically endangered in 2002, more than half were from Asia, with the remaining species being distributed geographically among North America, Mesoamerica, South America, the Mediterranean, Africa, and Australasia (Buhlmann et al. 2002). Outside of Asia, terrestrial terrapins and tortoises are imperiled by excessive harvesting combined with loss of suitable habitat (Baillie et al. 2004). Collection and removal of turtles from North America, mostly for the export trade, has also been significant (Franke and Telecky 2001; Ceballos and Fitzgerald 2004) and could understandably be implicated in the decline of some species in the wild. Sea turtles, all of which are classified as imperiled, are subject to unique threats, with declines attributed to mortality from incidental by-catch, harvesting of turtles and eggs for consumption, and degradation of nesting and foraging habitat (Lutcavage et al. 1997; Spotila 2004).

### 3.3.2 CROCODYLIANS

A quarter of a century ago, every crocodylian species in the world was categorized as endangered or threatened. Ironically, because the fate of only 7 of the 23 species remains uncertain at the beginning of the 21st century, the group is considered by many conservationists to be a major success story. Conservation efforts by the Crocodile Specialist Group of the Species Survival Commission–IUCN are generally viewed as the cause for an upturn in the status of two-thirds of the crocodylian species that have traditionally suffered from the pressures of harvesting and habitat loss (IUCN 2009).

Despite the noted conservation successes in reducing declines and extinction threats for some crocodylian species, the status of 7 species, including the Chinese alligator (*Alligator sinensis*), the black caiman (*Melanosuchus niger*), the Indian gharial (*Gavialis gangeticus*), and 4 species of crocodile, remains one of teetering on the brink of extinction. In mainland Asia, only about 150 Chinese alligators are estimated to be present in their native habitat, and the Siamese crocodile (*Crocodylus siamensis*) is effectively extirpated from its native home of Thailand (formerly Siam), with few populations persisting in other parts of Southeast Asia. Only a single established population of the



Philippine crocodile (*Crocodylus mindorensis*) is known to exist at this time, with the species now occupying less than 20% of its former range of most of the Philippines Archipelago. Likewise, the Indian gharial is considered to be critically endangered despite strong and effective conservation programs in some parts of the country. In fact, nearly 80 gharials, representing 6% of the known population, died from unknown causes in an Indian forest preserve from December 2007 to January 2008 (Mahmood 2008). Initial reports identifying liver cirrhosis in the dead animals suggested that parasites or environmental contaminants may have played a role in the deaths.

In the western hemisphere, no fewer than 3 crocodylian species remain on the list of special concern, although some seem to be recovering through regional conservation efforts. In tropical South America, the most endangered species are the black caiman and the Orinoco crocodile (*Crocodylus intermedius*), a species whose decline has been attributed in part to the unrestricted use of pesticides for agricultural purposes. The Cuban crocodile (*Crocodylus rhombifer*), whose geographic range once included several islands in the West Indies, is now represented in the wild only from highly localized areas in Cuba. Nevertheless, many conservationists point to the American alligator (*Alligator mississippiensis*) as one of the greatest models of successful recovery for any threatened vertebrate.

Most of the remaining 16 species of crocodylians — including 3 native to Africa, 2 found in Australia, and others at scattered locations in tropical America and the Pacific — are considered to be vulnerable only if the strict conservation programs in place are discontinued. Also, although many of these remaining species are not considered to be threatened with extinction throughout their geographic ranges, some still have critically endangered regional populations.

### 3.3.3 SQUAMATES: LACERTILIANS

Like many other reptiles, the status of most lizard species and populations is largely unknown (Figure 3.1). However, based on current information, lizards appear to have a small proportion of imperiled species (IUCN 2009). This is due in large part to life history attributes that make many lizards less susceptible to decline from anthropogenic factors. Notably, many lizards occur at high population densities, have short generation times, high fecundity, and are not as long-lived as other reptiles. Consequently, lizards may sometimes adjust rapidly to environmental change or rebound quickly from short-term population reductions. In fact, some lizard species fare well in human-modified or early successional habitats (e.g., *Anolis* spp., *Hemidactylus* spp.). In several cases, these life history attributes have contributed to the successful establishment of exotic lizards introduced into areas outside of their native ranges. For example, there are more than 30 species of nonnative lizards in Florida, representing over two-thirds of the total lizard fauna in that state (Meshaka et al. 2005). Ultimately, however, several lizard species are declining and have been classified as threatened or worse under the IUCN Red List system (IUCN 2009). Moreover, the lack of data on the status of many lizard populations may further jeopardize their long-term persistence.

Causes for the endangerment of lizards vary widely, but life history characteristics greatly influence the degree to which different factors threaten species. Imperiled species are those that typically have attributes such as endemism, restricted geographic ranges, large body size, long lives, late maturity, or low fecundity, which make them susceptible to population declines from anthropogenic factors. For instance, the slow-maturing, long-lived, and often endemic giant land iguanas of the Caribbean (*Cyclura* and *Brachylophus* spp.) are among the most threatened lizards globally (Pianka and Vitt 2003). In some cases, populations of Caribbean iguanas have dwindled to as few as 100 individuals. Much of their decline has been attributed to the historical harvest of these lizards for food and the introduction of nonnative pests such as mongooses, rats, goats, and pigs onto their island homes. The fate of these lizards now depends almost entirely on human intervention to preserve habitat, control introduced predators, and increase recruitment to avoid permanent extinction. Many varanoid lizards are also slow maturing and long-lived, and several of them are currently protected in parts of their ranges due to population declines (Pianka and Vitt 2003; Pianka et al. 2004). Although phylogenetically distinct from lizards, tuatara (*Sphenodon* spp.) share these “slow”



life history characteristics and have long incubation periods that place them in similar jeopardy (McIntyre 1997).

At least a few smaller, "typical" lizards are also threatened or critically endangered despite having high fecundity and early sexual maturity. The largest contributor to the endangerment of these species is often their endemism, restricted ranges, or highly specialized habitat requirements. High fecundity and early sexual maturity may safeguard them from declines associated with harvesting but greatly increase their risk of decline from habitat loss. In fact, habitat loss is listed as a contributing factor in the imperilment of more than half of squamates currently recognized as near threatened or worse on the IUCN Red List (IUCN 2009). For instance, the Coachella Valley fringe-toed lizard (*Uma inornata*), the island night lizard (*Xantusia riversiana*), and several island geckos from Madagascar and the Caribbean (*Phelsuma* spp. and *Sphaerodactylus* spp.) are threatened or critically endangered due to combinations of restricted geographic ranges and habitat loss. Nevertheless, some species, such as Madagascar's Antsingy leaf chameleon (*Brookesia perarmata*), also presumably suffer from heavy collection, which has led to restrictions on their international trade (CITES 2003).

### 3.3.4 SQUAMATES: SERPENTS

Despite recent reports of reptile declines, the global status of snake populations has received relatively little attention. Indeed, along with lizards, snakes have yet to receive comprehensive review by the IUCN. According to the 2004 IUCN Global Species Assessment, only 3.4% of squamate species had been evaluated, compared with 67% of turtles, 90% of mammals, 100% of birds, and 100% of amphibians (Baillie et al. 2004). Five years later, the online IUCN Red List still demonstrates that the status of most squamates is unknown (Figure 3.1; IUCN 2009). Snakes are notorious for their cryptic behavior and low or sporadic activity, which seriously complicates efforts to assess population status (Parker and Plummer 1987). Thus, even for relatively well-studied species, population size or density often remains unknown (Dorcas and Willson 2009). As in lizards, risk of imperilment in snakes generally correlates more closely with life history attributes and geography than with taxonomy. Threatened species are most often those with specialized habitat requirements, small geographic ranges, or life history characteristics such as large body size, delayed sexual maturity, and/or low reproductive rates. Additionally, there is regional variation in the relative importance of threats faced by snakes such that major threats and taxa at risk vary among regions or continents.

Many snakes have specialized habitat requirements, making them particularly susceptible to habitat loss or degradation. For example, many of the most threatened snake species in the eastern United States, including the eastern indigo snake (*Drymarchon couperi*), eastern diamondback rattlesnake (*Crotalus adamanteus*), pine snake (*Pituophis melanoleucus*), and southern hognose snake (*Heterodon simus*), are those associated with the nearly eliminated longleaf pine ecosystem (Todd and Andrews 2008). Likewise, loss or degradation of wetland habitats has prompted federal listing of several North American snakes, including the wetland-associated eastern massasauga (*Sistrurus catenatus catenatus*), San Francisco garter snake (*Thamnophis sirtalis tetrataenia*), and copperbelly watersnake (*Nerodia erythrogaster neglecta*). Small geographic range, combined with specialized habitat requirements, puts species at risk from a variety of threats to their habitat. For example, the broad-headed snake (*Hoplocephalus bungaroides*), considered Australia's most endangered snake species, is restricted to small regions of rock outcrop habitat in eastern Australia and has suffered extensively from habitat degradation due to rock removal, collection for the pet trade, and canopy closure from fire suppression (Shine et al. 1998; Webb et al. 2002, 2005). Many of the most dramatic cases of imperilment due to small geographic range occur among snake species endemic to islands such as the Caribbean Lesser Antilles. The Lesser Antilles harbor 25 snake species, 87.5% of which are endemic, and are home to some of the rarest snakes in the world (e.g., the Antigua racer, *Alsophis antiguae*; Daltry et al. 2001). The region has suffered between



6 and 11 historical extirpations and at least one historical extinction, primarily due to predation by introduced mongooses (Henderson 2004). Although data are lacking, similar declines may also be occurring in other island archipelagos across the globe.

Across all animal taxa, life history attributes such as large body size, delayed sexual maturity, and low reproductive output contribute to imperilment, and many of the most threatened snake species also share these characteristics (Meffe and Carroll 1997). Among snakes, body size correlates strongly with home range size (Reed and Shine 2002; Reed 2003). Thus, larger species typically need larger tracts of suitable habitat and move more extensively than smaller species, presumably putting them at greater risk from road mortality or other threats (Andrews and Gibbons 2008). Many taxonomic groups of snakes have intrinsically slow growth and low reproductive rates, making them particularly susceptible to overharvesting and less able to recover from short-term population declines. For example, many species of European and Asian vipers (*Vipera* spp.) are considered threatened (IUCN 2009), in part because their "slow" life history characteristics put them at risk from persecution, collection, and habitat loss. Among Australian elapid snakes, Reed and Shine (2002) found that the characteristics that correlated most strongly with species endangerment were foraging strategy (ambush foragers were most imperiled) and mating system (species with female-biased sexual size dimorphism and lacking male-male combat were frequently threatened). They postulated that snakes employing ambush foraging had more specific habitat requirements and exhibited "risky" life history attributes such as low reproductive rates and slow growth rates. Likewise, large female body size potentially increased vulnerability of females to anthropogenic sources of mortality. Their results suggest that factors contributing to endangerment in snakes may differ substantially from other taxa (especially endotherms) and are not always intuitive (Reed and Shine 2002). Unfortunately, similar macroecological analyses have not been performed for other snake groups or geographic regions.

Substantial regional differences exist in the threats affecting snake populations and, consequently, in the status of snake populations. Throughout most temperate regions of Europe and North America, the paramount threat to snake populations is apparently habitat loss and degradation. However, road mortality, persecution, and collection for the pet trade have been implicated in the decline of some species, and causes of apparent declines in others remain enigmatic. Little data exist on the status of snake populations in tropical regions of the world; however, as with other taxa, snakes are undoubtedly suffering as a result of rampant habitat destruction occurring in tropical regions. A relatively novel, but poorly understood threat to snakes in these regions is the phenomenon of cascading declines discussed previously. In Asia, snakes face greater pressure from exploitative use than in other regions of the world, with estimated millions of snakes harvested annually from China and other regions of Southeast Asia (Zhou and Jiang 2004, 2005; Brooks et al. 2007). Finally, the myriad introduced exotic species in Australia (e.g., foxes, feral cats, cane toads) pose serious threats to the fragile ecosystems of that island continent. Indeed, as noted previously, cane toads alone have been suggested to threaten as much as 30% of Australia's terrestrial snake species (Phillips et al. 2003).

Although our knowledge of the status of global snake populations remains woefully inadequate, increasing awareness of the importance of snakes as top predators in many ecosystems (e.g., Ineich 2007) and advances in methodology for studying snake populations (Dorcas and Willson in press) will undoubtedly increase our ability to effectively conserve snake populations in future decades.

### 3.4 CONCLUSION

Continuing to determine the status, distribution, and basic ecology of many reptiles is of paramount importance. Although low detectability of reptiles, and subsequently poor awareness of declines in their populations, may hamper research and conservation efforts, ongoing advances in field methodology, mark-recapture analyses, and our understanding of reptile life histories and behaviors should



continue to improve our knowledge of the status and distribution of many reptiles. As recognition of declining reptile populations increases, determining causes of those declines should also become a primary goal. Some populations and species may be affected by one or a few factors, but multiple interacting stressors or threats likely affect many reptile populations. Numerous studies have demonstrated direct effects that environmental contaminants have on reptiles. But the many ways that environmental contaminants could exacerbate ongoing declines from other threats, such as disease and parasitism, habitat loss, and introduced invasive species, remain underappreciated. The hope that reptile declines will tail off or that highly imperiled species will be able to claw their way back from near extinction rests on a full understanding of the plight of reptiles, threats to their populations, causes for their declines, and effective mobilization of conservation resources.

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### REFERENCES

- Alford RA, Richards SJ. 1999. Global amphibian declines: a problem in applied ecology. *Ann Rev Ecol Syst* 30:133–165.
- Anderson RM, May RM. 1978. Regulation and stability of host-parasite population interactions. I. Regulatory processes. *J Anim Ecol* 47:219–247.
- Andrews KM, Gibbons JW. 2005. How do highways influence snake movement? Behavioral responses to roads and vehicles. *Copeia* 2005:771–781.
- Andrews KM, Gibbons JW. 2008. Roads as catalysts of urbanization: snakes on roads face differential impacts due to inter- and intraspecific ecological attributes. In: Mitchell JC, Jung RE, editors, *Urban herpetology, herpetological conservation*. Vol. 3. Salt Lake City (UT): Society for the Study of Amphibians and Reptiles, p 145–153.
- Araújo MB, Thuiller W, Pearson RG. 2006. Climate warming and the decline of amphibians and reptiles in Europe. *J Biogeogr* 33:1712–1728.
- Aresco MJ. 2003. Highway mortality of turtles and other herpetofauna at Lake Jackson, Florida, USA and the efficacy of a temporary fence/culvert system to reduce road kills. In: Irwin CL, Garrett P, McDermott KP, editors, *2003 Proceedings of the International Conference on Ecology and Transportation*. Raleigh (NC): Center for Transportation and the Environment, North Carolina State University, p 433–449.
- Aresco MJ. 2005. The effect of sex-specific terrestrial movements and roads on the sex ratio of freshwater turtles. *Biol Conserv* 123:37–44.
- Aresco MJ, Gunzburger MS. 2004. Effects of large-scale sediment removal on populations of herpetofauna in Florida wetlands. *J Herpetol* 38:275–279.
- Ashley EP, Kosloski A, Petrie SA. 2007. Incidence of intentional vehicle-reptile collisions. *Human Dimen Wildl* 12:137–143.
- Baillie JEM, Hilton-Taylor C, Stuart SN, editors. 2004. *2004 IUCN red list of threatened species: a global species assessment*. Cambridge (UK): IUCN Publications Services.
- Bergeron CM, Husak JF, Unrine JM, Romanek CS, Hopkins WA. 2007. Influence of feeding ecology on blood mercury concentrations in four species of turtles. *Environ Toxicol Chem* 26:1733–1741.
- Bishop JM. 1983. Incidental capture of diamondback terrapin by crab pots. *Estuaries* 6:426–430.
- Bonnet XR, Shine R, Lourdais O. 2002. Taxonomic chauvinism. *Trends Ecol Evol* 17:1–3.
- Brooks ML, Pyke DA. 2001. Invasive plants and fire in the deserts of North America. In: Galley KEM, Wilson TP, editors, *Proceedings of the Invasive Species Workshop: the role of fire in the control and spread of invasive species*. Fire Conference 2000: The First National Congress on Fire Ecology, Prevention, and Management. Miscellaneous Publication 11. Tallahassee (FL): Tall Timbers Research Station, p 1–14.
- Brooks SE, Allison EH, Reynolds JD. 2007. Vulnerability of Cambodian water snakes: initial assessment of the impact of hunting at Tonle Sap Lake. *Biol Conserv* 139:401–414.
- Buhlmann KA. 1995. Habitat use, terrestrial movements and conservation of the turtle *Deirochelys reticularia* in Virginia. *J Herpetol* 29:173–181.



- Buhlmann KA, Coffman G. 2001. Fire ant predation of turtle nests and implications for the strategy of delayed emergence. *J Elisha Mitchell Sci Soc* 117:94–100.
- Buhlmann KA, Hudson R, Rhodin AGJ. 2002. A global action plan for conservation of tortoises and freshwater turtles. Strategy and funding prospectus. Washington (DC): Turtle Conservation Fund, Conservation International and Chelonian Research Foundation.
- Burke VJ, Gibbons JW. 1995. Terrestrial buffer zones and wetland conservation: a case study of freshwater turtles in a Carolina bay. *Conserv Biol* 9:1365–1369.
- Cadi A, Joly P. 2003. Competition for basking places between the endangered European pond turtle (*Emys orbicularis galloitalica*) and the introduced red-eared slider (*Trachemys scripta elegans*). *Can J Zool* 81:1392–1398.
- Cadi A, Joly P. 2004. Impact of the introduction of the red-eared slider (*Trachemys scripta elegans*) on survival rates of the European pond turtle (*Emys orbicularis*). *Biodiv Conserv* 13:2511–2518.
- Campbell KR, Campbell TS. 2000. Lizard contaminant data for ecological risk assessment. *Rev Environ Contam Toxicol* 165: 39–116.
- Campbell KR, Campbell TS. 2001. The accumulation and effects of environmental contaminants on snakes: a review. *Environ Monitor Assess* 70:253–301.
- Carpenter AI, Rowcliffe JM, Watkinson AR. 2004. The dynamics of the global trade in chameleons. *Biol Conserv* 120:291–301.
- Carr AF. 1952. Handbook of turtles: the turtles of the United States, Canada, and Baja California. Ithaca (NY): Cornell University Press.
- Case TJ, Bolger DT, Petren K. 1994. Invasions and competitive displacement among house geckos in the tropical Pacific. *Ecology* 75:464–477.
- Ceballos CP, Fitzgerald LA. 2004. The trade in native and exotic turtles in Texas. *Wildl Soc Bull* 32:1–12.
- Chaloupka M, Balazs G. 2005. Modelling the effect of fibropapilloma disease on the somatic growth dynamics of Hawaiian green sea turtles. *Marine Biol* 147:1251–1260.
- CITES. 2003. Notifications to the parties. Available from: <http://www.cites.org>.
- Covacevich J, Archer M. 1975. The distribution of the cane toad, *Bufo marinus*, in Australia and its effects on indigenous vertebrates. *Memoirs Queensland Museum* 17:305–310.
- Cox J. 2007. FWC bans entombment of gopher tortoises. *Naples Daily News*, June 13.
- Daltry JC, Bloxam Q, Cooper G, Day ML, Hartley J, Henry M, Lindsay K, Smith BE. 2001. Five years of conserving the “world’s rarest snake,” the Antigua racer *Alsophis antiguae*. *Oryx* 35:119–127.
- Daszak P, Cunningham AA, Hyatt AD. 2000. Emerging infectious diseases of wildlife — threats to biodiversity and human health. *Science* 287:443–449.
- Daszak P, Cunningham AA, Hyatt AD. 2003. Infectious disease and amphibian population declines. *Div Distrib* 9:141–150.
- Dobson AP, Hudson PJ. 1986. Parasites, disease and the structure of ecological communities. *Trends Ecol Evol* 1:11–15.
- Dodd Jr CK. 1988. Disease and population declines in the flattened musk turtle *Sternotherus depressus*. *Am Midl Nat* 119:394–401.
- Dodd Jr CK. 1990. Effects of habitat fragmentation on stream-dwelling species, the flattened musk turtle *Sternotherus depressus*. *Biol Conserv* 54:33–45.
- Dodd Jr CK, Seigel RA. 1991. Relocation, repatriation, and translocation of amphibians and reptiles: are they conservation strategies that work? *Herpetologica* 47:336–350.
- Donaldson WL, Price AH, Morse J. 1994. The current status and future prospects of the Texas horned lizard (*Phrynosoma cornutum*) in Texas. *Texas J Sci* 46: 97–113.
- Dorcas ME, Willson JD. 2009. Innovative methods for studies of snake ecology and conservation. In: Mullin SJ, Seigel RA, editors, *Snakes: applied ecology and conservation*. Ithaca (NY): Cornell University Press, 5–37.
- Dorcas ME, Willson JD, Gibbons JW. 2007. Population decline and demographic changes of a diamondback terrapin population over three decades. *Biol Conserv* 137:334–340.
- Driscoll DA. 2004. Extinction and outbreaks accompany fragmentation of a reptile community. *Ecol Appl* 14:220–240.
- Dudgeon D, Arthington AH, Gessner MO, Kawabata Z-I, Knowler DJ, Lévêque C, Naiman RJ, Prieur-Richard A-H, Soto D, Stiassny MLJ, Sullivan CA. 2006. Freshwater biodiversity: importance, threats, status and conservation challenges. *Biol Rev* 81:163–182.
- Dunham AE. 1993. Population responses to environmental change: operative environments, physiologically structured models, and population dynamics. In: Kareiva PM, Kingsolver JG, Huey RB, editors, *Biotic interactions and global change*. Sunderland (MA): Sinauer Associates, p 95–119.



- DuRant SE, Hopkins WA, Talent LG. 2007a. Impaired terrestrial and arboreal locomotor performance in the western fence lizard (*Sceloporus occidentalis*) after exposure to an AChE-inhibiting pesticide. *Environ Pollut* 149:18–24.
- DuRant SE, Hopkins WA, Talent LG. 2007b. Energy acquisition and allocation in an ectothermic predator exposed to a common environmental stressor. *Comp Biochem Physiol C Toxicol Pharmacol* 145:442–448.
- Eisenberg JF. 1983. The gopher tortoise as a keystone species. In: Bryant RJ, Franz R, editors, *The gopher tortoise: a keystone species*, Proceedings of the 4th Annual Meeting, Gopher Tortoise Council, Valdosta, GA, p 1–4.
- Ernst CH, Barbour RW, Lovich JE. 1994. *Turtles of the United States and Canada*. Washington (DC): Smithsonian Institution Press.
- Fonnesbeck CJ, Dodd Jr CK. 2003. Estimation of flattened musk turtle (*Sternotherus depressus*) survival, recapture, and recovery rate during and after a disease outbreak. *J Herpetol* 37:602–607.
- Franke J, Telecky TM. 2001. *Reptiles as pets: an examination of the trade in live reptiles in the United States*. Washington (DC): Humane Society of the United States.
- Frazer NB, Greene JL, Gibbons JW. 1993. Temporal variation in growth rate and age at maturity of male painted turtles, *Chrysemys picta*. *Am Midl Nat* 130:314–324.
- Fritts TH, Rodda GH. 1998. The role of introduced species in the degradation of island ecosystems: a case history of Guam. *Ann Rev Ecol Syst* 29:113–140.
- Gardner TA, Barlow J, Peres CA. 2007. Paradox, presumption and pitfalls in conservation biology: the importance of habitat change for amphibians and reptiles. *Biol Conserv* 138:166–179.
- Gibbons JW. 1987. Why do turtles live so long? *Bioscience* 37:262–269.
- Gibbons JW. 1988. The management of amphibians, reptiles, and small mammals in North America, Proceedings of the Symposium, Flagstaff, AZ, July 19–21, 1988, p 4–10.
- Gibbons JW, Scott DE, Ryan TJ, Buhlmann KA, Tuberville TD, Metts BS, Greene JL, Mills T, Leiden Y, Poppy S, Winne CT. 2000. The global decline of reptiles, déjà vu amphibians. *Bioscience* 50: 653–666.
- Gibbs JP, Steen DA. 2005. Historical trends in turtle sex ratios in the United States: long-term implications of road mortality. *Conserv Biol* 19:552–556.
- Glor RE, Flecker AS, Benard MF, Power AG. 2001. Lizard diversity and agricultural disturbance in a Caribbean forest landscape. *Biodiv Conserv* 10:711–723.
- Goin JW. 1992. Requiem or recovery? *Texas Parks Wildl* August:28–35.
- Guertin DS, Easterling WE, Brandle JR. 1997. Climate change and forests in the Great Plains. *Bioscience* 47:287–295.
- Guillette LJ, Gross TS, Masson GR, Matter JM, Percival HF, Woodward AR. 1994. Developmental abnormalities of the gonad and abnormal sex hormone concentrations in juvenile alligators for contaminated and control lakes in Florida. *Environ Health Perspect* 102:680–688.
- Henderson RW. 2004. Lesser Antillean snake faunas: distribution, ecology, and conservation concerns. *Oryx* 38:311–320.
- Hokit DG, Branch LC. 2003. Habitat patch size affects density of Florida scrub lizard (*Sceloporus woodi*). *J Herpetol* 37:257–265.
- Holem RR, Hopkins WA, Talent LG. 2006. Effect of acute exposure to malathion and lead on sprint performance of the western fence lizard (*Sceloporus occidentalis*). *Arch Environ Contam Toxicol* 51:111–116.
- Hopkins WA. 2000. Reptile toxicology: challenges and opportunities on the last frontier in vertebrate ecotoxicology. *Environ Toxicol Chem* 19:2391–2393.
- Hopkins WA, Roe JH, Snodgrass JW, Staub BP, Jackson BP, Congdon JD. 2005. Effects of chronic dietary exposure to trace elements on banded water snakes (*Nerodia fasciata*). *Environ Toxicol Chem* 21:906–913.
- Hopkins WA, Rowe CL, Congdon JD. 1999. Elevated trace element concentrations and standard metabolic rate in banded water snakes (*Nerodia fasciata*) exposed to coal combustion wastes. *Environ Toxicol Chem* 18:1258–1263.
- Hopkins WA, Winne CT. 2006. Influence of body size on swimming performance of four species of neonatal natricine snakes acutely exposed to a cholinesterase-inhibiting pesticide. *Environ Contam Toxicol* 25:1208–1213.
- Hopkins WA, Winne CT, DuRant SE. 2005. Differential swimming performance of two natricine snakes exposed to a cholinesterase-inhibiting pesticide. *Environ Pollut* 133:531–540.
- Hoyer RF, O'Donnell RP, Mason RT. 2006. Current distribution and status of sharp-tailed snakes (*Contia tenuis*) in Oregon. *Northwestern Nat* 87:195–202.



- Hudson PJ, Dobson AP, Newborn D. 1998. Prevention of population cycles by parasite removal. *Science* 282:2256–2258.
- Ineich I, Bonnet X, Brischoux F, Kulbicki M, Seret B, Shine R. 2007. Anguilliform fishes and sea kraits neglected predators in coral-reef ecosystems. *Marine Biol* 151:793–802.
- IPCC. 2007. Climate change 2007: the physical science basis. Contribution of Working Group I to the Fourth Assessment Report of the Intergovernmental Panel on Climate Change (Solomon S, Qin D, Manning M, Chen Z, Marquis M, Averyt KB, Tignor M, Miller HL, editors). Cambridge (UK): Cambridge University Press.
- Irwin L, Irwin K. 2005. Global threats affecting the status of reptile populations. In: Gardner SC, Oberdorster E, editors, *Toxicology of reptiles*. Boca Raton (FL): CRC Press, p. 9–34.
- IUCN. 2009. 2008 IUCN red list of threatened species. Available from: <http://www.iucnredlist.org>. Accessed January 2009.
- Janzen FJ. 1994. Climate change and temperature-dependent sex determination in reptiles. *Proc Natl Acad Sci USA* 91:7487–7490.
- Kanowski JJ, Reis TM, Catterall CP, Piper SD. 2006. Factors affecting the use of reforested sites by reptiles in cleared rainforest landscapes in tropical and subtropical Australia. *Restor Ecol* 14:67–76.
- King RB, Queral-Regil A, Stanford KM. 2006b. Population size and recovery criteria of the threatened Lake Erie watersnake: integrating multiple methods of population estimation. *Herpetol Monogr* 20:83–104.
- King RB, Ray JM, Stanford KM. 2006a. Gorging on gobies: beneficial effects of alien prey on a threatened vertebrate. *Can J Zool* 84:108–115.
- Kjoss VA, Litvaitis JA. 2001. Community structure of snakes in a human-dominated landscape. *Biol Conserv* 98:285–292.
- Kofron CP. 1991. Aspects of ecology of the threatened ringed sawback turtle, *Graptemys oculifera*. *Amphibia Reptilia* 12:161–168.
- Lagueux CJ. 1991. Economic analysis of sea turtle eggs in a coastal community on the Pacific coast of Honduras. In: Robinson JG, Redford KH, editors, *Neotropical wildlife use and conservation*. Chicago (IL): University of Chicago Press, p 136–144.
- Lampo M, de Leo GA. 1998. The invasion ecology of the toad *Bufo marinus*: from South America to Australia. *Ecol Appl* 8:388–396.
- Lewis RL, Crowder LB, Shaver DJ. 2003. The impact of turtle excluder devices and fisheries closures on loggerhead and Kemp's ridley strandings in the western Gulf of Mexico. *Conserv Biol* 17:1089–1097.
- Lewis RL, Freeman SA, Crowder LB. 2004. Quantifying the effects of fisheries on threatened species: the impact of pelagic longlines on loggerhead and leatherback sea turtles. *Ecol Lett* 7:221–231.
- Lutcavage ME, Plotkin P, Witherington B, Lutz PL. 1997. Human impacts on sea turtle survival. In: Lutz PL, Musick JA, editors, *The biology of sea turtles*. Boca Raton (FL): CRC Press, p 387–410.
- Mahmood S. 2008. Deaths of rare crocodile in India stir alarm. *New York Times* online. Available from: <http://www.nytimes.com/2008/01/22/science/22croc.html?ref=science>. Accessed January 2008.
- Matthews KR, Knapp RA, Pope KL. 2002. Garter snake distributions in high-elevation aquatic ecosystems is there a link with declining amphibian populations and nonnative trout introductions? *J Herpetol* 36:16–22.
- Mazzoni R, Cunningham AC, Daszak P, Apolo A, Perdomo E, Speranza G. 2003. Emerging pathogen of wild amphibians in frogs (*Rana catesbeiana*) farmed for international trade. *Emerg Infect Dis* 9:995–998.
- McCoy ED, Mushinsky HR, Lindzey J. 2006. Declines of the gopher tortoise on protected lands. *Biol Conserv* 128:120–127.
- McIntyre M. 1997. *Conservation of the Tuatara*. Wellington (AU): Victoria University Press.
- Meffe GK, Carroll RC, editors. 1997. *Principles of conservation biology*. 2nd ed. Sunderland (MA) Sinauer Associates.
- Meik JM, Jeo RM, Mendelson III JR, Jenks KE. 2002. Effects of bush encroachment on an assemblage of diurnal lizard species in central Namibia. *Biol Conserv* 106:29–36.
- Meshaka Jr WE, Butterfield BP, Hauge JB. 2005. *Exotic amphibians and reptiles of Florida*. Melbourne (FL) Krieger Publishing Co.
- Mieres MM, Fitzgerald LA. 2006. Monitoring and managing the harvest of tegu lizards in Paraguay. *J Wildl Manage* 70:1723–1734.
- Mittermeier RA, Carr JL, Swingland IR, Werner TB, Mast RB. 1992. Conservation of amphibians and reptiles. In: Adler K, editor, *Herpetology: current research on the biology of amphibians and reptiles*. St. Louis (MO): SSAR Publications, p. 59–80.
- Mount RH. 1981. The red imported fire ant, *Solenopsis invicta* (Hymenoptera: Formicidae), as a possible serious predator on some native Southeastern vertebrates: direct observations and subjective impressions. *Alabama Acad Sci* 52:71–78.



- Park K. 2004. Assessment and management of invasive alien predators. *Ecol Soc* 9. Available from: <http://www.ecologyandsociety.org/vol9/iss2/art12/>.
- Parker WS, Plummer MV. 1987. Population ecology. In: Seigel RA, Collins JT, Novak SS, editors, *Snakes: ecology and evolutionary biology*. New York: McGraw-Hill, p 253–301.
- Pechmann JHK. 2003. Natural population fluctuations and human influences: null models and interactions. In: Semlitsch RD, editor, *Amphibian conservation*. Washington (DC): Smithsonian Institution Press, p 85–93.
- Phillips BL, Brown GP, Shine R. 2003. Assessing the potential impact of cane toads on Australian snakes. *Conserv Biol* 17:1738–1747.
- Pianka ER, King DR, King RA. 2004. *Varanoid lizards of the world*. Bloomington (IN): Indiana University Press.
- Pianka ER, Vitt LJ. 2003. *Lizards: windows to the evolution of diversity*. Los Angeles: University of California Press.
- Pimentel D, Lach L, Zuniga R, Morrison D. 2000. Environmental and economic costs of nonindigenous species in the United States. *Bioscience* 50:53–65.
- Pounds JA, Bustamante MR, Coloma LA, Consuegra JA, Fogden MPL, Foster PN, La Marca E, Masters KL, Merino-Viteri A, Puschendorf R, Ron SR, Sanchez-Azofeifa GA, Still CJ, Young BE. 2006. Widespread amphibian extinctions from epidemic disease driven by global warming. *Nature* 439:161–167.
- Power ME, Tilman D, Estes JA, Menge BA, Bond WJ, Mills LS, Daily G, Castilla JC, Lubchenco J, Paine RT. 1996. Challenges in the quest for keystones. *Bioscience* 46:609–620.
- Reed RN. 2003. Interspecific patterns of species richness, geographic range size, and body size among New World venomous snakes. *Ecography* 26:107–117.
- Reed RN. 2005. An ecological risk assessment of nonnative boas and pythons as potentially invasive species in the United States. *Risk Anal* 25:753–766.
- Reed RN, Gibbons JW. 2003. Conservation status of live United States nonmarine turtles in domestic and international trade. Report to Division of Scientific Authority, United States Fish and Wildlife Service.
- Reed RN, Shine R. 2002. Lying in wait for extinction: ecological correlates of conservation status among Australian Elapid snakes. *Conserv Biol* 16:451–461.
- Roe JH, Gibson J, Kingsbury BA. 2006. Beyond the wetland border: estimating the impact of roads for two species of water snakes. *Biol Conserv* 130:161–168.
- Roe JH, Kingsbury BA, Herbert NR. 2003. Wetland and upland use patterns in semi-aquatic snakes: implications for wetlands conservation. *Wetlands* 23:1003–1014.
- Roe JH, Kingsbury BA, Herbert NR. 2004. Comparative snake ecology: conservation of mobile animals that use temporally dynamic resources. *Biol Conserv* 118:79–89.
- Romero LM, Wikelski W. 2002. Severe effects of low-level oil contamination on wildlife predicted by the corticosterone-stress response: preliminary data and a research agenda. *Spill Sci Technol Bull* 7:309–313.
- Sala OE, Chapin FSI, Armesto JJ, Berlow E, Bloomfield J, Dirzo R, Huber-Sanwald E, Huenneke LJ, Jackson RB, Kinzin A, Leemans R, Lodge DM, Mooney HA, Oesterheld M, Poff NL, Sykes MT, Walker BH, Walker M, Wall DH. 2000. Global biodiversity scenarios for the year 2100. *Science* 287:1770–1774.
- Schlaepfer MA, Hoover C, Dodd Jr CK. 2005. Challenges in evaluating the impact of the trade in amphibians and reptiles on wild populations. *Bioscience* 55:256–264.
- Seigel RA, Gibbons JW, Lynch TK. 1995. Temporal changes in reptile populations: effects of a severe drought on aquatic snakes. *Herpetologica* 51:424–434.
- Seigel RA, Smith RB, Seigel NA. 2003. Swine flu or 1918 pandemic? Upper respiratory tract disease and the sudden mortality of gopher tortoises (*Gopherus polyphemus*) on a protected habitat in Florida. *J Herpetol* 37:137–144.
- Semenza JC, Tolbert PE, Rubin CH, Guillette Jr LJ, Jackson RJ. 1997. Reproductive toxins and alligator abnormalities at Lake Apopka, FL. *Environ Health Perspect* 105:1030–1032.
- Shelby JA, Mendonca MT. 2001. Comparison of reproductive parameters in male yellow-blotched map turtles (*Graptemys flavimaculata*) from a historically contaminated site and a reference site. *Comp Biochem Physiol C Toxicol Pharmacol* 129:233–242.
- Shine R, Ambariyanto, Harlow PS, Mumpuni. 1999. Reticulated pythons in Sumatra: biology, harvesting and sustainability. *Biol Conserv* 87:349–357.
- Shine R, Webb JK, Fitzgerald M, Sumner J. 1998. The impact of bush-rock removal on an endangered snake species, *Hoplocephalus bungaroides* (Serpentes: Elapidae). *Wildl Res* 1998:285–295.
- Spotila JR. 2004. *Sea turtles: a complete guide to their biology, behavior, and conservation*. Baltimore: Johns Hopkins University Press.



- Steen DA, Aresco MJ, Beilke SG, Compton BW, Condon EP, Dodd Jr CK, Forrester H, Gibbons JW, Greene JL, Johnson G, Langen TA, Oldham MJ, Oxier DN, Saumure RA, Schueler FW, Sleeman JM, Smith LL, Tucker JK, Gibbs JP. 2006. Relative vulnerability of female turtles to road mortality. *Anim Conserv* 9:269–273.
- Stevenson DJ, Dyer KJ, Willis-Stevenson BA. 2003. Survey and monitoring of the eastern indigo snake in Georgia. *Southeastern Natural* 2:393–408.
- Still CJ, Foster PN, Schneider SH. 1999. Simulating the effects of climate change on tropical montane cloud forests. *Nature* 398:608–610.
- Suarez AV, Bolger DT, Case TJ. 1998. Effects of fragmentation and invasion on native ant communities in coastal southern California. *Ecology* 79:2041–2056.
- Suarez AV, Case TJ. 2002. Bottom-up effects on persistence of a specialist predator: ant invasions and horned lizards. *Ecol Appl* 12:291–298.
- Suarez AV, Holway DA, Case TJ. 2001. Patterns of spread in biological invasions dominated by long-distance jump dispersal: insights from Argentine ants. *Proc Natl Acad Sci USA* 98:1095–1100.
- Thompson R, Starzomski BM. 2007. What does biodiversity actually do? A review for managers and policy makers. *Biodiv Conserv* 16:1359–1378.
- Thorbjarnarson JB, Perez N, Escalona T. 1997. Biology and conservation of aquatic turtles in Cinaruco-Capanaparo National Park, Venezuela. In: Van Abbema J, editor, Conservation, restoration, and management of tortoises and turtles — an international conference. Purchase (NY): New York Turtle and Tortoise Society and WCS Turtle Recovery Program, State University of New York, p 109–112.
- Timmerman WW, Martin WH. 2003. Conservation guide to the eastern diamondback rattlesnake *Crotalus adamanteus*. *Herpetol Circ* 32.
- Todd BD, Andrews KM. 2008. Response of a reptile guild to forest harvesting. *Conserv Biol* 22:753–761.
- Todd BD, Rothermel BB, Reed RN, Luhring TM, Schlatter K, Trenkamp L, Gibbons JW. 2008. Habitat alteration increases invasive fire ant abundance to the detriment of amphibians and reptiles. *Biol Invasions* 10:539–546.
- Todd BD, Winne CT, Willson JD, Gibbons JW. 2007. Getting the drift: examining the effects of timing, trap type, and taxon on herpetofaunal drift fence surveys. *Am Midland Natural* 158:292–305.
- Toledo LF, Ribeiro RS, Haddad CFB. 2007. Anurans as prey: an exploratory analysis and size relationships between predators and their prey. *J Zool* 271:170–177.
- Tuberville TD, Bodie JR, Jensen JB, LaClaire L, Gibbons JW. 2000. Apparent decline of the southern hog-nosed snake, *Heterodon simus*. *J Elisa Mitchell Sci Soc* 116:19–40.
- Valentine LE. 2006. Habitat avoidance of an introduced weed by native lizards. *Austral Ecol* 31:732–735.
- Ware S, Frost C, Doerr PD. 1993. Southern mixed hardwood forest: the former longleaf pine forest. In: Martin WH, Boyce SG, Echternacht AC, editors, Biodiversity of the southeastern United States. New York: John Wiley & Sons, p 447–493.
- Webb JK, Brook BW, Shine R. 2002. Collectors endanger Australia's most threatened snake, the broad-headed snake *Hoplocephalus bungaroides*. *Oryx* 36:170–181.
- Webb JK, Shine R, Pringle RM. 2005. Canopy removal restores habitat quality for an endangered snake in a fire suppressed landscape. *Copeia* 2005:894–900.
- Weldon C, du Preez L, Hyatt AD, Muller R, Speare R. 2004. Origin of the amphibian chytrid fungus. *Emerg Infect Dis* 10:2100–2105.
- Whiles MR, Lips KR, Pringle CM, Kilham SS, Bixby RJ, Brenes R, Connelly S, Colon-Gaud JC, Hunte-Brown M, Huryn AD, Montgomery C, Peterson S. 2006. The effects of amphibian population declines on the structure and function of neotropical stream ecosystems. *Front Ecol Environ* 4:27–34.
- Whitfield SM, Bell KE, Philippi T, Sasa M, Bolanos F, Chaves G, Savage JM, Donnelly MA. 2007. Amphibian and reptile declines over 35 years at La Selva, Costa Rica. *Proc Natl Acad Sci USA* 104:8352–8356.
- Wilcove DS, Rothstein D, Dubow J, Phillips A, Losos E. 1998. Quantifying threats to imperiled species in the United States. *Bioscience* 48:607–615.
- Willcox AS, Nambu DM. 2007. Wildlife hunting practices and bushmeat dynamics of the Banyangi and Mbo people of Southwestern Cameroon. *Biol Conserv* 134:251–261.
- Willson JD, Winne CT, Dorcas ME, Gibbons JW. 2006. Post-drought responses of semi-aquatic snakes inhabiting an isolated wetland: insights on different strategies for persistence in a dynamic habitat. *Wetlands* 26:1071–1078.
- Winne CT, Willson JD, Todd BD, Andrews KM, Gibbons JW. 2007. Enigmatic decline of a protected population of eastern kingsnakes, *Lampropeltis getula*. *Copeia* 2007:507–519.
- Wojcik DP, Allen CR, Brenner RJ, Forsys EA, Jouvenaz DP, Lutz RS. 2001. Red imported fire ants: impact on biodiversity. *Am Entomol* 47:16–23.



- Zachos J, Pagani M, Sloan L, Thomas E, Billups K. 2001. Trends, rhythms, and aberrations in global climate 65 Ma to present. *Science* 292:686–687.
- Zhou Z, Jiang Z. 2004. International trade status and crisis for snake species in China. *Conserv Biol* 18:1386–1394.
- Zhou Z, Jiang Z. 2005. Identifying snake species threatened by economic exploitation and international trade in China. *Biodiv Conserv* 14:3525–3536.
- Zim HS, Smith HM. 1953. *Reptiles and amphibians: a guide to familiar American species*. New York: Golden Press.
- Zug GR, Vitt LJ, Caldwell JP. 2001. *Herpetology: an introductory biology of amphibians and reptiles*. New York: Academic Press.